Endangered Species Act - Section 7 Consultation

Biological Opinion

Action Agency:	National Oceanic and Atmospheric Administration (NOAA), National Marine Fisheries Service (NMFS), Southeast Regional Office (SERO), Sustainable Fisheries Division (F/SER2)
Activity:	Endangered Species Act (ESA) Section 7 Consultation on the authorization and management of the Puerto Rico fishery under the Puerto Rico Fishery Management Plan (FMP), the St. Thomas/St. John fishery under the St. Thomas/St. John FMP, and the St. Croix fishery under the St. Croix FMP SERO-2019-04047
Consulting Agency:	NOAA, NMFS, SERO, Protected Resources Division (F/SER3)
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List of Frequently Used Acronyms

BMP	Best management practice
CFR	Code of Federal Regulations
CPUE	Catch Per Unit Effort
DPS	Distinct Population Segment
DWH	Deepwater Horizon
DTRU	Dry Tortugas Recovery Unit
ESA	Endangered Species Act
FDEP	Florida Department of Environmental Protection
FP	Fibropapillomatosis disease
FWRI	Fish and Wildlife Research Institute
GCRU	Greater Caribbean Recovery Unit
ITS	Incidental Take Statement
NA	North Atlantic
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Association
NWA	Northwest Atlantic
PRM	Post-release mortality
RC	Restoration Center
RPMs	Reasonable and Prudent Measures
SA	South Atlantic
SAV	Submerged aquatic vegetation
SCL	Straight carapace length
SEFSC	Southeast Fisheries Science Center
STSSN	Sea Turtle Stranding and Salvage Network
USFWS	U.S. Fish and Wildlife Service

Units of Measurement

°C	Degrees Celsius
°F	Degrees Fahrenheit
cm	Centimeter(s)
ft	Feet
ft^2	Square feet
in	Inch(es)
g	Grams
kg	Kilograms
lb	Pound(s)
mi	Mile(s)
mi^2	Square mile(s)

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Introduction

Section 7(a)(2) of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. § 1531 et seq.), requires each federal agency to insure that any action that it authorizes, funds, or carries out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of any critical habitat of such species. To fulfill this obligation, Section 7(a)(2) requires federal agencies to consult with the appropriate Secretary on any action they propose that "may affect" listed species or designated critical habitat. The National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service (USFWS) share responsibilities for administering the ESA.

Consultations on most listed marine species and their designated critical habitat are conducted between the action agency and NMFS. The consultation is concluded after NMFS concurs with an action agency that its action is not likely to adversely affect listed species or critical habitat or 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 its critical habitat. If jeopardy or destruction or adverse modification is found to be likely, the Opinion identifies reasonable and prudent alternatives (RPAs) to the action as proposed, if any, that can avoid jeopardizing listed species or resulting in the destruction/adverse modification of critical habitat. The Opinion states the amount or extent of incidental take of the listed species that may occur, specifies reasonable and prudent measures (RPMs) that are required to minimize the impacts of incidental take and monitoring to validate the expected effects of the action, and recommends conservation measures to further conserve the species.

This document represents NMFS's Opinion on the effects of approving the proposed Puerto Rico Fishery Management Plan (FMP), the St. Thomas/St. John FMP, and the St. Croix FMP (evaluation of the 3 actions is batched in one Opinion) in the U.S. Caribbean Exclusive Economic Zone (EEZ) on threatened and endangered species and designated critical habitat, in accordance with Section 7 of the ESA. This consultation considers the operation of these fisheries as they are managed under their respective proposed fishery management plans. NMFS has dual responsibilities as both the action agency under the Magnuson-Stevens Fishery Conservation and Management Act (MSA) (16 U.S.C. §1801 et seq.) and the consulting agency under the ESA. For the purposes of this consultation, the Southeast Regional Office (SERO) Sustainable Fisheries Division (SFD) is considered the action agency and the consulting agency is the SERO Protected Resources Division (PRD).

This Opinion has been prepared in accordance with Section 7 of the ESA and regulations promulgated to implement that section of the ESA. It is based on information provided in the proposed FMPs for each island group, as well as information provided in recovery plans, NMFS databases, and other relevant published and unpublished scientific and commercial data cited in the Literature Cited section of this document. During this consultation, we conducted electronic searches of the general scientific literature. We also contacted subject matter experts (e.g., NMFS science center staff) for informaton. These searches specifically sought to identify data or other information that supports a particular conclusion (for example, a study that suggests a species will respond to a stimulus in a certain way) as well as data that does not support our conclusion. When data are equivocal, or in the face of uncertainty, our decisions are designed to avoid the risks of inaccurately concluding that an action is not likely to have an adverse effect on listed species.

1.0 Consultation History

Historical Fishery Management Context

To date, federal fisheries throughout the U.S. Caribbean EEZ have been managed under four U.S. Caribbean-wide FMPs: the FMP for the Reef Fish Fishery of Puerto Rico and the U.S. Virgin Islands (USVI) (Reef Fish FMP); the FMP for the Spiny Lobster Fishery of Puerto Rico and the USVI (Spiny Lobster FMP); the FMP for the Queen Conch Resources of Puerto Rico and the USVI (Queen Conch FMP); and, the FMP for Corals and Reef Associated Plants and Invertebrates of Puerto Rico and the USVI (Coral FMP). NMFS has consulted on these FMPs since the late 1980s. The most recent consultations on the Reef Fish FMP and Spiny Lobster FMP were completed in 2011 (NMFS 2011a; NMFS 2011b). The FMPs managed the species across the U.S. Caribbean island areas.

While the current consultation is on 3 new, stand alone Caribbean FMPs, we mention these previous FMPS as they provide the foundation for the creation of the new island-based FMPs. While the new FMPs make changes to some of the management measures (e.g., species to be managed and annual catch limits (ACLs)), they retain many others (e.g., closed areas and seasons, recreational bag limits, size limits). Fishing activities (e.g., gear and overall effort) will be largely unchanged and this Opinion will draw upon some of the information learned from the earlier consultations.

Creation of New Island Based FMPs- The Current Proposed Actions

The Council has proposed the creation of three new independent island-based FMPs (the Puerto Rico FMP, St. Thomas/St. John FMP, and the St. Croix FMP), and has submitted these FMPs to NMFS for approval. These FMPs will replace the previous FMPs. Since the implementation of the new FMPs constitute federal actions, SFD is initiating consultation on them.

A complete description of the proposed actions was received from SFD on April 14, 2020. On April 14, 2020, SERO initiated formal consultation. This Opinion batches the consultations for the 3 FMPs together. These are all separate federal actions, however they are related, as each is derived from the former species-based plans applicable across the Caribbean EEZ. The types of gear and anticipated impacts to ESA-listed species and designated critical habitat are similar across the FMPs. For consultation efficiency and comprehensive evaluation of overall effects to the species and critical habitat affected (all FMPs affect the same populations of protected species and designated critical habitat), the new FMPs are considered together in this single batched consultation.

2.0 Description of the Proposed Actions and Action Areas

The National Marine Fisheries Service (NMFS) proposes to approve and implement three fishery management plans (FMP) developed by the Caribbean Fishery Management Council (Council) that would manage fishery resources in the U.S. Caribbean exclusive economic zone (EEZ), and promulgate regulations implementing those plans. The three plans are (1) the Comprehensive FMP for the Puerto Rico EEZ (Puerto Rico FMP), (2) the Comprehensive FMP for the St. Thomas/St. John EEZ (St. Thomas/St. John FMP), and (3) the Comprehensive FMP for the St. Croix EEZ (St. Croix FMP) (hereafter referred to as the island-based FMPs). The following subsections summarize the overall characteristics of the Puerto Rico, St. Thomas/St. John, and St. Croix fisheries in the U.S. Caribbean, as managed under the proposed FMPs, that are relevant to the analysis of their potential effects on threatened and endangered species and designated critical habitat.

2.1 Overview of Management and Regulations

Responsibility for federal fishery management decision-making in the U.S. Caribbean is divided between the Secretary of Commerce (Secretary) and the Council. The Council is responsible for preparing, monitoring, and revising management plans for fisheries needing “conservation and management” within their jurisdiction as well as proposed regulations that the Council “deems necessary or appropriate” to implement the FMP. 16 U.S.C. §§ 1852(h)(1), 1853(c). The Magnuson-Stevens Fishery Conservation and Management Act (Magnuson-Stevens Act) is the principal federal statute governing the management of U.S. marine fisheries. The Secretary has responsibility for approving, disapproving, or partially approving the FMPs, after evaluating whether they are consistent with the Magnuson-Stevens Act and with other applicable laws. 16 U.S.C. § 1854(a)(3). In addition, the Secretary reviews the proposed regulations for consistency with the FMP and with applicable law and is responsible for promulgating appropriate regulations. 16 U.S.C. § 1854(b). The Secretary has delegated this authority to NMFS.

To date, the Council has managed federal fisheries throughout the U.S. Caribbean EEZ under four U.S. Caribbean-wide FMPs: the FMP for the Reef Fish Fishery of Puerto Rico and the U.S. Virgin Islands (USVI) (Reef Fish FMP); the FMP for the Spiny Lobster Fishery of Puerto Rico and the USVI (Spiny Lobster FMP); the FMP for the Queen Conch Resources of Puerto Rico and the USVI (Queen Conch FMP); and, the FMP for Corals and Reef Associated Plants and Invertebrates of Puerto Rico and the USVI (Coral FMP).

The three island-based FMPs proposed by the Council would replace the four U.S. Caribbean-wide FMPs and transition from U.S. Caribbean-wide to island-based management. Each plan would govern resources in U.S. Caribbean EEZ off each corresponding management area: Puerto Rico (Appendix A, Table A.1), St. Thomas/St. John (Appendix A, Table A.3), and St. Croix (Appendix A, Table A.5). The proposed action to approve and implement those plans would modify regulations at 50 CFR part 622 under the authority of the Magnuson-Stevens Act.

Each island-based FMP contains a description of the history of management and implementing regulations for the Reef Fish, Spiny Lobster, Queen Conch, and Coral fisheries (see Appendix C in each island-based FMP) and all management measures that will be applicable to fishery resources in each FMP’s geographic scope following approval and implementation of the FMPs (see Chapter 5 in each FMP).[1] Each island-based FMP would subsume pertinent fishery management measures from the Caribbean-wide FMPs that remain applicable for the new FMP, and would modify select management measures specific to each island management area, as necessary and appropriate. Modified management measures in each island-based FMP include the species to be managed, how the species are organized for management (as individual species, or in species groups with or without one or more indicator species), management reference points and status determination criteria for species for which management is continued, and updated framework measures.[2] In addition, each island-based FMP specifies management measures for species new to management, including management reference points and status determination criteria, and identifies and describes essential fish habitat for those newly managed species.

Each new FMP would manage a select number of reef fish and pelagic fish species, spiny lobster and queen conch, and all coral, sea cucumber, and sea urchin species within the management area (Table 2.1). The Puerto Rico FMP would also manage three ray species. Species would be managed with annual catch limits, annual catch targets for the pelagic species, and accountability measures. Additionally, each FMP would prohibit harvest of species that are ecologically important as habitat (all corals) or habitat engineers (midnight, blue, rainbow parrotfish), species that were previously classified as overfished in U.S. Caribbean federal waters based on NMFS determination (Nassau grouper, goliath grouper, and queen conch), and other species that the Council determined were particularly vulnerable to fishing activities (sea cucumbers, sea urchins, and, for Puerto Rico only, rays). The FMPs would also prohibit harvest of certain species in particular areas or during particular times of year, prohibit the use of certain gear types, describe certain allowable gear types, set size and bag limits, and prohibit anchoring in specific areas. These measures and others pertinent to understanding how the fisheries operate and for evaluating potential effects on ESA-listed species and designated critical habitat are described in Table 2.2 (Puerto Rico), Table 2.3 (St. Thomas/St. John), and Table 2.4 (St. Croix). The list of management measures in these tables is not exhaustive. For a complete list of management measures, please see Chapter 5 in each of the island-based FMPs.

Table 2.1. Species included for management under the Puerto Rico FMP, St. Thomas/St. John FMP, and St. Croix FMP. Species marked with a dash (-) would not be included in the respective island-based FMP.

Species Name	Common Name	Management Category	Puerto Rico FMP	St. Thomas/ St. John FMP	St. Croix FMP
Lobatus gigas	Queen conch	Queen Conch	Managed	Managed	Managed
Panulirus argus	Caribbean spiny lobster	Spiny Lobster	Managed	Managed	Managed
Apsilus dentatus	Black snapper	Reef Fish	Managed	Managed	Managed

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[1] "Chapter 5: Conservation and Management Measures" in each FMP was prepared prior to NMFS's decision as to whether to approve, disapprove, or partially approve the FMPs and to promulgate implementing regulations, if the FMPs are approved in whole or in part. If there are any differences between Chapter 5 and the regulations implementing the FMPs, the regulations will control.

[2] Framework measures provide the Council and NMFS the flexibility to more expeditiously adjust management measures to respond to changing fishery conditions or new scientific information. Each island-based FMP would expand or modify the range of existing management measures that can be implemented by the Council by framework process.

Species Name	Common Name	Management Category	Puerto Rico FMP	St. Thomas/ St. John FMP	St. Croix FMP
Lutjanus buccanella	Blackfin snapper	Reef Fish	Managed	Managed	Managed
Lutjanus vivanus	Silk snapper	Reef Fish	Managed	Managed	Managed
Rhomboplites aurorubens	Vermilion snapper	Reef Fish	Managed	Managed	Managed
Lutjanus synagris	Lane snapper	Reef Fish	Managed	Managed	Managed
Lutjanus analis	Mutton snapper	Reef Fish	Managed	Managed	Managed
Ocyurus chrysurus	Yellowtail snapper	Reef Fish	Managed	Managed	Managed
Pristipomoides aquilonaris	Wenchman	Reef Fish	Managed	-	-
Pristipomoides macrophthalmus	Cardinal snapper	Reef Fish	Managed	-	-
Etelis oculatus	Queen snapper	Reef Fish	Managed	Managed	Managed
Lutjanus jocu	Dog snapper	Reef Fish	Managed	-	-
Lutjanus apodus	Schoolmaster	Reef Fish	Managed	-	Managed
Lutjanus griseus	Gray snapper	Reef Fish	-	-	Managed
Lutjanus cyanopterus	Cubera snapper	Reef Fish	Managed	-	-
Epinephelus striatus	Nassau Grouper	Reef Fish	Managed	Managed	Managed
Epinephelus itajara	Goliath grouper	Reef Fish	Managed	Managed	Managed
Epinephelus guttatus	Red hind	Reef Fish	Managed	Managed	Managed
Mycteroperca bonaci	Black grouper	Reef Fish	Managed	Managed	Managed
Epinephelus morio	Red grouper	Reef Fish	Managed	Managed	Managed
Mycteroperca tigris	Tiger grouper	Reef Fish	Managed	Managed	Managed
Mycteroperca venenosa	Yellowfin grouper	Reef Fish	Managed	Managed	Managed
Hyporthodus flavolimbatus	Yellowedge grouper	Reef Fish	Managed	Managed	-
Cephalopholis fulva	Coney	Reef Fish	Managed	Managed	Managed
Cephalopholis cruentatus	Graysby	Reef Fish	Managed	-	Managed
Epinephelus adscensionis	Rock hind	Reef Fish	Managed	-	Managed
Hyporthodus mystacinus	Misty grouper	Reef Fish	Managed	Managed	Managed
Mycteroperca interstitialis	Yellowmouth grouper	Reef Fish	Managed	Managed	-
Scarus coeruleus	Blue parrotfish	Reef Fish	Managed	Managed	Managed
Scarus coelestinus	Midnight parrotfish	Reef Fish	Managed	Managed	Managed
Scarus guacamaia	Rainbow parrotfish	Reef Fish	Managed	Managed	Managed
Scarus vetula	Queen parrotfish	Reef Fish	Managed	Managed	Managed
Scarus taeniopterus	Princess parrotfish	Reef Fish	Managed	Managed	Managed
Sparisoma chrysopterum	Redtail parrotfish	Reef Fish	Managed	Managed	Managed
Sparisoma viride	Stoplight parrotfish	Reef Fish	Managed	Managed	Managed
Sparisoma aurofrenatum	Redband parrotfish	Reef Fish	Managed	Managed	Managed
Scarus iseri	Striped parrotfish	Reef Fish	Managed	Managed	Managed
Sparisoma rubripinne	Redfin parrotfish	Reef Fish	-	Managed	Managed
Acanthurus coeruleus	Blue tang	Reef Fish	Managed	Managed	Managed
Acanthurus bahianus	Ocean surgeonfish	Reef Fish	Managed	Managed	Managed
Acanthurus chirurgus	Doctorfish	Reef Fish	Managed	Managed	Managed
Canthidermis sufflamen	Ocean triggerfish	Reef Fish	Managed	-	-
Balistes vetula	Queen triggerfish	Reef Fish	Managed	Managed	Managed

Species Name	**Common Name**	**Management Category**	**Puerto Rico FMP**	**St. Thomas/ St. John FMP**	**St. Croix FMP**
Balistes capriscus	Gray triggerfish	Reef Fish	Managed	-	-
Lachnolaimus maximus	Hogfish	Reef Fish	Managed	Managed	-
Halichoeres radiates	Puddingwife	Reef Fish	Managed	-	-
Bodianus rufus	Spanish hogfish	Reef Fish	Managed	-	-
Holacanthus ciliaris	Queen angelfish	Reef Fish	Managed	Managed	Managed
Pomacanthus arcuatus	Gray angelfish	Reef Fish	Managed	Managed	Managed
Pomacanthus paru	French angelfish	Reef Fish	Managed	Managed	Managed
Haemulon plumierii	White grunt	Reef Fish	Managed	Managed	Managed
Haemulon album	Margate	Reef Fish	-	Managed	-
Haemulon sciurus	Bluestriped grunt	Reef Fish	-	Managed	Managed
Caranx hippos	Crevalle jack	Reef Fish	Managed	-	-
Alectis ciliaris	African pompano	Reef Fish	Managed	-	-
Elagatis bipinnulata	Rainbow runner	Reef Fish	Managed	-	-
Caranx crysos	Blue runner	Reef Fish	-	Managed	-
Calamus bajonado	Jolthead porgy	Reef Fish	-	Managed	-
Archosargus rhomboidalis	Sea bream	Reef Fish	-	Managed	-
Calamus penna	Sheepshead porgy	Reef Fish	-	Managed	-
Calamus calamus	Saucereye porgy	Reef Fish	-	Managed	-
Holocentrus rufus	Longspine squirrelfish	Reef Fish	-	-	Managed
Sphyraena barracuda	Great barracuda	Pelagic	Managed	-	-
Lobotes surinamensis	Tripletail	Pelagic	Managed	-	-
Coryphaena hippurus	Dolphin	Pelagic	Managed	Managed	Managed
Coryphaena equiselis	Pompano dolphin	Pelagic	Managed	-	-
Euthynnus alletteratus	Little tunny	Pelagic	Managed	-	-
Thunnus atlanticus	Blackfin tuna	Pelagic	Managed	-	-
Scomberomorus cavalla	King mackerel	Pelagic	Managed	-	-
Scomberomorus regalis	Cero mackerel	Pelagic	Managed	-	-
Acanthocybium solandri	Wahoo	Pelagic	Managed	Managed	Managed
Manta birostris	Giant manta ray	Rays	Managed	-	-
Aetobatus narinari	Spotted eagle ray	Rays	Managed	-	-
Hypanus americanus	Southern stingray	Rays	Managed	-	-
Multiple species*		Corals	Managed	Managed	Managed
Multiple species*		Sea Urchins	Managed	Managed	Managed
Multiple species*		Sea Cucumbers	Managed	Managed	Managed

* All coral, sea urchin, and sea cucumber species that occur in each management area would be managed under each FMP. Appendix E of each FMP includes a list of known coral and echinoderm species included in the FMP.

Table 2.2. Summary of federal regulations in the Puerto Rico management area.

Puerto Rico Federal Management Measures
Prohibited Species
No person may fish for or possess goliath grouper, Nassau grouper, blue parrotfish, midnight parrotfish, rainbow parrotfish, giant manta ray, spotted eagle ray, southern stingray, or queen conch in or from the Puerto Rico EEZ. Such fish caught in the Puerto Rico EEZ must be released immediately with a minimum of harm.
No person may fish for or possess any species of coral (e.g., stony corals, octocorals, black corals), sea urchins, or sea cucumbers in or from the Puerto Rico EEZ. The taking of a managed coral, sea urchin, or sea cucumber in the Puerto Rico EEZ is not considered unlawful possession provided it is returned immediately to the sea in the general area of fishing.
Area Closures
From December 1 through the last day of February, each year, fishing is prohibited in those parts of Tourmaline Bank and Abrir La Sierra Bank areas off western Puerto Rico that are in federal waters (see map and Appendix A, Table A.2).
From October 1 through March 31, each year, no person may fish for or possess Puerto Rico reef fish species in or from those parts of the Bajo de Sico area off western Puerto Rico that are in federal waters (see map and Appendix A, Table A.2).
Seasonal Species Closures
From December 1 through the last day of February, each year, no person may fish for or possess red hind in or from the Puerto Rico EEZ west of 67°10′ W. longitude (see map).
From October 1 through December 31, each year, no person may fish for or possess vermilion, black, silk, or blackfin snapper in or from the Puerto Rico EEZ.
From April 1 through June 30, each year, no person may fish for or possess lane or mutton snapper in or from the Puerto Rico EEZ.
From February 1 through April 30, each year, no person may fish for or possess yellowfin, red, tiger, black or yellowedge grouper in or from the Puerto Rico EEZ.
Gear Prohibitions and/or Harvest Restrictions
Fishing with pots, traps, bottom longlines, or gill or trammel nets is prohibited in federal waters of Bajo de Sico, Tourmaline Bank, and Abrir la Sierra Bank year-round.
An explosive (except an explosive in a powerhead where a powerhead is an allowable gear) may not be used to fish in the Puerto Rico EEZ. A vessel fishing in the Puerto Rico EEZ for a species managed under the Puerto Rico FMP, may not have on board any dynamite or similar explosive substance.
A poison, drug, or other chemical may not be used to fish for Puerto Rico reef fish in the Puerto Rico EEZ.
A powerhead may not be used in the Puerto Rico EEZ to harvest Puerto Rico reef fish.
A gillnet or trammel net may not be used in the Puerto Rico EEZ to fish for Puerto Rico reef fish or spiny lobster.
A fish trap[3] used or possessed in the Puerto Rico EEZ must have an escape mechanism as defined and comply with minimum mesh size regulations as described in the Puerto Rico FMP.
Finfish must be maintained with head and fins intact.[4]
A spear, hook, or similar device may not be used in the Puerto Rico EEZ to harvest a spiny lobster.

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[3] In the Puerto Rico EEZ, a fish trap is defined as a trap and its component parts (including the lines and buoys), regardless of the construction material, used for or capable of taking finfish, except a trap used in the directed fishery for spiny lobster.

[4] Exceptions for bait and fish consumed at sea. See Chapter 5 of the Puerto Rico FMP for more details. Shark, swordfish, and tuna species are subject to specific requirements applicable to highly migratory species.

Puerto Rico Federal Management Measures
A spiny lobster trap used or possessed in the Puerto Rico EEZ must have an escape mechanism as described in the Puerto Rico FMP.
Egg-bearing spiny lobster must be returned to the water unharmed. An egg-bearing spiny lobster may not be stripped, scraped, shaved, clipped, or in any other manner molested, in order to remove the eggs.
A spiny lobster must be maintained with head and carapace intact.
Anchoring Restrictions
Anchoring by fishing vessels is prohibited year-round in federal waters of Bajo de Sico.
The owner or operator of any fishing vessel, recreational or commercial that fishes for or possesses Puerto Rico reef fish in or from the Puerto Rico EEZ must ensure that the vessel uses only an anchor retrieval system that recovers the anchor by its crown, thereby preventing the anchor from dragging along the bottom during recovery. For a grapnel hook, this could include an incorporated anchor rode reversal bar that runs parallel along the shank, which allows the rode to reverse and slip back toward the crown. For a fluke or plow type anchor, a trip line consisting of a line from the crown of the anchor to a surface buoy would be required
Size Limits
Yellowtail snapper minimum size limit of 12 in (30.5 cm) total length.
Spiny lobster minimum size limit of 3.5 in (8.9 cm) carapace length.
Bag Limits
Aggregate bag limit for snapper, grouper and parrotfish combined: 5 fish per person/day, of which no more than 2 may be parrotfish, or, if 3 or more persons are aboard, 15 fish total per vessel/day, of which no more than 6 may be parrotfish.
Aggregate bag limit for angelfish, grunts, wrasses, jacks, triggerfish, surgeonfish, combined: 5 fish per person/day, of which no more than 1 may be surgeonfish, or, if 3 or more persons are aboard, 15 fish total per vessel/day, of which no more than 4 may be surgeonfish.
Spiny lobster: 3 spiny lobsters per person/day, not to exceed 10 spiny lobsters per vessel/day, whichever is less.

Table 2.3. Summary of federal regulations in the St. Thomas/St. John management area.

St. Thomas/St. John Federal Management Measures
Prohibited Species
No person may fish for or possess goliath grouper, Nassau grouper, blue parrotfish, midnight parrotfish, rainbow parrotfish, or queen conch in or from the St. Thomas/St. John EEZ. Such fish caught in the St. Thomas/St. John EEZ must be released immediately with a minimum of harm.
No person may fish for or possess any species of coral (e.g., stony corals, octocorals, black corals), sea urchins, or sea cucumbers in or from the St. Thomas/St. John EEZ. The taking of a managed coral, sea urchin, or sea cucumber in the St. Thomas/St. John EEZ is not considered unlawful possession provided it is returned immediately to the sea in the general area of fishing.
Area Closures
Fishing for any species is prohibited year-round in the Hind Bank Marine Conservation District, south of St. Thomas (see map and Appendix A, Table A.4).
From February 1 through April 30, each year, no person may fish for or possess any species of fish, except for highly migratory species, in or from the Grammanik Bank closed area, south of St. Thomas (see map and Appendix A, Table A.4).
Seasonal Species Closures
From October 1 through December 31, each year, no person may fish for or possess vermilion, black, silk, or blackfin snapper in or from the St. Thomas/St. John EEZ.

St. Thomas/St. John Federal Management Measures
From April 1 through June 30, each year, no person may fish for or possess lane or mutton snapper in or from the St. Thomas/St. John EEZ.
From February 1 through April 30, each year, no person may fish for or possess yellowfin, red, tiger, black, or yellowedge grouper in or from the St. Thomas/St. John EEZ.
Gear Prohibitions and/or Harvest Restrictions
Fishing with pots, traps, bottom longlines, gillnets, or trammel nets is prohibited year-round in Grammanik Bank area.
An explosive (except an explosive in a powerhead where a powerhead is an allowable gear) may not be used to fish in the St. Thomas/St. John EEZ. A vessel fishing in the St. Thomas/St. John EEZ for a species managed under the St. Thomas/St. John FMP, may not have on board any dynamite or similar explosive substance
A poison, drug, or other chemical may not be used to fish for St. Thomas/St. John reef fish in the St. Thomas/St. John EEZ.
A powerhead may not be used in the St. Thomas/St. John EEZ to harvest St. Thomas/St. John reef fish.
A gillnet or trammel net may not be used in the St. Thomas/St. John EEZ to fish for St. Thomas/St. John reef fish or spiny lobster.
A fish trap[5] used or possessed in the St. Thomas/St. John EEZ must have an escape mechanism as defined and comply with minimum mesh size regulations as described in the St. Thomas/St. John FMP.
Finfish must be maintained with head and fins intact.[6]
A spear, hook, or similar device may not be used in the St. Thomas/St. John EEZ to harvest a spiny lobster.
A spiny lobster trap used or possessed in the St. Thomas/St. John EEZ must have an escape mechanism as described in the St. Thomas/St. John FMP.
Egg-bearing spiny lobster must be returned to the water unharmed. An egg-bearing spiny lobster may not be stripped, scraped, shaved, clipped, or in any other manner molested, in order to remove the eggs.
A spiny lobster must be maintained with head and carapace intact.
Anchoring Restrictions
Anchoring by fishing vessels is prohibited year-round in the Hind Bank Marine Conservation District, south of St. Thomas.
The owner or operator of any fishing vessel, recreational or commercial that fishes for or possesses St. Thomas/St. John reef fish in or from the St. Thomas/St. John EEZ must ensure that the vessel uses only an anchor retrieval system that recovers the anchor by its crown, thereby preventing the anchor from dragging along the bottom during recovery. For a grapnel hook, this could include an incorporated anchor rode reversal bar that runs parallel along the shank, which allows the rode to reverse and slip back toward the crown. For a fluke or plow type anchor, a trip line consisting of a line from the crown of the anchor to a surface buoy would be required
Size Limits
Yellowtail snapper minimum size limit of 12 in (30.5 cm) total length.
Spiny lobster minimum size limit of 3.5 in (8.9 cm) carapace length.
Bag Limits
Aggregate bag limit for snapper, grouper and parrotfish combined: 5 fish per person/day, of which no more than 2 may be parrotfish, or, if 3 or more persons are aboard, 15 fish total per vessel/day, of which no more than 6 may be parrotfish.

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[5] In the St. Thomas/St. John EEZ, a fish trap is defined as a trap and its component parts (including the lines and buoys), regardless of the construction material, used for or capable of taking finfish, except a trap used in the directed fishery for spiny lobster.

[6] Exceptions for bait and fish consumed at sea. See Chapter 5 of the St. Thomas/St. John FMP for more details. Shark, swordfish, and tuna species are subject to specific requirements applicable to highly migratory species.

St. Thomas/St. John Federal Management Measures
Aggregate bag limit for angelfish, grunts, wrasses, jacks, porgies, triggerfish, surgeonfish, combined: 5 fish per person/day, of which no more than 1 may be surgeonfish, or, if 3 or more persons are aboard, 15 fish total per vessel/day, of which no more than 4 may be surgeonfish.
Spiny lobster: 3 spiny lobsters per person/day, not to exceed 10 spiny lobsters per vessel/day, whichever is less.

Table 2.4. Summary of federal regulations in the St. Croix management area.

St. Croix Federal Management Measures
Prohibited Species
No person may fish for or possess goliath grouper, Nassau grouper, blue parrotfish, midnight parrotfish, or rainbow parrotfish in or from the St. Croix EEZ. Such fish caught in the St. Croix EEZ must be released immediately with a minimum of harm.
No person may fish for or possess any species of coral (e.g., stony corals, octocorals, black corals), sea urchins, or sea cucumbers in or from the St. Croix EEZ. The taking of a managed coral, sea urchin, or sea cucumber in the St. Croix EEZ is not considered unlawful possession provided it is returned immediately to the sea in the general area of fishing.
Area Closures
From March 1 through June 30, each year, all fishing is prohibited in the Mutton Snapper Spawning Aggregation Area in federal waters off St. Croix (see map and Appendix A, Table A.6).
From December 1 through the last day of February, each year, fishing is prohibited in the Lang Bank area east of St. Croix (see map and Appendix A, Table A.6).
Seasonal Species Closures
From October 1 through December 31, each year, no person may fish for or possess vermilion, black, silk, or blackfin snapper in or from the St. Croix EEZ.
From April 1 through June 30, each year, no person may fish for or possess lane or mutton snapper in or from the St. Croix EEZ.
From February 1 through April 30, each year, no person may fish for or possess yellowfin, red, tiger, or black grouper in or from the St. Croix EEZ.
No person may fish for or possess a queen conch in or from the St. Croix EEZ, except from November 1 through May 31 in the area east of 64°34′ W. longitude (see map).
Gear Prohibitions and/or Harvest Restrictions
Fishing with pots, traps, bottom longlines, gillnets, or trammel nets is prohibited year-round in the Lang Bank and Mutton Snapper Spawning Aggregation areas in federal waters off St. Croix.
An explosive (except an explosive in a powerhead where a powerhead is an allowable gear) may not be used to fish in the St. Croix EEZ. A vessel fishing in the St. Croix EEZ for a species managed under the St. Croix FMP, may not have on board any dynamite or similar explosive substance
A poison, drug, or other chemical may not be used to fish for St. Croix reef fish in the St. Croix EEZ.
A powerhead may not be used in the St. Croix EEZ to harvest St. Croix reef fish.
A gillnet or trammel net may not be used in the St. Croix EEZ to fish for St. Croix reef fish or spiny lobster.
A fish trap[7] used or possessed in the St. Croix EEZ must have an escape mechanism as defined and comply with minimum mesh size regulations as described in the St. Croix FMP.

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[7] In the St. Croix EEZ, a fish trap is defined as a trap and its component parts (including the lines and buoys), regardless of the construction material, used for or capable of taking finfish, except a trap used in the directed fishery for spiny lobster.

St. Croix Federal Management Measures
Finfish must be maintained with head and fins intact[8].
A spear, hook, or similar device may not be used in the St. Croix EEZ to harvest a spiny lobster.
A spiny lobster trap used or possessed in the St. Croix EEZ must have an escape mechanism as described in the St. Croix FMP.
Egg-bearing spiny lobster must be returned to the water unharmed. An egg-bearing spiny lobster may not be stripped, scraped, shaved, clipped, or in any other manner molested, in order to remove the eggs.
A spiny lobster must be maintained with head and carapace intact.
No person may harvest queen conch in the St. Croix EEZ by diving while using a device that provides a continuous air supply from the surface.
Queen conch in or from the St. Croix EEZ must be maintained with meat and shell intact.
Anchoring Restrictions
The owner or operator of any fishing vessel, recreational or commercial that fishes for or possesses St. Croix reef fish in or from the St. Croix EEZ must ensure that the vessel uses only an anchor retrieval system that recovers the anchor by its crown, thereby preventing the anchor from dragging along the bottom during recovery. For a grapnel hook, this could include an incorporated anchor rode reversal bar that runs parallel along the shank, which allows the rode to reverse and slip back toward the crown. For a fluke or plow type anchor, a trip line consisting of a line from the crown of the anchor to a surface buoy would be required
Size Limits
Yellowtail snapper minimum size limit of 12 in (30.5 cm) total length.
Redband parrotfish minimum size limit of 8 in (20.3 cm) fork length.
Princess, queen, striped, redtail, stoplight, and redfin parrotfish minimum size limit of 9 in (22.9 cm) fork length.
Spiny lobster minimum size limit of 3.5 in (8.9 cm) carapace length.
Queen conch minimum size limit is 9 in (22.9 cm) in length, that is, from the tip of the spire to the distal end of the shell, or 3/8 in (9.5 mm) in lip width at its widest point.
Commercial Trip Limit
The trip limit for queen conch in or from the St. Croix EEZ is 200 queen conch per day.
Bag Limits
Aggregate bag limit for snapper, grouper and parrotfish combined: 5 fish per person/day, of which no more than 2 may be parrotfish, or, if 3 or more persons are aboard, 15 fish total per vessel/day, of which no more than 6 may be parrotfish.
Aggregate bag limit for angelfish, grunts, triggerfish, surgeonfish, squirrelfish, combined: 5 fish per person/day, of which no more than 1 may be surgeonfish, or, if 3 or more persons are aboard, 15 fish total per vessel/day, of which no more than 4 may be surgeonfish.
Spiny lobster: 3 spiny lobsters per person/day, not to exceed 10 spiny lobsters per vessel/day, whichever is less.
Queen conch: 3 queen conch per person/day or, if more than 4 persons are aboard, not to exceed 12 queen conch per vessel/day.

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[8] Exceptions for bait and fish consumed at sea. See Chapter 5 of the St. Croix FMP for more details. Shark, swordfish, and tuna species are subject to specific requirements applicable to highly migratory species.

2.2 Description of the Puerto Rico Fishery

The Puerto Rico fishery is small-scale or artisanal in nature, and is comprised of commercial, recreational, and subsistence participants. All fishery resources are consumed on the island; there is little or no export. Commercial fishing provides sustenance and employment, while recreational fishing provides food and leisure activity for local residents and visitors. In general, commercial and recreational fishermen target similar species of fish and shellfish, including reef fish, offshore pelagic fish, and spiny lobster, among others. Persons engaged in commercial and recreational fishing also engage in subsistence fishing, or fishing for household consumption. The following description provides a general characterization of the Puerto Rico fishery operating in federal waters, for purposes of this consultation.

The Puerto Rico FMP would manage 51 species of reef fish, 9 pelagic finfish species, 3 species of rays, spiny lobster, queen conch, and all species of coral, sea urchin, and sea cucumber that occur in federal waters off Puerto Rico (Table 2.1). This list of species was tailored to the specific characteristics of the Puerto Rico EEZ.

2.2.1 Commercial Fishing in Puerto Rico

Fishing permits are not required to commercially harvest species proposed for management under the Puerto Rico FMP from federal waters off Puerto Rico. However, under Puerto Rico law, to sell fish caught in waters off Puerto Rico, a commercial fishing license is required from Puerto Rico's Department of Natural and Environmental Resources (DNER). As an obligation of the license, commercial fishermen are required to submit monthly catch reports to the DNER, which contain landings information for all fish caught by commercial fishermen, in both federal (9-200 nautical miles) and commonwealth (less than 9 nautical miles from shore) waters.

Geographically, Puerto Rico's west coast is the most productive area due to relatively shallow and extended shelf; by contrast, the north coast is the least productive because it has a narrow insular shelf (Suárez Caábro 1979).

Puerto Rico Commercial Fishermen

In 2018, there were 1,277 commercially licensed fishermen in Puerto Rico (764 full-time, 134 part-time, and 379 beginner fishermen) (D. Matos, DNER Fisheries, pers. comm. to M. Lopez, NMFS SERO, February 2019). The number of commercial fishermen that submitted catch reports in 2016 (pre-hurricane Maria) was 811 and in 2018 (post-hurricane Maria)[9] was 720. The total number of fishing trips reported in 2016 was 29,292 and in 2018 was 26,349. Of the catch reports submitted, 33% of the 2016 fishermen and 36% of the 2018 fishermen reported operating primarily in federal waters and 11% of the total fishing trips in each year were reported in federal waters. Additionally, a percentage of the commercial landings data were reported from an "unknown" location (43% of the 2016 commercial fishermen; 33% of the 2018 commercial fishermen; 10% of the trips taken in 2016; 4% of the trips taken in 2018), either

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[9] September 20, 2017, Hurricane Maria made a direct hit on Puerto Rico as a powerful Category 4 hurricane and caused great devastation and destruction throughout the whole island. Months after the hurricane hit, 60% of the population was without electrical power and commercial fishing activities were operating at 33% of its normal capacity pre-Maria (CFMC 2019a).

because the fishermen were not certain if the fishing location was in Commonwealth or federal waters, or because the location field on the catch report was left blank. It is possible that some of those "unknowns" occurred in federal waters, but the percentage is unquantifiable at this time.

Commercial fishing is a daily activity, in that fishermen leave in the morning or the evening and generally return to shore within twenty-four hours of departure. Most commercial fishing operations are manned by a captain and one helper, although diving operations generally have a captain and two helpers. Most commercial catches are landed in fishing centers, which can also serve as places to process and market fish, repair and store fishing gear and equipment, and socialize and exchange information with other fishermen (Valdés-Pizzini 1990; Griffith et al. 1992). On average, commercial fishermen take four trips per week and devote 33 hours per week to fishing and fishing-related activities (Agar et al. 2020).

Puerto Rico Commercial Fishing Vessels

In 2018, a total of 671 commercial fishing vessels and 3 commercial charter vessels were registered with the U.S. Coast Guard and thus available to operate in federal waters.

Commercial fishing vessels in Puerto Rico are relatively small, averaging 20 ft in length (Matos-Caraballo and Agar 2011). The majority of vessels are composed of a fiberglass hull or, less often, fiberglass and wood, with even fewer made of wood (Matos-Caraballo and Agar 2011). Most vessels feature a single outboard gas engine with an average 80 horsepower (Matos-Caraballo and Agar 2011). The vessels can also have one or two electric winches used on the shelf edge or in deep fishing banks to capture deep-water snappers (i.e., silk and queen snappers), and may have global position equipment (GPS) and depth sensors which aid in the identification of fishing areas (Valdés-Pizzini 2011).

Puerto Rico Commercial Fishing Gear and Targeted Species

Commercial fishermen target multiple species using multiple gear types during the same fishing trip. Nearly two-thirds of fishermen (63.2%) use at least three gear types during a fishing trip (Griffith et al. 2007). The information from Griffith et al. 2007 is general to all fishing off Puerto Rico, and this analysis assumes that commercial fisherman operating in federal waters use the same gear types and the same amounts. Historically trap gear dominated the catch by commercial fishermen in Puerto Rico, but their use has declined over time leading to a more balanced fishery using lines, traps, and spears (Appeldoorn et al. 2015). Gear types principally used by commercial fishermen in federal waters are hook-and-line[10], fish and lobster traps, spears, snares, and SCUBA. Eleven of the 720 commercial fishermen that submitted catch reports in 2018 used gillnets in federal waters to catch pelagic species (e.g., king mackerel, cero mackerel, and great barracuda) and bait species (e.g., ballyhoo and herrings). Additionally, commercial fishermen use cast nets and gillnets in Commonwealth waters to catch bait species (e.g., ballyhoo) that are then likely used to catch federally managed species (e.g., dolphin and wahoo) in federal waters.

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[10] Hook-and-line gear includes automatic reel, bandit gear, buoy gear, handline, longline (pelagic and bottom longlines), and rod and reel. 50 CFR 622.2.

In 2008, Matos-Caraballo and Agar (2011) found hook-and-line the most commonly used and productive gear in Commonwealth and federal water combined. The five most popular hook-and-line methods in all waters off Puerto Rico include vertical bottom lines,[11] followed by handlines, troll lines, rod and reels, and longlines (Agar and Shivlani 2016). For Puerto Rico, Agar and Shivlani (2016) found that vertical bottom lines (40%) and handlines (32%) are responsible for the largest share of the total landings (combined Commonwealth and federal) in 2014, using hook-and-line gear. Less than 2% of the fishermen using hook-and-line gear stated they primarily fished in federal waters; 39% stated that they fished equally in both federal and Commonwealth waters (Agar and Shivlani 2016). Fishermen use vertical bottom lines in federal waters to target deep-water snapper and grouper species such as queen, silk, cardinal, vermilion, blackfin and black snappers and misty grouper. The majority of the vertical bottom line fishing activity is centered on the west coast of Puerto Rico. Commercial fishermen using vertical bottom lines to target queen and cardinal snappers tend to drift fish, whereas those that target silk, black, blackfin, vermilion, and wenchman snappers tend to fish while anchored because the species are found in shallower waters (600 ft; 183 m) relative to queen and cardinal snappers, which are found in deeper waters (800-1,400 ft; 244-427 m) (Agar and Shivlani 2016). Fishermen use handlines in federal waters to target yellowtail, mutton and lane snappers, red hind grouper, and migratory pelagic species like dolphin, and king and cero mackerel (Agar and Shivlani 2016). Commercial fishermen using handlines reported fishing over a variety of habitats and depths depending on the species targeted (Agar and Shivlani 2016). For example, those targeting yellowtail snapper mainly fished between 40-120 ft (12-37 m) over muddy and hard bottoms, whereas those targeting silk snapper fished between 80-1,500 ft (24-457 m) over hard bottom. Most hook-and-line fishermen stated that they fished year-round on a full-time basis, spending, on average, 38 hours per week on fishing and fishing related activities, such as boat and engine maintenance and fish marketing (Agar and Shivlani 2016).

In Puerto Rico, traps are used in both federal and Commonwealth waters to harvest reef fish, deep-water snappers, and spiny lobster. In 2008, Matos-Caraballo and Agar (2011) found that fish traps accounted for 48% of the trap units in Commonwealth and federal waters combined (n = 4,574), followed by lobster traps at 40% (n = 3,842), and then deep-water snapper traps at 12% (n = 1,181). Fish traps are used to catch spiny lobster and various reef fish, such as silk snapper, mutton snapper, lane snapper, hogfish, yellowtail snapper, white grunt, red hind, and parrotfish species, among others, while lobster traps mainly catch spiny lobsters (Agar et al. 2017). Fish traps were more common than lobster traps because of their versatility in catch, with 66% of commercial fishermen using fish traps and 20% using a combination of fish and lobster traps in Commonwealth and federal waters combined (Agar et al. 2017). Agar et al. (2017) found that commercial fishermen in Puerto Rico had between 4 and 210 fish and lobster traps in the water, with an average of 51 traps, and that approximately 14% of the trap fishermen operated in both federal and Commonwealth waters; no fishermen reported fishing traps solely in federal waters. Traps are deployed at various depths (5–100 fathoms; 9-183 m) and habitats (e.g., seagrass beds and hard, sandy and muddy bottoms) depending on the weather conditions and the species that are targeted (Schärer et al. 2004). Use of trap gear has declined in recent years (i.e., fishermen have switched to fishing with other gear types), though traps remain popular among small-scale

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[11] A vertical bottom line is a specific type of hook-and-line gear that is deployed vertically in the water column with one or more hooks attached near the 1-8 pound lead weight at the terminal end of the line and fished at depths ranging from 50-150 fathoms (300-900 ft; 91-274 m).

fishermen (Agar et al. 2017). A major factor that influenced the reduction of trap use was the loss of gear due to the cutting of buoys by vessels and hurricanes. In the east coast of Puerto Rico, approximately 500 fish traps were estimated to be lost due to Hurricane María in 2017 (R. Espinosa, Conservación Conciencia, pers. comm. to M. Lopez, NMFS SERO, 2019). In addition, the high price of materials necessary to construct the traps has also contributed to the decline in use.

Commercial diving (in both Commonwealth and federal waters) is the fastest growing and most valuable fishery in Puerto Rico (Agar and Shivlani 2016). In federal waters, divers selectively target a diverse group of highly valued species such as spiny lobster and miscellaneous reef fish (e.g., snappers, parrotfish, wrasses) (Agar and Shivlani 2016). Of the total number of active commercial divers in Puerto Rico (n = 219)[12], approximately 25% operated in both federal and Commonwealth waters; no fishermen reported diving solely in federal waters (Agar and Shivlani 2017). Diving teams are composed of a boat tender and one or more divers and operate in water depths ranging from 10 to 150 ft (3–45.7 m) (Agar and Shivlani 2017). Besides using their hands, divers in federal waters use snares to catch spiny lobster and spears to catch reef fish, such as snappers, hogfish, and triggerfish (Agar and Shivlani 2017).[13] In federal waters, divers operate over hard bottoms and reef areas when fishing for spiny lobster and reef fish (Agar and Shivlani 2017).

2.2.2 Recreational Fishing in Puerto Rico

In Puerto Rico, recreational fishing activities in federal waters are conducted from private vessels (including rentals) and charter vessels. Recreational fishermen frequently target the same species as commercial fishermen and use hook-and-line and SCUBA gear to harvest those fish, but are not allowed to sell their catch per Commonwealth law. Instead, recreational fishermen pursue fishing for purposes ranging from subsistence fishing (solely for household consumption) to sport fishing (solely for pleasure with little to no consumption of catch). Recreational fishing effort is highest from May through August and lowest from November through February, with participation averaging between 8 and 12 days per month (Griffith et al. 2007).

No federal licenses or permits are required for the recreational harvest of Council-managed species in federal waters off Puerto Rico. Since 2010, all fishermen fishing recreationally in U.S. Caribbean federal waters, including the Puerto Rico EEZ, are required to register through the National Angler Registry. Although Puerto Rico fishing regulations state that a license for all recreational fishermen 13 years and older is required (excluding fishermen on charter or head boats), this requirement is not currently enforced. Most for-hire vessels operating in both state and federal waters are required to obtain a U.S. Coast Guard Operator of Uninspected Passenger Vessels License (as a minimum), insurance, the DNER charter fishing license, a permit from the Puerto Rico Tourism Company.

Although recreational and sport fishing activities in Puerto Rico are prevalent, data on the recreational catch and effort, species composition of the catch, and biological data on the species targeted and harvested are mostly lacking. The only continuous attempt at gathering these data

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[12] The total number of active commercial divers consisted of divers who reported landings at least once between 2011 and 2013 from Commonwealth and federal waters combined.

[13] A spear, hook, or similar device may not be used in federal waters to harvest spiny lobster.

from the recreational fishing sector dates to 2000, when the Marine Recreational Fishery Statistics Survey (MRFSS) was implemented in Puerto Rico. The MRFSS approach involved interviewing recreational fishermen as they returned to the dock to obtain information on their catch and effort during that fishing trip. Because all fishermen cannot be interviewed as they return from every trip, a second component of the MRFSS approach was a telephone-based survey of the island's population to determine what percentage of the population fishes recreationally and how frequently they do so. By then multiplying the intercept catch-per-trip estimates by the survey estimates of total effort, an estimate of total recreational catch for each stock could be derived. That product then comprised the estimate of recreational landings by year for each stock. The MRFSS program is now referred to as the Marine Recreational Information Program (MRIP), although data collection activities were suspended in Puerto Rico in 2017, and have not resumed to date due to impacts from Hurricane María.

Puerto Rico Recreational Fishermen

Recreational fishermen operating in Commonwealth and federal waters in Puerto Rico outnumber commercial fishermen by over 100 to one, but they land around the same number of pounds as the commercial catch (Griffith et al. 2007). However, the effort expended by an individual recreational fishermen is much less than that expended by a commercial fishermen, with recreational fishermen taking, on average, less than one trip per month (Griffith et al. 2007). The recreational charter industry is unevenly spread around Puerto Rico, with the San Juan area, the Northeast, and the Southwest regions supporting the greatest number of charter boats and other regions witnessing an occasional fishermen entering the industry seasonally or on a temporary basis, usually to supplement commercial fishing (Griffith et al. 2007).

NMFS' MRIP estimated 124,674 recreational fishermen took a total of 653,614 fishing trips in Puerto Rico in 2016, of which 11% (n = 70,196) were in federal waters (MRIP Query August 2020).

Puerto Rico Recreational Fishing Vessels

In 2018, 23,715 recreational (i.e., non-commercial) vessels in Puerto Rico were registered with the U.S. Coast Guard, including 37 personal watercraft. The number of recreational fishing vessels is unknown. The majority of the recreational vessels were less than 16 ft (44%) or ranged from 16 to 26 ft (42%). Only a small percentage, generally less than 10%, of recreational boaters are recreational fishermen (Griffith et al. 2007).

Puerto Rico Recreational Fishing Gear and Targeted Species

The majority of recreational fishermen use at least two gear types and over one-third of the fishermen use three gear types (Griffith et al. 2007). Griffith et al. (2007) reported that recreational fishermen predominantly use hook-and-line (54.4%) and SCUBA gear (10.4%). Recreational fishermen using hook-and-line gear tend to catch snapper-grouper species, including silk snapper and yellowtail snapper, and SCUBA divers tend to catch lobster (Griffith et al. 2007). Recreational hook-and-line fishermen also target and catch offshore pelagic species, including dolphin, tunas, mackerels, and wahoo. Dolphinfish and tuna species dominate the catch in federal waters off Puerto Rico (Griffith et al. 2007). Recreational fishermen using SCUBA gear also spear reef fish, particularly snappers and groupers, and coastal pelagic fish.

The information from Griffith et al. 2007 is general to all fishing off Puerto Rico, and this analysis assumes that recreational fisherman operating in federal waters use the same gear types and the same amounts.

MRIP records obtained from the Southeast Fisheries Science Center (SEFSC) on January 12, 2018, indicate that marine recreational fishermen caught 13 federally managed finfish species in federal waters off Puerto Rico in 2016.[14] The most caught finfish in 2016, in both number of fish and estimated number of pounds was dolphin, followed by wahoo.

2.2.3 Subsistence Fishing in Puerto Rico

Some fishermen only fish to provide food for household consumption (Griffith et al. 2007). Subsistence fishing remains an important aspect of fishing in federal waters off Puerto Rico, but is not fully understood with respect to participation, catch, or disposition of that catch. Subsistence fishermen mostly consist of people from working class backgrounds who target snapper-grouper species (40%) and pelagic species such as dolphin (7.4%) and king mackerel (5.9%), but almost no shellfish (Griffith et al. 2007). Their gear varieties are similar to those of recreational fishermen, but with fewer who use SCUBA gear.

2.3 Description of the St. Thomas/St. John Fishery

The St. Thomas/St. John fishery is small-scale or artisanal in nature, and is comprised of commercial, recreational, and subsistence participants. All fishery resources are consumed on the island; there is little or no export. Commercial fishing provides sustenance and employment, while recreational fishing provides food and leisure activity for local residents and visitors. In general, commercial and recreational fishermen target similar species of fish and shellfish, including reef fish, coastal and offshore pelagic fish, and spiny lobster, among others. Subsistence fishing, or fishing for household consumption, remains a component of both the commercial and recreational fishing sectors.

The best available fishing information for St. Thomas/St. John stems largely from the most recent census of licensed commercial fishermen in the USVI (Kojis et al. 2017) and from a survey of boat-based recreational fishermen in the USVI (Kojis and Tobias 2016). Unless otherwise noted, the information from Kojis et al. 2017 and Kojis and Tobias 2016 is general to all fishing off St. Thomas/St. John, and this analysis assumes that information in these studies pertains to commercial and recreational fishing in federal waters. Therefore, the following description provides a general characterization of the St. Thomas/St. John fishery operating in federal waters, for purposes of this consultation.

The St. Thomas/St. John FMP would manage 45 species of reef fish, 2 pelagic finfish species, spiny lobster, queen conch (Table 2.1), and all species of coral, sea urchin, and sea cucumber that occur in federal waters off St. Thomas/St. John. This list of species was tailored to the specific characteristics of the St. Thomas/St. John EEZ.

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[14] Federally managed finfish species caught by recreational fishermen in federal waters included: blackfin snapper, blackfin tuna, Dolphin, great barracuda, king mackerel, little tunny, mutton snapper, rainbow runner, red hind, silk snapper, tripletail, wahoo, and yellowtail snapper.

2.3.1 Commercial Fishing in St. Thomas/St. John

Fishing permits are not required to commercially harvest species proposed for management under the St. Thomas/St. John FMP in federal waters off St. Thomas/St. John. However, to sell fish caught in federal waters in St. Thomas/St. John, a commercial fishing license is required from the Department of Planning and Natural Resources (DPNR) of the USVI.[15] Commercial fishermen are required to report their catch and effort for every trip for both federal (3-200 nautical miles) and territorial (less than 3 nautical miles from shore) waters (CFMC 2011). Commercial fishing licenses are only issued to U.S. citizens who are permanent residents of the USVI for at least one year. On August 24, 2001, the DPNR implemented a moratorium on issuance of new commercial fishing licenses, which remains in effect. License renewals are only issued to fishermen who have held a commercial fishing license within three years of June 2001 and have complied with catch reporting requirements.

The St. Thomas and St. John shelf is wide, up to eight miles on the southern side of the islands and 20 miles on the northern side, so a relatively large part of the fishing grounds are located in federal waters. St. Thomas/St. John fishermen report their landings by the location code where the fish were caught, which span both federal and territorial waters, with 18 different sites reported on the north, south and east end of St. Thomas and four sites on St. John (Kojis et al. 2017).

St. Thomas/St. John Commercial Fishermen

The most recent census of licensed commercial fishermen in the USVI reported a total of 113 commercial fishermen on St. Thomas and six on St. John for a total of 119 (Kojis et al. 2017). Of those 119 fishermen, 4.6% said they fished primarily in federal waters and 42.5% said they fished equally in federal and territorial waters. The number of commercial fishermen that submitted catch reports in 2016 (pre-hurricanes Irma and Maria) was 65 and in 2018 (post-hurricanes Irma and Maria)[16] was 67. The total number of fishing trips reported in 2016 was 2,482 and in 2018 was 1,752.

Of the catch reports submitted, 72% of the 2016 fishermen and 60% of the 2018 fishermen reported operating primarily in federal waters and 46% of the 2016 fishing trips and 39% of the 2018 fishing trips were reported in federal waters. Additionally, a percentage of the commercial landings data were reported from an “unknown” location (57% of the 2016 commercial fishermen; 9% of the 2018 commercial fishermen; 7% of the trips taken in 2016), either because the fishermen were not certain if the fishing location was in Commonwealth or federal waters, or because the location field on the catch report was left blank.[17] It is possible that some of those “unknowns” occurred in federal waters, but that percentage is unquantifiable at this time.

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[15] The DPNR requires commercial fishing licenses for (1) all commercial fishermen, (2) any person who uses a pot, trap, set-net, or haul seine, (3) any person who sells, trades, or barters any part of their catch (including charter boat operators who sell or trade their catch), and (4) commercial fishing helpers who must obtain a helper’s permit to assist a licensed commercial fisher (the licensed commercial fisher must be onboard when the helper is fishing).

[16] On September 6, 2017, Category Five Hurricane Irma passed directly over St. John and St. Thomas and two weeks later on September 20, 2017, the eye of Category Five Hurricane Maria passed just offshore of the southwestern tip of St. Croix before turning north and crossing Puerto Rico. The combined effects of the two storms had a significant impact on the territories’ infrastructure, including that the commercial fishing industries. Damage was caused to fishing-related infrastructure, ports, docks, fishing businesses, vessels, and fishing gear.

[17] None of the fishing trips in 2018 were reported from an unknown location.

Approximately 89% of commercial fishermen in St. Thomas/St. John reported that they fish year-round, while the remaining 11% fished seasonally (Kojis et al. 2017). Commercial fishermen fished an average of 8.2 hours per trip and 8.4 trips per month, for an average total of 68.9 hours per month (Kojis et al. 2017). Fishermen spent on average 3.7 hours per week maintaining and repairing vessels, 3.7 hours per week maintaining and repairing gear, and 8.4 hours per week selling their catch.

St. Thomas/St. John Commercial Fishing Vessels

In 2018, 102 commercial fishing vessels were registered with the DPNR (H. Forbes, Director, DPNR Environmental Enforcement Division, pers. comm. to P. Opay, NMFS SERO, December 16, 2019) and thus available to operate in federal waters.

According to the 2016 census, the 104 commercial fishing vessels in St. Thomas/St. John ranged from 14-47 ft in length, with an average length of 24.6 ft (Kojis et al. 2017). The St. Thomas/St. John fleet was comprised of older vessels, the majority of which were built in 1981-1990 and constructed of fiberglass and wood. The majority of fishing vessels have a single gas-powered engine ranging in horsepower (hp) from 20-300 for outboard engines and 40-671 hp for inboard engines. Inboard engines were generally in the 151-300 size range while the majority of outboard engines were 101-150 hp. Most fishermen reported using electronic equipment such as depth finders and GPS, but most relied on cell phones for communication rather than a marine radio or EPIRB.

St. Thomas/St. John Commercial Fishing Gear and Targeted Species

Commercial fishermen target a variety of species using multiple gear types (Kojis et al. 2017). Most fishermen in St. Thomas/St. John (80.8%) used more than one method of fishing (e.g., trap fishing, line fishing, net fishing, or diving), with line fishing as the most common fishing method used at 77.8% (Kojis et al. 2017). Trap fishing was the second most common type of fishing (43.4%) followed by diving (32.3%). Gear types principally used by commercial fishermen in federal waters are fish and lobster traps, hook-and-line[18], spears, snares, and SCUBA (Kojis et al. 2017). Commercial fishermen use cast nets and gillnets in territorial waters to catch bait species (e.g., ballyhoo) that are then likely used to catch federally managed species (e.g., dolphin and wahoo) in federal waters.

In St. Thomas/St. John, commercial landings for trap gear have consistently been greater than landings reported for the other gear types. Fishermen reported owning 6,287 traps (55% fish traps and 45% lobster traps) (Kojis et al. 2017). However, studies show that fishermen do not use all of their traps all the time (Sheridan et al. 2006; Kojis et al. 2017). The average number of fish traps and plastic and wire lobster traps deployed per fishermen were 176 (fish), 328 (plastic), and 86 (wire) (Kojis et al. 2017). Kojis et al. (2017) also reported the average number of traps hauled per trip across federal and territorial waters: 50.2 for fish traps, 16.0 for wire lobster traps, and 47.3 for plastic lobster traps. Most fishermen in St. Thomas used trap lines (average of 13 traps per line; range 4-25 traps per line) and mechanized pot haulers with traps made of wire materials with identical size specifications per trap: 4 ft (122 cm) length and width, 1.5 ft (46

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[18] Hook-and-line gear includes automatic reel, bandit gear, buoy gear, handline, longline, and rod and reel. 50 CFR 622.2.

cm) height, and 2 in (5 cm) mesh (Sheridan et al. 2006). A survey of trap fishermen found that most traps were deployed on the southwest and southeast coasts of St. Thomas while few traps were deployed on the northeast coast of St. Thomas and the north coast of St. John (Sheridan et al. 2006). In St. Thomas/St. John, fish traps are used to catch spiny lobster and various reef fish, such as queen triggerfish, red hind, gray angelfish, doctorfish, white grunt, saucereye porgy, mutton snapper, and parrotfish species, among others, while lobster traps primarily catch spiny lobsters. Of the 19 fishermen that reported using spiny lobster traps, approximately 16% deployed the spiny lobster traps in federal waters and 53% deployed in both federal and territorial waters (Kojis et al. 2017). Of the 33 fishermen that reported using fish traps, approximately 9% deployed the fish traps in federal waters and 42% deployed in both federal and territorial waters (Kojis et al. 2017). St. Thomas fishermen reported that they deployed traps at an average depth of 156 ft (47.5 m), with two fisherman deploying traps in deeper depth ranges of 89-318 ft and 180-600 ft (27-97 m and 55–183 m) (Sheridan et al. 2006).

Of all the hook-and-line gear used by fishermen in St. Thomas/St. John, yo-yo reels (handlines) were used the most, followed by rod and reels (Kojis et al. 2017). In federal waters, handlines were used to primarily target yellowtail snapper, blue runner, and red hind grouper, while rod and reels were used to primarily target dolphin and wahoo. Of the 69 fishermen that reported using handlines, approximately 3% fished in federal waters and 33% fished in both federal and territorial waters (Kojis et al. 2017). Of the 43 fishermen that reported using rod and reels, approximately 5% fished in federal waters and 49% fished in both federal and territorial waters (Kojis et al. 2017). The number of hooks per line ranged from 1-2 for handlines and rod and reels (Kojis et al. 2017). Three fishermen reported using multi-hook (10 hooks per line) vertical set line used to target deep-water snapper-grouper species, of which one fished in federal waters and one in both federal and territorial waters (Kojis et al. 2017).

Only 32 (32%) of the St. Thomas/St. John fishermen fished by diving (Kojis et al. 2017). Most skin and SCUBA divers used snares for catching spiny lobster and spears for spearing fish (e.g., gray angelfish and red hind grouper). Of the 27 skin diving fishermen, none operated solely in federal waters and four (15%) operated in federal and territorial waters (Kojis et al. 2017). Similarly, of the 12 fishermen that used SCUBA gear, none operated solely in federal waters and two (17%) operated in federal and territorial waters. Kojis et al. (2017) state that diving fishermen enter the water with a buoyed dive flag attached to them by a long line, and the boat captain follows the diver by following the dive flag and diver's bubbles, which suggests that anchoring does not occur during fishing activities involving skin or SCUBA diving.

2.3.2 Recreational Fishing in St. Thomas/St. John

Recreational fishermen are persons who primarily fish to provide food for themselves or their families and those who catch and release fish. Van Beukering et al. (2011) found that recreational fishermen in the USVI fish for enjoyment (33%), for food for themselves (26%) or for friends and family (13%), for social interaction (14%) or tradition (11%).

Three categories of recreational fishing occur in the USVI: (1) big game fishing conducted from large (greater than 30 ft [9 m]) vessels, which primarily target billfish; (2) private boat fishing conducted from smaller boats, which primarily target reef fish and offshore pelagic fish; and (3) shoreline fishing (beach, pier, dock, etc.), which primarily target reef fish (Kojis and Tobias 2016).

There are no federal licenses or permits are required for the recreational harvest of any Council-managed species in federal waters off St. Thomas/St. John. Since 2010, all fishermen fishing recreationally in U.S. Caribbean federal waters, including the St. Thomas/St. John EEZ, are required to register through the National Angler Registry. In 2019, the USVI government established a recreational fishing license for the Territory, offered for four categories: individual, charter/for-hire, private vessel, and tournament. However, no information (e.g., number of license holders by category or location [territorial/federal waters]) is available at this time.

The unique topography of St. Thomas/St. John, south of the six-mile-deep Puerto Rico Trench, makes this management area known for some of the hardiest game fish in the world, with several fishing tournaments held in St. Thomas/St. John each year. Many recreational fishermen fish along these drop-offs. Two well-known fishing areas off St. Thomas are the North Drop, about 20 miles north of St. Thomas, and the South Drop, 8 miles south of St. Thomas. Migrating schools of small fish gather in these areas, which attracts larger pelagic species such as wahoo and dolphin.

St. Thomas/St. John Recreational Fishermen

Kojis and Tobias (2016) found that of 37% of the registered boat owners[19] fished recreationally and that the majority (81%) of recreational fishing in the USVI occurs from boats. The majority of recreational fishermen reported that they fished in territorial waters (85%) while about half (56%) reported fishing in federal waters (Kojis and Tobias 2016). Van Beukering et al. (2011) surveyed recreational fishermen in the USVI and found that 20% of the households on St. Thomas and 10% of the households on St. John are involved in recreational fishing, and that most households fish less than once a week, usually on the weekends.

The most common time of day that boat-based recreational fishermen landed fish was 9 am to 9 pm with a peak landing period from 3–6 pm. The mean length of an average recreational fishing trip was 4.4 hours with a mean of 3.3 trips per month (Kojis and Tobias 2016). Twenty-two percent of St. Thomas/St. John respondents participated in recreational fishing tournaments (Kojis and Tobias 2016). Dolphin, wahoo, and billfish tournaments occur in the spring and summer during migrations of the pelagic species.

The MRFSS/MRIP recreational data collection program does not operate in St. Thomas/St. John. The program was instituted in 2000, but after one year of operation was discontinued due to logistical problems. Although a recreational data collection program does not operate in St. Thomas/St. John, the DPNR's Division of Fish and Wildlife (DFW) has made available three different ways for recreational fishermen to submit data for recreational fishing activities including traditional catch report forms, sport-fishing logbooks for charter companies or for those that fish often, and through a digital reporting form,[20] which can easily be accessed by scanning a quick response (QR) code. Recreational fishermen are asked to report their catch by area fished, but the reporting areas sometimes span both federal and territorial waters. The DFW has established a website to facilitate the collection of recreational fishing data, including the

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[19] Of the 769 boat owners from the 2013 boat registration list, 378 of the 400 randomly selected registrants responded to the survey (Kojis and Tobias 2016).

[20] USVI Recreational Fishery Data Collection & Licensing 2019 presentation.

number of fishermen fishing as a resident, visitor, charter boat operator, or tournament participant and has updated their Commercial and Recreational Fishers' Information Handbook to reflect the updates for recreational fishing in the USVI. Preliminary numbers of recreational fishermen are not available at this time, but this program is an improvement on collecting recreational fisheries data that could be used for monitoring recreational fishing activities in federal waters.

To date, the number of recreational (i.e., non-commercial) fishermen on St. Thomas/St. John has not been quantified. However, a telephone survey conducted from December 1998 to July 1999 estimated the number of recreational fishermen on St. Thomas/St. John to be 7,705 (Mateo 2004). Approximately 50% of those recreational fishermen fished exclusively from boats, while 38% fished from boats and from shore (Mateo 2004). The recreational fishermen fishing from boats (up to 6,781 fishermen; 7,705 x 0.88) could fish in both territorial and federal waters.

St. Thomas/St. John Recreational Fishing Vessels

In 2018, a total of 2,285 recreational vessels were registered, 10-15% of which were estimated to be fishing vessels (H. Forbes, DPNR Environmental Enforcement Division, pers. comm. to P. Opay, NMFS SERO, December 16, 2019). The mean length of power boats used for recreational fishing was 21.4 ft (6.5 m). Sailboats were on average nearly twice as large with a mean length of 39.7 ft (12.1 m) (Kojis and Tobias 2016).

The charter boat fleet often fish the northern shelf edge, particularly the North Drop, to catch billfish and other deep-water pelagic species. This fleet caters to wealthy recreational fishermen (Kojis et.al. 2017). A number of resident charter vessels also have commercial licenses and commercially fish when not chartered.

St. Thomas/St. John Recreational Fishing Gear and Targeted Species

In the USVI, most recreational fishermen use hook-and-line gear such as handlines (also called a yo-yo reel) and rod and reels, and that the most common fishing technique used by recreational fishermen was trolling (Kojis and Tobias 2016). Kojis and Tobias (2016) found that recreational fishermen in St. Thomas/St. John participated in inshore and offshore trolling to target dolphinfish and jack species; shallow-water bottom and drift line fishing to target yellowtail snapper and blue runner; and deep-water buoy and line fishing to target deep-water snapper and grouper species such as blackfin snapper, queen snapper, and misty grouper. Skin and SCUBA divers spearfished for reef fish species (e.g., snappers, groupers, triggerfish, grunts, and porgies) and hand collected spiny lobster (Kojis and Tobias 2016). Forty-one percent of surveyed fishermen used cast nets to collect baitfish (Kojis and Tobias 2016).

2.3.3 Subsistence Fishing in St. Thomas/St. John

Subsistence fishing characterizes both commercial and recreational fishing in USVI but is not fully understood with respect to participation, catch, or disposition of that catch. Subsistence fishermen in St. Thomas and St. John fish to provide food for their family and household. Five of the surveyed commercial fishermen responded that they primarily fish for food (Kojis et al. 2017) and 42% of surveyed recreational fishermen considered themselves to be subsistence fishermen (Kojis and Tobias 2016).

2.4 Description of the St. Croix Fishery

The St. Croix fishery is small-scale or artisanal in nature, and is comprised of commercial, recreational, and subsistence participants. All fishery resources are consumed on the island; there is little or no export. Commercial fishing provides sustenance and employment, while recreational fishing provides food and leisure activity for local residents and visitors. In general, commercial and recreational fishermen target similar species of fish and shellfish, including reef fish, coastal and offshore pelagic fish, and spiny lobster, among others. Subsistence fishing, or fishing for household consumption, remains a component of both the commercial and recreational fishing sectors.

The best available fishing information for St. Croix stems largely from the most recent census of licensed commercial fishermen in the USVI (Kojis et al. 2017) and from a survey of boat-based recreational fishermen in the USVI (Kojis and Tobias 2016). Unless otherwise noted, the information from Kojis et al. 2017 and Kojis and Tobias 2016 is general to all fishing off St. Thomas/St. John, and this analysis assumes that information in these studies pertains to commercial and recreational fishing in federal waters. Therefore, the following description provides a general characterization the St. Croix fishery operating in federal waters, for purposes of this consultation.

The St. Croix FMP would manage 41 species of reef fish, 2 pelagic finfish species, spiny lobster, queen conch (Table 2.1), and all species of coral, sea urchin, and sea cucumber that occur in federal waters off St. Croix. This list of species was tailored to the specific characteristics of the St. Croix EEZ.

2.4.1 Commercial Fishing in St. Croix

Fishing permits are not required to commercially harvest species proposed for management under the St. Croix FMP in federal waters off St. Croix. However, to sell fish caught in federal waters in St. Croix, a commercial fishing license if required from the DPNR of the USVI.[21] Commercial fishermen are required to report their catch and effort for every trip for both federal (3-200 nautical miles) and territorial (less than 3 nautical miles from shore) waters (CFMC 2011). Commercial fishing licenses are only issued to U.S. citizens who are permanent residents of the USVI for at least one year. No licenses are issued to minors under 17 years old, except with written consent. On August 24, 2001, the DPNR implemented a moratorium on issuance of new commercial fishing licenses, which remains in effect. License renewals are only issued to fishermen who have held a commercial fishing license within three years of June 2001 and have complied with catch reporting requirements.

Kojis et al. (2017) reported that the majority of St. Croix fishermen reported fishing in territorial waters (59.1%) and theorized that this may be because the shelf around St. Croix (except for part of Lang Bank) is narrow and lies within territorial waters. The shelf edge drops off abruptly to depths greater than 1,000 ft (350 m). Lang Bank is frequently exposed to heavy seas, which often deters fishermen from accessing the bank, especially the offshore portion that lies in

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[21] The DPNR requires commercial fishing licenses for (1) all commercial fishermen, (2) any person who uses a pot, trap, set-net, or haul seine, (3) any person who sells, trades, or barters any part of their catch (including charter boat operators who sell or trade their catch), and (4) commercial fishing helpers who must obtain a helper's permit to assist a licensed commercial fisher (the licensed commercial fisher must be onboard when the helper is fishing).

federal waters, and thus, the insular shelf around St. Croix is the main fishing grounds for commercial fishermen (Kojis et al. 2017). Given the smaller shelf surrounding St. Croix, deep-water is relatively close to shore and, therefore, deep-water snapper and pelagic fish are much more accessible to the island's small boat fishery (Kojis et al. 2017). Commercial fishing in St. Croix is a daily activity, in that fishermen leave in the morning or the evening and generally return to shore within 12 hours of departure.

St. Croix Commercial Fishermen

The most recent census of licensed commercial fishermen in the USVI reported 141 commercial fishermen on St. Croix (Kojis et al. 2017). Of those 141 fishermen, 14.6% said they fished primarily in federal waters and 26.4% said the fished equally in federal and territorial waters. The number of commercial fishermen submitting catch reports in 2016 (pre-hurricanes Irma and Maria) was 74 and 2018 (post-hurricanes Irma and Maria)[22] was 43. The total number of fishing trips reported in 2016 was 2,489 and in 2018 was 802.

Of the catch reports submitted, 69% of the 2016 fishermen and 67% of the 2018 fishermen reported operating primarily in federal waters and 36% of the 2016 fishing trips and 42% of the 2018 fishing trips were reported in federal waters. Additionally, a percentage of the commercial landings data were reported from an "unknown" location (42% of the 2016 commercial fishermen; 21% of the 2018 commercial fishermen; 10% of the trips taken in 2016, and 4% of the trips taken in 2018), either because the fishermen were not certain if the fishing location was in Commonwealth or federal waters, or because the location field on the catch report was left blank. It is possible that some of those "unknowns" occurred in federal waters, but the percentage is unquantifiable at this time.

Approximately 96% of commercial fishermen in St. Croix reported that they fish year-round, while the remaining 4% fished seasonally (Kojis et al. 2017). Commercial fishermen fished an average of 6.6 hours per trip and 11.5 trips per month, for an average total of 75.9 hours per month (Kojis et al. 2017). Fishermen spent on average 2.9 hours per week maintaining and repairing vessels, 2.9 hours per week maintaining and repairing gear, and 6.1 hours per week selling their catch.

St. Croix Commercial Fishing Vessels

In 2018, 137 commercial fishing vessels were registered with the DPNR (H. Forbes, DPNR Environmental Enforcement, pers. comm. to P. Opay, NMFS SERO, December 16, 2019) and thus available to operate in federal waters.

According to the 2016 census, the 129 commercial fishing vessels on St. Croix ranged from 12-45 ft in length, with an average length of 21.9 ft (Kojis et al. 2017). The St. Croix fleet was comprised of older vessels, the majority of which were built in 1971-1980 and constructed of fiberglass and wood. The majority of fishing vessels have a single gas-powered engine ranging from 3.5-300 hp for outboard engines and from 15-500 hp for inboard engines. Inboard engines

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[22] On September 6, 2017, Category Five Hurricane Irma passed directly over St. John and St. Thomas and two weeks later on September 20, 2017, the eye of Category Five Hurricane Maria passed just offshore of the southwestern tip of St. Croix before turning north and crossing Puerto Rico. The combined effects of the two storms had a significant impact on the territories' infrastructure, including that the commercial fishing industries. Damage was caused to fishing-related infrastructure, ports, docks, fishing businesses, vessels, and fishing gear.

were generally in the 151-300 hp size range while the majority of outboard engines were 76-100 hp. Most fishermen reported using electronic equipment such as depth finders and GPS, but most relied on cell phones for communication rather than a marine radio or EPIRB.

St. Croix Commercial Fishing Gear and Targeted Species

Commercial fishermen target a variety of species using multiple gear types (Kojis et al. 2017). Most fishermen in St. Croix (70%) used more than one method of fishing (e.g., trap fishing, line fishing, net fishing, or diving), with line fishing as the most common fishing method used at 91.8% (Kojis et al. 2017). Fishing while diving was the second most common type of fishing (62.7%) followed by trap fishing (31.8%). Gear types principally used by commercial fishermen in federal waters are hook-and-line[23], fish traps, spears, snares, and SCUBA (Kojis et al. 2017). Commercial fishermen use cast nets in territorial waters to catch bait species (e.g., ballyhoo) that are then likely used to catch federally managed species (e.g., dolphin and wahoo) in federal waters. A small percentage of fishermen in St. Croix use gillnets (2 of 109 fishermen) and umbrella nets (1 of 109 fishermen) to catch baitfish species (e.g., mackerel scad) in federal and territorial waters.

In St. Croix, commercial landings reported using dive gear have consistently been greater than landings reported for hook-and-line or trap gear. Half of the fishermen surveyed in St. Croix reported owning their own SCUBA gear, and almost all reported using snares or spears during diving operations (Kojis et al. 2017). Fishermen commonly used SCUBA gear to spear reef fish, hand gather queen conch, and snare spiny lobsters (Kojis et al. 2017). In federal waters, the top species targeted by divers using spear gear are parrotfish species (e.g., stoplight, redfin, and queen parrotfish), queen triggerfish, red hind grouper, and schoolmaster snapper. The average dive time reported by fishermen using SCUBA gear was 3.7 hours. Of the 59 fishermen that used SCUBA gear, none operated solely in federal waters and 22 (37%) operated in federal and territorial waters combined (Kojis et al. 2017). Kojis et al. (2017) state that diving fishermen enter the water with a buoyed dive flag attached to them by a long line, and the boat captain follows the diver by following the dive flag and diver's bubbles, which suggests that anchoring does not occur during fishing activities involving skin or SCUBA diving.

Of all the hook-and-line gear used by fishermen in St. Croix, yo-yo reels (handlines) were used the most, followed by vertical set, multi-hook lines (used in deep-water habitats), and rod and reels (Kojis et al. 2017). Two types of vertical set lines were used by fishermen in St. Croix to fish in deep water, off the shelf edge: single hook for pelagic species (e.g., dolphin and wahoo) and multi-hook for deep-water snapper (e.g., silk, vermilion, queen, wenchman, and blackfin) and grouper species (Kojis et al. 2017). Of the 100 fishermen that reported using handlines, approximately 3% fished in federal waters and 50% fished in both federal and territorial waters (Kojis et al. 2017). Of the 42 fishermen that reported using multi-hook, vertical set lines, none fished solely in federal waters and 79% fished in both federal and territorial waters (Kojis et al. 2017). The number of hooks per line ranged from 1-7 for handlines and from 3-40 for multi-hook, vertical set lines (Kojis et al. 2017). No longlines, surface or bottom, were used in federal waters in St. Croix.

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[23] Hook-and-line gear includes automatic reel, bandit gear, buoy gear, handline, longline, and rod and reel. 50 CFR 622.2.

Fishermen in St. Croix reported owning 855 traps (96% of which are fish traps) (Kojis et al. 2017). However, studies show that fishermen do not use all of their traps all the time (Sheridan et al. 2006; Kojis et al. 2017). The average number of fish traps and wire lobster traps deployed per fisher were 120 (fish) and 20[24] (wire) (Kojis et al. 2017). The average number of traps hauled per trip was not reported for St. Croix, so the average number of traps owned per fisherman was used: 34.3 for fish traps and 20 for lobster traps (Kojis et al. 2017). Most fishermen in St. Croix used single traps marked with buoys (as opposed to trap lines) and set and hauled traps by hand. When buoys were missing, most fishermen would use dive gear to recover the missing traps (Sheridan et al. 2006). St. Croix fishermen placed the largest numbers of traps on the south coast, fewer traps on the east and northeast coasts, and none on the west and northwest coasts at mean fishing depth of 58 ft and range of 24-100 ft (average 17.7 m; range 7.3-30.5 m) (Sheridan et al. 2006). All traps were made of wire materials, but the reported size specifications of the traps varied (Sheridan et al. 2006). Trap lengths ranged from 4-6 ft (122-183 cm), trap widths ranged from 3-4 ft (91-122 cm), and all reported trap heights were 1.5 ft (46 cm). In St. Croix, fish traps are used to catch spiny lobster and various reef fish, such as blackfin snapper, silk snapper, schoolmaster, red hind grouper, and queen triggerfish, among others, while lobster traps primarily catch spiny lobsters. Only one fishermen reported using lobster traps, and all 20 traps were deployed in federal waters (Kojis et al. 2017). Of the 24 fishermen that reported using fish traps, approximately 4% deployed the fish traps in federal waters and 17% deployed the traps in both federal and territorial waters (Kojis et al. 2017).

2.4.2 Recreational Fishing in St. Croix

Recreational fishermen are persons who primarily fish to provide food for themselves or their families and those who catch and release fish. Van Beukering et al. (2011) found that recreational fishermen in the USVI fish for enjoyment (33%), for food for themselves (26%) or for friends and family (13%), for social interaction (14%) or tradition (11%).

Three categories of recreational fishing occur in the USVI: (1) big game fishing conducted from large (greater than 30 ft [9 m]) vessels, which primarily target billfish; (2) private boat fishing conducted from smaller boats, which primarily target reef fish and offshore pelagic fish; and (3) shoreline fishing (beach, pier, dock, etc.), which primarily target reef fish (Kojis and Tobias 2016).

There are no federal licenses or permits required for the recreational harvest of any Council-managed species in federal waters off St. Croix. Since 2010, all fishermen fishing recreationally in U.S. Caribbean federal waters, including the St. Croix EEZ, are required to register through the National Angler Registry. In 2019, the USVI government established a recreational fishing license for the Territory, offered for four categories: individual, charter/for-hire, private vessel, and tournament. However, no information (e.g., number of license holders by category or location [territorial/federal waters]) is available at this time.

The diverse coastal and marine environment is home to coral reefs, salt ponds, and mangrove forests. Common benthic habitats around St. Croix include coral reef, hard bottom, and seagrass (Kendall et al. 2001). This diversity supports a variety of species including sharks and sea

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[24] The average number of traps deployed was not available, so the assumption was made that the 20 traps owned by the single fisherman in St. Croix were all deployed during fishing operations.

turtles, reef fish, wahoo and tuna, which in turn creates ample fishing and SCUBA diving opportunities. St. Croix's narrow shelf limits the area available for harvest of reef fish when compared to St. Thomas and St. John, but the deep-waters closer to shore allow fishermen easier access to pelagic fish (Feingold 2014).

St. Croix Recreational Fishermen

Kojis and Tobias (2016) found that 37% of the registered boat owners[25] fished recreationally and that the majority (97%) of recreational fishing in the USVI occurs from boats. The majority of recreational fishermen reported that they fished in territorial waters (85%) while about half (56%) reported fishing in federal waters (Kojis and Tobias 2016). Van Beukering et al. (2011) surveyed recreational fishermen in the USVI and found that 27% of the households on St. Croix are involved in recreational fishing, and that most households fish less than once a week, usually on the weekends.

The most common time of day that boat-based recreational fishermen landed fish was 9 am to 9 pm with a peak landing period from 3 – 6 pm. The mean length of an average recreational fishing trip was 4.4 hours with a mean of 3.3 trips per month (Kojis and Tobias 2016). Six percent of St. Croix respondents participated in recreational fishing tournaments (Kojis and Tobias 2016). Dolphin, wahoo, and billfish tournaments occur in the spring and summer during migrations of pelagic species.

The MRFSS/MRIP recreational data collection program does not operate in St. Croix. The program was instituted in 2000, but after one year of operation was discontinued due to logistical problems. Although a recreational data collection program does not operate in St. Croix, the DPNR's DFW has made available three different ways for recreational fishermen to submit data for recreational fishing activities including traditional catch report forms, sport-fishing logbooks for charter companies or for those that fish often, and through a digital reporting form,[26] which can easily be accessed by scanning a QR code. Recreational fishermen are asked to report their catch by area fished, but the reporting areas sometimes span both federal and territorial waters. The DFW has established a website to facilitate the collection of recreational fishing data, including the number of fishermen fishing as a resident, visitor, charter boat operator, or tournament participant and has updated their Commercial and Recreational Fishers' Information Handbook to reflect the updates for recreational fishing in the USVI. Preliminary numbers of recreational fishermen are not available at this time, but this program is an improvement on collecting recreational fisheries data that could be used for monitoring interactions of recreational activities in federal waters.

To date, the number of recreational (i.e., non-commercial) fishermen on St. Croix has not been quantified (Goedeke et al. 2016). However, a telephone survey conducted from December 1998 to July 1999 estimated the number of recreational fishermen on St. Croix to be 3,294 (Mateo

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[25] Of the 769 boat owners from the 2013 boat registration list, 378 of the 400 randomly selected registrants responded to the survey (Kojis and Tobias 2016).

[26] USVI Recreational Fishery Data Collection & Licensing 2019 presentation.

2004). Approximately 19% of those recreational fishermen fished exclusively from boats, while 21% fished from boats and from shore (Mateo 2004). The recreational fishermen fishing from boats (up to 1,318 fishermen; 3,294 x 0.40) could fish in both territorial and federal waters.

St. Croix Recreational Fishing Vessels

In 2018, a total of 769 recreational vessels were registered, 10-15% of which were estimated to be fishing vessels (Director Forbes, Environmental Enforcement Division, pers. comm. to P. Opay, NMFS SERO, December 16, 2019). The mean length of power boats used for recreational fishing was 21.4 ft (6.5 m). Sailboats were on average nearly twice as large with a mean length of 39.7 ft (12.1 m) (Kojis and Tobias 2016).

St. Croix Recreational Fishing Gear and Targeted Species

In the USVI, most recreational fishermen use hook-and-line gear such as handlines (also called a yo-yo reel) and rod and reels, and that the most common fishing technique used by recreational fishermen was trolling (Kojis and Tobias 2016). Kojis and Tobias (2016) found that recreational fishermen in St. Croix participated in offshore trolling and bottom fishing to target dolphinfish and jack species; shallow-water bottom and drift line fishing to target yellowtail snapper and blue runner; and deep-water buoy and line fishing to target deep-water snapper and grouper species such as blackfin snapper, queen snapper, and misty grouper. Skin and SCUBA divers spearfished for reef fish species (e.g., snappers, groupers, triggerfish, and grunts) and hand collected spiny lobster (Kojis and Tobias 2016).

2.4.3 Subsistence Fishing in St. Croix

Subsistence fishing characterizes both commercial and recreational fishing in USVI but is not fully understood with respect to participation, catch, or disposition of that catch. Subsistence fishermen in St. Croix fish to provide food for their family and household. One commercial fishermen responded that they primarily fish for food (Kojis et al. 2017) and 42% of surveyed recreational fishermen considered themselves to be subsistence fishermen (Kojis and Tobias 2016).

2.5 Action Area

The action area for an opinion is defined as all of the areas affected by the federal action and not merely the immediate area involved in the action, 50 CFR 402.02. Therefore, to determine the action area for each FMP included in this opinion, we reviewed the area where the federally-managed fishing occurs, accounting for closed areas, and the surrounding areas where effects of the managed activities could occur.

The U.S. Caribbean is located in the eastern portion of the Caribbean archipelago, about 1,100 miles (mi) (1,770 kilometers [km]) east-southeast of Miami, Florida (Olcott 1999). The region is composed of the Commonwealth of Puerto Rico in the Greater Antilles and St. Thomas/St. John and St. Croix, USVI in the Lesser Antilles.

The U.S. EEZ in the Caribbean covers an area of approximately 75,687 square miles (mi^2) (196,029 square kilometers [km^2]), which, for management purposes, is divided into the Puerto Rico, St. Thomas/St. John, and St. Croix management areas (Figure 2.1) (Table 2.5). In the

Caribbean Sustainable Fisheries Act Amendment (CFMC 2005), fishable habitat was defined as those waters less than or equal to 100 fathoms (600 feet [ft]; 183 meters [m]) (Figure 2.2), and that definition remains unchanged under the proposed FMPs.[27] The total area of fishable habitat (less or equal to 100 fathoms) in the U.S. Caribbean is estimated to be 2,932 mi^2 (7,594 km^2). However, only 403 mi^2 (1,045 km^2) of that area is located in federal waters.

For each FMP, the action area considered for this consultation corresponds to the area of fishable habitat (less or equal to 100 fathoms) located in federal waters (Table 2.5). However, due to the connectivity of fish and invertebrate stocks between commonwealth, territorial, and federal waters, the proposed action may also affect listed species and critical habitat in commonwealth and territorial waters. Because ESA-listed corals and *Acropora* critical habitat in commonwealth or territorial waters could be affected by fishing for herbivorous fish in federal waters, the action area also includes the locations in non-federal waters where the corals or *Acropora* critical habitat are likely affected.

Table 2.5. Area estimates for U.S. Caribbean, Puerto Rico, St. Thomas/St. John, and St. Croix exclusive economic zones (EEZ) and amount of fishable habitat within each EEZ and territorial waters.

Location	Total EEZ	Total Fishable Habitat Area	Fishable Habitat in EEZ Waters	Fishable Habitat in Territorial Waters
U.S. Caribbean	75,687 mi^2 (196,029 km^2)	2,932 mi^2 (7,594 km^2)	403 mi^2 (1,045 km^2)	2,529 mi^2 (6,549 km^2)
Puerto Rico	65,368 mi^2 (169,303 km^2)	2,248 mi^2 (5,823 km^2)	158 mi^2 (410 km^2)	2,090 mi^2 (5,413 km^2)
St. Thomas/ St. John	1,103 mi^2 (2,856 km^2)	539 mi^2 (1,396 km^2)	219 mi^2 (567 km^2)	320 mi^2 (829 km^2)
St. Croix	9,216 mi^2 (23,870 km^2)	145 mi^2 (375 km^2)	26 mi^2 (68 km^2)	119 mi^2 (307 km^2)

Puerto Rico

The island of Puerto Rico includes the adjacent inhabited islands of Vieques and Culebra as well as various other isolated islands without permanent populations including Mona, Monito, and Desecheo. Puerto Rico is surrounded on three sides by deep ocean waters. The Mona Passage separates Puerto Rico from Hispaniola to the west and is about 75 mi (120 km) wide and more than 3,300 ft (1,000 m) deep. The Puerto Rico Trench borders the northern coast and is 28,000 ft (8,500 m) deep, and to the south the sea bottom descends to the 16,400 ft (5,000 m) deep Venezuelan Basin of the Caribbean Sea. To the east, Puerto Rico shares the shallow-water shelf platform with St. Thomas and St. John, which extends east towards the British Virgin Islands.

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[27] Although some fishing activities occur in federal waters deeper than 100 fathoms (600 ft; 183 m), such as fishing for deep-water snappers or pelagic species like dolphin and wahoo, at this time the fishing effort and area estimates for activities occurring in federal waters at depths greater than 100 fathoms are difficult to quantify. Thus, for purposes of this consultation, the fisheries described are assumed to operate within the current description of fishable habitat: those waters less than or equal to 100 fathoms.

Puerto Rico EEZ waters are located 9-200 nautical miles (17-370 km) from the coast of the island and cover approximately 65,368 mi^2 (169,303 km^2). The total area of fishable habitat (less or equal to 100 fathoms) in all waters off Puerto Rico (commonwealth and federal waters, combined) is estimated to be 2,248 mi^2 (5,823 km^2). However, only 158 mi^2 (410 km^2) of that area, approximately 7%, is located in federal waters.

St. Thomas/St. John

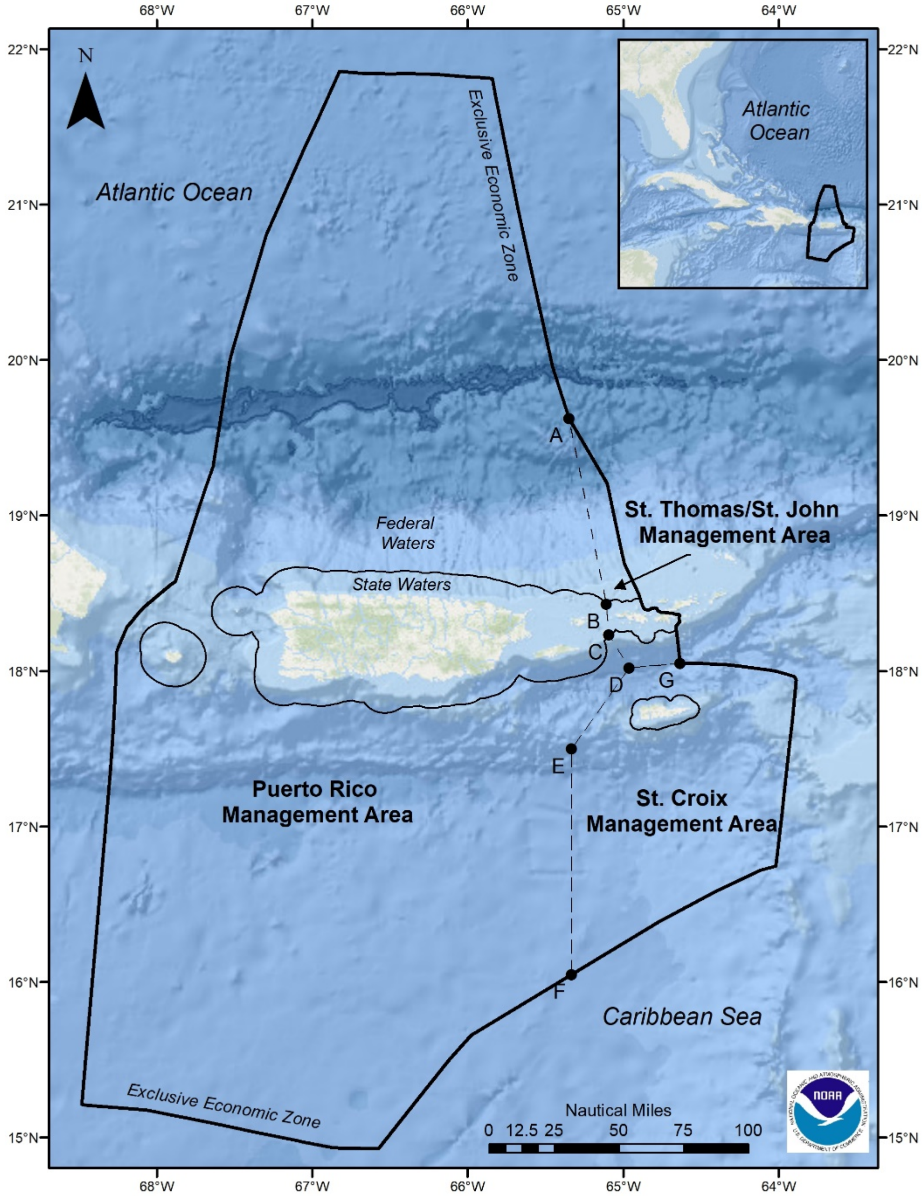
The islands of St. Thomas and St. John are bordered by the Atlantic Ocean to the north and the Caribbean Sea to the south. The shelf on St. Thomas/St. John is wide, up to eight miles on the southern side of the islands and 20 miles on the northern side, so a relatively large part of the shelf, the primary fishing grounds, lies in federal waters.

St. Thomas/St. John EEZ waters are located 3-200 nautical miles (6-370 km) from the coast of the island and cover approximately 1,103 mi^2 (2,856 km^2). The total area of fishable habitat (less or equal to 100 fathoms) in all waters off St. Thomas/St. John (territorial and federal waters, combined) is estimated to be 539 mi^2 (1,396 km^2). Of that area, 219 mi^2 (567 km^2), approximately 41%, is located in federal waters.

St. Croix

St. Croix is located about 40 nautical miles (74 km) south of St. Thomas and St. John and is entirely surrounded by the Caribbean Sea. The island of St. Croix is on a different geological platform than the islands of St. Thomas/St. John and Puerto Rico, and is separated from those islands by a 2.5 mi (4 km) deep trench. St. Croix is surrounded by a gradually descending narrow shelf around the island, which allowed the growth of a fringing reef around much of the coastline (Van Beukering et al. 2011).

St. Croix EEZ waters are located 3-200 nautical miles (6-370 km) from the coast of the island and cover approximately 9,216 mi^2 (23,870 km^2). The total area of fishable habitat (less or equal to 100 fathoms) in all waters off St. Croix (territorial and federal waters, combined) is estimated to be 145 mi^2 (375 km^2). Of that area, 26 mi^2 (68 km^2), approximately 18%, is located in federal waters.



Service Layer Credits: Esri, Garmin, GEBCO, NOAA NGDC, and other contributors

Figure 2.1. Location of the U.S. Caribbean exclusive economic zone with subdivisions for the three island-specific management areas (dashed line) and boundaries for federal (thick solid line) and state waters (thin solid line).

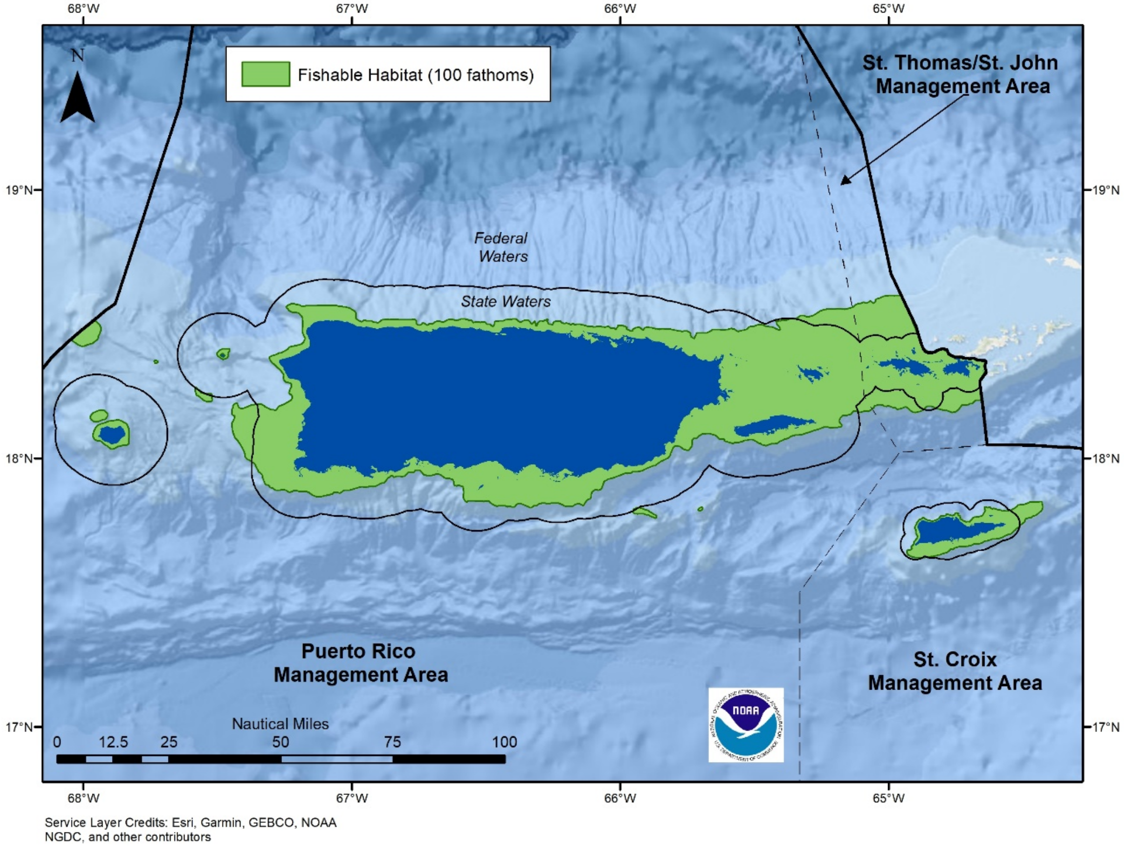


Figure 2.2. Fishable habitat defined as those waters less or equal to 100 fathoms (green shaded area) located in federal (thick solid line) and state (thin solid line) waters for each of the three management areas (dashed line) in the U.S. Caribbean exclusive economic zone. The blue shaded area represents land.

3.0 Status of Listed Species and Critical Habitat

The following endangered and threatened species and designated critical habitat under the jurisdiction of NMFS may occur in or near the action area of the fisheries considered in this Opinion (Table 3.1 and 3.2).

Table 3.1. ESA-Listed and Proposed to be Listed Species That May Occur In U.S. Caribbean Federal Waters and Assessed in this Consultation

Marine Mammals		**Status**
Sei whale	*Balaenoptera borealis*	Endangered
Sperm whale	*Physeter macrocephalus*	Endangered
Fin whale	*Balaenoptera physalus*	Endangered
Sea Turtles		
Green (North Atlantic and South Atlantic DPSs)	*Chelonia mydas*	Threatened
Hawksbill sea turtle	*Eretmochelys imbricata*	Endangered
Leatherback sea turtle	*Dermochelys coriacea*	Endangered
Loggerhead sea turtle (NW Atlantic DPS)	*Caretta caretta*	Threatened
Invertebrates		
Elkhorn coral	*Acropora palmata*	Threatened
Staghorn coral	*Acropora cervicornis*	Threatened
Rough cactus coral	*Mycetophyllia ferox*	Threatened
Pillar coral	*Dendrogyra cylindrus*	Threatened
Lobed star coral	*Orbicella annularis*	Threatened
Mountainous star coral	*Orbicella faveolata*	Threatened
Boulder star coral	*Orbicella franksi*	Threatened
Fish		
Scalloped hammerhead shark (Central and Southwest Atlantic DPS)	*Sphyrna lewini*	Threatened
Nassau grouper	*Epinephelus striatus*	Threatened
Oceanic whitetip shark	*Carcharhinus longimanus*	Threatened
Giant Manta Ray	*Manta birostris*	Threatened

Table 3.2. Designated Critical Habitat In or Near U.S. Caribbean Federal Waters and Assessed in this Consultation

Species	**Unit(s) and/or Area(s)**
Green sea turtle	Culebra Island
Hawksbill sea turtle	Mona and Monita Island
Leatherback sea turtle	Sandy Point, St Croix, U.S. Virgin Islands
Staghorn and elkhorn corals	Area 2: Puerto Rico and Associated Islands, Area 3: St, John/St. Thomas, U.S. Virgin Islands, and Area 4: St. Croix, U.S. Virgin Islands

3.1 Analysis of Species Not Likely to be Adversely Affected

We have determined that the proposed actions being considered in this Opinion are not likely to adversely affect the following listed species under the ESA: sperm, sei, and fin whales; the Northwest Atlantic DPS of loggerhead sea turtle; and green, hawksbill and leatherback sea turtle critical habitats. The following discussion summarizes our rationale for these determinations.

3.1.1 Sperm, Sei, and Fin Whales

Endangered sperm whales may occur in U.S. Caribbean federal waters. In waters surrounding Puerto Rico and the U.S. Virgin Islands, NMFS winter ship surveys indicate that sperm whales inhabit continental slope and oceanic waters (Roden and Mullin 2000; Swart and Burks 2000; Swartz et al. 2002). Earlier sightings from the northeastern Caribbean have been reported by Erdman (1970), Erdman et al. (1973) and Taruski and Winn (1976), and these and other sightings from Puerto Rican and the Virgin Island waters are summarized by Mignucci-Giannoni (1988). Of the 2,016 cetacean sightings catalogued by Mignucci-Giannoni (1998) in the waters of Puerto Rico, U.S. Virgin Islands, and British Virgin Islands, based on published and unpublished records that were collected up until 1989, 43 sperm whales were sighted. Based on the sightings, Mignucci-Giannoni (1998) suggested that they occur from late fall through winter and early spring but are rare from April to September. The only two records of whales with calves were in late December and early January (Mignucci-Giannoni 1998). Mignucci-Giannoni (1998) reports that sperm whales were observed 64.2% of the time near the shelf edge, 26.4% offshore, and 9.4% over the continental shelf. Mignucci-Giannoni (1998) did not define these categories in units of depth, therefore we could not clearly determine whale presence in the action area, but we assume there some overlap. Sightings were recorded for Isla de Mona, Mona Passage, off Rincón, off San Juan and Loíza, south of Ponce, south of Isla de Vieques, north of St. Croix, along the southern shelf edge of the north-ern Virgin Islands, between St. Thomas and St. Croix, and off Anegada.

Endangered fin and sei whales may also occur in U.S. Caribbean federal waters. The 2,016 cetacean sightings catalogued by (Mignucci-Giannoni 1998) included three fin whales, two sei whales, and 14 additional *Balaenoptera spp.* whales sightings that were not identified to species due to difficulty in distinguishing at sea between fin whales and sei whales. The *Balaenoptera spp.* in Puerto Rico were observed only north of Isla de Mona, and south of Cayo Ratones in Salinas. Most sightings of *Balaenoptera spp*. from the Virgin Islands include north of Whale Banks, off Anegada, Virgin Gorda, Tortola, south of St. John and St. Thomas, and west and east of St. Croix. Sightings were equally distributed over the shelf (30.8%), near the shelf edge (30.8%) and in offshore waters (38.4%). Mignucci-Giannoni (1998) did not define these categories in units of depth, therefore we could not clearly determine whale presence in the action area, but we assume there some overlap.

In the 2020 List of Fisheries Final Rule, NMFS classified fisheries associated with the proposed actions as Category III fisheries, meaning that the annual mortality and serious injury of a stock resulting from each fishery is less than or equal to one percent of the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimum sustainable population (85 FR

21079). While it is possible that the proposed actions could overlap with the areas where these species are found, based on review of available data from 1978-2017 there were no records of fin or sei whale strandings in the USVI or Puerto Rico. While there were a few reports of sperm whale strandings, none of those had any documented signs of interactions with fisheries (L. Stratton, NMFS SEFSC, pers. comm. to P. Opay, NMFS SERO, November 14, 2017). Additionally, there are no documented interactions between these whales and these fisheries. NMFS expects that interactions with these species would be extremely unlikely to occur.

The depth at which sperm whales are generally found greatly reduces the likelihood of overlap between these whales and the proposed actions, Watkins et al. (2002) tracked sperm whale dives by radio tag telemetry and found that depth of the area they used typically was at least 200 m greater than the depth of the whale's dive (which means overall depth is greater than where FMP fishing would occur). For this further reason, we expect that interactions with sperm whales would be extremely unlikely to occur.

3.1.2 Loggerhead Sea Turtle

Loggerhead sea turtles are found in subtropical northern and southern oceans with only a few seen in the tropics. Although loggerhead sea turtles may be present in the action area, these sea turtles are uncommon in the U.S. Caribbean and are not reported to occur outside of nesting season and then only rarely. Hillis-Star et al. (1998) notes loggerhead turtles in the U.S. Caribbean are mainly transitory and only occasionally seen. For example, in Puerto Rico, loggerhead sea turtle nests have been reported by DNER in Loíza, Humacao, Vieques, and Culebra but nesting is infrequent. Only 2 loggerhead nests have been reported on the west coast of Puerto Rico. DNER stranding data (2008) contains one report of a loggerhead that was injured off the west coast of Puerto Rico in an attempt to hunt the animal, but the animal was rehabilitated and released; NMFS is unaware of any other stranding records of loggerhead sea turtles from the U.S. Caribbean. Until 2003, the southern and eastern most records of loggerhead sea turtles in the United States was Culebra. In 2003, 2 loggerhead sea turtles were identified on STX. The first one was a sea turtle that had been attacked by sharks and was found by snorkelers. The second positive identification was of a nesting sea turtle on Buck Island.

In areas where loggerhead sea turtles are abundant (e.g. Gulf of Mexico), they are subject to hook-and-line capture via their attraction to baited hooks (e.g., NMFS 2009). Loggerhead sea turtles are also typically vulnerable to entanglement in trap lines because of their attraction to, or attempts to feed on, species caught in the traps and epibonts growing on traps, trap lines, and floats (NMFS and FWS 1991b). However, given their rarity and mainly transitory nature in the action area, we believe adverse effects on loggerhead sea turtle interactions via gear authorized by the FMPs in the U.S. Caribbean are extremely unlikely.

3.1.3 Leatherback Sea Turtle

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

Leatherbacks face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (plastics, petroleum products, petrochemicals, etc.), ecosystem alterations (nesting beach development, beach nourishment and shoreline stabilization, vegetation changes, etc.), poaching, global climate change, fisheries interactions, natural predation, and disease.

While leatherbacks are vulnerable to entanglement in fishing gear, SEFSC (unpublished) data that reflects how the fisheries will operate under the proposed action indicates that this species is very unlikely to be caught in the federally-managed fisheries under the Puerto Rico, St. Thomas/St. John, and St. Croix FMPs.

3.1.4 Giant Manta Ray

This species has not been caught in the commercial fisheries in the federal waters off Puerto Rico, St. Thomas/St. John, or St. Croix. Additionally, the species has not been caught in the recreational fishery in federal waters off Puerto Rico. Recreational data are not collected in the U.S. Virgin Islands. However, Puerto Rico provides the closest proxy available to estimate what take may occur in St. Thomas/St. John and St. Croix, which has not caught giant manta rays. NMFS believes that the likelihood that this species would be caught in federal waters of the FMP areas is extremely low.

We believe that the risk of giant manta ray being struck by a vessel associated with the proposed actions is also extremely low. While giant manta rays can be frequently observed traveling just below the surface and will often approach or show little fear toward vessels, few instances of confirmed or suspected strandings of giant manta ray are attributed to vessel strike injury. This lack of documented mortalities could be the result of other factors that influence carcass detection (e.g., wind, currents, scavenging, decomposition, etc.); however, giant manta rays appear to be quick and agile enough to avoid most moving vessels, as is anecdotally evidenced by videos showing high speed vessels passing over giant manta rays and the ray being able to avoid the interaction. In general, information about interactions, whether lethal or nonlethal, between vessels and giant manta rays is limited, particularly within, or near, the action area. Available information indicates the threat of vessel strike on giant manta ray is predominantly an issue in shallow, coastal waters and in proximity to inlets where giant manta ray frequent, likely to facilitate feeding. Vessels associated with the proposed action could transit through some of these areas, for example inlets. Due to the expected low concentration of animals in the action area, very limited reports of vessel interactions, and ability to avoid moving vessel traffic outside of confined spaces, the likelihood that this species would interact with vessels associated with the proposed actions will be extremely low.

3.1.5 Critical Habitat

Sea Turtle Critical Habitat

Critical habitat for green and hawksbill sea turtles occurs in the Puerto Rico FMP action area, and leatherback critical habitat occurs in the St. Croix FMP action area. Critical habitat for the green sea turtle is designated in the waters surrounding the island of Culebra, Puerto Rico, from the mean high water line seaward to 3 nautical miles (Figure 3.1.A). These waters include Culebra's outlying keys including Cayo Norte, Cayo Ballena, Cayos Geniquí, Isla Culebrita, Arrecife Culebrita, Cayo de Luis Peña, Las Hermanas, El Mono, Cayo Lobo, Cayo Lobito, Cayo Botijuela, Alcarraza, Los Gemelos, and Piedra Steven (63 FR 46693, September 2,1998). Critical habitat for the hawksbill sea turtle has been designated in the waters surrounding the islands of Mona and Monito, Puerto Rico, from the mean high water line seaward to 3 nautical miles (63 FR 46693, September 2, 1998) (Figure 3.1.B). Critical habitat for the leatherback sea turtle has been designated in the waters adjacent to Sandy Point on the southwest corner of St. Croix, USVI and in waters from the 100-fathom curve shoreward to the level of mean high tide, with boundaries at 17°42′12"N and 64°50′00″W (44 FR 17710, March 23, 1979) (Figure 3.1.C). The critical habitats for green and hawksbill sea turtles were designated to provide protection mainly for important developmental and resting habitats. Critical habitat for leatherback sea turtles was designated to provide protection to sea turtles using the designated waters for courting, breeding, and as access to and from nesting areas on Sandy Point Beach, St. Croix, USVI. The commonwealth and territorial waters around Puerto Rico and USVI are included as part of the action areas because managing fisheries in federal waters under the proposed actions affects local herbivorous fish populations, which can affect corals and *Acropora* critical habitat outside of federal waters.

The critical habitat for green, hawksbill or leatherback sea turtles is not likely to be adversely affected by the proposed actions. The critical habitat for green sea turtles and hawksbill sea turtles lies entirely within Puerto Rico's waters, and over 99% of the critical habitat designated in the action area for leatherback sea turtles, due to the bathymetry around St. Croix, lies within USVI waters. Thus, fishing activities under the proposed actions would not overlap with the critical habitat areas and the proposed actions are unlikely to have any measurable effect on sea turtles use of these areas. Green and hawksbill sea turtles depend on healthy coral reefs, but their dependence is based on its importance as foraging and shelter habitats. Fish are not a common prey item of sea turtles so effects from reef fish harvest will not affect how the area functions as forage habitat. The proposed actions are not expected to result in changes in abundance of seagrasses (which are the principal dietary component of juvenile and adult green sea turtles) or sponges (which hawksbill sea turtles feed on). Effects on the ledges and caves of coral reefs in critical habitat areas and the shelter function they provide also are extremely unlikely from the proposed actions.

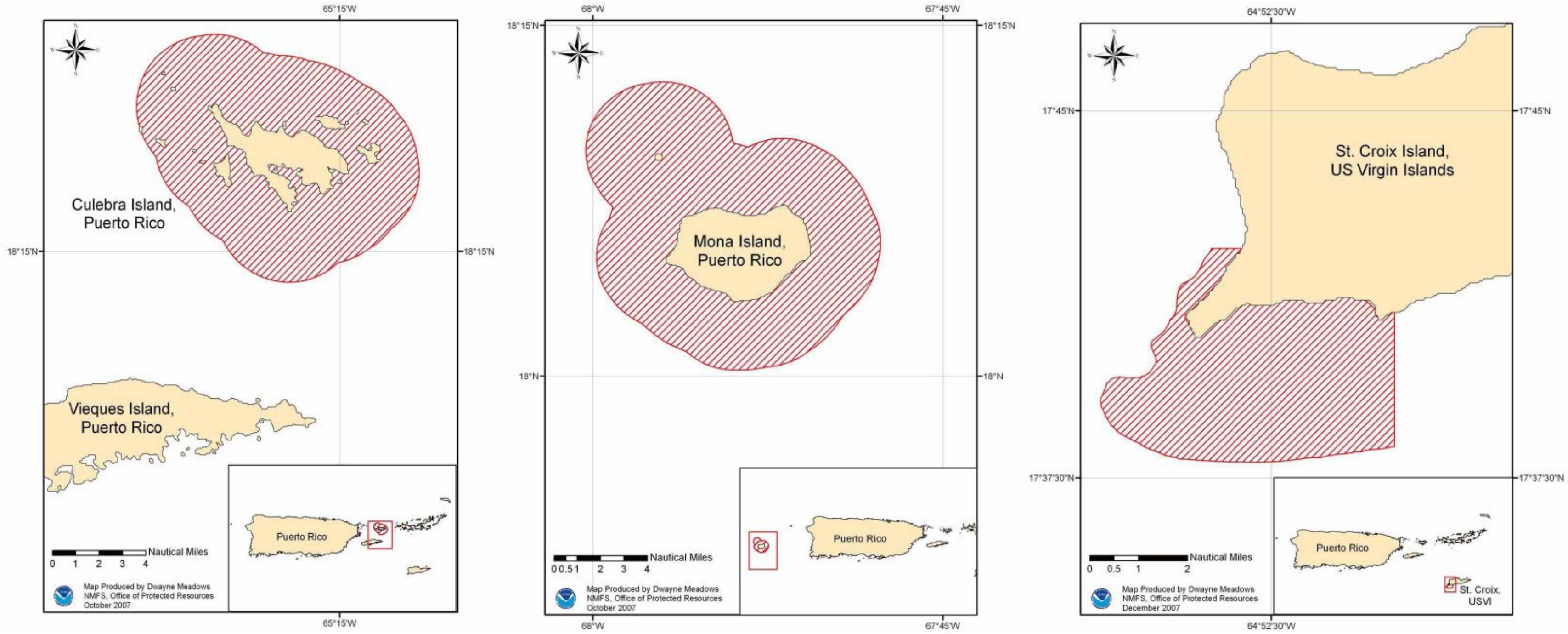


Figure 3.1 Sea Turtle Critical Habitat in the Caribbean
(A) Green Sea Turtle Critical Habitat (B) Hawksbill Sea Turtle Critical Habitat (C) Leatherback Sea Turtle Critical Habitat

3.2 Species and Critical Habitat Likely to be Adversely Affected

ESA-listed corals (i.e., elkhorn, staghorn, boulder star, mountainous star, lobed star, rough cactus, and pillar corals), eklhorn and staghorn coral critical habitat, the North Atlantic (NA) and South Atlantic (SA) DPSs of green sea turtles, hawksbill sea turtles, Nassau grouper, the Central and Southwest Atlantic DPS of scalloped hammerhead shark, and the oceanic whitetip shark may be adversely affected by the proposed actions. The remaining sections of this Opinion will focus solely on these species.

The following subsections are synopses of the best available information on the status of the species that are likely to be adversely affected by one or more components of the proposed actions, including information on the distribution, population structure, life history, abundance, and population trends of each species and threats to each species. The biology and ecology of these species as well as their status and trends inform the effects analysis for this opinion. Additional 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
https://www.fisheries.noaa.gov/welcome

3.2.1 General Threats Faced by All Sea Turtle Species

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

Fisheries

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

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

Non-Fishery In-Water Activities

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

Coastal Development and Erosion Control

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

Environmental Contamination

Multiple municipal, industrial, and household sources, as well as atmospheric transport, introduce various pollutants such as pesticides, hydrocarbons, organochlorides (e.g., dichlorodiphenyltrichloroethane [DDT], polychlorinated biphenyls [PCB], and perfluorinated chemicals [PFC]), and others that may cause adverse health effects to sea turtles (Garrett 2004;

Grant and Ross 2002; Hartwell 2004; Iwata et al. 1993). Acute exposure to hydrocarbons from petroleum products released into the environment via oil spills and other discharges may directly injure individuals through skin contact with oils (Geraci 1990), inhalation at the water's surface, and ingesting compounds while feeding (Matkin and Saulitis 1997). Hydrocarbons also have the potential to impact prey populations, and therefore may affect listed species indirectly by reducing food availability in the action area.

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

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

Climate Change

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

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

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

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

Other Threats

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

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

3.2.2 Green Sea Turtles

Information Relevant to All DPSs

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

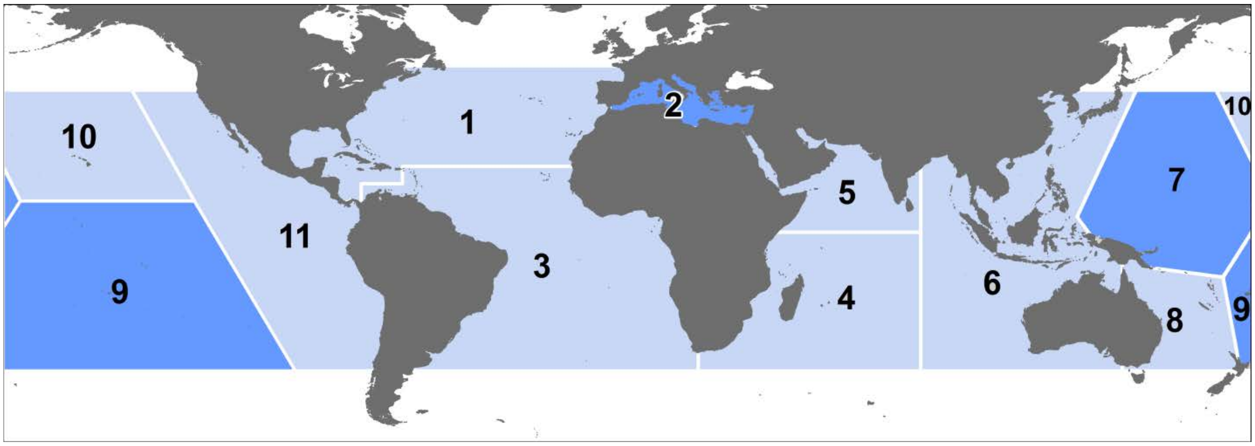


Figure 3.2. Threatened (light) and endangered (dark) green turtle DPSs: 1. North Atlantic, 2. Mediterranean, 3. South Atlantic, 4. Southwest Indian, 5. North Indian, 6. East Indian-West Pacific, 7. Central West Pacific, 8. Southwest Pacific, 9. Central South Pacific, 10. Central North Pacific, and 11. East Pacific.

Species Description and Distribution

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

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

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

North Atlantic DPS Distribution

The NA DPS boundary is illustrated in Figure 3.2. Four regions support nesting concentrations of particular interest in the NA DPS: Costa Rica (Tortuguero), Mexico (Campeche, Yucatan, and Quintana Roo), U.S. (Florida), and Cuba. By far the most important nesting concentration for green turtles in this DPS is Tortuguero, Costa Rica. Nesting also occurs in The Bahamas, Belize, Cayman Islands, Dominican Republic, Haiti, Honduras, Jamaica, Nicaragua, Panama, Puerto Rico, Turks and Caicos Islands, and North Carolina, South Carolina, Georgia, and Texas, U.S.A. In the eastern North Atlantic, nesting has been reported in Mauritania (Fretey 2001).

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

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

South Atlantic DPS Distribution

The SA DPS boundary is shown in Figure 3.2, and includes the U.S. Virgin Islands in the Caribbean. The SA DPS nesting sites can be roughly divided into four regions: western Africa, Ascension Island, Brazil, and the South Atlantic Caribbean (including Colombia, the Guianas, and Aves Island in addition to the numerous small, island nesting sites).

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

Life History Information

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

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

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

Status and Population Dynamics

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

North Atlantic DPS

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

Tortuguero, Costa Rica is by far the predominant nesting site, accounting for an estimated 79% of nesting for the DPS (Seminoff et al. 2015). Nesting at Tortuguero appears to have been increasing since the 1970's, when monitoring began. For instance, from 1971-1975 there were approximately 41,250 average annual emergences documented and this number increased to an average of 72,200 emergences from 1992-1996 (Bjorndal et al. 1999). Troëng and Rankin (2005) collected nest counts from 1999-2003 and also reported increasing trends in the population consistent with the earlier studies, with nest count data suggesting 17,402-37,290 nesting females per year (NMFS and USFWS 2007a). Modeling by Chaloupka et al. (2008) using data sets of 25 years or more resulted in an estimate of the Tortuguero, Costa Rica population's growing at 4.9% annually.

In the continental United States, green sea turtle nesting occurs along the Atlantic coast, primarily along the central and southeast coast of Florida where an estimated 200-1,100 females nest each year (Meylan et al. 1994; Weishampel et al. 2003). Occasional nesting has also been documented along the Gulf Coast of Florida (Meylan et al. 1995). Green sea turtle nesting is documented annually on beaches of North Carolina, South Carolina, and Georgia, though nesting is found in low quantities (nesting databases maintained on www.seaturtle.org).

In Florida, index beaches were established to standardize data collection methods and effort on key nesting beaches. Since establishment of the index beaches in 1989, the pattern of green sea turtle nesting has generally shown biennial peaks in abundance with a positive trend during the 10 years of regular monitoring (Figure 3.3). According to data collected from Florida's index nesting beach survey from 1989-2019, green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high in 2019 with almost 41,000 nests on the Index Nesting Beaches (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/). Two consecutive years of nesting declines in 2008 and 2009 caused some concern, but this was followed by increases in 2010 and 2011, and a return to the trend of biennial peaks in abundance thereafter (Figure 3.3). Modeling by Chaloupka et al. (2008) using data sets of 25 years or more has resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9% at that time. Increases have been even more rapid in recent years.

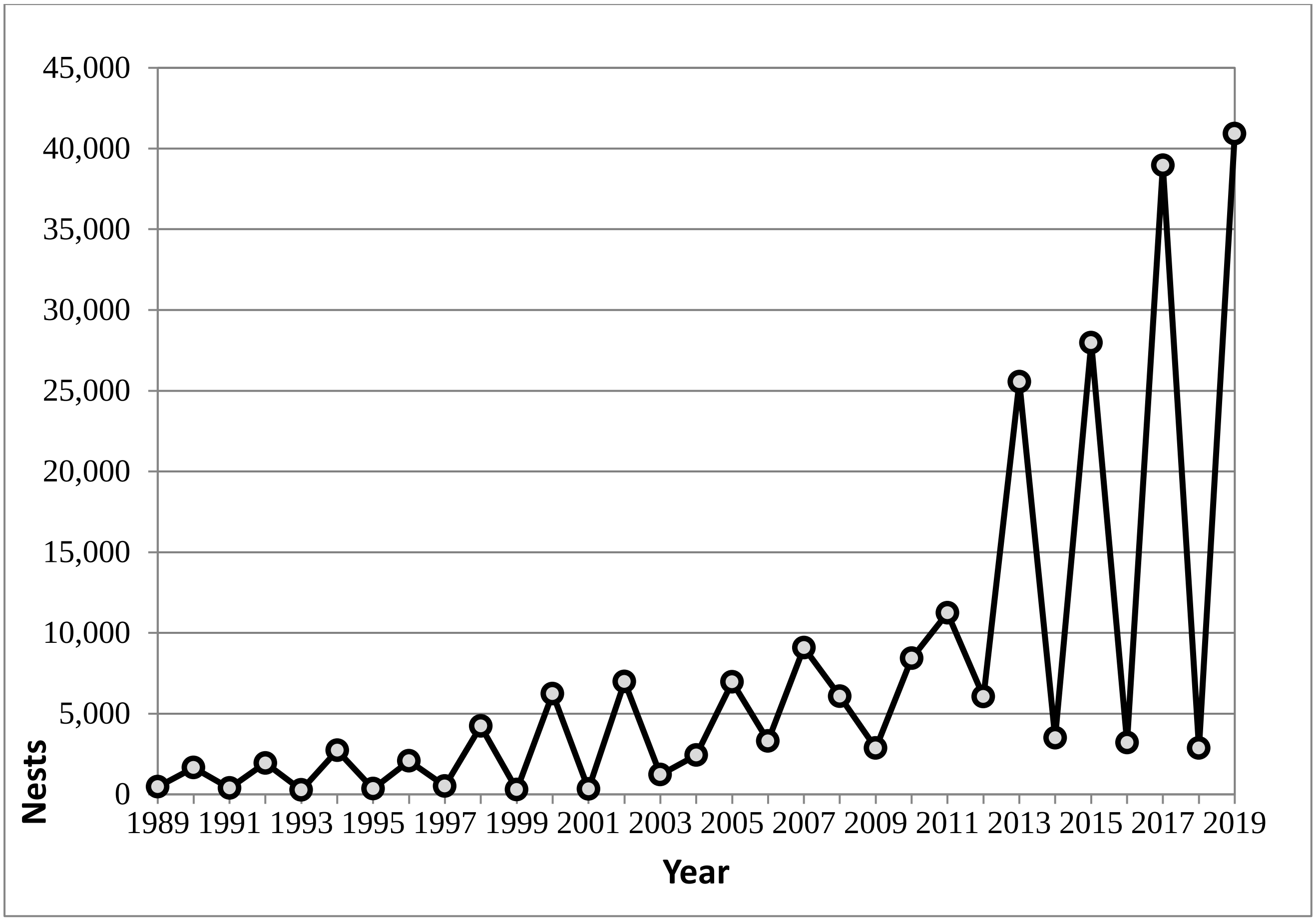


Figure 3.3. Green sea turtle nesting at Florida index beaches since 1989

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

South Atlantic DPS

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

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

Threats

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

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

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

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

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

3.2.3 Hawksbill Sea Turtle

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

Species Description and Distribution

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

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

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

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

Life History Information

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

Female hawksbills return to the beaches where they were born (natal beaches) every 2-3 years to nest (Van Dam et al. 1991; Witzell 1983) and generally lay 3-5 nests per season (Richardson et al. 1999). Compared with other sea turtles, the number of eggs per nest (clutch) for hawksbills can be quite high. The largest clutches recorded for any sea turtle belong to hawksbills (approximately 250 eggs per nest) (Hirth and Latif 1980), though nests in the U.S. Caribbean and Florida more typically contain approximately 140 eggs (USFWS hawksbill fact sheet, http://www.fws.gov/northflorida/SeaTurtles/Turtle%20Factsheets/hawksbill-sea-turtle.htm). Eggs incubate for approximately 60 days before hatching (USFWS hawksbill fact sheet). Hatchling hawksbill sea turtles typically measure 1-2 in (2.5-5 cm) in length and weigh approximately 0.5 oz (15 g).

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

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

Status and Population Dynamics

There are currently no reliable estimates of population abundance and trends for non-nesting hawksbills at the time of this consultation; therefore, nesting beach data is currently the primary information source for evaluating trends in global abundance. NMFS and USFWS (2013) reported that an estimated total of 22,004 to 29,035 hawksbills nest each year among 88 sites included in their evaluation (across the Atlantic, Indian, and Pacific Oceans). This is a rough estimate of total annual reproductive effort since not all nesting sites have been surveyed and included in the evaluation, some data are for single years, and some represent a professional judgment of the estimate of annual reproductive output. Among the 63 sites for which historic trends could be assessed, all 63 showed a decline during the long-term period of > 20 to 100 years. Among the 41 sites for which recent trend data are available, the picture is somewhat more optimistic with 10 (24%) increasing, 3 (7%) stable, and 28 (68%) decreasing. Although greatly depleted from historic levels, nesting populations in the Atlantic Ocean in general are doing better than in the Indo-Pacific.

The most significant hawksbill nesting in Puerto Rico occurs on Mona Island, and nesting also occurs on Culebra Island, Vieques Island, and some mainland beaches. Nesting populations appeared to be in decline until the early 1990s, but all have increase during the periods they were surveyed: Mona Island (1974-2005), +539%; Caja de Muertos (1995-2003), +23%; Culebra Island (1993-2005), +190; and Humacao (1987-2004), +930% (NMFS and FWS 2007) (NMFS and USFWS 2007b). Mona Island now hosts some 280-467 nesting females annually (van Dam et al. in press). The most significant nesting the USVI occurs at Buck Island Reef National Monument of St. Croix, but occurs elsewhere in the USVI. At Buck Island Reef National Monument, during the period from 1988 to 2006 hawksbill nesting increased by +143% to 56 nesting females annually. However, similar increase have not been recorded at St. John, perhaps due to the proximity of the legal turtle harvest in the British Virgin Islands (Mortimer and Donnelly (2008).

Nesting concentrations in the Pacific Ocean appear to be performing the worst of all regions despite the fact that the region currently supports more nesting hawksbills than either the Atlantic or Indian Oceans, a greater proportion of the nesting sites are declining.

Threats

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

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

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

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

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

3.2.4 Overview of Status of Corals

There are 7 species of corals (elkhorn, staghorn, lobed star, boulder star, mountainous star, pillar, and rough cactus) that occur on shallow coral reefs (see Figure 3.4) widely throughout wider-Caribbean, including south Florida, Puerto Rico, U.S. Virgin Islands, and the Gulf of Mexico (only star corals). Due to their broad distribution and sessile nature, these species may occur within the action area. This Section will address the general threats to all coral species. It will also summarize information on the dynamics relating to herbivorous fish and coral. Sections 3.2.6 – 3.2.12 will address the distribution, life history, population structure, abundance, population trends, and unique threats to each species of coral.

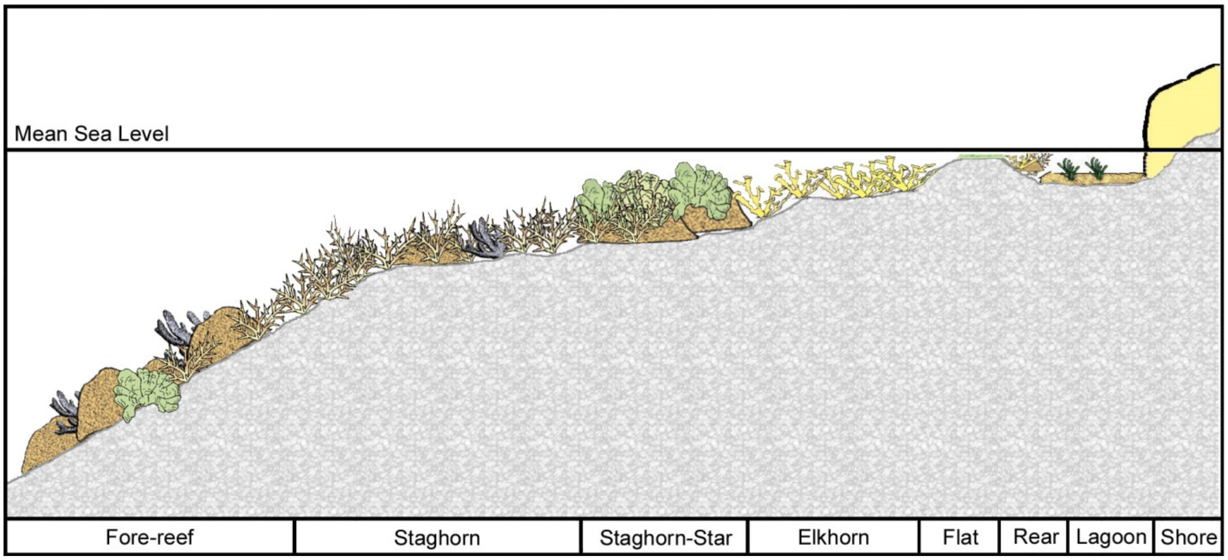


Figure 3.4. Reef zonation schematic example modified from several reef zonation-descriptive studies (Bak 1977; Goreau 1959).

General Threats Faced by All Coral Species

Corals face numerous natural and man-made threats that shape their status and affect their ability to recover. Many of the threats are either the same or similar in nature for all listed 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.

Ocean Warming

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

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

Ocean Acidification

Ocean acidification is a result of global climate change caused by increased CO_2 in the atmosphere and dissolving into 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 GHGs in the atmosphere since the Industrial Revolution, ocean acidification has already occurred throughout the world’s oceans, including in the Caribbean, and is predicted to increase considerably between now and 2100. Along with ocean warming and disease, we consider ocean acidification to be one of the most important threats posing extinction risks to coral species between now and the year 2100, although individual susceptibility varies among the listed corals.

Diseases

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

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’s 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.

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.

Trophic Effects of Reef Fishing

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

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

Emerging Hypothesis Regarding Environmental Conditions Favoring Algae Over Coral

There is increasing evidence that the concentrations of dissolved organic carbon (DOC) in waters of Caribbean Basin are adversely affecting corals in the region. There are 2 types of DOC, ‘labile’ which can be used easily by most marine organisms, and ‘refractory’ which relatively few organisms can use directly. DOC adversely affects corals in 2 ways: (1) indirectly by promoting the growth of algae, and (2) directly by affecting the health of coral colonies. DOC can adversely affect corals indirectly by promoting the growth of algae since they compete directly for space with coral. Algae and sponges can also directly affect corals by shading them, restricting water and gas flow across corals, or releasing alleochemicals (Loh et al. 2015), which are essentially chemical weapons.

Since corals also use DOC, its excessive amount is the problem, not its simple availability in the environment. Not only are corals in competition for space with algae, there is also evidence that DOC may reduce the overall health of corals. Corals do not have skins, bones, or an immune system to protect them; rather coral tissues are entirely exposed to the environment (J. Bohnsack, NMFS, to A. Herndon, NMFS, pers. comm. June 2016). Corals protect themselves by developing a balanced group of symbiotic organisms including the coral tissue itself, microbes (Bacteria and Archaea), fungus, and viruses (Barott et al. 2012) that live within the mucus layer of the coral tissue. Increased DOC can act as food for different species of microbes that can throw this relationship out of balance leading to disease and other ill effects in coral. Thus, as more DOC becomes available, the environment surrounding a coral can grow increasingly hostile. This also reduces their ability to compete for space because resources cannot be allocated toward growth or competing for space when they must be allocated toward healing.

Dynamics Relating to Herbivorous and Corallivorous Fish and Coral

In coral reef systems, herbivores play an important role in coral reef resilience by limiting the establishment and growth of algae, which can impede coral recruitment. In the Caribbean, herbivorous fish (e.g., parrotfish and surgeonfish) and echinoids (e.g., sea urchins) are equally important in maintaining algal levels (Green and Bellwood 2009).

Herbivorous reef fish have a range of feeding methods and consume a variety of plant material (e,g, macroalgae, epilithic algal turf, detrital material, algal mats and associated organisms such as bacteria). As such, species are categorized into functional groups based on how they feed, what they consume, and their impact to the substrate.

Parrotfish have a unique feeding method that differentiates them from other herbivorous fishes (Bonaldo et al. 2014). Surgeonfish tend to bite the upper portions of algae, which allows the basal portions of the algae to regrow rapidly. In contrast, the parrotfish scrape the substratum when biting, removing the top and basal portions of the algae as well as portions of the substrate. This feeding method has a greater impact on algal communities when compared to other herbivorous fish groups and may facilitate the settlement of corals by providing areas of bare substrate. Thus, as grazers of reef substrate, parrotfish perform a fundamentally different role from that of other herbivorous fish, highlighting the critical importance of parrotfish to the function they play for healthy coral reefs.

The role parrotfish play in the coral-algae relationship is largely dependent upon their *functional grazing group* ("grazing groups"). Species are classified into 1 of 3 grazing groups based on broadly defined diets and feeding methods: (1) scraping grazers, (2) excavating or bioeroding grazers or (3) macroalgal browsers. *Scraping grazers* graze relatively frequently, cropping down existing algal filaments and fighting its spread; they may or may not remove calcium carbonate. *Excavating or bioerodering grazers* take relatively deep, infrequent bites and usually remove portions of the hard bottom and reef substrate along with the algae attached, exposing bare substrate. Bites from these species often remain algae free longer than bites from scraping grazers (Bonaldo et al. 2014). *Macroalgae browsers* appear to be the only group of species that graze, to a significant degree, on established fleshy macroalgae (Bellwood and Choat 1990; Bellwood et al. 2004; Bonaldo et al. 2014). Table 3.3 lists parrotfish (Adam et al. 2015b) and surgeonfish species (ocean surgeonfish, doctorfish, blue tang) by grazing group. Surgeonfish are functionally classified as 'grazer/detritivores' that intensely graze algae, but do not scrape or excavate the substrate (Green and Bellwood 2009).

Table 3.3. Herbivorous Reef Fish Species of the U.S. Caribbean Managed Under the Proposed Actions and Their Functional Grazing Group

Scientific Name	Common Name	Grazing Group
Scarus coelestinus	Midnight parrotfish	Excavating or bioeroding grazers
Scarus coeruleus	Blue parrotfish	Scraping grazer
Scarus guacamaia	Rainbow parrotfish	Excavating or bioeroding grazers
Scarus iseri	Striped parrotfish	Scraping grazer
Scarus taeniopterus	Princess parrotfish	Scraping grazer
Scarus vetula	Queen parrotfish	Scraping grazer
Sparisoma aurofrenatum	Redband parrotfish	Macroalgal browser
Sparisoma chrysopterum	Redtail parrotfish	Macroalgal browser
Sparisoma rubripinne	Yellowtail/Redfin parrotfish	Macroalgal browser
Sparisoma viride	Stoplight parrotfish	Excavating or bioeroding grazers
Acanthurus bahianus	Ocean surgeonfish	Grazer/detritivore
Acanthurus chirurgus	Doctorfish	Grazer/detritivore
Acanthurus coeruleus	Blue tang	Grazer/detritivore

Together, these species appear to play complementary roles in algal grazing, meaning they graze a wider range of algae found on reefs than any 1 species or genus would alone. For example, some parrotfish (scraping, excavating or bioeroding grazers) feed on algal turf, preventing it from becoming established or growing into frondose seaweeds in areas where algae does not currently exist (Adam et al. 2015b). Conversely, other species of parrotfish (macroalgal browsers) eat already established macroalgae and prevent it from spreading and overgrowing corals, but are less effective at suppressing turf algae (Adam et al. 2015b). From an ecosystem perspective, the preferred diet and feeding patterns of each functional group complement one another. Each group targets and removes types of algae in ways others do not.

Burkepile and Hay (2008) provide a specific example of complementary feeding. They evaluated how different combinations of redband parrotfish (*Sparisoma aurofrenatum)*, ocean surgeonfish (*Acanthurus bahianus*) and princess parrotfish (*Scarus taeniopterus)* affected different algae species. By manipulating which species occurred together, they were able to evaluate how the diet preferences and grazing method affected algal communities. The authors concluded that increased herbivorous fish species richness led to reduced biomass and diversity of algae (Burkepile and Hay 2008). They stated that considering all herbivorous fish as a single functional group underplays the specific role each species plays in the coral-algae relationship. They even indicated redband parrotfish (*S. aurofrenatum),* had a disproportionately large effect on the algae communities in their study (Burkepile and Hay 2008).

In the Burkepile and Hay (2008) study, the redband parrotfish was most effective in controlling macroalgae in areas where it was already well established. The princess parrotfish was effective in controlling filamentous algae and preventing the establishment of macroalgae in areas where it had not yet become established (Burkepile and Hay 2008). Both types of algae can adversely affect corals and *Acropora* critical habitat, yet neither species of parrotfish was effective in controlling both types of algae on their own, meaning neither species is capable of mediating algae growth and spread and promoting coral recruitment by itself (Adam et al. 2015a).

Complementary feeding is clearly beneficial, but there are also benefits to species having very similar diet preferences, this creates *functional redundancy*. Functional redundancy likely increases resilience within a specific functional grazing group to the loss of single species (Adam et al. 2015a; Walker 1992). For example, when a species declines due to overfishing or disease, species with similar dietary preferences and feeding mechanisms may be able to compensate for the loss or reduction in algal grazing caused when that species becomes less abundant (Adam et al. 2015a).

The importance of redundant and complementary dietary preferences is reinforced when you consider that some herbivorous fishes may actually be food-limited on algae-dominated reefs since macroalgae frequently have physical or chemical defenses that only certain species of herbivorous fish can eat (Adam et al. 2015a). For example, parrotfish that feed on turf algae may not be able to eat frondose seaweeds, meaning that even when frondose seaweeds are abundant certain guilds of parrotfish may actually suffer from food shortages. Increasing herbivore food limitation would further reinforce an algae-dominated state (Adam et al. 2015a).

Impact of Corallivorous Parrotfish on Corals in the U.S. Caribbean

Something else that must be considered when discussing the role parrotfish play in the coral-algae relationship is *corallivory*. Caribbean parrotfish are considered largely herbivorous with most species not consuming corals at all. However, the adults of a few species such as stoplight (*Sparisoma viride*), redband *(Sparisom aurofrenatum*), midnight *(Scarus coelestinus*), and rainbow *(Scarus guacamaia*) have been documented consuming live corals in the Caribbean (Glynn 1997; Miller and Hay 1998; Randall 1967a; Randall 1974; Rotjan and Lewis 2006). However, the overall proportion of coral in the diet of these species is very low. Bites of live coral are generally no more 1-2% of those species' diets; though up to 5% of stoplight parrotfish bites have been documented occurring on live coral (Bonaldo et al. 2014; Bruggemann et al. 1994a; Bruggemann et al. 1994b; Rotjan and Lewis 2006). In some cases, it appears that the parrotfish were targeting coral boring organisms, not actual coral tissue (Rotjan and Lewis 2005), while in others it appears that parrotfishes actively avoided newly settled corals (Birkeland 1977).

Because parrotfishes bite relatively deep into coral skeletons, *corallivory* by these species, is likely to have a greater impact on corals than feeding by other corallivores (e.g., butterfly fish) that bite or pick individual polyps when feeding (Bonaldo et al. 2014). The tissue damage caused by the feeding of a butterfly fish (i.e., loss of a single polyp) may regenerate within 7–10 days, whereas parrotfish grazing scars can take from 40 days to 3 years to heal (Bak and Steward-Van Es 1980; Bonaldo et al. 2011; Bruckner and Bruckner 1998; Jayewardene and Birkeland 2006; Sanchez et al. 2004). It appears that corallivorous parrotfish prefer to graze on slower growing massive and submassive corals, such as *Porites* and *Orbicella* (Bellwood et al. 2003; Hoey and Bellwood 2008). Although parrotfish may have negative effects on corals (Littler et al. 1989; Miller and Hay 1998; Mumby 2009a; Rotjan and Lewis 2005), records of total colony mortality caused by parrotfish are relatively rare (Bruckner and Bruckner 1998; Francini-Filho et al. 2008; Rotjan and Lewis 2005). Even those coral species grazed most intensely (e.g., *Orbicella* complex) appear to have a high capacity for tissue regeneration with little evidence that processes of corallivory exceed the ability of coral to regenerate tissue from predation, even on small ramets of coral (Mumby 2009a; Sanchez et al. 2004). Venera-Ponton et al. (2011) found that predation upon the corals *Porites astreoides* by parrotfish neither lessened survivorship nor resulted in negative growth for any of the grazed colonies.

While parrotfish grazing can cause coral recruit mortality through accidental predation, their grazing also clears substrate to create suitable settlement habitat (Bonaldo et al. 2014). Studies from the Great Barrier Reef in Australia, indicate the coral larvae mortality increased in areas where parrotfish were present, relative to those areas where parrotfish were excluded (Trapon et al. 2013). Conversely, several studies have found that the densities of juvenile corals increased as the population density of parrotfish increased (Hoey et al. 2011; Hughes et al. 2007; Penin et al. 2011). This is likely because increases in grazing activity increase the amount of suitable settlement sites and prevent smothering of newly settled corals by algae and sediment on the algae (Bonaldo et al. 2014). However, it is possible that at higher parrotfish and coral larvae densities, these positive effects may be outweighed by the effects of increased mortality through incidental grazing (Bonaldo et al. 2014). Others have concluded that corallivory may constitute a source of mortality in coral recruits, but those negative impacts are outweighed by the positive effects of removing algal competitors (Mumby et al. 2007a; Mumby et al. 2007b).

Phase-Shift and Resilience

The ability of a reef to resist a phase shift depends on its *resilience*. Resilience refers to a coral reef systems ability to undergo disturbance, acclimate to changes, or recover, without completely switching to an alternative persistent state (Hughes et al. 2010). Consider a “healthy reef” with competition occurring between coral and algae, but generally in a high coral, low algae condition. A resilient reef will maintain the high coral, low algae state even when exposed to impacts (e.g., sedimentation, increasing water temperatures, fishing, storms) because those effects act relatively slowly across the entire system, and corals often have time to compensate.

It is the combination of chronic and acute stressors that often overwhelm a reef’s capacity for resilience and leads to phase shifts. Phase shifts in complex systems, like a reef, are generally spurred when a threshold is reached, not during a slow linear decline. So while a high coral, low algae reef may be able to withstand the chronic stress of sedimentation for years without shifting phase, those adverse effects likely degraded the system. It might not be until an acute stressor affects the system that the threshold for phase shift is breached (Hughes et al. 2010). A reef suffering from chronic sedimentation may finally undergo a phase shift following a hurricane because chronic sedimentation impaired the reefs potential to replenish itself to such an extent that it simply cannot recover. Because of the role parrotfish play in removing and moderating algal growth, managing their harvest is a way marine resource managers can help promote reef resilience.

Phase Shift Impacts to Herbivorous Fish Habitat

Coral and reef structure composed of dead coral skeletons can provide important habitat for many reef species, including herbivorous fish (Adam et al. 2014). Because dead coral can remain in place even after the coral tissue has died, tissue loss does not immediately result in habitat loss (Alvarez-Filip et al. 2009). Dead coral skeletons are even considered a feature essential to the conservation of *Acropora* coral by providing a surface for larval settlement (73 FR 72210, November 26, 2008).

The amount of structural complexity of a reef is referred to as *rugosity*. Coral scientists measure it on a on a scale ranging from 1 (completely flat) to infinity; though values above 3 are considered very rare (Alvarez-Filip et al. 2009). The rugosity of reefs in the Caribbean has declined dramatically since the late 1960s. Alvarez-Filip et al. (2009) reviewed 464 records from 200 reefs across the Caribbean for the period 1969-2008. Those reports indicate from 1969-2008, the number reefs with a rugosity index value of 2 or more declined from 45% of sites to 2% of sites.

Alvarez-Filip et al. (2009, 2011a) indicate that the dramatic loss of *Acropora* over the past decades has led to increasing numbers of “weedy” corals (i.e., *Porites* spp. and *Agaricia* spp.) and larger slower-growing boulder corals (i.e., *Orbicella* complex). Alvarez-Filip et al. (2009, 2011a) indicate *Orbicella* spp., also in decline, are now the primary reef builders left in the Caribbean Basin. However, these boulder corals will lead to reefs with less structure relative to reefs composed of branching species like *Acropora*. These boulder corals also grow more slowly than *Acropora*, meaning their capacity to compete for space with fast-growing algae is lower (Alvarez-Filip et al. 2011; Hughes 1994)).

Acropora were the primary coral species creating the structural framework of high rugosity reefs. The replacement of these species by weedy and boulder corals reduces the overall complexity of reefs. This loss of reef structural complexity may mean a loss of habitat for certain species, some of which are commercially important in the U.S. Caribbean (e.g., snappers, groupers, parrotfish, and lobsters). Paddack et al. (2009) conducted a meta-analysis of reef fish density using 48 studies across 318 reefs in the Caribbean, for the period 1955-2007. The authors report that reef fish densities from 1996-2007 have been declining throughout the Caribbean. This includes decreasing densities of commercially important snapper, groupers, and spiny lobsters. They concluded the declines were not the result of fishing or the recovery of large predators, but rather habitat loss, particularly the loss of structurally complex reefs (Paddack et al. 2009).

The relationship between herbivorous fish and coral appears to be particularly important, with each potentially affecting the availability of habitat for the other. For example, since only adult corals spawn, once adult corals are lost, coral cover and reef complexity are likely to decline because the loss of adults creates a bottleneck constraining the rate at which adults are replaced (Mumby 2009b). Since herbivorous fish require at least some level of predator protection, when reef complexity degrades or collapses, herbivory is also likely to diminish or cease (Adam et al.

2015a; Hoey and Bellwood 2011; Madin et al. 2010; Randall 1965b). Declining coral cover also liberates new space for macroalgal colonization. Without a commensurate increase in the amount parrotfish grazing, increasing levels of macroalgae essentially decreases the intensity of grazing (i.e., the rate at which each patch of substrate is regrazed tends to decline as available area increases) (Mumby 2009b; Williams et al. 2001). This reduction or cessation in herbivory establishes yet another feedback loop. The drop in grazing intensity allows macroalgal cover to increase further, thus strengthening the bottleneck in the coral population, and further reinforcing the feedback loop (Mumby 2009b). This diminishes the likelihood of reef architectural complexity on a reef increasing over time, leading to less habitat for herbivorous fish, which likely reduces grazing Such a situation may all but lock in an algal dominated state (Adam et al. 2015b; Blackwood et al. 2011; Bozec et al. 2013; Hughes et al. 2007).

While acknowledging the potentially important influence a reef's structural framework may have on herbivorous fish, Bonaldo et al. (2014) caution against making generalizations. The authors explain that due to the complexities of those relationships and local disturbance factors (e.g., coral bleaching, storms), it is difficult to come to any generalization on the role reef structure plays on parrotfish distribution among reefs.

Bonaldo et al. (2014) report that advances in herbivorous fish tracking have allowed researchers (Howard et al. 2013; Welsh and Bellwood 2012a; Welsh and Bellwood 2012b) to determine the home ranges of many herbivorous fish are smaller than initially believed. As a consequence, grazing on a given reef likely becomes very dependent on the local populations of herbivorous fish (Bonaldo et al. 2014; Howard et al. 2013; Welsh and Bellwood 2012a). Relatively small home ranges may have an additional consequence, depending on the relationship between reef structure and herbivorous fish abundance. If herbivorous fish are limited to structurally-complex reefs, and they are likely to stay relatively close to those habitats, it suggests that grazing in areas lacking that structure will decline over time as herbivorous fish prefer to remain near structurally-complex reefs.

Importance and Grazing Capacity of Herbivorous Fish Versus Sea Urchins

As coral cover decreases on reefs, the space available for colonization by algae increases. One of the ecological factors influencing the speed at which algae can grow and spread is herbivory. Historically in the Caribbean, herbivorous fish (i.e., surgeonfish and parrotfish) and sea urchins (*Diadema antillarum, "Diadema"*) were the primary herbivores playing a role in the coral-algae relationship. However, the overall importance of these species to the coral-algae relationship is not equal. Burkepile and Hay (2008) indicate that while surgeonfish clearly play a role in the coral-algae relationship, their overall grazing contribution appears to be less than parrotfish and sea urchins. Teasing out the importance of *Diadema* and parrotfish to the coral/algae relationship is more difficult.

A common evaluation of the role of *Diadema* and parrotfish on grazing focuses on the periods immediately following a well-documented *Diadema* collapse in the early 1980s. Immediately prior to the collapse, *Diadema* was undoubtedly the primary grazer on most Caribbean reefs. Following the mass mortality, Caribbean reefs started to experience changes in coral and algal cover. Immediately following the mass mortality event, the amount of algae increased on reefs where *Diadema* had been the dominant grazer, and a phase shift from coral-dominated to algal-dominated reefs began; this shift occurred rapidly in places (Carpenter 1990; Levitan 1988). For *Acropora* species, the *Diadema* mass mortality and subsequent phase shift toward more algae-dominated reefs was compounded by an outbreak of white band disease (WBD), which appears to have preferentially targeted *Acropora* species and is the only coral disease to date that has been documented to cause major changes in the composition and structure of reefs (Humann and Deloach 2003). Evaluating the importance of *Diadema* grazing at this period of time clearly suggests that the species played the most significant role in coral-algae relationship.

However, going back further in time suggests potentially different roles for *Diadema* and parrotfish in the coral-algae relationship. Historically, herbivorous fish grazing may have been as, if not more, important on Caribbean reefs than *Diadema* (Adam et al. 2015a). Some studies suggest that urchins were the most important grazing animals in reef environments prior to the 1980s (Ogden et al. 1973; Woodley 1979), Levington 1982). However, others suggest urchins were the most important grazers on reefs only because herbivorous fish populations had been significantly reduced by fishing (Hay 1984; Hay et al. 1983; Lewis 1986; Lewis and Wainwright 1985; Wanders 1977).

For example, Hay (1984) compared the contributions of *Diadema* and parrotfish to overall grazing of seagrass on "lightly to unfished" reefs and "heavily" fished reefs in the USVI, Panama, Honduras, Belize, and the Bahamas. Based on the information reported, the author suggested that herbivorous fish were the grazers of primary importance on "lightly to unfished" reefs, while *Diadema* were the primary grazers on "heavily" fished reefs. Ogden et al. (1973) also determined *Diadema* were the primary grazers at their study site (St. Croix, USVI). However, they noted their surprise at the close association *Diadema* maintained with the reef, even though the major predators of *Diadema* near their study site "ha[d] been severely overfished in recent years" (Ogden et al. 1973). This suggests the importance of *Diadema* in the coral-algae relationship prior to its collapse in the 1980s may have been more the result of previously important herbivorous fish and *Diadema* predators being removed via fishing and not because *Diadema* filled a unique ecological role (Adam et al. 2015a; Hay 1984).

Hughes (1994) documented the shift from a coral-dominated to algal-dominated reef in Jamaica. The author suggested the framework of this decline as: (1) fishing reduced herbivory by herbivorous fish, (2) *Diadema* became abundant (Hay 1984), (3) Hurricane Allen caused extensive damage to corals in 1980, (4) *Diadema* die off in 1983, (5) fleshy algae begin to spread following *Diadema* die off, (6) bleaching events (1987, 1989, and 1990) further reduce coral cover, (7) intense cover of fleshy algae caused by reduced herbivory inhibits settlement of corals (McManus 2000). In the Caribbean, Idjadi et al. (2010) found that it appears that *Diadema* are effective at enhancing scleractinian coral recruitment and growth.

While it does not appear that *Diadema* fill a unique ecological role, they do possess traits that make them very effective grazers. Unlike herbivorous fish that appear to have specific diet preferences, *Diadema* eat a wider variety of plants and algae (Ogden 1976). *Diadema* also have a tendency to graze very intensively over small areas (Carpenter 1984). Collectively, these traits allowed *Diadema* to effectively provide the ecological functions of several herbivorous fish species at once (Adam et al. 2015a). In essence, *Diadema* were "super grazers" that individually removed the vast majority of several types and species of algae in small areas. As their overall numbers increased in response to reduced predation and increasing food (i.e., algae), the amount of area grazed was substantial and their importance as grazers became significant. The large numbers of *Diadema* on reefs prior to their collapse may have helped mask ecological impacts caused by the removal of herbivorous fish (Adam et al. 2015a).

Whether *Diadema* masked any ecological effects from fishing is difficult to tease out from the available data. There is evidence from throughout the Caribbean that as larger, more desirable fish species (e.g., snappers and grouper) are removed from the system, other species are targeted, often times parrotfish (Friedlander and Beets 2008; Mumby et al. 2012). Even in the U.S. Caribbean there is evidence that fishing pressure on all stocks was beginning to cause concern for fishery managers as early as the 1980s. In their original 1985 fishery management plan for the shallow-water reef fish fishery, the Caribbean Fisheries Management Council (CFMC) expressed concern over declining snapper and grouper stocks (CFMC and NMFS 1985). One of the original objectives of that FMP was to "[p]revent the harvest of individuals of species of high value (e.g. snappers, groupers, and others) which are less than optimum size" (CFMC and NMFS 1985). The FMP even mentions a dramatic increase in demand for local fish occur just after World War II (WWII) as a result of an increase in military personnel moving to the islands following the war (CFMC and NMFS 1985). This suggests that fishery stock information from prior to WWII would be necessary to properly evaluate changes in the fishery over time.

By 1990, the CFMC had to amend the original FMP because "the parrotfish, which was considered second or third class in most sectors of this [shallow-water reef fish] fishery, is now sold as first class and is one of the most frequently landed species displacing the less abundant snappers and groupers" (CFMC and NMFS 1990).

It is worth noting that limited *Diadema* recovery has been reported in some areas. However, *Diadema*'s future role as an important grazer on Caribbean reefs is unclear. It is unclear how quickly *Diadema* recovery may occur or whether Caribbean-wide populations will ever reach the pre-die off levels.

Additionally, *Diadema* are far more destructive bioeroders than parrotfish. Only a few species of parrotfish erode significant volumes of reef substrate when feeding. *Diadema*, conversely, burrow into and erode the reef matrix (Bellwood et al. 2004; Ogden 1977). In high densities *Diadema* could undercut and dislodge massive corals. If they do reach pre-die off populations levels and densities again, they may cause significant issues in reef stability. Evidence of urchins' capacity to destroy reefs has been documented in the Galapagos Islands and elsewhere in the East Pacific (Eakin 1996; Glynn 1988).

Finally, the *Diadema* die-off appears to have been caused by infectious disease that spread quickly throughout the entire Caribbean basin (Adam et al. 2015a; Lessios 1988). While the actual causative agent of the epidemic has never been identified, the outbreak may have been worse because of the unusually high densities of the urchin due to the elimination of its predators and competitors. Infectious disease appears to be a common cause of boom and bust dynamics in other echinoderms (Adam et al. 2015a; Uthicke et al. 2009), suggesting that relying on urchins to be the primary herbivores on reefs may set the stage for repeated, catastrophic losses of herbivore populations (Adam et al. 2015a).

Scientific Debate Regarding the Significance of Herbivore-Mediated Algal Impacts

The scientific literature includes arguments about the significance of the herbivore-mediated algal impacts to corals and coral reefs (Hughes et al. 2010, Bruno et al. 2009, Aronson and Precht 2006). In particular, there is debate on the extent that phase shifts (i.e., shifts from high coral and low macroalgal cover to high macroalgal and low coral cover reefs) are occurring and how widespread and generalizable the reef phase shift problem is. Comparisons of reef health within and across regions also raise questions about the relative significance of herbivore-mediated algal impacts overall.

Bruno et al. (2009) evaluated the conditions of coral reefs from four global regions (i.e., the Greater Caribbean, the Florida Keys, the Great Barrier Reef [GBR], and the Indo-Pacific) for evidence that phase shifts from coral dominated reef systems to macroalgae dominated reef systems were occurring regionally and worldwide, and whether phase-shifts reported for certain locations was broadly generalizable. When determining what constituted a phase-shifted reef, the authors state that macroalgae or coral do not necessarily need to occupy a majority (50%) of the benthos to be considered dominating or defining a community. Rather, they stated that for their study they considered reefs with 25% or more macroalgal cover to be algal dominated (Bruno et al. 2009). Based on that threshold, their meta-analysis of previous survey data concluded that "the replacement of corals by macroalgae as the dominant benthic functional group is less common and less geographically extensive than assumed." However, their study did indicate macroalgal growth appears to be disproportionately affecting the Caribbean and, to a lesser extent, the Florida Keys relative of the other regions of the world (i.e., the Indo-Pacific and GBR). Twenty percent of all the reefs surveyed in the study (n=1851) had macroalgal cover of 25% or more; however, 52% of those reefs occurred in the Caribbean, even though the region accounts for only 8% of the world's reefs. The Caribbean also had the highest average macroalgae cover (23.6%) across reefs from all regions, followed by the Florida Keys. Average coral cover in the Keys, however, was the lowest of any region and was 60% less than the Caribbean reefs.

Resiliency

Bruno et al. (2019) suggest that while scientists have advocated for local actions such as fishing restrictions on herbivorous fish as a mechanism to mitigate local stressors to limit the effects of climate change on reef-building corals, they found little empirical support for the notion of managed resilience. They outline some reasons why the protection of herbivorous fish (especially parrotfish) has had little effect on coral resilience, suggesting that it is possible (among other things) that the impacts of local stressors are often swamped by the much greater effect of ocean warming on corals. However, they acknowledge that other coral reef scientists believe that marine protected areas and overall attenuations of local stressor intensity indirectly confer resilience to reef ecosystems in general and coral populations in particular (i.e., the managed-resilience hypothesis). Controlling local stressors is thought to improve coral resistance to and recovery from disturbances such as storms, disease outbreaks, and mass bleaching caused by ocean warming. They further acknowledge that some scientists believe that increased resilience is conferred to corals via a range of physiological mechanisms, through stronger immunity and better health, and ecological processes, e.g., by increasing grazing by herbivorous fish and thereby lowering competition with space-monopolizing macroalgae.

However, Bruno et al. (2019) conclude that "the managed-resilience paradigm has virtually no empirical support." These competing arguments demonstrate the uncertainty regarding corals' resiliency to grazing. Given this uncertainty, including which dynamics are mostly likely at play in Puerto Rico and the USVI, NMFS acts in a conservative manner and assumes that healthy herbivorous fish populations are beneficial to coral reefs and habitat. Because of the fundamentally different way that parrotfish interact with macroalgae on coral reefs compared to other herbivorous fish species (i.e., surgeonfish), we assume parrotfish are of utmost importance when considering the effects of herbivory on corals and *Acropora* critical habitat.

3.2.5 Elkhorn coral (*Acropora palmata*)

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).

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).

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). However, growth rates in Curaçao have been reported to be slower today than they were several decades ago (Brainard et al. 2011c). Annual linear extension 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[28] species that reproduces sexually after the full moon of July, August, 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 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.

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

Status and 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 appears to be 2 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 ± 0.08 colonies per m^2). The eastern Caribbean populations are characterized by denser (0.30 ± 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 2 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 2 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. 2011c). 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. 2011c). 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 US 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 US 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 US 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 US 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 at reef sites has remained relatively stable at approximately 1% of reef sites 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.

Threats

A summary of threats to all corals is provided in Section 3.2.5 General Threats Faced by All Coral Species. Detailed information on the threats to elkhorn coral can be found in the Final Listing rule (79 FR 53852; 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.

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.

3.2.6 Staghorn coral (*Acropora cervicornis*)

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).

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).

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. 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.

Status and 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 were 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).

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. 2011c). 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. 2011c). 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). There are some small pockets of remnant robust populations such as in southeast Florida (Vargas-Angel et al. 2003), Honduras (Keck et al. 2005; Riegl et al. 2009), and Dominican Republic (Lirman et al. 2010). Additionally, Lidz and Zawada (2013) observed 400 colonies of staghorn coral along 44 miles (70.2 km) of transects near Pulaski Shoal in the Dry Tortugas where the species had not been seen since the cold water die-off of the 1970s.

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 US 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 US 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 US 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 US 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.

Threats

A summary of threats to all corals is provided in Section 3.2.5 General Threats Faced by All Coral Species. Detailed information on the threats to staghorn coral can be found in the Final Listing rule (79 FR 53852; 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.

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.

3.2.7 Pillar Coral (*Dendrogyra cylindrus*)

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

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. (2011c) 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).

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[29] 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

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

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.

Status and 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 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 3.5).

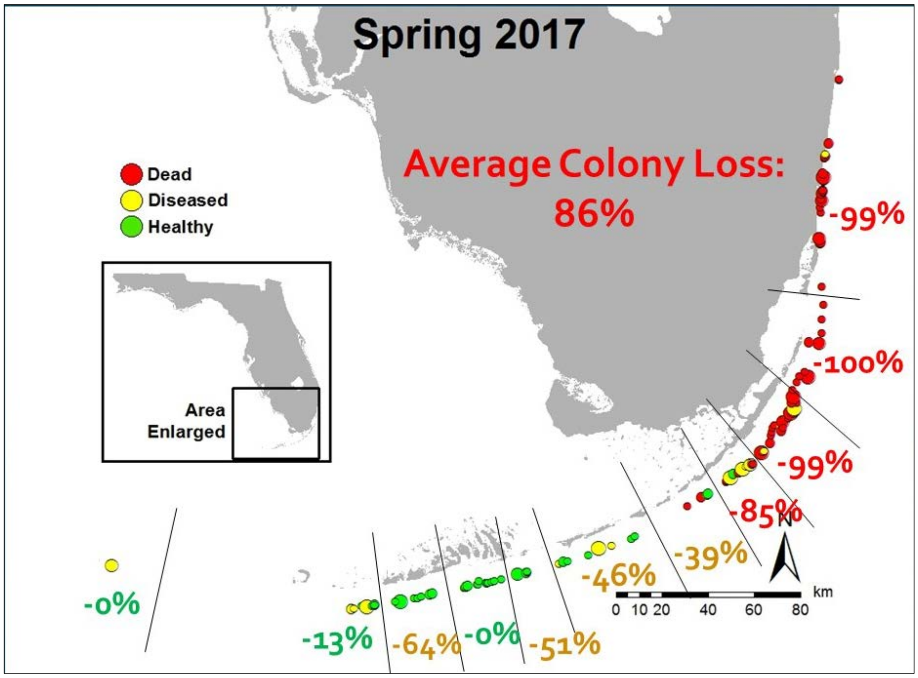

Figure 1.5 Condition of known pillar coral colonies in Florida between 2014 and 2017 (Figure curtesy of K. Neely and C. Lewis).

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 US 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 US 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 US Virgin Islands, though qualitative observations indicate that damage was also widespread but variable by site.

Other than the declining population in Florida, there are two reports of population trends from the Caribbean. In monitored photo-stations in Roatan, Honduras, cover of pillar coral increased slightly from 1.35% 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.

Threats

A summary of threats to all corals is provided in Section 3.2.5 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 53852; 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. 2011c). 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 3.5).

Pillar coral appears to be moderately capable of removing sediment from its tissue (Brainard et al. 2011c). However, pillar coral may be more sensitive to turbidity due to its high reliance on nutrition from photosynthesis (Brainard et al. 2011c) 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. 2011c), 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.

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.

3.2.8 Rough Cactus Coral (*Mycetophyllia ferox*)

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

Species Description and Distribution

Rough cactus coral forms a thin, encrusting plate that is weakly attached to substrate. Rough cactus coral is taxonomically distinct (i.e., separate species), though difficult to distinguish in the field from other *Mycetophyllia* species. Maximum colony size is 20 in (50 cm) in diameter.

Rough cactus coral occurs in the western Atlantic Ocean and throughout the wider Caribbean Sea. It has not been reported in the Flower Garden Banks (Gulf of Mexico) or in Bermuda. It inhabits reef environments in water depths of 16-295 ft (5-90 m), including shallow and mesophotic habitats (e.g., > 100 ft [30 m]).

Life History Information

Rough cactus coral is a hermaphroditic brooding[30] species. Colony size at first reproduction is greater than 15 in^2 (100 cm^2). Recruitment of rough cactus coral appears to be very low, even in studies from the 1970s. Rough cactus coral has a lower fecundity compared to other species in its genus (Morales Tirado 2006). Over a 10-year period, no colonies of rough cactus coral were observed to recruit to an anchor-damaged site in the U.S. Virgin Islands, although adults were observed on the adjacent reef (Rogers and Garrison 2001). No other life history information appears to exist for rough cactus coral.

Status and Population Dynamics

Information on rough cactus 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.

Rough cactus corals are uncommon and typically occur at an average density of <0.001 to 0.02 colonies per m^2. In benthic surveys conducted in the US Virgin Islands between 2002 and 2018, rough cactus corals were encountered in less than half of the survey years, and density was $\leq$0.001 colonies per m^2 at the 1% to 2% of sites where they occurred (NOAA, unpublished data). Rough cactus corals were present at 8% of sites surveyed in Puerto Rico in 2008, but in surveys conducted between 2010 and 2018, they were found at 1% to 4% of surveyed sites at an average density of <0.001 to 0.004 colonies per m^2 (NOAA, unpublished data). Rough cactus corals were encountered in 2% to 10% of sites surveyed in Florida between 1999 and 2006, but in

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[30] Simultaneously containing both sperm and eggs, which are fertilized within the parent colony and grows for a period of time before release.

surveys between 2007 and 2017, they were only encountered in three survey years and at only 1% of sites at an average density of <0.001 colonies per m^2 (NOAA, unpublished data). Density of rough cactus coral in southeast Florida and the Florida Keys was approximately 0.8 colonies per approximately 100 ft^2 (10 m^2) between 2005 and 2007 (Wagner et al. 2010). In a survey of 97 stations in the Florida Keys, rough cactus coral declined in occurrence from 20 stations in 1996 to 4 stations in 2009 (Brainard et al. 2011c). At 21 stations in the Dry Tortugas, rough cactus coral declined in occurrence from 8 stations in 2004 to 3 stations in 2009 (Brainard et al. 2011c). Taken together, these data indicate that the species has declined in Florida and potentially also in Puerto Rico over the past one to two decades.

A recent coral disease event has greatly affected coral populations in Florida. This unprecedented, multi-year disease event, which began in 2014, swept through Florida and caused massive mortality from St. Lucie Inlet in Martin County to Looe Key in the lower Florida Keys. The effects of this widespread disease have been severe, causing mortality of millions of coral colonies across several species, including *Mycetophyllia* species. At study sites in southeast Florida, prevalence of disease was recorded at 67% of all coral colonies and 81% of colonies of those species susceptible to the disease (Precht et al. 2016). No species-specific information is available for the effects of disease on rough cactus coral, but in a survey of 134 sites conducted between October 2017 and April 2018, 9% of *Mycetophyllia* species were affected (Neely 2018). This disease prevalence is a snapshot in time and does not represent the total proportion of *Mycetophyllia* species affected by the disease outbreak.

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

In other areas of the Caribbean, rough cactus coral is also uncommon. In a survey of Utila, Honduras between 1999 and 2000, rough cactus coral was observed at 8% of 784 surveyed sites and was the 36th most commonly observed out of 46 coral species; other *Mycetophyllia* species were seen more commonly (Afzal et al. 2001). In surveys of remote southwest reefs of Cuba, rough cactus coral was observed at 1 of 38 reef-front sites, where average abundance was 0.004 colonies per approximately 108 ft^2 (10 m^2); this was comparatively lower than the other 3 *Mycetophyllia* species observed (Alcolado et al. 2010). Between 1998 and 2004, rough cactus coral was observed at 3 of 6 sites monitored in Colombia, where their cover ranged from 0.3-0.4% (Rodriguez-Ramirez et al. 2010). In Barbados, rough cactus coral was observed on 1 of 7 reefs surveyed, and the average cover was 0.04% (Tomascik and Sander 1987).

Rough cactus coral has been reported to occur on a low percentage of surveyed reefs and is one of the least common coral species observed. On reefs where rough cactus coral is found, it generally occurs at abundances of less than 1 colony per approximately 100 ft^2 (10 m^2) and cover of less than 0.1%. Low encounter rate and percent cover coupled with the tendency to include *Mycetophyllia* spp. at the genus level make it difficult to discern population trends of rough cactus coral from monitoring data. However, reported losses of rough cactus coral from monitoring stations in the Florida Keys and Dry Tortugas (63-80% loss) and decreased encounter frequency in Puerto Rico indicate the population has declined.

Threats

A summary of threats to all corals is provided in Section 3.2.5 General Threats Faced by All Coral Species. Detailed information on the threats to rough cactus coral can be found in the Final Listing Rule (79 FR 53851; September 10, 2014); however, a brief summary is provided here. Rough cactus coral is highly susceptible to disease, and susceptible to ocean warming, acidification, trophic effects of fishing, nutrients, and sedimentation.

Rough cactus coral has some susceptibility to ocean warming. However, the available information does not support a more precise description of susceptibility to this threat. The bleaching reports available specifically for rough cactus coral and at the genus level indicate similar trends of relatively low bleaching observed in 1995, 1998, and 2010 (less than 25%). Further in the more severe 2005 bleaching event, higher beaching levels (50-65%) or no bleaching, were observed in different locations in its range. Reproductive failure and a disease outbreak were reported for the genus after the 2005 bleaching event. Although bleaching of most coral species is spatially and temporally variable, understanding the susceptibility of rough cactus coral is somewhat confounded by the species' low sample size in any given survey due to its low encounter rate.

Rough cactus coral is highly susceptible to disease. Reports in the Florida Keys indicate rough cactus coral is very susceptible to white plague, and reports of high losses and correlation with higher temperatures date back to the mid-1970s (Dustan 1977). Although heavy impacts of disease on rough cactus coral have not been reported in other locations, an outbreak of white plague was credited with causing heavy mortality at the genus level in Puerto Rico after the 2005 bleaching event (Wilkinson 2008).

Rough cactus coral may be susceptible to nutrient enrichment as evidenced by its absence from eutrophic sites in one location. However, there is uncertainty about whether the absence is a result of eutrophic conditions or a result of uncommon or rare occurrence. Therefore, we conclude that rough cactus coral likely has some susceptibility to nutrient enrichment. However, the available information does not support a more precise description of susceptibility.

Summary of Status

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

3.2.9 Boulder star coral (Orbicella franksi)

On September 10, 2014, NMFS listed boulder star coral as threatened (79 FR 53851). 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 single species –*Montastraea annularis*– with varying 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*. The 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 star coral species complex rather than individual species because visual distinction can be difficult where colony structure cannot be discerned (e.g., small colonies or photographic methods). Information from these studies is reported for the species complex. Where species-specific information is available, it is reported. Information about boulder star coral 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, with the additional specifications of *Orbicella faveolata* and *Orbicela franksi*.

Species Description and Distribution

Boulder star coral is distinguished by large, unevenly arrayed polyps that give the colony its characteristic irregular surface. Colony form is variable, and the skeleton is dense with poorly developed annual bands. Colony diameter can reach up to 16 ft (5 m) with a height of up to 6.5 ft (2 m).

Boulder star coral is distributed in the western Atlantic Ocean and throughout the Caribbean Sea including in the Bahamas, Bermuda, and the Flower Garden Banks. Boulder star coral tends to have a deeper distribution than the other 2 species in the *Orbicella* species complex. It occupies most reef environments and has been reported from water depths ranging from approximately 16-165 ft (5-50 m), with the species complex reported to 250 ft (90 m). *Orbicella* species are a common, often dominant, component of Caribbean mesophotic reefs (e.g., >100 ft [30 m]), suggesting the potential for deep refugia for boulder star coral.

Life History 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. Boulder star coral is reported to be the slowest of the 3 species in the complex (Brainard et al. 2011b). They grow more slowly in deeper water and in less clear water.

All 3 species of the star coral complex are hermaphroditic broadcast spawners, with spawning concentrated on 6-8 nights following the full moon in late August, September, or early October, depending on timing of the full moon and location. Boulder star coral spawning is reported to be about 1- 2 hours earlier than lobed star coral and mountainous star coral. All 3 species are largely self-incompatible (Knowlton et al. 1997; Szmant et al. 1997). Fertilization success measured in the field was generally below 15% for all 3 species, as it was closely linked to the number of colonies concurrently spawning. In Puerto Rico, minimum size at reproduction for the star coral species complex was 13 in^2 (83 cm^2).

Successful recruitment by the star coral species complex appears to always have been rare. Only a single recruit of *Orbicella* was observed over 18 years of intensive observation of approximately 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. Of 351 colonies of boulder star coral tagged in Bocas del Toro, Panama, larger colonies were noted to spawn more frequently than smaller colonies between 2002 and 2009 (Levitan et al. 2011).

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

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.

Status and Population Dynamics

Information on boulder 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 were data exist.

Reported density is variable by location and habitat and is reported to range from 0.002 to 10.5 colonies per ~100 ft^2 (10 m^2). Benthic surveys conducted in Florida between 1999 and 2017 recorded an average density of 0.01 to 0.36 colonies per m^2, and boulder star coral was observed at 5% to 45% of surveyed sites (NOAA, unpublished data). In Puerto Rico, boulder star coral was observed at 3% to 50% of sites, and average density ranged from 0.002 to 0.13 colonies per m^2 in surveys conducted between 2008 and 2018 (NOAA, unpublished data). In the US Virgin Islands, boulder star coral was present at a density of 0.02 to 0.24 colonies per m^2 at 19% to 69% of sites surveyed between 1999 and 2018 (NOAA unpublished data). Limited surveys in the Flower Garden Banks reported a relatively stable density of 0.91 to 1.05 colonies per m^2 between 2010 and 2015, and boulder star coral was present at 90% to 100% of surveyed sites (NOAA, unpublished data). In a survey of 31 sites in Dominica between 1999 and 2002, boulder star coral was present in 7% of the sites at less than 1% cover (Steiner 2003a). On remote reefs off southwest Cuba, colony density was 0.08 colonies per ~100 ft^2 (10 m^2) at 38 reef-crest sites and 1.05 colonies per ~100 ft^2 (10 m^2) at 30 reef-front sites (Alcolado et al. 2010). The number of boulder star coral colonies in Cuba with partial colony mortality were far more frequent than those with no mortality across all size classes, except for 1 (i.e., less than ~20 in [50 cm]) that had similar frequency of colonies with and without partial mortality (Alcolado et al. 2010).

Abundance at some sites in Curaçao and Puerto Rico appeared to be stable over an 8-10 year period. In Curaçao, abundance was stable between 1997 and 2005, with partial mortality similar or less in 2005 compared to 1998 (Bruckner and Bruckner 2006). Abundance was also stable between 1998-2008 at 9 sites off Mona and Desecheo Islands, Puerto Rico. In 1998, 4% of all corals at 6 sites surveyed off Mona Island were boulder star coral colonies, and approximately 5% were boulder star corals in 2008; at Desecheo Island, about 2% of all coral colonies were boulder star coral in both 2000 and 2008 (Bruckner and Hill 2009).

Recent events have greatly impacted boulder star 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, including boulder 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 US 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 10-14% of boulder star corals were impacted (NOAA 2018). In Florida, approximately 23% of boulder star corals surveyed at 57 sites were impacted (Florida Fish and Wildlife Conservation Commission, unpublished data). Survey data are not available for the US Virgin Islands, though qualitative observations indicate that damage was also widespread but variable by site.

In some locations, colony size has decreased over the past several decades. Bruckner conducted a survey of 185 sites (2010 and 2011) in 5 countries (The Bahamas, Bonaire, Cayman Islands, Puerto Rico, and St. Kitts and Nevis) and reported the size of boulder star coral and lobed star coral colonies as significantly smaller than mountainous star coral. The total mean partial mortality of boulder star coral was 25%. Overall, the total live area occupied by boulder star coral declined by a mean of 38%, and mean colony size declined from 210 in^2 to 131 in^2 (1356 cm^2 to 845 cm^2). At the same time, there was a 137% increase in small tissue remnants, along with a decline in the proportion of large (1,500 to 30,000 cm^2), completely alive colonies.

Mortality was attributed primarily to outbreaks of white plague and yellow band disease, which emerged as corals began recovering from mass bleaching events. This was followed by increased predation and removal of live tissue by damselfish (Bruckner 2012).

Overall, abundance of boulder star coral appears stable in some locations and has declined in others. Although boulder star coral remains common, the buffering capacity of its life history strategy that has allowed it to remain abundant has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. We anticipate that population abundance is likely to decrease in the future with increasing threats.

Threats

A summary of threats to all corals is provided in Section 3.2.5 General Threats Faced by All Coral Species. Detailed information on the threats to boulder star coral can be found in the Final Listing Rule (79 FR 53852; September 10, 2014); however, a brief summary is provided here. Boulder star coral is highly susceptible to ocean warming, disease, ocean acidification, sedimentation, and nutrients, and susceptible to trophic effects of fishing.

Available information indicates that boulder star coral is highly susceptible to warming temperatures with a reported 88-90% bleaching frequency. Reported bleaching-related mortality from one study is high at 75%. There is indication that new algal symbiotic species establishment occurs after bleaching in boulder star coral.

In a 2010 cold-water event that affected south Florida, boulder star coral ranked as the 14th most susceptible coral species out of the 25 most abundant coral species. Average partial mortality was 8% in surveys from Martin County to the lower Florida Keys after the 2010 cold-water event compared to 0.4% average mortality during summer surveys between 2005 and 2009.

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

Boulder star coral is often reported as among the species with the highest disease prevalence. Although there are few quantitative studies of the effects of disease on boulder star coral, there is evidence that partial mortality can average about 25-30% and that disease can cause shifts to smaller size classes. Thus, we conclude that boulder star coral is highly susceptible to disease.

Genus information indicates sedimentation negatively affects primary production, growth rates, calcification, colony size, and abundance. Genus level information also indicates boulder star coral is likely susceptible to nutrient enrichment through reduced growth rates and lower recruitment. Additionally, nutrient enrichment has been shown to increase the severity of yellow band disease in boulder star coral. Thus, we conclude that boulder star coral survival is highly susceptible to sedimentation and nutrient enrichment.

Summary of Status

Boulder star coral has undergone declines most likely from disease and warming-induced bleaching. There is evidence of synergistic effects of threats for this species including increased disease severity with nutrient enrichment. Boulder 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 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 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 boulder star coral is limited to areas with high localized human impacts and predicted increasing threats. Its depth range of approximately 16-165 ft (5-50 m), possibly up to 295 ft (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. Boulder 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. However, we anticipate that the population abundance is likely to decrease in the future with increasing threats.

3.2.10 Lobed star coral (*Orbicella annularis*)

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

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.

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, 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 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.

Status and 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 were 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 1 year after the bleaching event. Population growth rates declined even further 2 years after the bleaching event, but they returned and then stabilized at the lower rate the following year.

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 US 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 US 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 US 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.

Threats

A summary of threats to all corals is provided in Section 3.2.5 General Threats Faced by All Coral Species. Detailed information on the threats to lobed star coral can be found in the Final Listing Rule (79 FR 53852; 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.

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.

3.2.11 Mountainous star coral (*Orbicella faveolata*)

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

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.

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, 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.

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.

Status and 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 were 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 US 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 US 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 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, 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 US 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 US 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.

Threats

A summary of threats to all corals is provided in Section 3.2.5 General Threats Faced by All Coral Species. 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.

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.

3.2.12 Elkhorn and Staghorn Coral Critical Habitat

On November 26, 2008, a Final Rule designating *Acropora* critical habitat was published in the Federal Register (73 FR 72210). Within the geographical area occupied by a listed species, critical habitat consists of specific areas on which are found those physical or biological features essential to the conservation of the species. The feature essential to the conservation of *Acropora* species (also known as essential feature) is substrate of suitable quality and availability, in water depths from the mean high water line to 98 ft (30 m), to support successful larval settlement, recruitment, and reattachment of fragments. Substrate of suitable quality and availability means consolidated hard bottom or dead coral skeletons free from fleshy macroalgae or turf algae and sediment cover. Areas containing these features 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 km^2) of marine habitat; the Puerto Rico area, which comprises approximately 1,383 mi^2 (3,582 km^2) of marine habitat; the STJ/STT area, which comprises approximately 121 mi^2 (313 km^2) of marine habitat; and the STX area, which comprises approximately 126 mi^2 (326 km^2) of marine habitat (Figure 3.6 A and B). The total area covered by the designation is thus approximately 2,959 mi^2 (7,664 km^2).

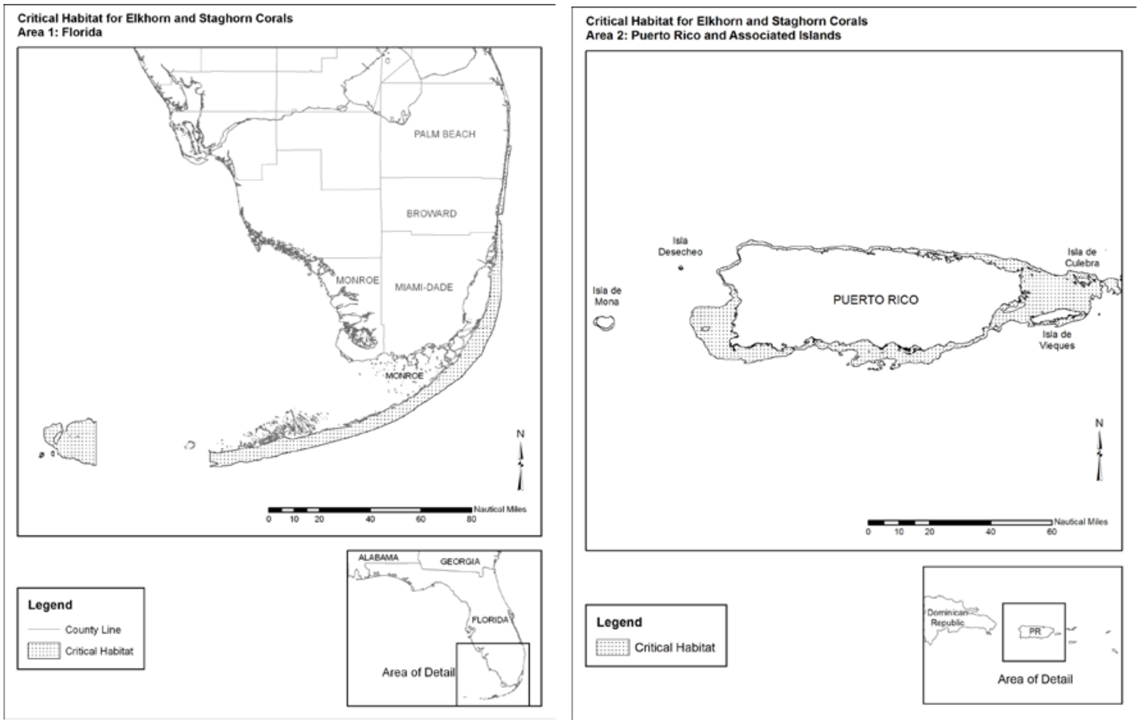


Figure 3.6 A. Designated *Acropora* Critical Habitat Area 1 and 2

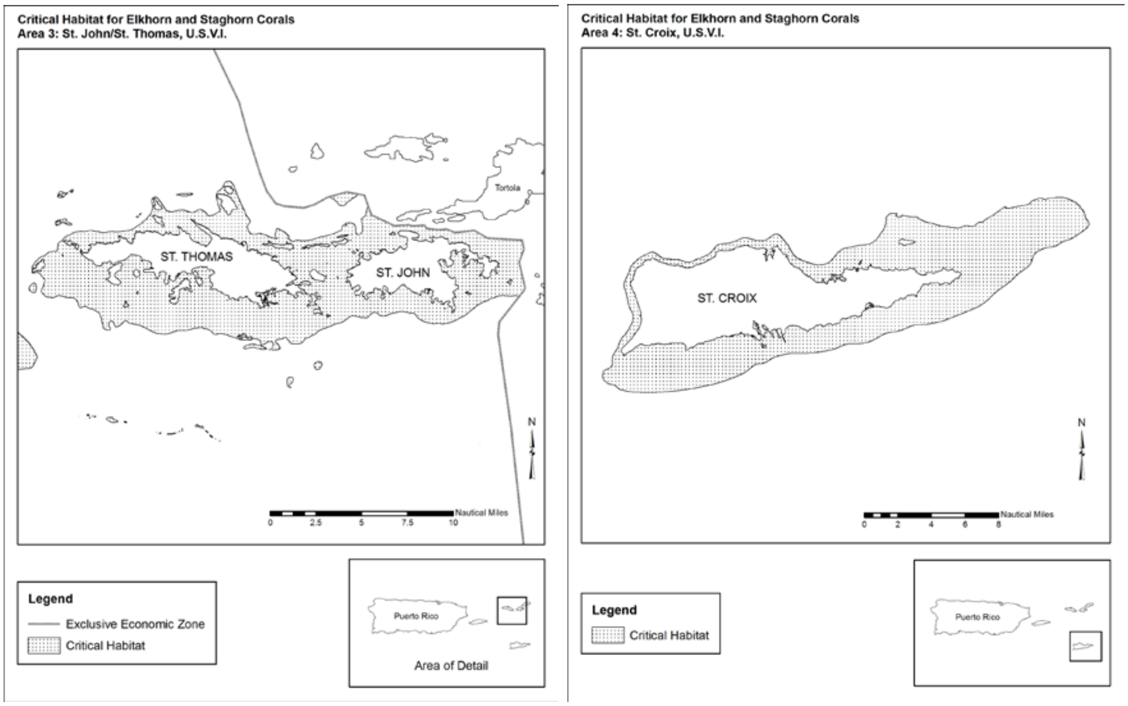


Figure 3.6 B. Designated *Acropora* Critical Habitat Area 3 and 4

The essential feature can be found unevenly dispersed throughout the 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 and algal 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 TCRMP 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.

3.2.13 Scalloped Hammerhead Shark – Central and Southwest Atlantic DPS

Four of 6 identified DPSs of scalloped hammerhead shark (*Sphyrna lewini*) were listed under the ESA by NMFS effective September 2, 2014 (79 FR 38213, July 3, 2014) (Figure 3.7). The Central and Southwest Atlantic and the Indo-West Pacific DPSs were listed as threatened, while the Eastern Atlantic and Eastern Pacific DPSs were listed as endangered. The Central and Southwest Atlantic DPS is bounded to the north by 28°N latitude, to the east by 30°W longitude, and to the south by 36°S latitude. All waters of the Caribbean Sea are within this DPS boundary, including The Bahamas' EEZ off the coast of Florida, the U.S. EEZ off Puerto Rico and the U.S. Virgin Islands, and Cuba's EEZ.

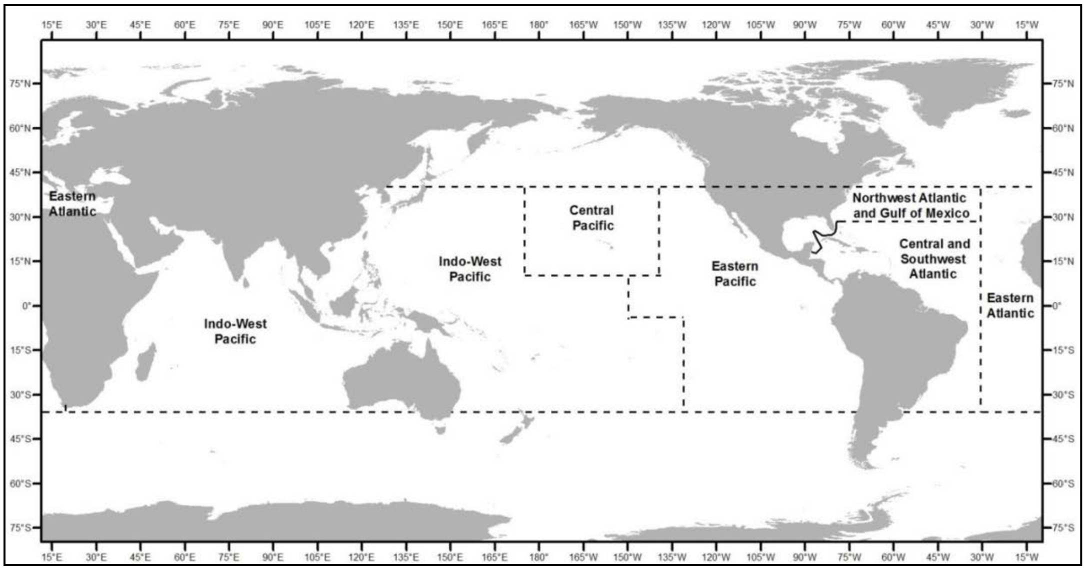


Figure 3.7. Scalloped hammerhead shark DPS boundaries (Source: 78 FR 20717; April 5, 2013).

Note: The Northwest Atlantic/Gulf of Mexico and Central Pacific DPSs are not listed under the ESA.

Species Description and Distribution

All hammerhead sharks belong to the family *Sphyrnidae* and are classified as ground sharks (order Carcharhiniformes). The hammerhead sharks are recognized by their laterally expanded head that resembles a hammer, hence the common name "hammerhead." The scalloped hammerhead shark is distinguished from other hammerheads by a noticeable indentation on the center and front portion of the head, along with 2 more indentations on each side of this central indentation, giving the head a "scalloped" appearance. It has a broadly arched mouth, and the back of the head is slightly swept backward.

The scalloped hammerhead shark is found throughout the world and lives in coastal warm temperate and tropical seas. It occurs over continental shelves and the shelves surrounding islands, as well as adjacent deep waters, but it is seldom found in waters cooler than 22°C (Compagno 1984; Schulze-Haugen et al. 2003). It ranges from the intertidal and surface waters to depths of up to approximately 1,475-1,675 ft (450-512 m) (Klimley 1993; Sanches 1991), with occasional dives even deeper (Jorgensen et al. 2009). It has also been documented entering enclosed bays and estuaries (Compagno 1984). In the western Atlantic Ocean, the scalloped hammerhead's range extends from the northeast coast of the U.S. (New Jersey) to Florida and on to Brazil, including the Gulf of Mexico and Caribbean Sea.

Scalloped hammerhead sharks are highly mobile and partly migratory, and are likely the most abundant of the hammerhead species (Maguire et al. 2006). These sharks have been observed making migrations along the edges of continents as well as between oceanic islands in tropical waters (Bessudo et al. 2011; Diemer et al. 2011; Duncan and Holland 2006; Kohler and Turner 2001). Although scalloped hammerhead sharks are highly mobile, this species rarely crosses entire oceans (Diemer et al. 2011; Duncan and Holland 2006; Kohler and Turner 2001). The median distance between mark and recapture of 3,278 tagged adult sharks along the eastern U.S. was less than 65 miles (100 km) (Kohler and Turner 2001). Tagging studies reveal the tendency for scalloped hammerhead sharks to aggregate around and travel to and from core areas or "hot spots" within locations (Duncan and Holland 2006; Hearn et al. 2010; Holland et al. 1993) (Bessudo et al. 2011). However, other studies indicate they are also capable of traveling long distances (e.g., 1,206 miles [1,941 km] (Bessudo et al. 2011); 1,038 mi [1,671 km] (Kohler and Turner 2001); 390 miles [629 km] (Diemer et al. 2011).

Both juveniles and adult scalloped hammerhead sharks occur as solitary individuals, pairs, or in schools (Compagno 1984). Adult aggregations are most common offshore over seamounts and near islands, especially near the Galapagos, Malpelo, Cocos and Revillagigedo Islands, and within the Gulf of California (Bessudo et al. 2011; CITES 2010; Compagno 1984; Hearn et al. 2010). Neonate and juvenile aggregations are more common in nearshore nursery habitats (Bejarano-Álvarez et al. 2011; Diemer et al. 2011; Duncan and Holland 2006). It has been suggested that juveniles inhabit these nursery areas for up to or more than 1 year as they provide valuable refuges from predation (Duncan and Holland 2006).

The scalloped hammerhead shark is a high trophic level predator (Cortés 1999) and an opportunistic feeder with a diet that includes a wide variety of bony fish, octopi/cuttlefish/squid, crabs/lobsters, and rays (Bush 2003; Compagno 1984) (Júnior et al. 2009; Noriega et al. 2011).

*Life History Informati*on

The scalloped hammerhead shark gives birth to live young (i.e., "viviparous"), with a gestation period of 9-12 months (Branstetter 1987; Stevens and Lyle 1989), which may be followed by a 1-year resting period (Liu and Chen 1999). Generally, females attain maturity around 6.5-8 ft (2.0-2.5 m) TL, while males reach maturity at smaller sizes (range 4-6.5 ft [1.3-2.0 m] TL). The available information specific to the Central and Southwest Atlantic DPS indicates females attain maturity when they reach around 7.5 ft (greater than 240 cm) TL, while males reach maturity at 6-6.5 ft (1.8-2.0 m) TL (Hazin et al. 2001).

The age at maturity differs by region. In Brazil (part of the Central and Southwest Atlantic DPS), males reach sexual maturity between 6.3 and 8.1 years, females at 15.2 years (Hazin et al. 2001). However, pupping does not appear to vary temporally by region and may be partially seasonal (Harry et al. 2011a; Harry et al. 2011b). Neonates are present year round, with abundance peaking during the spring and summer months (Adams and Paperno 2007; Bejarano-Álvarez et al. 2011; Duncan and Holland 2006; Harry et al. 2011a; Harry et al. 2011b; Noriega et al. 2011). Females move inshore to birth, with litter sizes anywhere between 1 and 41 live pups. No relationship between litter size and female shark length was identified by Hazin et al. (2001) for animals off the northeastern coast of Brazil. The DPS-specific information indicates pups are generally greater than 1.2 ft (0.38 m) at birth (Hazin et al. 2001).

While it appears that maturity, age, and growth estimates vary by region, it is unclear whether these differences are truly biological or the result of differences in the interpretations of aging methodology (Piercy et al. 2007). Scalloped hammerhead sharks develop opaque bands on their vertebrae which are used to estimate age. Assuming annual band formation for animals in the Atlantic, and adjusting age maturity estimates from the Pacific accordingly, the average age at maturity for female scalloped hammerheads is around 12.8 years and 8.1 years for males. Based on analysis of the available data, the scalloped hammerhead shark can be characterized as a long-lived (i.e., at least 20-30 years) (Dudley and Simpfendorfer 2006), late-maturing, and relatively slow-growing species (Branstetter 1990). Within the DPS, Kotas et al. (2011) estimate the maximum age of females as 31.5 years and 29.5 years for males.

Status and Population Dynamics

Data from multiple sources indicate that the Atlantic population (including the Central and Southwest Atlantic DPS) of scalloped hammerheads has experienced severe declines over the past few decades. Miller et al. (2014) concluded that abundance numbers for this DPS are unavailable but likely similar to, and probably worse than, those found in the Northwest Atlantic and Gulf of Mexico DPS. It is likely that scalloped hammerheads in the Northwest Atlantic and Gulf of Mexico were overfished beginning in the early 1980s and experienced periodic overfishing from 1983-2005 (Jiao et al. 2011). Other studies have also observed similar decreases in scalloped hammerhead shark populations along the Atlantic coast. For example, Baum et al. (2003) calculated that the northwest Atlantic population of scalloped hammerhead shark has declined by 89% since 1986; however, this study is controversial due to its sole reliance on pelagic longline (PLL) logbook data. Off the southeastern U.S. coast, Beerkircher et al. (2002) found significant declines in nominal CPUE for scalloped hammerhead shark between 1981-1983 (CPUE = 13.37 in Berkeley and Campos 1988) and 1992-2000 (CPUE = 0.48).

For the northwest Atlantic and Gulf of Mexico DPS, models estimated the virgin population size to be between 142,000 and 169,000 individuals (range 116,000-260,000) (Hayes et al. 2009). Those models also estimated populations of 24,850-27,900 individuals in 2005 (most recent year estimated) (Hayes et al. 2009).

In a stock assessment for the scalloped hammerhead shark, (Hayes et al. 2009)concluded that the northwestern Atlantic and Gulf of Mexico scalloped hammerhead shark stock has been depleted by approximately 83% since 1981. Miller et al. (2014) concluded that abundance numbers for the Central and Southwest Atlantic DPS are unavailable but likely similar to, and probably worse than, those found in the Northwest Atlantic and Gulf of Mexico DPS. It is likely that scalloped hammerheads in the Central and Southwest Atlantic DPS have experienced at least that level of decline since the early 1980s.

Threats

Scalloped hammerhead sharks are both targeted and taken as bycatch in many global fisheries. They are targeted by semi-industrial, artisanal, and recreational fisheries, and caught as bycatch in PLL tuna and swordfish fisheries and purse seine fisheries. There is a lack of information on the fisheries prior to the early 1970s, with only occasional mentions in historical records. Significant catches of scalloped hammerheads have gone, and continue to be, unrecorded in many countries outside the U.S. Brazil, the country that reports one of the highest scalloped hammerhead landings in South America, maintains heavy industrial fishing of this species off its coastal waters. In the late 1990s, Amorim et al. (1998) remarked that heavy fishing by longliners led to a decrease in this population off the coast of Brazil. According to the FAO global capture production database, Brazil reported a significant increase in catch of scalloped hammerhead during this period, from 30 metric tons (mt) in 1999 to 508 mt by 2002, before decreasing to a low of 87 mt in 2009. Information from PLL and bottom gillnet fisheries targeting several species of hammerhead sharks off southern Brazil indicates declines of more than 80% in CPUE from 2000 to 2008, with the targeted hammerhead fishery abandoned after 2008 due to the rarity of the species (FAO 2010). Scalloped hammerhead is also commonly landed by artisanal fishers in the Central and Southwest Atlantic, with concentrated fishing effort in nearshore and inshore waters, areas likely to be used as nursery grounds. In the Caribbean, specific catch and landings data are unavailable; however, scalloped hammerhead shark is often a target of artisanal fisheries off Trinidad and Tobago and Guyana, and anecdotal reports of declines in abundance, size, and distribution shifts of sharks suggest significant fishing pressure on overall shark populations in this region (Kyne et al. 2012).

The exploitation of this DPS continues to go largely unregulated. In Brazilian waters, there are very few fishery regulations that help protect hammerhead populations. For example, the minimum legal size for a scalloped hammerhead caught in Brazilian waters is approximately 24 in (60 cm) total length (TL); however, scalloped hammerhead shark pups may range from 15-23 in (38 - 55 cm). As the pup sizes are very close to this minimum limit, the legislation is essentially ineffective, and as such, large catches of both juveniles and neonates have been documented from this region (CITES 2010; Kotas et al. 2008). Lack of enforcement of existing regulations also hamper regulatory effectiveness.

In addition, scalloped hammerheads are likely underreported in catch records as many records do not account for discards (e.g., where the fins are kept, but the carcass is discarded) or reflect dressed weights instead of live weights. Also, many catch records do not differentiate between the hammerhead species, or shark species in general, and thus species-specific population trends for scalloped hammerheads are not readily available.

Although scalloped hammerhead meat is considered essentially unpalatable (due to its high urea concentration), some countries still consume the meat domestically or trade it internationally, including Colombia, Mexico, and Uruguay (CITES 2010; Vannuccini 1999). However, it is thought that the current volume of scalloped hammerhead shark traded meat and products is insignificant when compared to the volume of its fins in international trade (CITES 2010).

3.2.14 Nassau Grouper

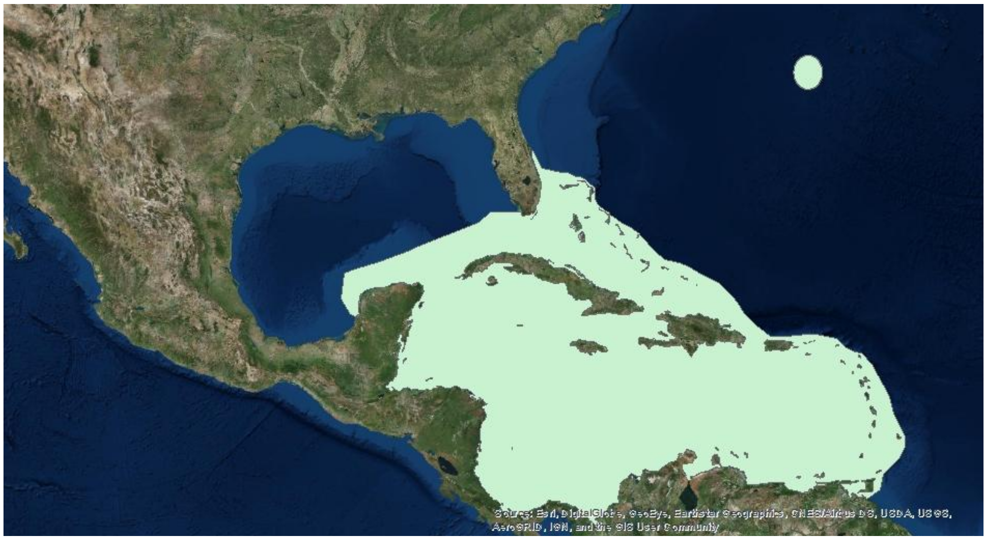
NMFS listed the Nassau grouper as threatened under the ESA effective July 29, 2016 (81 FR 42268, June 29, 2016). This section provides a summary of key biological information as presented in the June 29, 2016, listing rule as well as the Biological Report (Hill and Sadovy de Mitcheson 2013).

Species Description and Life History
The Nassau grouper, *Epinephelus striatus* (Bloch 1792), is a moderate-sized serranid fish with large eyes and a robust body. Coloration is variable, but adult fish are generally buff, with five dark brown vertical bars, a large black saddle blotch on top of the base of the tail, and a row of black spots below and behind each eye. Color pattern can also change within minutes from almost white to bicolored to uniformly dark brown, according to the behavioral state of the fish (Carter et al. 1994; Colin 1992; Heemstra and Randall 1993; Longley 1917). A distinctive bicolor pattern is seen when two adults or an adult and large juvenile meet and is frequently observed at spawning aggregations Heemstra and Randall (1993). There is also a distinctive dark tuning-fork mark that begins at the front of the upper jaw, extends back between the eyes, and then divides into two branches on top of the head behind the eyes. Another dark band runs from the tip of the snout through the eye and then curves upward to meet its corresponding band from the opposite side just in front of the dorsal fin. Juveniles exhibit a color pattern similar to adults (Silva et al. 2002).

As with many serranids, the Nassau grouper is slow-growing and long-lived; estimates range up to a maximum of 29 years (Bush et al. 1996). Using length-frequency analysis, which tends to exclude younger animals, a theoretical maximum age at 95% asymptotic size is 16 years. Individuals of more than 12 years of age are not common in fisheries, with more heavily fished areas yielding much younger fish on average. Most studies indicate a rapid growth rate for juveniles, which has been estimated to be about 10 mm/month TL for small juveniles, and 8.4-11.7 mm/month TL for larger juveniles (Beets and Hixon 1994) (Eggleston 1995). Maximum size is about 122 cm TL and maximum weight is about 25 kg (Heemstra and Randall 1993; (Humann and DeLoach 2002); (Froese 2010). Generation time (the interval between the birth of an individual and the subsequent birth of its first offspring) is estimated as 9-10 years (Sadovy and Eklund 1999). Data from scales and otoliths indicate that fish reach sexual maturity in approximately 4-7 years (Hill and Sadovy de Mitcheson 2013).

Distribution

The Nassau grouper's confirmed distribution currently includes "Bermuda and Florida (USA), throughout the Bahamas and Caribbean Sea" (e.g., (Heemstra 1993)). The occurrence of Nassau grouper from the Brazilian coast south of the equator as reported in Heemstra (1993) is "unsubstantiated" (Craig et al. 2011). The Nassau grouper has been documented in the Gulf of Mexico, at Arrecife Alacranes (north of Progreso) to the west off the Yucatan Peninsula, Mexico (Hildebrand et al. 1964). Nassau grouper is generally replaced ecologically in the eastern Gulf by red grouper (*E. morio*) in areas north of Key West or the Tortugas (Smith 1971). They are considered a rare or transient species off Texas in the northwestern Gulf of Mexico (Gunter and Knapp 1951) in (Hoese and Moore 1998). The first confirmed sighting of Nassau grouper in the Flower Garden Banks National Marine Sanctuary, which is located in the northwest Gulf of Mexico approximately 180 km southeast of Galveston, Texas, was reported by (Foley et al. 2007b). Many earlier reports of Nassau grouper up the Atlantic coast to North Carolina have not been confirmed. The Biological Report (Hill and Sadovy de Mitcheson 2013) provides a detailed description of the distribution, summarized in Figure 3.8.



Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community

Figure 3.8 Confirmed distribution of Nassau grouper currently includes Bermuda, Florida (USA), the Bahamas, and Caribbean Sea. Image courtesy Phil Caldwell.

Basic Biology

Habitat and Depth Use Information

The Nassau grouper is primarily a shallow-water, insular fish species that has long been valued as a major fishery resource throughout the wider Caribbean, South Florida, Bermuda, and the Bahamas (Carter et al. 1994). This species is considered a reef fish, but it transitions through a series of developmental habitats. As larvae, the Nassau grouper is planktonic. After an average of 35-40 days and at an average size of 32 mm TL, larvae recruit from an oceanic environment into demersal habitats (Colin 1992); (Eggleston 1995). Following settlement, juvenile Nassau grouper inhabit macroalgae (primarily *Laurencia* spp.), coral clumps (*Porites* spp.), and seagrass beds (Dahlgren 1998; Eggleston 1995). Recently-settled Nassau grouper have also been collected from rubble mounds at 18 m depth (Colin et al. 1997). Post-settlement, small Nassau grouper have been reported with discarded queen conch shells (*Strombus gigas*) and other debris around Thalassia beds (Eggleston 1995; Randall 1983).

Juvenile Nassau grouper (12-15 cm TL) are relatively solitary and remain in specific areas for months (Bardach 1958). Juveniles of this size class are associated with macroalgae, and both natural and artificial reef structure. As juveniles grow, they move progressively to deeper areas and offshore reefs (Colin et al. 1997; Tucker et al. 1993). Schools of 30-40 juveniles (25-35 cm TL) were observed at 8-10 m depths in the Cayman Islands (Tucker et al. 1993). No clear distinction can be made between types of adult and juvenile habitats, although a general size segregation with depth occurs, with smaller Nassau grouper in shallower inshore waters (3.7-16.5 m) and larger individuals more common near deeper (18.3-54.9 m) offshore banks (Bardach 1958; Bardach et al. 1958; Cervigón 1994; Radakov et al. 1975; Silva Lee 1974; Thompson and Munro 1978).

Adult Nassau grouper tend to be relatively sedentary and are generally associated with high-relief coral reefs or rocky substrate in clear waters to depths of 130 m. Generally, adults are most common at depths less than 100 m (Hill and Sadovy de Mitcheson 2013) except when at spawning aggregations where they are known to descend to depths of 255 m (Starr et al. 2007).

Foraging Information

Adult Nassau grouper are unspecialized, bottom-dwelling, ambush-suction predators (Randall 1965a; Thompson and Munro 1978). Numerous studies describe adult Nassau grouper as piscivorous (Carter et al. 1994; Eggleston et al. 1998; Randall 1965a; Randall 1967b; Randall and Brock 1960). Feeding can take place around the clock although most fresh food is found in stomachs collected in the early morning and at dusk (Randall 1967b). Young Nassau grouper (20.2-27.2 mm standard length [SL]) feed on a variety of plankton, including pteropods, amphipods, and copepods (Greenwood 1991; Grover et al. 1998).

Spawning Behavior and Habitat

The effects of fishing in relation to spawning behavior is an important issue for this species (please refer to the Population Dynamics and Status and the Threats sections that follow).

Nassau grouper form spawning aggregations at predictable locations around the winter full moons, or between full and new moons (Aguilar-Perera 1994; Carter et al. 1994; Colin 1992; P.L. 1992; Smith 1971; Tucker et al. 1993; Tucker and Woodward 1994). Aggregations consist of hundreds, thousands, or, historically, tens of thousands of individuals. Some aggregations have persisted at known locations for periods of 90 years or more (see references in Hill and Sadovy de Mitcheson 2013). Pair spawning has not been observed.

About 50 individual spawning aggregation sites have been recorded, mostly from insular areas in the Bahamas, Belize, Bermuda, British Virgin Islands, Cayman Islands, Cuba, Honduras, Jamaica, Mexico, Puerto Rico, Turks and Caicos, and the U.S.V.I.; however, many of these may no longer form. Recent evidence suggests that spawning is occurring at what may be reconstituted or novel spawning sites in both Puerto Rico and the U.S.V.I. (Hill and Sadovy de Mitcheson 2013). Suspected or anecdotal evidence also identifies spawning aggregations in Los Roques, Venezuela (Boomhower et al. 2010) and Old Providence in Colombia's San Andrés Archipelago (Prada et al. 2004). Spawning aggregation sites have not been reported in the Lesser Antilles, Central America south of Honduras, or Florida.

"Spawning runs," or movements of adult Nassau grouper from coral reefs to spawning aggregation sites, were first described in Cuba in 1884 by Vilaro Diaz, and later by (Guitart-Manday and Juárez-Fernandez 1966). Nassau grouper migrate to aggregation sites in groups numbering between 25 and 500, moving parallel to the coast or along shelf edges or even inshore reefs (Aguilar-Perera and Aguilar-Davila 1996; Carter et al. 1994; Colin 1992; Nemeth et al. 2009). Distance traveled by Nassau grouper to aggregation sites is highly variable; some fish move only a few kilometers, while others move up to several hundred kilometers (Bolden 2000; Carter et al. 1994; Colin 1992). Ongoing research in the Exuma Sound, Bahamas has tracked migrating Nassau grouper up to 200 km, with likely estimates of up to 330 km, as they move to aggregation sites (Hill and Sadovy de Mitcheson 2013).

Observations suggest that individuals can return to their original home reef following spawning. Bolden (2001) reported 2 out of 22 tagged fish returning to home reefs in the Bahamas one year after spawning. Sonic tracking studies around Little Cayman Island have demonstrated that spawners may return to the aggregation site in successive months with returns to their residential reefs in between (Semmens et al. 2007). Larger fish are more likely to return to aggregation sites and spawn in successive months than smaller fish (Semmens et al. 2007). It is not known how Nassau grouper select and locate aggregation sites or why they aggregate to spawn. Spawning aggregation sites are typically located near significant geomorphological features, such as projections (promontories) of the reef as little as 50 m from the shore, and close to a drop-off into deep water over a wide (6-60 m) depth range (Aguilar-Perera 1994; Beets and Friedlander 1999; Burnett-Herkes 1975; Carter 1989; Colin 1992; Colin et al. 1987; Craig 1966; Fine 1990; Olsen and LaPlace 1979; Smith 1972). Sites are characteristically small, highly circumscribed areas, measuring several hundred meters in diameter, with soft corals, sponges, stony coral outcrops, and sandy depressions (Aguilar-Perera 1994; Beets and Friedlander 1999; Burnett-Herkes 1975; Carter 1989; Colin 1992; Colin et al. 1987; Craig 1966; Fine 1990; Olsen and LaPlace 1979; Smith 1972).

The link between spawning sites and settlement sites is also not well understood. Spawning aggregations form around the full moon between December and March (reviewed in Sadovy and Eklund (1999)), though this may occur later (May-August) in more northerly latitudes (Bardach 1958; Burnett-Herkes 1975; Gorce and (eds.) 1939; Smith 1971). The formation of spawning aggregations is triggered by a very narrow range of water temperatures between 25-26°C. While day length has also been considered as a trigger for aggregation formation (Carter et al. 1994; Colin 1992; Tucker et al. 1993), temperature is evidently a more important stimulus (Hill and Sadovy de Mitcheson 2013). Spawning occurs for up to 1.5 hours around sunset for several days (Whaylen et al. 2007). At spawning aggregation sites, Nassau grouper tend to mill around for a 1-2 days in a "staging area" adjacent to the core area where spawning activity later occurs (Colin 1992; Kadison et al. 2010; Nemeth 2012). Courtship is indicated by 2 behaviors that occur late in the afternoon: “following” and “circling” (Colin 1992). The aggregation then moves into deeper water shortly before spawning (Carter et al. 1994; Colin 1992; Tucker et al. 1993). Progression from courtship to spawning may depend on aggregation size, but generally fish move up into the water column, with an increasing number exhibiting the bicolor phase (Carter et al. 1994; Colin 1992).

Repeated spawning occurs at the same site for up to 3 consecutive months generally around the full moon or between the full and new moons (Aguilar-Perera 1994; Carter et al. 1994; Colin 1992; Smith 1971; Tucker et al. 1993; Tucker and Woodward 1994). Participation by individual fish across the months is unknown. It is unknown whether a single, mature female will spawn continuously throughout the spawning season or just once per year.

Population Dynamics and Status

Few formal stock assessments have been conducted for the Nassau grouper, likely because of limited data. The most recent published assessment, conducted in The Bahamas, suggests that fishing effort in The Bahamas needs to be reduced from the 1998 to 2001 level in order to avoid overexploitation of stocks relative to biological reference points (Cheung et al. 2013).

During the first U.S. survey of the fishery resources of Puerto Rico, the Nassau grouper was noted as a common and very important food fish, reaching a weight of 50 lb (22.7 kg) or more ((Evermann 1900). By 1970, Nassau grouper was still the fourth most common shallow-water species landed in Puerto Rico ((Thompson 1978), and it was common in the reef fish fishery of the Virgin Islands, where an aggregation in the 1970s contained an estimated 2,000-3,000 individuals (Olsen and LaPlace 1979) (Olsen and LaPlace 1979). During the 1980s, port sampling in the U.S.V.I. showed that Nassau grouper accounted for 22% of grouper landings with 85% of the Nassau grouper catch coming from spawning aggregations (D. Olsen, Chief Scientist – St. Thomas Fishermen’s Association, pers. comm. to J. Rueter, NMFS SERO PRD, October, 2013). By 1981, “the Nassau grouper ha(d) practically disappeared from the local catches and the ones that d(id) appear (were)-small compared with previous years” (CFMC and NMFS 1985) and by 1986, the Nassau grouper was considered commercially extinct in the U.S. Virgin Islands/Puerto Rico region (Bohnsack et al. 1986). About 1,000 kg were landed from the Reef Fish fishery during the latter half of the 1980s in Puerto Rico, most of them were less than 500 mm, indicating they were likely sexually immature (Sadovy 1997).

Although there are few data on historic abundance of Nassau grouper off the U.S. mainland, it appears that abundance was once high in southern Florida (Springer and McErlean 1962). Anecdotal reports from spearfishers noted large daily catches in the 1950s (Bohnsack 1990). Interviews of Florida Keys’ residents suggested that Nassau grouper were once caught in much greater numbers from the upper Florida Keys and the Bahamas (Sadovy and Eklund 1999). Starck (1968) reported Nassau grouper frequently at Alligator Reef in the Florida Keys.

Historically, Nassau grouper was a component of the grouper fishery in Florida, suggesting once healthy (sub)population(s) in southeastern U.S. mainland waters (Sadovy and Eklund 1999). In contrast, now the species is rarely encountered (Sadovy and Eklund 1999). In the Dry Tortugas, where Nassau grouper were once abundant, only one individual was recorded in 1994 out of 183 point censuses and none in 37 predator censuses (Sluka et al. 1998). On Elbow Reef, Florida

Keys, mean Nassau grouper densities were 0.01- 0.04 fish per 100 m^2 in 1993-94 (Sluka et al. 1998), with few seen on census dives through the Florida Keys. Censuses comparing areas protected and unprotected from fishing indicated that Nassau grouper, where protected, had a higher density and were one of the dominant grouper species observed (Sluka et al. 1997). Despite 10-20 years of no-take protection of the Nassau grouper in the Florida Keys, Nassau grouper has made no appreciable recovery and numbers remain extremely low (Semmens et al. 2007, Don DeMaria pers. comm. 2012 *In* Hill and Sadovy de Mitcheson 2013).

Little is known about the dynamics of unexploited stocks of Nassau grouper, although some of the data from the 1980s give us some insight (Carter et al. 1994). Spawning stock biomass per recruit has not been quantified for the species, but landings data clearly show a chronological trend from abundance to rarity in many areas (e.g., (Sadovy 1997). Of particular concern has been the rapid and extreme decline in numbers taken from traditional aggregation sites (Sala et al. 2001). In general, slow-growing, long-lived species (such as snappers and groupers) with limited spawning periods and, possibly, with only a narrow recruitment window are susceptible to overexploitation ((Bannerot et al. 1987; Polovina and Ralston 1987). Hodgson and Liebeler (2002) noted that Nassau grouper were absent from 82% of shallow Caribbean reefs (3–10 m) during a 5-year period (1997-2001) of underwater surveys for the ReefCheck project. This is derived from underwater surveys in most countries in the range of the species.

Because insufficient stock assessments or population estimates exist, NMFS (81 FR 42268, June 29, 2016) considered the changes in spawning aggregations as a proxy for the status of the current population. NMFS believes the status of spawning aggregations is likely to be reflective of the overall population because adults migrate to spawning aggregations for the only known reproductive events. Historically, 50 spawning aggregation sites had been identified throughout the Caribbean (Sadovy De Mitcheson et al. 2008). Of these 50, less than 20 probably still remain (Sadovy De Mitcheson et al. 2008). Numbers of fish at aggregation sites once numbered in the tens of thousands (30,000 – 100,000 fish; Smith 1972), however they have now been reduced to less than 3,000 at those sites where counts have been made (Sadovy De Mitcheson et al. 2008). In many areas of its range, the species is now considered commercially extinct and numerous spawning aggregations have been extirpated with no signs of recovery (81 FR 42268, June 29, 2016). Based on the size and number of current spawning aggregations, the Nassau grouper population appears to be significantly reduced from its historical size.

Threats

The most serious threats to Nassau grouper are fishing at spawning aggregations and inadequate law enforcement. These threats are considered high risk threats to the species, and are currently affecting the status of Nassau grouper, putting it at a heightened risk of extinction. Nassau grouper are fished commercially and recreationally throughout the year by handline, longline, fish traps, spear guns, and gillnets (NMFS General Canvas Landing System). Aggregations are mainly exploited by handlines or by fish traps, although gillnets were being used in Mexico in the early to mid-1990s (Aguilar-Perera 2004). Sadovy and Eklund (1999) show declines in landings, catch per unit effort (CPUE) and, by implication, abundance in the late 1980's and early 1990's throughout its range, which has led Nassau grouper to now be considered commercially extinct in a number of areas (Sadovy and Eklund 1999). Recent reports from throughout the Nassau grouper's range document continued population declines and loss of aggregations (Sadovy de Mitcheson 2012). The aggregative reproduction style - gathering at predictable sites in large concentrations to spawn during a few weeks (over a few months) each year - makes the Nassau grouper vulnerable as a target of fishing like many other reef species that form large aggregations to spawn. In many places, aggregation-fishing once produced most of the annual landings of the species (e.g.,(Claro 1990)). Because Nassau grouper are only known to reproduce in spawning aggregations, removing ripe individuals during spawning has the potential to greatly influence population dynamics and future fishery yields (Shapiro 1987). The fact that much of the catch in many countries historically came from spawning aggregations (Aguilar-Perera 1994; Olsen and LaPlace 1979; Sadovy and Eklund 1999) likely magnified the effects to the extent that targeted aggregations have collapsed in many countries (Sadovy de Mitcheson 2012). Its declines have compromised the ecological function of a major top predator in the reef ecosystem (Mumby et al. 2006; Mumby et al. 2012; Randall 1967b). Law enforcement in many foreign countries is less than adequate, thus rendering the regulations ineffective; although many countries have taken regulatory measures to conserve Nassau grouper, the species faces an ongoing threat due to the inadequacy of regulatory mechanisms to prevent or remediate the impacts of other threats that are elevating the species' extinction risk, particularly fishing of spawning aggregations. Overutilization in the form of historical harvest has reduced population size and led to the collapse of spawning aggregations in many locations. While some countries have made efforts to curb harvest, fishing at spawning aggregation sites remains a "high risk" threat, and this risk is exacerbated by the inadequacy of regulatory control and law enforcement, which leads to continued overutilization (low abundance), reduced reproductive output, and reduced recruitment (more details found in 81 FR 42268, June 29, 2016).

There is currently no fishery for Nassau grouper in the United States, and possession is prohibited (for additional details of the history, see Sadovy and Eklund (1999)). Nassau grouper may show up as bycatch in various fisheries around south Florida and the CFMC FMPs as discussed in Section 5 of this opinion. Barotrauma from rapid decompression, increased time in warm surface waters, and increased exposure to predation threats may result in species mortality in the absence of a directed fishery (Bartholomew and Bohnsck 2005).

Other factors also pose a threat to the status of this species. Poor spatial population structure/connectivity is an increasing risk for Nassau grouper and is due, in part, to the declining number and size of spawning aggregations, which affects population structure. Population growth rate/productivity issues also present an increasing risk for the species. The nature of these factors could contribute to the species becoming endangered over the foreseeable future.

Abundance of Nassau grouper has diminished dramatically over the past several decades. This decline is a direct impact of historical harvest and the overfishing of spawning aggregations. The current reduced population abundance of Nassau grouper is a threat to the status of the species over the foreseeable future if abundance continues to decline.

In the final rule listing Nassau grouper (81 FR 42268, June 29, 2016), NMFS considered climate change as a threat to Nassau grouper including global warming, sea level rise, and ocean acidification. Although Nassau grouper occur across a range of temperatures, spawning occurs when sea surface temperatures range between 25°C-26°C (Colin 1992; Tucker and Woodward

1996). Because Nassau grouper spawn in a narrow window of temperatures, a rise in sea surface temperature outside that range could impact spawning or shift the geographic range of the species to overlap with waters within the required temperature parameters. Increased sea surface temperatures have also been linked to coral habitat loss through bleaching and disease, as well as possible negative effects to coral and coral reefs due to sea level rise (Munday et al. 2008). Further, increased global temperatures are also predicted to change parasite-host relationships and may present additional unknown concerns (Harvell et al. 2002; Marcogliese 2001). Another potential effect of climate change could be the loss of structural habitat in coral reef ecosystems as ocean acidification is anticipated to affect the integrity of coral reefs (Munday et al. 2008). While climate change has the potential to pose a threat to this species there is currently insufficient information to determine how it is affecting the extinction risk of the Nassau grouper now or in the foreseeable future.

3.2.15 Oceanic Whitetip Shark

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

Species Description and Distribution

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

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

The oceanic whitetip shark is distributed worldwide in epipelagic tropical and subtropical waters between 30º North latitude and 35º South latitude (Baum et al. 2006). In the Western Atlantic, oceanic whitetips occur from Maine to Argentina, including the Caribbean and Gulf of Mexico.

The oceanic whitetip shark is a highly migratory species of shark that is usually found offshore in the open ocean, on the outer continental shelf, or around oceanic islands in deep water, occurring from the surface to at least 152 meters (m) depth. Essential Fish Habitat (EFH) for the oceanic whitetip shark includes localized areas in the central Gulf of Mexico and Florida Keys, and depths greater than 200 m in the Atlantic (from southern New England to Florida, Puerto Rico and the U.S. Virgin Islands. Although the oceanic whitetip can be found in decreasing numbers out to latitudes of 30° N and 35° S, with abundance decreasing with greater proximity to continental shelves, it has a clear preference for open ocean waters between 10° S and 10° N (Backus et al. 1956; Bonfil et al. 2008; Compagno 1984; Strasburg 1958). The species can be found in waters between 15°C and 28°C, but it exhibits a strong preference for the surface mixed layer in water with temperatures above 20 °C, and is considered a surface-dwelling shark. It is however, capable of tolerating colder waters down to 7.75°C for short periods as exhibited by brief, deep dives into the mesopelagic zone below the thermocline (>200 m), presumably for foraging (Howey-Jordan et al. 2013; Howey et al. 2016). However, exposures to these cold temperatures are not sustained (Musyl et al. 2011; Tolotti et al. 2015) and there is some evidence to suggest the species tends to withdraw from waters below 15°C (e.g., the Gulf of Mexico in winter; Compagno 1984).

Little is known about the movement or possible migration paths of the oceanic whitetip shark. Although the species is considered highly migratory and capable of making long distance movements, tagging data provides evidence that this species also exhibits a high degree of philopatry (i.e., site fidelity) in some locations. To date, there have been three tagging studies conducted on oceanic whitetip sharks in the Atlantic. In the Atlantic, young oceanic whitetip sharks have been found well offshore along the southeastern coast of the U.S., suggesting that there may be a nursery in oceanic waters over this continental shelf (Compagno 1984; Bonfil et al. 2008). In the southwestern Atlantic, the prevalence of immature sharks, both female and male, in fisheries catch data suggests that this area may serve as potential nursery habitat for the oceanic whitetip shark (Coelho et al. 2009; Frédou et al. 2015; Tambourgi et al. 2013; Tolotti et al. 2015). Juveniles seem to be concentrated in equatorial latitudes, while specimens in other maturational stages are more widespread (Tambourgi et al. 2013). Pregnant females are often found close to shore, particularly around the Caribbean Islands. For more information on oceanic whitetip distribution, see Young et al. (2016).

*Life History Informati*on

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

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

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

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

Status and Population Dynamics

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

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

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

Threats

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

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

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

4.0 Environmental Baseline

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

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

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

4.1 Status of Species and Critical Habitat in the Action Areas

As stated in Section 2, the proposed actions would occur in the federal waters off the coasts of Puerto Rico and the USVI.

Sea Turtles

Based on the information discussed above, and their habitat and eating preferences, green and hawksbill sea turtles may be located in the action area for each of the proposed actions and may be affected by the fishing activities associated with the proposed actions. All of these species are migratory, traveling for forage grounds or reproduction purposes. The federal waters around Puerto Rico and the USVI may be used by these sea turtles. These same individuals may eventually migrate into oceanic waters, as well as other areas of the Gulf of Mexico, Caribbean Sea, and North Atlantic Ocean at certain times of the year, and thus may be impacted by activities occurring there; therefore, turtles in the action areas are exposed to threats discussed in Section 3. Sea turtle nesting also occurs along some parts of the coast of Puerto Rico and the

USVI. The status of the species of sea turtles (including the DPSs where applicable) in the action areas, as well as the threats to these species, are best reflected in their range-wide statuses and supported by the species accounts in Section 3 (Status of Species).

Corals and Acropora Critical Habitat

The elkhorn, staghorn, boulder star, mountainous star, lobed star, rough cactus, and pillar corals, and *Acropora* critical habitat may be located in the action areas and may be affected by the fishing activities associated with the proposed actions. *Acropora*, *Mycetophyllia, Dendrogyra,* and *Orbicella,* and *Acropora* critical habitat in the action areas face the same threats as corals and critical habitat outside the action areas. The coral reefs of Puerto Rico and the USVI have changed over the past decades, and coral cover has declined on many reefs (NMFS 2011a) (Brainard et al. 2011d). Unfortunately, in the summer of 2017, much of the action areas were impacted by hurricane activity, which further damaged and weakened these reef systems. NOAA issued a “Status of Puerto Rico’s Coral Reefs in the Aftermath of Hurricanes Irma and Maria” assessment report (NOAA 2018) that summarized the situation as follows:

> “In February 2018, the Federal Emergency Management Agency (FEMA) assigned the National Oceanic and Atmospheric Administration (NOAA) to conduct coral reef assessments and emergency triage restoration activities in support of the National Disaster Recovery Framework Natural and Cultural Resources Recovery Support Function. A total of 414,354 m^2 of coral reef and over 80,000 corals were surveyed at 153 sites across Puerto Rico between February 25 and May 7, 2018. Approximately, 5,400 coral fragments or broken coral colonies were reattached (triage) to the reef at 32 sites in the Northeast, North, and Vieques regions. Overall, an average of 11% of Puerto Rico’s corals were damaged by the hurricanes; however, some sites experienced far more severe damage (up to 100%). The major reef-building and ESA-listed corals were the most severely impacted species: pillar coral (*Dendrogyra cylindrus*), elkhorn coral (*Acropora palmata*), lobed star coral (*Orbicella annularis*), and staghorn coral (*A. cervicornis*). The Northeast (including Culebra), North, Vieques, and West regions showed the highest levels of damaged corals, as might be predicted by the highest wave energies experienced in these regions due to the paths of the hurricanes. However, within a region there was considerable variability of damage between sites, likely due to particular site’s exposure (i.e., orientation with respect the dominant wave direction) or amount and species of corals at that site.
>
> Triage activities salvaged thousands of at-risk corals; however, thousands more are still likely at-risk. Further, in many cases, the fragments or loose colonies may have been removed completely from the reef site by the waves and are lost from the system. Thus, some sites would benefit from replanting the reef with propagated corals from nurseries to restore, or potentially enhance the protective services the reefs provide.”

It is likely that the USVI reef system was similarly negatively impacted by the storms. Based on the best available information, we believe that status of these species and critical habitat for the action areas are no better, and likely somewhat currently worse, than described in Section 3.

Fish

The Nassau grouper, the Central and Southwest Atlantic DPS of scalloped hammerhead shark, and oceanic whitetip shark occur in the action areas and are subject to threats (e.g., capture in fisheries) that contributed to the species' listing status and are discussed in Section 3. The status of these species in the action areas, as well as the threats to these species, is supported by the species accounts in Section 3.

4.2 Factors Affecting Species in the Action Areas

Sea Turtles

The following analysis examines actions that may affect these species or their environments specifically within the action areas. Sea turtles found in the immediate project areas may travel widely throughout the Atlantic, Gulf of Mexico, and Caribbean Sea, and individuals found in the action areas can potentially be affected by activities anywhere within this wide range. These impacts outside of the action areas are discussed and incorporated as part of the overall status of the species as detailed in Status of Species section, above. Past and present activities that shape the environmental baseline for sea turtles in the action areas of this consultation primarily include poaching, boat strikes, incidental capture and mortality in fisheries, and ingestion and entanglement in marine debris. Other activities affecting sea turtle in the action areas include marine pollution, vessel and military activities, dredging, permits allowing take under the ESA, and research and education activities.

Existing data is not robust enough to fully assess the overall impact of each state, federal, and private action or other human activity in the action areas in their entirety. However, to the extent those impacts have manifested themselves at the population level, such past impacts are subsumed in the information presented on the status and trends of the species considered. Additionally, the benefits to sea turtles as a result of recovery activities already implemented may not be evident in the status and trend of the population for years given the relatively late age to maturity for sea turtles, and depending on the age class(es) affected.

Sea Turtle Harvest and Poaching

Boulon (2000) summarized historic sea turtle harvest in the action areas and poaching information through 1999. During the nineteenth century, the sea turtle fishery in Puerto Rico and USVI was subsistence only. However, exceptions to this occurred in the early twentieth century, for example in USVI when hawksbill sea turtles were taken only for scutes and meat was discarded (Fleming 2001). For more please refer to Eckert (1992). Much of the harvest occurred on the beaches adjacent to the action areas. A substantial green sea turtle fishery for food and export to Europe also existed historically (Fleming 2001). Fleming (2001b) provided a report on exploitation and trade of sea turtles in the Caribbean, including Puerto Rico and USVI. This report provides valuable information on the past exploitation of sea turtles in the U.S. Caribbean.

Fleming (2001b) reports that despite protective legislation in Puerto Rico, an unquantifiable but persistent demand for sea turtle products, meat and eggs in particular, persisted since the harvest and use of these was outlawed in the 1970s. Fishers took sea turtles opportunistically to sell to friends or restaurants, or for personal consumption. There was a steady black market largely organized to fill existing orders from specific buyers (Fleming 2001b). While meat and eggs have not appeared on restaurant menus since the 1980s, these were offered to specific customers in certain establishments in coastal areas (Fleming 2001b). Although there are no complete data on take of sea turtles in Puerto Rico, when the publication was written one estimate was 1,000-1,500 adult, subadult, and juvenile marine turtles were poached annually for personal consumption or sale to restaurants, markets, and trusted individuals (Fleming 2001b). Eggs of all species have been collected for food (Fleming 2001b). There has been a dramatic decrease in poaching of eggs and slaughter of nesting females due to the presence of sea turtle community groups since 2012, although it is possible some poaching is occurring undetected. However, inwater feeding areas still suffer from poaching issues. For example, in 2018 slaughtering of hawksbill was recorded in several keys of the south coast of Puerto Rico (C. Diez, Programa de Especies Protegidas-DRNA-PR, pers. comm. to P. Opay, NMFS SERO PRD, March 27, 2019).

Fleming (2001b) reports that with the exception of use for educational, scientific or display purposes, there was no legal harvest, use, or trade of sea turtles in the USVI around the year 2000. However, as with Puerto Rico, even despite protective legislation, there was demand for sea turtle meat and eggs and their use domestically continues (Fleming 2001b).

Low levels of poaching of sea turtles and eggs has been known to occur in St. Thomas/St. John (Fleming 2001b). Fishers in Frenchtown, on St. Thomas, traditionally harvested sea turtles and eggs. Fleming (2001b) reports that at the time of the report fishers still periodically poached in the USVI, but more often travelled to the British Virgin Islands to take turtles. Fleming (2001b) indicates that most of the marine turtle nesting in the USVI occurs on St. Croix, and more eggs and sea turtle poaching has occurred on St. Croix than elsewhere in the USVI. Fleming (2001b) also suggests that at the time of writing this was also likely partly attributable to a more depressed economy and a larger Hispanic population, that retained a cultural practice of eating eggs and sea turtles (Eckert 1989; Fleming 2001b).

While poaching of eggs, juveniles, and adult sea turtles in or near the action areas has declined dramatically, and the presence of researchers on nesting beaches has helped reduce poaching, the threat of poaching still occurs. Insufficient enforcement capabilities of protective laws in non-protected areas greatly limit the effectiveness of legal protection.

Federally Managed Fishing

The reef fish and spiny lobster fisheries in the U.S. Caribbean managed by the CFMC may affect sea turtles in the action area. These fisheries have been managed since 1984 (spiny lobster) and 1985 (reef fish) under the Reef Fish and Spiny Lobster FMPs and implementing regulations. The future operation of these fisheries under the island-based FMPs are part of the proposed action under consultation. This section considers effects of the fisheries to date. The fishery for offshore pelagic species, managed by the NMFS, Office of Sustainable Fisheries, Highly Migratory Species (HMS) Management Division also occurs in the EEZ portion of the action areas, and the HMS fishery may affect sea turtles.

Threatened and endangered sea turtles are adversely affected by several types of fishing gears that have been used in federally managed fisheries operating within the action areas for decades. Gillnet, hook-and-line gear (i.e., longlines and vertical line), and pot fisheries have all been documented as interacting with sea turtles. Available information suggests sea turtles can be captured in any of these gear types when the operation of the gear overlaps with the distribution of sea turtles, but gillnets are believed to have the most frequent interactions. In addition to active fishing gear, lost and abandoned gear may be especially deadly.

For the fisheries within the action areas that are federally-managed, impacts have been evaluated under Section 7.

Atlantic HMS (Swordfish, Tuna, Billfish, and Shark) Fishery
The fishery for Atlantic HMS is known to incidentally capture large numbers of leatherback and loggerhead sea turtles, particularly in the pelagic longline component. Pelagic longline, pelagic driftnet, bottom longline, and/or purse seine gear have all been documented taking sea turtles. Thousands of sea turtles have been caught in this fishery throughout the Atlantic since 1992, and a portion of these interactions occurred in the Caribbean. A subset of these animals were landed dead, and another subset likely experienced post-release mortality, a number which was substantial (NMFS 2004). A permanent prohibition on the use of driftnet gear in the swordfish fishery was published in 1999. NMFS reinitiated consultation on the pelagic longline component of this fishery (NMFS 2004) because the authorized number of incidental takes for loggerheads and leatherbacks sea turtles, species not likely to be adversely affected by the proposed actions under consultation in this Opinion, were exceeded. The resulting Biological Opinion stated the long-term continued operation this sector of the fishery was likely to jeopardize the continued existence of leatherback sea turtles, but reasonable and prudent alternatives were identified allowing for the continued authorization of the pelagic longline fishing that would not jeopardize leatherback sea turtles. Reinitiation of consultation has been conducted again and a biological opinion issued in 2020; jeopardy to any species is not expected. In the U.S. Caribbean, commercial tuna and swordfish fishermen primarily use pelagic longline (PLL), rod and reel, and handline gear (NMFS 2012). Longline vessels targeting HMS in the Caribbean set fewer hooks per set, on average and fish deeper in the water column than the fleets in other areas (e.g., Northeast Distant). Appendix B lists takes that have been most recently authorized through 2020. For the species likely to be adversely affected by the proposed actions under consultation in this Opinion, green sea turtles and hawksbill sea turtles, the incidental take statement anticipates that going forward up to 21 of any combination of the NA green, SA green, hawksbill, Kemp's ridley, or olive ridley sea turtles will be taken by the fishery ever 3 years, of which 8 takes will be lethal.

Caribbean Reef Fish Fishery
The reef fish fishery in waters around Puerto Rico and the USVI uses pots and traps, hook and line, longline, and spearguns. The proposed actions transition management of fisheries in the U.S. Caribbean, including the reef fish fishery, from Caribbean-wide FMPs to island-based FMPs. The future operation of the reef fish fishery, as managed under island-based FMPs, is the subject of this consultation so is not part of the environmental baseline. However, its past effects on sea turtles are part of the environmental baseline. Appendix B reports the takes currently authorized for the fishery.

Caribbean Spiny Lobster Fishery
The spiny lobster fishery in waters around Puerto Rico and the USVI uses pots and traps, and hand-harvest. Due to the predominance of fishable habitat in state waters, NMFS assumes that

most of the commercial harvest occurs in state waters, but fishery statistics do not allow accurate separation of harvest in the EEZ from harvest in state waters (Matos-Caraballo 2002). The proposed actions transition management of fisheries in the U.S. Caribbean, including the spiny lobster fishery, from Caribbean-wide FMPs to island-based FMPs. The future operation of the spiny lobster fishery, as managed under island-based FMPs, is the subject of this consultation so is not part of the environmental baseline. However, its past effects on sea turtles are part of the environmental baseline. Appendix B reports the takes currently authorized for the fishery.

Territorially Managed Fishing

Recreational fishing as regulated by Puerto Rico and the USVI and can affect ESA-listed sea turtle species considered in this opinion. Commercial fishery components for the territorial sectors (managed by the territories), and the size of the fishing area for those sectors are larger than the area managed by the federal FMPs. Therefore, their impacts are likely greater than those of the federally-managed fisheries. Pressure from territorially managed fishing is likely to continue, but with effects that are hard to quantify.

Vessel Traffic

Commercial and recreational vessels can adversely affect sea turtles through propeller and vessel strikes. Many records of vessel interactions have been documented within the action areas. Vessel strikes can result in direct injury or death through collision (concussive) impacts or propeller wounds. A sea turtle’s spine and ribs are fused to the shell, which is a living part of their body that grows, sheds, and bleeds. Rapidly moving vessels can cause fractures in the head or carapace, and injuries to the carapace can fracture the spinal column and cause buoyancy problems. Abnormally buoyant sea turtles are unable to dive for food or escape predators or future vessel strikes. Propellers cut through the shell and sever or damage the spine and internal organs. Chronic or partially healed propeller wounds also may be associated with secondary problems such as emaciation and increased buoyancy (Walsh 1999).

NMFS and the U.S. Coast Guard (USCG) have completed an informal Section 7 consultation for the Caribbean Marine Event Program for all annually occurring marine events in USVI and Puerto Rico. As a result of this consultation, the USCG now includes as permit conditions measures to avoid and minimize potential impacts to sea turtles and their habitat from marine events, especially events involving motorized vessels such as speedboat races.

The proliferation of vessels is associated with the proliferation and expansion of docks, the expansion and creation of port facilities, and the expansion and creation of marinas. As part of the ESA Section 7 consultation process for dock, port, and marine construction activities under the jurisdiction of the U.S. Army Corps of Engineers (USACE), NMFS also considers the impacts of the vessel traffic from the operation of these facilities and any measures to avoid and minimize adverse impacts to sea turtles.

It is difficult to definitively evaluate the potential risk to sea turtles stemming from specific vessel traffic from any action because numerous factors affect the potential for vessel strike and vessel strike rates, including vessel type and speed, environmental factors, amount of vessel traffic, and sea turtle abundance. This difficulty is compounded by a general lack of information on vessel use trends, particularly with regard to offshore vessel traffic in the action areas.

The proportion of vessel-struck sea turtles that survive or die is unknown. In many cases, it is not possible to determine whether documented injuries on stranded animals resulted in death or were post-mortem injuries. Sea turtles in the wild are documented with healed injuries; thus, we know at least some sea turtles survive without human intervention, but many vessel strikes are likely fatal.

Marine Debris and Pollution

Marine debris, including abandoned, lost, or otherwise discarded fishing gear (ALDFG) can pose a serious threat to sea turtles in the action areas. Sea turtles have been found to ingest a wide variety of abiotic debris items such as plastics. ALDFG can kill sea turtles via entanglement, ingestion, or ghost fishing as lost gear continues to function undetected.

Sources of pollutants include atmospheric loading of pollutants such as PCBs and stormwater runoff from coastal towns and cities into rivers and canals emptying into bays and the ocean. McKenzie et al. (1999) measured concentrations of chlorobiphenyls and organochlorine pesticides in sea turtles tissues collected from the Mediterranean (Cyprus, Greece) and European Atlantic waters (Scotland) between 1994 and 1996. Decreasing lipid contaminant burdens with sea turtle size were observed in green turtles, most likely attributable to a change in diet with age. No information on detrimental threshold concentrations is available and little is known about the consequences of exposure of organochlorine compounds to sea turtles. Research is needed into how chlorobiphenyl, organochlorine, and heavy-metal accumulation affect the short- and long-term health of sea turtles and what effect those chemicals have on the number of eggs laid be females.

Nutrient loading from land-based sources, such as coastal communities and agricultural operations, stimulate plankton blooms in closed or semi-closed estuarine systems. The effects on larger embayments are unknown. Water quality monitoring studies by DPNR’s Division of Environmental Protection (DEP) in waters around USVI indicate that surface waters are affected by increasing point and non-point source pollution from failing septic systems, discharges from vessels, failure of BMPs on construction sites, and failure of on-site disposal methods (Rothenberger et al. 2008). These factors result in increased sedimentation and nutrient transport, bacterial contamination, and trash and other debris entering surface and nearshore waters from developed areas. The effects of these water quality declines on species such as sea turtles are unknown. However, it is clear that water quality degradation leads to habitat degradation of coral reefs and other coralline communities, as well as seagrass beds. Thus, some indirect effect on green and hawksbill sea turtles due to degradation of foraging habitat quality could occur.

The development of marinas and docks in inshore waters can negatively impact nearshore habitats. Fueling facilities at marinas can sometimes discharge oil, gas, and sewage into sensitive estuarine and coastal habitats. Although these contaminant concentrations do not likely affect the more pelagic waters, the species of sea turtles analyzed in this Biological Opinion travel between nearshore and offshore habitats and may be exposed to and accumulate these contaminants during their life cycles.

Oil and Gas exploration has not been conducted in the U.S. Caribbean. However, HOVENSA, (formerly Hess Oil Virgin Islands Corp, now the Limetree Bay refinery) located on St. Croix was among the top ten largest refineries in the world and the second largest in the United States. Established in the 1960s, the oil refinery was capable of processing up to a half million barrels of oil a day. Leaks from oil process and storage resulted in plumes of oil floating on top of the groundwater underlying the facility. Hurricane Hugo in 1999 produced a considerable number of small spills from damaged vessels and shore structures in the USVI, but no major spills have occurred there (ITOPF 2006a). Several major spills (i.e., 2 spills of approximately 2,500 tons and 1 of approximately 5,000 tons) have occurred in Puerto Rico waters and, despite the deployment of considerable amounts of equipment, large areas of the coast were oiled and large scale operations were undertaken to recover sunken oil (ITOPF 2006b). Oil spills can impact sea turtles directly through three primary pathways: ingestion - when animals swallow oil particles directly or consume prey items that have been exposed to oil; absorption – when animals come into direct contact with oil; and inhalation - when animals breath volatile organics released from oil, or from “dispersants” applied by response teams in an effort to increase the rate of degradation of the oil in seawater.

Military Activities

Military ordnance detonation has adversely affected sea turtles in the action areas. The Navy conducted military exercises between 1941 and 2003, including ship-to-shore and aerial bombing with live ammunition via its Atlantic Fleet Weapons Training Facility on the island of Vieques. Various types of explosive and non-explosive ordnance were used for aerial and naval bombardment. Although active use of the range has ended, cleanup of unexploded ordinance is continuing.

Natural Disturbances

Stochastic (i.e., random) events, such as hurricanes, occur in Puerto Rico and the USVI and can affect sea turtles in the action areas. These events are by nature unpredictable, and their effect on the recovery of the species is unknown; yet, they have the potential to directly impede recovery if animals die as a result or indirectly if important habitats are damaged. Storms result in breakage of sessile benthic organisms from extreme wave action and storm surges. Intense storms that cover a broad area can eliminate or damage large expanses of reef or result in blowouts and loss of seagrass habitats. For example, major hurricanes have caused significant losses in coral cover and changes in the physical structure of many reefs in USVI. There have been over 10 hurricanes that have affected the reefs of USVI between 1979 and 2017 (Drayton et al. 2004). Hurricane David in 1979 caused a reduction in mean coral cover along transects at Flat Cay Reef, STT, from 65% to 44% and Hurricane Hugo in 1989 caused a 30-40% decline in coral cover along transects and within quadrats in Great Lameshur Bay, STJ (Rogers et al. 2008b). Tropical storms and hurricanes in 2004, 2008, and 2010 also resulted in severe flooding across USVI. This flooding also caused significant sedimentation of areas resulting in additional degredation of reef habitats. The most recent hurricane in 2017 caused serious damage (as discussed in 4.1 of this section). In addition to affecting the sessile benthic organisms themselves, these changes in the structure of the reef affect species like sea turtles, in particular greens and hawksbills. In-water habitat for green and hawksbill sea turtles is temporarily lost or temporarily or permanently degraded (depending on the magnitude of the storm).

Climate Change

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

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

Conservation Actions Benefitting Turtles

NMFS has implemented a series of regulations aimed at reducing the potential for incidental capture and mortality of sea turtles from commercial fisheries near or in the action areas. These include sea turtle release gear requirements for Caribbean fisheries, including long line and trap gears. Under Section 6 of the ESA, we may enter into cooperative research and conservation agreements with states to assist in recovery actions of listed species.

Sea Turtle Research, Monitoring, Outreach and Education

The Puerto Rico DNER conducts research on hawksbill sea turtles in the area of Mona and occasionally Desecheo Island, and on green sea turtles in Culebra. The Puerto Rico DNER also monitors beaches around Puerto Rico, including those on Vieques in coordination with the Navy. The Sea Turtle Program of Puerto Rico is a multi-agency collaboration between DNER together with several nongovernmental organizations and other agencies (Sea Grant-UPR, Rio Piedras-UPR, Mayaguez-UPR, Chelonia, WIDECAST, United States Fish and Wildlife Service). The main goal is to educate the public, and to investigate, recuperate and protect the species. Nesting beach surveys are conducted on several sites along the coast of Puerto Rico and adjacent islands. The species targeted for these surveys are the leatherback (April-July) and hawksbill (August-December). Since 1992, in-water surveys have been conducted for hawksbill turtles at Mona Island and Desecheo and for green turtles at Culebra.

Several USVI research projects have been ongoing in the action areas for decades. Since 1981 leatherback sea turtle nesting has been protected and monitored at the USFWS Sandy Point National Wildlife Refuge in St. Croix using saturation tagging protocols. Nests in danger of erosion are relocated to low-risk beach zones, which has resulted in increased hatch success and an increasing nesting population. Other sea turtle species are monitored by project staff as well. Since 1988, hawksbill sea turtle nesting on Buck Island, St. Croix, has been monitored by National Parks Service (NPS) staff using saturation tagging protocols. Since 1994, in-water capture of juvenile hawksbill sea turtles by the NPS at Buck Island, St. Croix, has also provided information on growth rates, movement patterns, habitat use, sex ratios, and general ecology. Also since 1994, saturation tagging protocols during peak green and hawksbill nesting season have been used on East End Beaches, St. Croix, that are owned by The Nature Conservancy.

Reducing Threats from Pelagic Longline and Other Fisheries

On July 6, 2004, NMFS published a final rule to implement management measures to reduce bycatch and bycatch mortality of sea turtles in the Atlantic pelagic longline fishery (69 FR 40734). The management measures include mandatory circle hook and bait requirements, and mandatory possession and use of sea turtle release equipment to reduce bycatch mortality. The rulemaking, based on the results of the 3-year Northeast Distant Closed Area research experiment and other available sea turtle bycatch reduction studies, is expected to have significant benefits to endangered and threatened sea turtles.

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

Sea Turtle Handling and Resuscitation Techniques

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

Sea Turtle Rescue and Rehabilitation

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

NMFS and the USVI have established stranding procedures to rescue and rehabilitate any live stranded sea turtles. The Sea Turtle Assistance and Rescue (STAR) network responds to sea turtle strandings on St. Croix. STAR is a volunteer network composed of local agency personnel, non-governmental organizations, veterinarians, and private individuals. STAR is managed through the West Indies Marine Animal Research and Conservation Service.

Central and Southwest Atlantic DPS of Scalloped Hammerhead Sharks, and Oceanic Whitetip Sharks

Federally Managed Fishing

The reef fish and spiny lobster fisheries in the U.S. Caribbean managed by the CFMC may affect scalloped hammerhead, and oceanic whitetip sharks in the action areas. These fisheries have been managed since 1984 under the Reef Fish and Spiny Lobster FMPs and implementing regulations. The future operation of these fisheries under the island-based FMPs are part of the proposed action under consultation. This section considers effects of the fisheries to date. The fishery for offshore pelagic species, managed by the NMFS, Office of Sustainable Fisheries, HMS Management Division also occurs in the EEZ portion of the action area and may affect these species.

Scalloped hammerhead sharks and oceanic whitetip sharks are adversely affected by types of fishing gears that have been used within the action area for decades. Gillnet and hook-and-line gear (i.e., longlines and vertical line) have all been documented as interacting with these species. These species can be captured in any of these gear types when the operation of the gear overlaps with their distribution. In addition to active fishing gear, lost and abandoned gear may be especially deadly.

For the fisheries within the action area that are federally managed, impacts have been evaluated under Section 7.

Atlantic HMS (Swordfish, Tuna, Billfish, and Shark) Fishery

The federally managed fishery for Atlantic HMS, including the fisheries for swordfish, tuna, billfish, and sharks, are known to incidentally capture sharks and rays. A subset of these animals likely experience post-release mortality, but the exact number is unclear. In the U.S. Caribbean, commercial tuna and swordfish fishermen primarily use pelagic longline (PLL), rod and reel, and handline gears (NMFS 2012). Longline vessels targeting HMS in the Caribbean set fewer hooks per set, on average and fish deeper in the water column than the fleets in other areas (e.g., Northeast Distant). The oceanic whitetip shark’s vertical and horizontal distribution significantly increases its exposure to industrial fisheries, including pelagic longline and purse seine fisheries. NMFS just concluded assessing the potential take of scalloped hammerhead sharks and oceanic whitetip sharks in the Caribbean EEZ waters, and estimates mortality of up to 249 scalloped hammerhead- central and southwest Atlantic DPS sharks, and 498 oceanic whitetip sharks every three year period (NMFS 2020).

Reef Fish Fishery

The reef fish fishery in waters around Puerto Rico and the USVI uses hook and line and longline that has taken scalloped hammerhead and oceanic whitetip sharks rays in the past. The proposed actions transition management of fisheries in the U.S. Caribbean, including the reef fish fishery, from Caribbean-wide FMPs to island-based FMPs. The future operation of the reef fish fishery, as managed under island-based FMPs, is the subject of this consultation so is not part of the environmental baseline. However, its past effects on scalloped hammerhead sharks and oceanic whitetip sharks are part of the environmental baseline. The exact number of these species taken is not known, but estimated to likely to have been in the dozens of animals. Mortality of the rays was expected to have been low, but mortality of the other species high.

Fisheries Independent Monitoring (FIM)

FIM is a broad approach to evaluating marine communities and the species that comprise them. FIM also investigates habitat conditions for purposes of learning more about system-wide trends. FIM data are collected using statistically valid sampling techniques to supplement fisheries-dependent information obtained from anglers and commercial fishing operations. A variety of fishing gears and techniques ensure sampling of fisheries resources in a wide range of sizes and ages. One lethal take of scalloped hammerhead sharks is expected in the FIM project as the result of using hook and line gear every 5 years (NMFS 2016).

Territorially Managed Fishing

Recreational fishing as regulated by Puerto Rico and the USVI and can affect protected species considered in this opinion. Commercial fishery components for the territorial sectors (managed by the territories), and the size of the actual fishing area for those sectors are larger than the area managed by the federal FMPs. Therefore, their impacts are likely greater than those of the federal fisheries. Pressure from territorially managed fishing is likely to continue, but with effects that are hard to quantify.

Marine Pollution

Environmental pollutants may have negative impacts on shark and ray species. Many pollutants in the environment have the ability to bioaccumulate in fish species and have the potential to cause negative physiological impacts to shark species (Young et al. 2016). However, it is unclear what effects these pollutants are having on these species.

Climate Change

As discussed earlier in this Opinion, there is a large and growing body of literature on past, present, and future impacts of global climate change. Potential effects to the environment commonly mentioned include changes in sea temperatures and salinity (due to melting ice and increased rainfall), ocean currents, storm frequency and weather patterns, and ocean acidification. These changes have the potential to affect species behavior and ecology including migration, foraging, reproduction (e.g., success), and distribution.

Additional discussion of climate change can be found in the Status of the Species. However, more information is needed to better determine the full and entire suite of impacts of climate change on these species and specific predictions regarding impacts in the action area are not currently possible.

Conservation and Recovery Actions Benefiting Species

NMFS prohibits the retention of scalloped hammerhead sharks on Atlantic HMS commercially-permitted vessels that have PLL gear on board, and by recreational fishermen fishing with a General Category permit participating in an HMS tournament or those fishing under an HMS Angling or Charter/ Headboat permit when tuna or tuna-like species are also retained.

The increasing number of shark fin bans are one effort to conserve sharks. The concern regarding the practice of finning and its effect on global shark populations has been growing both domestically and internationally. The push to stop shark finning and curb the trade of shark fins is evident overseas, including in Asian countries, where the demand for shark fins is highest.

CITES listings are another effort to conserve the DPS. Member nations of CITES, referred to as “Parties,” voted in support of listing scalloped hammerhead sharks and oceanic whitetip sharks in Appendix II—an action that means increased protection, but still allows legal and sustainable trade.

There are also national and international organizations with shark-focused goals that include advocating the conservation of sharks through education and campaign programs and conducting shark research to fill data gaps regarding the status of shark species. Some of these organizations include: The Pew Environment Group, Oceana, Ocean Conservancy, Shark Trust, Bite-Back, Shark Project, Pelagic Shark Research Foundation, Shark Research Institute, and Shark Savers.

When the Central and Southwest Atlantic scalloped hammerhead DPS was listed as threatened, NMFS evaluated the needs of and threats to the DPS and determined that protective regulations were not currently necessary and appropriate for the conservation of the DPS. Similarly, NMFS did not propose such regulations for the oceanic whitetip shark, but may consider potential protective regulations pursuant to section 4(d) for the oceanic whitetip in a future rulemaking. Therefore, there are no prohibitions to take (e.g., capture) these species. For example, no permits to conduct research are required.

Nassau Grouper

Federally Managed Fishing

Fisheries in the action areas managed via CFMC that may affect Nassau grouper are the reef fish and spiny lobster fisheries.

Nassau groupers are adversely affected by vertical line and pot/trap fisheries. Available information suggests this species can be captured in any of these gear types when the operation of the gear overlaps with its distribution. In addition to active fishing gear, lost and abandoned gear may be especially deadly.

For the fisheries within the action areas that are federally managed, impacts have been evaluated under Section 7.

Reef Fish and Spiny Lobster Fisheries

The reef fish and spiny lobster fisheries in waters around Puerto Rico and the USVI use hook and line and trap gear that has taken Nassau grouper in the past. The proposed actions transition management of fisheries in the U.S. Caribbean, including the reef fish and spiny lobster fisheries, from Caribbean-wide FMPs to island-based FMPs. The future operation of the reef fish and spiny lobster fisheries, as managed under island-based FMPs, is the subject of this consultation so is not part of the environmental baseline. However, the fisheries' past effects on Nassau grouper are part of the environmental baseline. The exact number of this species taken is unknown, but estimated to have been hundreds of animals.

Territorially Managed Fishing

Recreational fishing as regulated by Puerto Rico and the USVI and can affect protected species considered in this opinion. Commercial fishery components for the territorial sectors (managed by the territories), and the size of the fishing areas for those sectors are larger than the area managed by the federal FMPs. Therefore, their impacts are likely greater than those of the federal fisheries. Pressure from territorially managed fishing is likely to continue, but with effects that are hard to quantify.

Marine Pollution

Environmental pollutants may have negative impacts on species. Many pollutants in the environment have the ability to bioaccumulate in fish species and have the potential to cause negative physiological impacts (e.g.,Young et al. 2016). However, it is unclear what effects these pollutants are having on the Nassau grouper.

Climate Change

As discussed earlier in this Opinion, there is a large and growing body of literature on past, present, and future impacts of global climate change. Potential effects to the environment commonly mentioned include changes in sea temperatures and salinity (due to melting ice and increased rainfall), ocean currents, storm frequency and weather patterns, and ocean acidification. These changes have the potential to affect species behavior and ecology including migration, foraging, reproduction (e.g., success), and distribution.

Additional discussion of climate change can be found in Section 3, Status of the Species. However, more information is needed to better determine the full and entire suite of impacts of climate change on the species, and specific predictions regarding impacts in the action area are not currently possible.

Conservation and Recovery Actions Benefiting Species

NMFS (2016) notes that general (throughout the species range) conservation efforts with the potential to address identified threats to Nassau grouper include, but are not limited to, fisheries management plans, education about overfishing and fishing of spawning aggregations, and projects addressing the health of coral reef ecosystems. While these can potentially benefit the species, many of these efforts are conducted outside the action areas.

Take and possession of Nassau grouper have been prohibited in federal waters since 1990. A ban on fishing for and possessing Nassau grouper has been in effect in the state of Florida since 1993, in Puerto Rico since 2004, and USVI since 2006.

As mentioned earlier, this species is newly listed under the ESA. No recovery plan currently exists for the Nassau grouper. NMFS will develop and implement a plan unless such a plan will not promote the conservation of the species.

Corals and *Acropora* Critical Habitat

Federally Managed Fishing

Fisheries in the action areas managed via the CFMC that may affect corals are the reef fish and spiny lobster fisheries.

Coral are adversely affected by pot or trap fisheries and associated activities (anchoring, deploying, retrieving). Available information suggests these species and their habitat can be negatively impacted when the operation of the gear overlaps with their distribution. In addition to active fishing gear, lost and abandoned gear may be damaging to corals and coral habitat.

For the fisheries within the action areas that are federally managed, impacts have been evaluated under Section 7.

Reef Fish and Spiny Lobster Fisheries

The reef fish and spiny lobster fishery activities in waters around Puerto Rico and the USVI have taken corals in the past. The proposed actions transition management of fisheries in the U.S. Caribbean, including the reef fish and spiny lobster fisheries, from Caribbean-wide FMPs to island-based FMPs. The future operation of the reef fish and spiny lobster fisheries, as managed under island-based FMPs, is the subject of this consultation so is not part of the environmental baseline. However, their past effects on coral are part of the environmental baseline. Up to 93 ft^2 of staghorn coral has been estimated to have been lethally affected every 3 years since 2011, from Caribbean spiny lobster fishery activities (NMFS 2011b). Up to 10,619 m^2 (0.0041 mi^2) of staghorn and elkhorn coral (combined) have been estimated to have been directly affected by the reef fish fishery every 3 years since 2011 (NMFS 2011). Additionally, the reef fish fishery has indirectly affected coral and *Acropora* critical habitat through its negative impacts (e.g., altering ecosystem functions and the resilience of these systems) on the algae control that has resulted from removal of herbivorous fish that inhabit the coral reefs of the Puerto Rico and the USVI (NMFS 2011a).

While we do not have similar calculations for the other coral species that are the subject of this Opinion, because they were listed after the most recent Opinions for the fisheries , we believe that given their similar life history characteristics, location in the action area of the fisheries, and susceptibility to the same stressors analyzed in the 2011 Opinions, that they too have also been negatively affected by the fisheries.

Federal Vessel Operations

Potential sources of adverse effects from federal vessel operations in the action areas include operations of the USCG, the Environmental Protection Agency (EPA), NOAA, and the NPS. Through the Section 7 process, where applicable, NMFS will continue to establish conservation measures for agency vessel operations to avoid or minimize adverse effects to listed species. Currently, they present the potential for some level of interaction.

Territorially Managed Fishing

Recreational fishing as regulated by Puerto Rico and the USVI can affect protected species considered in this opinion. Commercial fishery components for the territorial sectors (managed by the territories), and the size of the fishing area for the sectors are larger than the area managed by the federal FMPs. Therefore, their impacts are likely greater than those of the federal fisheries. Pressure from territorially managed fishing is likely to continue, but with effects that are hard to quantify.

Natural Disturbance

Hurricanes and large coastal storms can also significantly harm corals. Hurricanes are also sometimes beneficial, if they do not result in heavy storm surge, during years with high sea surface temperatures, as they lower the temperatures providing fast relief to corals during periods of high thermal stress (Heron et al. 2008). Hurricanes may also act to scour competing macroalgae off patches of reef. However, major hurricanes have caused significant losses in coral cover and changes in the physical structure of many reefs in the USVI. For example, there were ten hurricanes that affected the reefs of the USVI between 1979 and 2003 (Drayton et al. 2004). Hurricane David in 1979 caused a reduction in mean coral cover along transects at Flat Cay Reef, St. Thomas, from 65 to 44% and Hurricane Hugo in 1989 caused a 30 to 40% decline in coral cover along transects and within quadrats in Great Lameshur Bay, St. John (Rogers et al. 2008). Overall, an average of 11% of Puerto Rico’s corals were damaged by the 2017 hurricanes; however, some sites experienced far more severe damage (up to 100%).

Climate Change

As discussed earlier in this Opinion, there is a large and growing body of literature on past, present, and future impacts of global climate change. Potential effects to the environment commonly mentioned include changes in sea temperatures and salinity (due to melting ice and increased rainfall), ocean currents, storm frequency and weather patterns, and ocean acidification. These changes have the potential to affect species behavior and ecology including, but not limited to, reproduction (e.g., success), and distribution. Additional discussion of climate change can be found in Section 3 Status of the Species.

Conservation and Recovery Actions Benefiting Species

NMFS has implemented a Section 4(d) rule to establish “take” prohibitions for listed elkhorn and staghorn corals. These prohibitions include the import, export, or take of elkhorn or staghorn corals for any purpose, including commercial activities. The 4(d) rule has exceptions for some activities, including scientific research and species enhancement, and restoration carried out by authorized personnel. On November 26, 2008, NMFS published a final rule designating critical habitat for listed elkhorn and staghorn corals (73 FR 72210). The critical habitat designation requires that all actions with a federal nexus are evaluated to ensure that the adverse modification of critical habitat will not occur.

As described in further detail below, federal regulations implementing fishery management plans prepared by the CFMC prohibit the use of bottom-tending fishing gear in some seasonally and permanently closed fishing areas containing coral reefs in federal waters of the EEZ. The USVI and Puerto Rico are moving toward similar regulations for both commercial and recreational fishers in territorial waters, and the USVI has established a ban on the use of gill and trammel nets, with the exception of surface nets for catching bait fish. In addition to regulations, education and outreach activities as part of the NOAA Coral Reef Conservation Program (CRCP), as well as through NMFS' ESA program, are ongoing through the Southeast Regional Office. NOAA Restoration Center has also established a staff member in Puerto Rico to participate in vessel grounding response and carry out restoration activities.

A recovery plan for elkhorn and staghorn corals has been completed and an implementation effort initiated. A recovery outline is available for pillar, rough cactus, lobed star, mountainous star, and boulder corals, with a recovery plan to follow.

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 regulations implementing the Coral and Reef Associated Plants and Invertebrates FMP allow some harvest of specifically defined Caribbean coral reef resources (50 CFR Part 622, Subpart U and Appendix A), but prohibit harvest or possession of other corals (50 CFR 622.472; 622.2[31]). For species for which harvest is allowed, regulations prohibit the use of chemicals, plants, or plant-derived toxins and power-assisted tools to harvest a Caribbean coral reef resource (50 CFR 622.9(b); 50 CFR 622.471(a)). Explosives cannot be used to fish in the Caribbean EEZ. 50 CFR 622.9(a). Federal regulations also prohibit the use of pots and traps, gillnets and trammel nets, and bottom longlines year round in seasonally closed areas in the EEZ, including the mutton snapper spawning aggregation area off Puerto Rico; the red hind spawning aggregation areas off St. Croix and the west coast of Puerto Rico (Tourmaline Bank and Abrir La Sierra Bank); the Grammanik Bank off St. Thomas/St. John; and Bajo de Sico off Puerto Rico (50 CFR 622.435(b)(2)). Amendment 1 to the FMP for Corals and Reef Associated Plants and Invertebrates established a marine conservation district in federal waters southwest of St. Thomas, the Hind Bank Marine Conservation District, where fishing for any species and anchoring by fishing vessels is prohibited year-round (50 CFR 622.435(b)(1)). Anchoring by fishing vessels also is prohibited year-round in federal waters of Bajo de Sico, off the west coast of Puerto Rico (50 CFR 622.435(b)(3)). Among other things, these measures protect coral or hard bottom from fishing and anchoring activities. NMFS also conducts essential fish habitat (EFH) consultations. Through EFH consultations, NMFS works with federal agencies to conserve and enhance EFH, which includes corals.

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[31] The species of coral for which harvest and possession is prohibited are as follows: a gorgonian, that is, a Caribbean coral reef resource of the Class Anthozoa, Subclass Octocorallia, Order Gorgonacea; a live rock; or a stony coral, that is, a Caribbean coral reef resource of the Class Hydrozoa (fire corals and hydrocorals) or of the Class Anthozoa, Subclass Hexacorallia, Orders Scleractinia (stony corals) and Antipatharia (black corals); or a part thereof. 50 CFR 622.2.

The NOAA Coral Reef Conservation Program (CRCP) is authorized to protect and manage coral reefs through the implementation of the Coral Reef Conservation Act of 2000. NOAA works with the non-profit National Fish and Wildlife Foundation to build public-private partnerships to reduce and prevent degradation of coral reefs. Through the CRCP, NOAA conducts activities such as mapping, monitoring, assessment, research, and restoration that benefit coral reef ecosystems; enhancing public awareness of such ecosystems; assisting states to remove abandoned vessels and marine debris from reefs; and conducting cooperative management of coral reef ecosystems.

The Virgin Islands National Park, the Virgin Islands Coral Reef National Monument, and Buck Island Reef National Monument provide protection to corals.

The Commonwealth of Puerto Rico has several laws and proposed regulations that may aid in the conservation of corals. The most pertinent statute is the 2000 Law for the Protection, Conservation, and Management of Coral Reefs in Puerto Rico (Law 147). This law explicitly mandates the conservation and management of coral reefs in order to protect their functions and values, and provides for the creation of zoned areas in order to mitigate impacts from human activities. These zones will facilitate the DNER in controlling human activity, such as anchoring, that can directly impact coral. Law 147 also directs the DNER to identify and mitigate threats to coral reefs from degraded water quality due to pollution and additionally directs the DNER to designate priority areas as marine reserves. Marine reserves are defined as areas where all extractive activities are prohibited in order to help recover depleted fishery resources and protect biodiversity, and can protect coral by preventing impacts from fishery gear. There are currently an additional 13 natural reserves in Puerto Rico that have coral reefs within their boundaries, all of which are located on all coasts and offshore islands.

The Territory regulates activities that occur in terrestrial and marine habitats of the USVI. The V.1. Code prohibits the taking, possession, injury, harassment, sale, offering for sale, etc. of any indigenous species, including live rock (V.1. Code Title 12 and the Indigenous and Endangered Species Act of 1990). Additionally, the 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.

Section 6 of the ESA allows NMFS to enter into cooperative agreements with states to assist in recovery actions of ESA-listed species, including scientific research related to documenting species condition and trends in presence and abundance. DNER renewed its Section 6 agreement with NMFS in 2019 however, is not receiving any funds this fiscal year.

Recovery actions may also include the collection of fragments from coral colonies, their grow-out in nursery areas, and the outplanting of fragments. The DNER has issued memoranda of understanding to several coral nursery operators with coral nurseries in various areas around Puerto Rico. The DNER is also the entity responsible for permitting the use of coral species, including ESA-listed corals, in coral nurseries. NMFS completed ESA Section 7 consultation with the USACE for the issuance of a Regional General Permit, SAJ-112, that would authorize the installation and maintenance of coral nursery operations up to 1 acre (ac) in size that do not require the placement of fill, such as the installation of polyvinyl chloride (PVC) “trees.” NOAA’s Restoration Center also maintains coral nurseries in various locations around Puerto Rico and uses farmed corals in efforts to repair damage from vessel groundings on reefs.

The NOAA Coral Reef Conservation Program, through its internal grants, external grants, and grants to the Territory, Commonwealth, 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 USVI and the status of listed corals. SERO has also developed outreach materials regarding the listing of corals. These materials have been circulated to constituents during education and outreach activities and public meetings, and as part of other Section 7 consultations.

5.0 Effects of the Action

Section 7(a)(2) of the ESA and implementing regulations require biological opinions to evaluate the effects of federal actions to determine if it would be reasonable to expect them to appreciably reduce listed species' likelihood of surviving and recovering in the wild by reducing their reproduction, numbers, or distribution (16 U.S.C. 1536; 50 CFR 402.02). The term "species" includes any subspecies of fish or wildlife or plants, and any distinct population segment (DPS) of any species of vertebrate fish or wildlife, which interbreeds when mature. Section 7 of the ESA and its implementing regulations also require (as applicable) biological opinions to determine if federal actions would appreciably diminish the value of critical habitat as a whole for the conservation of listed species (16 U.S.C. 1536; 50 CFR 402.02).

In this section of the Opinion we assess the effects of the proposed actions on ESA-listed corals (i.e., elkhorn, staghorn, boulder star, mountainous star, lobed star, rough cactus, and pillar corals), *Acropora* critical habitat, the NA and SA DPSs of green sea turtles, hawksbill sea turtles, Nassau grouper, the Central and Southwest Atlantic DPS of scalloped hammerhead sharks, and oceanic whitetip sharks. 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 addition to effects from fishing activities themselves, effects on corals and *Acropora* critical habitat are expected from fishing for herbivorous reef fish, as managed under the FMPs. These fishing activities in federal waters may affect populations of herbivorous fishes in commonwealth and territorial waters because these populations are continuous, fluid, and move across jurisdictional boundaries. Since species travel back and forth between the U.S. EEZ and commonwealth or territorial waters, harvest of these species in the EEZ under the proposed actions may reduce their numbers or size, potentially changing the amount of algae grazed throughout the EEZ and commonwealth and territorial waters. Such an impact could affect the resilience of ESA-listed coral and *Acropora* critical habitat in EEZ and commonwealth and territorial waters.

We do not expect indirect consequences to other ESA-listed species (i.e., non-coral species) considered in this consultation. Indirect consequences could result from habitat degradation, reduction of prey or foraging base, for example. The operation of the federally-managed fisheries in the U.S. Caribbean (i.e., vessel operations, gear deployment and retrieval) is not expected to impact the water column or habitat in a manner that would have measurable effects on these non-coral species. Unlike mobile trawls and dredges that physically disturb habitat as they are dragged along the bottom, the gear types used in the U.S. Caribbean fisheries are suspended in the water column or essentially stationary on the bottom (bottom longline gear fits this latter category). Similarly, anchoring activities are not expected to impact the water column or habitat in a manner that would have measurable effects on these non-coral species.

While sea turtles may opportunistically prey on dead fish, fish, spiny lobster, or conch[32] are not the primary prey of green or hawksbill sea turtles; thus, a reduction of prey or foraging base for sea turtles is not likely to result from fishing under the proposed actions. Juvenile green sea turtles in the action area have a herbivorous diet, and by adulthood feed almost exclusively on seagrasses and algae. In the Caribbean, hawksbills almost exclusively feed on sponges.

Nassau grouper diet consists of fish, crustaceans, and other species. The Central and Southwest Atlantic DPS of scalloped hammerhead shark and oceanic whitetip sharks are high, trophic level predators and opportunistic feeders with a diet that includes a wide variety of bony fish, octopi, cuttlefish, squid, crabs, lobsters, and rays. While diets of the Nassau grouper, scalloped hammerhead shark, and oceanic whitetip shark could potentially include species harvested in fisheries managed under the Puerto Rico, St. Thomas/St. John, and St. Croix FMPs, these prey species are only a portion of a variety of species eaten by these species, and are a subset of a variety of food sources. The Puerto Rico, St. Thomas/St. John, and St. Croix FMPs manage harvest to ensure that overfishing does not occur and is addressed if it does occur. Given the variety of the diets of Nassau grouper, scalloped hammerhead, and oceanic whitetip shark the species harvested by the proposed actions represent only a portion of what the species might eat, and the fisheries management goals are to prevent overfishing in the fisheries, ensuring the forage or prey species continue to be available, NMFS believes that the fisheries will not impact the availability of forage for these species and no indirect consequences from the fisheries are expected.

Conservative Decisions- Providing the Benefit of the Doubt to the Species
The analysis in this section is based upon the best available commercial and scientific data on sea turtle, fish, and coral biology, and the effects of the proposed action. However, there may be instances where there is limited information upon which to make a determination. In those cases, in keeping with the direction from the U.S. Congress to provide the “benefit of the doubt” to threatened and endangered species [House of Representatives Conference Report No. 697, 96th Congress, Second Session, 12 (1979)], we will generally make determinations that provide the most conservative (conservation oriented) outcome for listed species.

5.1 Stressors

In order to assess the effects of the proposed action, we must first identify the “stressors” or components of the actions that could adversely affect the species that are the subject of this consultation. The proposed actions (fishing activities under three proposed FMPs) would subject the species to the following stressors that could adversely affect them (as applicable and discussed in these sections): 1) capture and/or entanglement in fishing gear or physical impacts from the gear; 2) vessel activity (boat strikes to sea turtles, anchoring to corals and *Acropora* critical habitat, and anchoring lines to sea turtles). Additionally, the harvest of herbivorous reef fish is expected to affect both *Acropora* critical habitat and ESA-listed coral species.

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[32] Under the proposed Puerto Rico and St. Thomas/St. John FMPs, harvest of conch would be prohibited. Harvest of conch would be allowed in federal waters off St. Croix under the St. Croix FMP.

5.2 Exposure

The exposure analyses identify the co-occurrence of ESA-listed species with the actions' stressors (and their effects) in space and time, and identify the nature of that co-occurrence. The analyses identify, as possible, the number, age or life stage, and gender of the individuals likely to be exposed to the actions' effects and the population(s) or subpopulations(s) those individuals represent. Nassau grouper, scalloped hammerhead sharks (Central and Southwest Atlantic DPS), and oceanic whitetip sharks of both genders and multiple age classes could be exposed to stressors associated with the proposed action. Adult, sub-adult, and juvenile green (NA and SA DPSs) and hawksbill sea turtles of both genders could be exposed to the stressors. Hatchlings are not expected to be affected. ESA-listed coral colonies and the essential features of *Acropora* critical habitat could be exposed to stressors from the proposed actions.

In the subsections below, we estimate the number and amount of each species that is likely to be incidentally taken or affected in the future, and the amount of *Acropora* critical habitat likely to be affected. While information relating to coral and *Acropora* critical habitat and the relationship to herbivorous fish and algae exists, it has limitations, is complicated, and is somewhat difficult to interpret and apply to estimate effects of the managed reef fish fisheries, specifically the effect of fishing for herbivorous fish. However, it is addressed to the extent the information allows. Harvest of herbivorous fish does not result in direct take of coral species and all effects relating to dynamics of herbivorous fish, algae, and coral and *Acropora* critical habitat are addressed solely in the Response section. Additionally, we have a limited and likely an incomplete representation of interactions that may be occurring between the species considered in this Opinion and the fisheries; however this information represents the best scientific data available.

The following represent the best available scientific data based on information from the NMFS SEFSC and other sources such as the Puerto Rico Department of Natural and Environmental Resources (DNER) and USVI Department of Planning and Natural Resources. In some instances, the information supplied did not provide actual numbers of animals taken (e.g., information was in pounds). The available information was then converted as necessary to number of animals as shown below. In the event no data were available from the SEFSC, numbers were estimated based on other available information as noted.

Additionally, the new island-based FMPs will not alter fishing practices in a way that would reduce the potential for interactions with ESA-listed species. Therefore, past data is the best available for analysis in this Opinion. Where detailed information on relative effort of federal and territorial fisheries is unavailable, NMFS believes the best assumption is that interactions would be proportional to fishable area in federal and commonwealth or territorial waters. We used data from 2012 forward when possible as it reflects how the fishery is expected to occur when the FMPs are implemented. The year 2012 is the first year that annual catch limits were implemented for federally managed fisheries in the Caribbean, and data following that time period best reflects how the fishery is expected to operate. However this data was not always available, in which case we used the best scientific information available.

5.2.1 Puerto Rico FMP

Sea Turtles- Fishing Gear Interactions-Commercial and Recreational Combined

Our analyses of the fishing gear effects on sea turtles focus on the effects of traps and hook-and-line gear on sea turtles. This is because we determined (e.g., NMFS 2005a) that other fishing methods in the U.S. Caribbean (i.e., by hand and spear[33]) are not likely to adversely affect sea turtles, and there is no new information to indicate otherwise. Commercial and recreational divers (either free diving or SCUBA-assisted) fishing with spears or by hand likely do occasionally encounter sea turtles, primarily over coral reefs. However, anecdotal information from such encounters indicates some sea turtles change their route to avoid coming in close proximity to divers, whereas others appear unaware of the presence of divers. Any behavioral effects on sea turtles from the presence of divers spearfishing or collecting fish by hand are expected to be insignificant. Given the selectivity of the gear and the careful aim divers exercise to strike a fish, divers spearfishing are easily able to avoid aiming in any direction where sea turtles are within their striking range.

Stranding data from for the U.S. Caribbean was provided by the Puerto Rico DNER to assist with the analysis in this Opinion. Actual fishery bycatch data from the Caribbean fisheries were not available for sea turtles. Strandings can be a valuable source of data. Stranding data are often used to monitor sea turtle nearshore mortality rates and sometimes are used as an indicator of the relative distributions and abundances of different species and sizes of sea turtles. They are also sometimes used to provide information on mass mortality events and potential mortality factors, fisheries impacts on sea turtles and other marine species, where mortality may be occurring, and to direct further observations. Likewise, when combined with other data, stranding information can also shed light on how anthropogenic impacts that occur at sea, and are otherwise difficult to study, are affecting aggregations.

Stranding data also have limitations. For example: (1) Not all sick or dead sea turtles strand; thus, sea turtle stranding data represent only a subset of all dead turtles, and the total proportion that strand is unknown. Factors affecting the likelihood of stranding include distance from shore, current and wind direction, bathymetry, marine scavengers, decomposition condition, presence of beaches, and accessibility of coastline. (2) Even if a sea turtle does strand, that does not mean it is necessarily discovered, reported, and documented. Whether or not a stranding is detected depends on the frequency of strandings in an area, frequency of beach monitoring, availability of volunteers to respond to a stranding event, and experience and training of those volunteers. (3) Decreases or increases in stranding numbers may not be due to decreases or increases in mortality rates. For example, mortality rates may remain unchanged but decreases or increases in local sea turtle populations may result in changes in the number of strandings. (4) Stranding information does not indicate where a potential mortality event (e.g., hooking, vessel strike) occurred, as a sea turtle could have been injured or killed at one location and then drifted with wind or currents for a considerable distance before being documented. (5) Last, when sea turtles do strand and are reported as such, often the cause of the stranding is unknown.

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[33] Under the current Spiny Lobster FMP, spears are unlawful for the harvest of spiny lobster (50 CFR 622.452(a)), and this prohibition would be retained in the Puerto Rico FMP.

In the U.S. Caribbean, strandings represent the best available information upon which to estimate potential interactions between fishing gear and sea turtles. We are not confident in our ability to monitor non-lethal effects occurring from these interactions. Thus, our analysis here is conservative and assumes that any interactions will result in mortality. TEWG (1998) estimates sea turtle strandings may represent as little as 5-6% of actual at-sea nearshore-mortality events.

Stranding data from 1985 to 2019[34] indicate that 14 green and 10 hawksbill sea turtles may have interacted with fishing gear (gear was associated with the stranding). To conservatively estimate possible effects to the species, NMFS assumes all these animals stranded as a result of fishing interactions.

Since strandings only represent as little as 5% of actual at sea events, NMFS calculated the likely total strandings as follows:

14/0.05 = 280 possible green sea turtle strandings (all gear, in territorial and federal waters)
10/0.05 = 200 possible hawksbill sea turtle strandings (all gear, in territorial and federal waters)

That means that over 35 years, the annual average:

280/35 = 8 greens are likely taken by fishing gear per year
200/35 = 5.71 hawksbills are likely taken by fishing gear per year

As discussed earlier in this Opinion, fishing activities managed under the Puerto Rico FMP occurs in 7% of the fishable area off of Puerto Rico. Adjusting by fishable area assumes an equal likelihood of interaction across fishable area in territorial and federal waters. In the absence of more specific data, this is reasonable because fishers use similar gears and similar effort across these waters, and turtle abundance is similar in areas where fishing occurs across all fishable habitat. Therefore, the expected take of sea turtles by fishing from the FMP activities must be adjusted as follows:

8 x 0.07 = 0.56, rounded to 1, green sea turtle lethally taken annually by FMP activities
5.71 x 0.07 = 0.40, rounded to 1, hawksbill sea turtle lethally taken annually by FMP activities

Sea Turtles- Vessel Strikes and Anchor Line-Commercial and Recreational Combined

Vessel traffic has the potential to adversely affect listed sea turtle species by direct strikes (i.e., impact with the vessel's hull or running gear), particularly those species that spend a significant amount of time near the surface like sea turtles. While this threat has been analyzed in some

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[34] The data was provided in one summary from PR DNER and NMFS was unable to parse out strandings for just the years 2012 forward, which is when annual catch limits were implemented and which represents how the fishery is expected to operate in future. Therefore, the calculations in this section use the entire data set. PR DNER data was considered the best available data since it includes data through 2019. Using this longer time series, which includes pre-2012 and therefore pre-annual catch limit information, potentially overestimates the interactions, but it is the best available information and it errs on the side of the species. For the USVI calculations, we relied on data from the Southeast Fisheries Science Center as that is the best available information.

coastal areas where there is significant overlap of sea turtles and vessel traffic (e.g., Florida), no analysis has been previously conducted for the action area due to the general lack of stranding data and vessel traffic patterns in the U.S. Caribbean. Therefore, we use Barnette (2018), which analyzed the probability of vessel strikes on sea turtles in coastal Florida waters, as a proxy in this Opinion.

Based on documented stranding data and vessel use patterns for various counties, Barnette (2018) estimated a vessel strike every 4,577-8,500 vessel trips under the most conservative of approaches (i.e., assuming low stranding returns and low number of annual vessel trips) and a vessel strike every 135,501 trips in areas with good stranding data and a high number of annual vessel trips. Because vessel traffic associated with recreational and commercial fishing activities within the action area (i.e., federal waters of the U.S. Caribbean) is likely not well represented by either of these estimate scenarios, we will use a mid-point of 65,642 trips to represent the threshold upon which we expect there may be a sea turtle mortality resulting from a vessel strike. We use this as a threshold and because we do not have data on the exact number of lethal interactions, we assume all strikes will lead to mortalities (to conduct a conservative worst case analysis).

NMFS' MRIP estimated 124,674 recreational fishermen took a total of 653,614 fishing trips in Puerto Rico in 2016, of which 11% (n = 70,196) were in federal waters (MRIP Query August 2020).

The number of commercial fishers in Puerto Rico has fluctuated significantly over the years. Nevertheless, based on the best available information provided by the Puerto Rico Department of Environmental and Natural Resources, there were 1,277 licensed fishers in Puerto Rico (764 full time, 134 part time, and 379 beginner fishers) in 2018 (CFMC 2019a). The number of commercial fishermen that submitted catch reports in 2016[35] was 811, with a total of 29,292 fishing trips reporting during the year. Of those 2016 catch reports, 11% of the total fishing trips were reported in federal waters. Additionally, 10% of the trips taken during the year were reported from an "unknown" location, either because the fishermen were not certain if the fishing location was in Commonwealth or federal waters, or because the location field on the catch report was left blank. It is possible that some of those "unknowns" occurred in federal waters. To err on the side of the species, assuming all of these "unknown" trips occurred in federal waters, we estimate that 6,152 of commercial fishing trips occurred in federal waters in 2016 (21% of 29,292 fishing trips).

In summary, an estimated 76,348 recreational and commercial fishing vessel trips are conducted in federal waters off Puerto Rico (70,196 + 6,152) per year. We assume these trips are associated with fishing for species managed under the Puerto Rico FMP and are part of the proposed action. This number of trips is over the threshold of 65,642vessel trips that we expect would result in a single sea turtle mortality due to vessel strike effects. We do not know which

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[35] We are relying on data from 2016 as this was the last year of complete data before the 2017 hurricane season, which affected fishing effort. By using 2016 information, we are anticipating a potential high effort scenario, if effort returns to the pre-hurricane levels.

species of sea turtle will be affected, therefore, to be conservative, we will assume 1 hawksbill and 1 green sea turtle will be taken annually. These are species expected to interact with the fishery based on the stranding data above.

With regard to anchor lines, NMFS expects that they would be taut (not loose and looping), and that entanglement with sea turtles would be extremely unlikely.

Scalloped Hammerhead Shark- Fishing Gear Interactions

Commercial

No scalloped hammerhead sharks were present in the 2012-2018 commercial landings data for federal waters.

Recreational

No scalloped hammerhead sharks were present in the recreational landings data[36] for federal waters.

Nassau Grouper- Fishing Gear Interactions

Commercial

Nassau grouper were not present in the 2012-2018 commercial landings data for federal waters.

Recreational

Nassau grouper were not present in the recreational landings data[37] for federal waters.

Oceanic Whitetip Shark- Fishing Gear Interactions

Commercial

No oceanic whitetip sharks were present in the 2012-2018 commercial landings data for federal waters.

Recreational

Oceanic white-tip sharks were present in the recreational landings data[38], which are reported in estimated counts of caught[39] and discarded fish. The annual estimate of the number of oceanic whitetip sharks caught in federal waters was 11 animals (65.382 / 6 = 10.897 rounded to 11).

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[36] Recreational landings data collection efforts were suspended in 2017 and have not resumed. Thus the year period used to calculate annual averages for recreational landings was 2012-2017 (n=6).

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[38] Recreational landings data collection efforts were suspended in 2017 and have not resumed. Thus the year period used to calculate annual averages for recreational landings was 2012-2017 (n=6).

[39] Fish caught refers to fish that were kept or killed.

Effects of Gear Deployment on *Acropora* Critical Habitat and Coral (Commercial and Recreational)

As noted in Section 2, we anticipate direct effects to ESA-listed corals and *Acropora* critical habitat from fishing associated with the Puerto Rico FMP will occur in the EEZ off the Puerto Rico, where the fishing occurs. Other effects to ESA-listed corals and *Acropora* critical habitat, including effects associated with the harvest of herbivorous reef fish, can occur throughout Commonwealth and federal waters, as harvested herbivorous fish may have otherwise traveled to Commonwealth waters and provided beneficial algae grazing to ESA-listed corals and *Acropora* critical habitat in those waters. As noted at the beginning of Section 5.2, those effects are discussed in the Response section below.

Fishermen in the Puerto Rico fishery may use the following gears that may affect corals and *Acropora* critical habitat: hook-and-line gear, including handlines and vertical bottom lines;[40] or SCUBA diving methods, including spear fishing for reef fish, and hand and snare collection of spiny lobster; and traps (fishing vessel anchoring will be discussed in the next section). Standard vertical line fishing practices have the potential to impact coral or *Acropora* critical habitat through hooks snagging colonies or the essential feature of *Acropora* critical habitat, consolidated hard substrate and dead coral skeleton, through the landing of weights or other pieces of gear on the benthos, or broken or discarded fishing line entangling coral colonies. However, fishermen in Puerto Rico deploy hook-and-line gear in the water column above the reef, so that the gear is not placed on the coral. Thus, we believe any adverse affects from vertical line fishing on colonies would be extremely unlikely to occur. Additionally, we believe hook-and-line gear used by Puerto Rico fishermen would not affect consolidated hard substrate or dead coral skeleton (the essential feature of *Acropora* critical habitat) in any manner that would appreciably alter the biological or physical characteristics that make them suitable for larval settlement or coral regeneration.

SCUBA-assisted fishing gear (spear guns, snares, or hand collection) is not thought to have impacts to the benthic features of a reef since divers generally do not intentionally contact the reef. Additionally, SCUBA-assisted fishing is highly selective, and the listed corals are not a target species; thus, we believe it is likely that divers would be able to avoid directly taking these species. Additionally, SCUBA divers that do encounter these colonies or dead coral skeletons when fishing for managed species would likely try to avoid them to reduce any chance of fishing gear becoming fouled. Thus, we believe adverse effects from SCUBA fishing on ESA-listed corals or *Acropora* critical habitat extremely unlikely.

The use of traps is known to cause physical damage to benthic habitats when they are set, hauled, lost, or abandoned (Chiappone et al. 2002, Sheridan et al. 2003, Mangi and Roberts 2006). Traps and trap lines can directly affect coral through breakage or abrasion, but traps can also destroy newly settled planulae during setting or hauling. Additionally, any space occupied by a trap temporarily prohibits that area from functioning as habitat because that space has been preempted by the trap making it unavailable for the settlement and growth of corals. The

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[40] Vertical bottom longlines are deployed vertically in the water column and are not considered to be bottom longline gear, which is deployed horizontally along the sea floor.

physical impacts of traps on coral and *Acropora* critical habitat in Puerto Rico may be estimated by analyzing the number of traps, the percentage of those traps reported to be used in coral or hardbottom areas, and the total area of coral cover in Puerto Rico.

This Opinion uses the calculation that uses the most conservative estimate (largest number) of traps (fish traps and lobster traps) used in federal waters off Puerto Rico in order to analyze the greatest possible impact that could occur during fishing activities managed under the Puerto Rico FMP. Those fish trap and lobster trap estimates were then used to estimate impacts to ESA-listed corals and *Acropora* critical habitat as summarized below.

Average fish trap dimensions in Puerto Rico are 4.1 ft (125 cm) by 3.4 ft (103 cm) (Scharer et al. 2004). Average lobster traps are 2.0 ft (60 cm) by 3.9 ft (120 cm) (Scharer et al. 2004). Agar et al. (2017) surveyed commercial fishermen in Puerto Rico between June 2014 and January 2016 and reported that a total 139 commercial fishermen use trap gear in commonwealth and federal waters off Puerto Rico. Of those 139 fishermen, an estimated 131 use fish trap gear and an estimated 36 use lobster trap gear.

Agar et al. (2017) found that commercial trap fishermen in Puerto Rico whether fishing in commonwealth or territorial waters, took an average of 2.3 trips per week, hauling an average of 30.2 fish traps per trip and 17.0 lobster traps per trip. Using these data, the total number of fish traps and lobster traps hauled by Puerto Rico commercial fishermen, fishing in commonwealth or federal waters, per week were estimated by multiplying the number of fishermen using each trap type by the average number of trips taken per week and then by the average number of fish traps or lobster traps hauled per trip. Then, that number was multiplied by 52 to estimate the total number of traps hauled per year (Table 5.1). While it may be unlikely that every fisherman would fish all 52 weeks in a year, this approach was chosen to avoid underestimating the potential impacts to the ESA-listed species and *Acropora* critical habitat.

Table 5.1. Estimated number of commercial fish traps and lobster traps hauled in Puerto Rico (commonwealth and federal waters combined) per year based on the average number of trips per week.

Trap Type	Number of Fishermen	Trips/Week	Traps/Trip	Total Traps Hauled/Week	Total Traps Hauled/Year
Fish	131	2.3	30.2	9,099	473,148
Spiny Lobster	36	2.3	17.0	1,408	73,216

Estimates of the number of recreational fishermen using trap gear in federal waters off Puerto Rico are not available at this time,[41] but are assumed to be zero or near zero. That assumption is

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[41] Gear type is not a parameter collected in the recreational landings data.

based on the fact that since traps are prohibited gear for recreational fishermen in commonwealth waters, it is extremely unlikely that recreational fishermen would use trap gear in federal waters. Therefore, the following discussion assumes trap use is in the commercial fisheries.

Effects of Traps on ESA-Listed Corals

The following sections calculate exposure of each of the ESA-listed coral species to trap gear associated with the proposed action.

Seven percent of fishable habitat in Puerto Rico is in federal waters. Approximately 473,148 fish traps and 73,216 lobster traps are set annually in the fishable habitat off Puerto Rico, in commonwealth and federal waters. Marshak et al. (2008) reported 45% of traps were in coral habitat[42] and Sheridan et al. (2005) found trap-caused damage at about 50% of all traps visited, including traps set in coral areas. Damage was evaluated within 5 meters of where the trap was set, and thus the study accounted for potential damage from trap movement. Thus, 106,458 (473,148 x 0.45 x 0.50) of the fish traps, and 16,474 (73,216 x 0.45 x 0.50) of the lobster traps that were placed and hauled from coral habitat in commonwealth and federal waters caused damage. Traps used in Puerto Rico average 13.94 sq. ft. (fish trap) and 7.80 sq ft. (lobster trap). Applying this information to the traps expected to cause damage, impacts are expected to be no more than 1,484,025 sq. ft. (106,458 x 13.94 sq. ft. fish trap) and 128,497 sq. ft. (16,474 x 7.80 sq. ft. lobster trap). However, only 7% of fishable habitat is in federal waters off Puerto Rico, and therefore we expect that only 7% of the trap fishing, and 7% of the damage from trap use, occurs in federal waters and is related to the proposed action. Therefore 103,882 (1,484,025 x 0.07) sq. ft. damage occurs from fish traps annually, and 8,995 (128,497 x 0.07) sq. ft. damage occurs from lobster traps annually for a total of 112,877 sq. ft. damage annually in federal waters off Puerto Rico. The estimates in this opinion are likely very conservative (over-estimates) because we assume that the whole area of the trap causes total damage to the coral when it is more likely that the corals are scraped, not completely lost. The species average coverage (percent of reef site occupied by living tissue of the species averaged across all reef sites surveyed) must be applied to calculate impacts to each listed species (average cover coral for this and other calculations is based on NOAA, unpublished data[43]).

Elkhorn Coral
percent cover = 0.1726%
112,877 sq. ft. x 0.001726 = 194.8 sq. ft.

Staghorn coral
percent cover = 0.1579%
112,877 sq. ft. x 0.001579 = 178.2 sq. ft.

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[42] Coral habitat includes areas colonized with corals and hardbottom areas that could be colonized by corals on the reef.

[43] These estimates are derived from benthic surveys conducted in the area by multiple groups that NOAA compiled into a database (not published, so not citable). Surveys were conducted differently by each group, but the values are for the percent of the area surveyed that was occupied by the species. For example, along a 10 m x 1 m transect, the percent cover of each species was calculated for the 10 m^2 area. We averaged all of those values to get an overall island-wide estimate of average percent cover of the species on the reef.

Rough cactus coral
percent cover = 0.0102%
112,877 sq. ft. x 0.000102 = 11.5 sq. ft.

Pillar coral
percent cover =0.0248%
112,877 sq. ft. x 0.000248 = 28.0 sq. ft.

Lobed star coral
percent cover =0.1173%
112,877 sq. ft. x 0.001173 = 132.4 sq. ft.

Mountainous star coral
percent cover =0.2738%
112,877 sq. ft. x 0.002738 = 309.1 sq. ft.

Boulder star coral
percent cover = 0.1423%
112,877 sq. ft. x 0.001423 = 160.6 sq. ft.

Effects of Traps on Acropora Critical Habitat

For this analysis, we conservatively calculate the impact on *Acropora* critical habitat from trap use and assume all trap gear set in the coral habitat area and on corals also is set on designated critical habitat. This overstates the impact of gear on species and critical habitat since gear are set on species or critical habitat, not both simultaneously, but is the best way to conservatively estimate potential effects. The essential feature of *Acropora* critical habitat includes substrate of suitable quality and availability, which is defined as consolidated hardbottom or dead coral skeleton that is free from fleshy macroalgae cover and sediment cover. Traps do not cause consolidated hardbottom to become unconsolidated, nor do they cause growth of macroalgae or cause sedimentation. However, traps placed on the consolidated hardbottom could temporarily preclude new settlement of planulae (free-swimming coral larvae), affecting the functionality of the essential feature. In addition, we believe that traps could damage dead coral skeletons. There are currently no data available to determine the number of dead coral skeletons occurring inside critical habitat within the EEZ of the U.S. Caribbean. Therefore, our analysis of trap effects acts conservatively and assumes that the entire area of critical habitat has dead coral skeletons. We use Marshak et al. (2008) that reported 45% of traps were in coral habitat and Sheridan et al. (2005) that indicate that of the traps deployed on various habitat sites, including coral habitat, traps cause damage at 50% of sites, to 13.7% of corals. For this analysis, we assume coral habitat includes coral critical habitat. We therefore assume our coral effects analysis is applicable to designated critical habitat for *Acropora*, and the damage to dead skeletons would be similar to damage that occurs when gear interacts with live branching coral. However, only 7% of fishable habitat is in federal waters off Puerto Rico (as discussed in the description of the proposed action), and we assume only 7% of the trap fishing that causes damage to critical habitat occurs in federal waters. Thus, 1,021 (473,148 x 0.45 x 0.50 x 0.1370 x 0.07) of the fish traps, and 158 (73,216 x 0.45 x 0.50 x 0.1370 x 0.07) of the lobster traps that were placed in and hauled from coral critical habitat in federal waters caused damage. Traps used in Puerto Rico average 13.94 ft^2 (fish trap) and 7.80 ft^2 (lobster trap). Applying this information to the traps expected to cause damage, impacts are expected to be no more than 14,233 ft^2 (1,021 x 13.94 ft^2 fish trap) and 1,232 ft^2 (158 x 7.80 ft^2 lobster trap) annually for a total of 15,465 ft^2 damage to coral critical habitat annually.

Effects of Fishing Vessel Anchors on ESA-Listed Coral and *Acropora* Critical Habitat

Here we examine the number of trips taken by fishing vessels, apply factors for the percentage of those vessels deploying anchors in areas with coral and *Acropora* critical habitat and the percentage of those anchors that do damage, and then analyze those findings based on an anchor footprint appropriate for the average vessel used in the Puerto Rico fishery.

Using the information we estimated above when analyzing the effect to sea turtles from vessel strikes and anchoring line from commercial and recreational fishing trips, combined, we assume 76,348 recreational and commercial trips occur per year in federal waters (70,196 recreational vessel trips and 6,152 commercial vessel trips). This number is an over-estimation of the number of anchors used since the majority of commercial fishermen operating in the managed fisheries in federal waters off Puerto Rico (>95%) do not use anchors during fishing activities (M. Hanke, Caribbean Fishery Management Council, pers. comm. to S. Stephenson, NMFS SERO, March 31, 2020). Additionally, most recreational fisheries in Puerto Rico do not require anchoring (Garcia-Moliner et al. 2002). NMFS assumes that 50% of the trips use anchors (this errs on the side of caution, considering "most" is 50% or more, but does not use the 95% figure, which would exclude almost all trips). Therefore, 76,348 x 0.50 =38,174 commercial and recreational trips occur annually that set anchors.

Effects on ESA-listed corals

Rogers et al. (1988) conducted a survey of boats in 1987 that revealed 14% of boats that anchored were anchored in coral communities and that about 40% of the anchors on these vessels caused damage. This study pertained to all boats observed in a particular area off of St. John, not just fishing vessels. In the absence of more specific data, it is a reasonable proxy for the potential number of fishing vessels anchored in coral communities in federal waters off Puerto Rico that cause damage. Therefore, if we multiply the total number of trips with anchoring (1 anchoring event per trip) taken in federal waters each year (38,174) by the percent anchored in coral communities (i.e., 14%), we arrive at an estimate of 5,345 anchors deployed in coral communities (38,174 x 0.14 = 5,344.36). If we then apply the percentage of anchors that cause damage to corals (i.e., 40%) to the estimated number of anchors deployed in coral communities, we arrive at an estimate of 2,138 anchors deployed each year that cause direct damage to corals (5,345 x 0.40 = 2,138). We then assume most 16- to 27-ft boats use a 4-lb. aluminum anchor with measurements of 24 in x 19 in (3.16 ft^2). Multiplying the anchor area by the estimated number of anchors deployed each year that damage to corals yields an estimate of the total area of coral damaged by anchors each year in federal waters (2,138 x 3.16 ft^2 = 6,756.08 ft^2). In assuming that the area of impact is limited to the size of the anchor, we are not assuming additional effects from anchor dragging. Anchor dragging could cause additional damage to reefs. However, fishing vessels operating in the managed fisheries in federal waters are relatively small, often with no place to take shelter from adverse weather. This makes it less likely that fishermen would choose to fish during weather or sea states that are most conducive to causing dragging. Thus, we believe anchor dragging is unlikely to be an issue of concern.

That estimated area of coral damaged by fishing vessel anchors can then be multiplied by the percent coverage of each ESA-listed corals species in Puerto Rico federal waters to calculate the impact of anchoring to each listed coral species.

Staghorn coral: 6,757 ft^2 per year x 0.1579% = 10.7 ft^2
Elkhorn coral: 6,757 ft^2 per year x 0.1726% = 11.7 ft^2
Rough cactus coral: 6,757 ft^2 per year x 0.0102% = 0.7 ft^2
Pillar coral: 6,757 ft^2 per year x 0.0248% = 1.7 ft^2
Lobed star coral: 6,757 ft^2 per year x 0.1173% = 7.9 ft^2
Mountainous star coral: 6,757 ft^2 per year x 0.2738% = 18.5 ft^2
Boulder star coral: 6,757 ft^2 per year x 0.1423% = 9.6 ft^2

Effects on Acropora Critical Habitat

It is necessary to consider the effects of anchoring on *Acropora* critical habitat by fishing vessels. NMFS uses the calculation of number of square feet per year of anchoring (6,757 ft^2) in coral communities just calculated for species (which incorporates setting on coral and how much is damaged) and applies it to critical habitat. Based on Rogers et al. (1988), we assume 14% of vessels that anchor do so in coral communities. We assume that the critical habitat is spread throughout that potential anchoring area, and that vessels are not anchored in critical habitat in any other area (i.e., the 14% figure covers anchoring in coral areas, including all *Acropora* critical habitat). Therefore, we assume that anchoring damaged 6,757 ft^2 of *Acropora* critical habitat annually.

Anchor dragging could cause additional damage to critical habitat. However, fishing vessels operating in the managed fisheries in federal waters are relatively small, often with no place to take shelter from adverse weather. This makes it less likely that fishermen would choose to fish during weather or sea states that are most conducive to causing dragging. Thus, we believe anchor dragging is unlikely to be an issue of concern.

5.2.2 St. Thomas/St. John FMP

Sea Turtles- Fishing Gear Interactions-Commercial and Recreational Combined

As discussed and explained in Section 5.2.1, our analyses of the fishing gear effects on sea turtles focus on the effects of traps and hook-and-line gear on sea turtles. This is because we determined that other fishing methods in the U.S. Caribbean (i.e., by hand and spear[44]) are not likely to adversely affect sea turtles, and there is no new information to indicate otherwise.

Stranding data from for the U.S. Caribbean was provided by the SEFSC to assist with the analysis of effects to species from the St. Thomas/St. John FMP activities. Actual fishery bycatch data from the Caribbean fisheries was not available for sea turtles. In the U.S. Caribbean, strandings represent the best available information upon which to estimate potential interactions between fishing vessels and sea turtles. We are not confident in our ability to monitor non-lethal effects occurring from these interactions. Thus, our analysis here is conservative and we assume that any interactions will result in mortality. Strandings can be a valuable source of data with the limitations discussed in the previous section 5.2.1, including but not limited to the fact that sea turtle strandings may represent as little as 5-6% of actual at-sea nearshore-mortality events.

SEFSC data from 2012 to 2017 (as noted at the outset, this data reflects the way the fishery currently fishes, and is the best available information), indicate that 2 green, 0 hawksbill, 0 leatherback sea turtles may have interacted with fishing gear (gear was associated with the stranding) in territorial and federal waters off St. Thomas/St. John. To conservatively estimate possible effects to the species, NMFS assumes all these animals stranded as a result of fishing interactions. Since strandings only represent as little as 5% of actual at sea events, NMFS calculated the likely total strandings as follows:

2/0.05 = 40 possible green sea turtle strandings (all gear, in territorial and federal waters)
0/0.05 = 0 possible hawksbill sea turtle strandings (all gear, in territorial and federal waters)
0/0.05 = 0 possible leatherback sea turtle strandings (all gear, in territorial and federal waters)

That means that over 6 years, the annual average:

40/6 =6.67 greens are likely taken by fishing gear per year
0/30 = 0 hawksbills are likely taken by fishing gear per year
0/30 = 0 leatherbacks are likely taken by fishing gear per year

As discussed earlier in this Opinion, fishing activities managed under the St. Thomas/St. John FMP occurs in 41% of the fishable area off of these islands. Adjusting by fishable area assumes an equal likelihood of interaction across fishable area in territorial and federal waters. In the absence of more specific data, this is reasonable because fishers use similar gears and similar

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[44] Under the current Spiny Lobster FMP, spears are unlawful for the harvest of spiny lobster (50 CFR 622.452(a)), and this prohibition would be retained in the St. Thomas/St. John FMP.

effort across these waters, and turtle abundance is similar in areas where fishing occurs across all fishable habitat. Therefore, the expected take of sea turtles by fishing from the FMP activities must be adjusted as follows:

6.67 x 0.41 = 2.73, rounded to 3, green sea turtles taken annually by FMP activities
0 hawksbill sea turtles taken annually by FMP activities
0 leatherback sea turtles taken annually by FMP activities

Sea Turtles- Vessel Strikes and Anchor Line-Commercial and Recreational Combined

Of the total fishable habitat off St. Thomas and St. John, only 219 mi^2 (41% of total fishable habitat) is found in the EEZ. The most recent census of licensed fishers in the USVI (Kojis et al. 2017) reported a total of 113 commercial fishermen on St. Thomas and six on St. John for a total of 119. Of those 119 fishermen, 4.6% said they fished primarily in federal waters and 42.5% said they fished equally in federal and territorial waters. The number of commercial fishermen submitting catch reports in 2016 was 65, with a total of 2,482 trips reported during the year. Of those 2016 catch reports, 46% of the total fishing trips were reported in federal waters. Additionally, 7% of the trips taken during the year were reported from an “unknown” location, either because the fishermen were not certain if the fishing location was in territorial or federal waters, or because the location field on the catch report was left blank. It is possible that some of those “unknowns” occurred in federal waters. To err on the side of the species, assuming all of these “unknown” trips occurred in federal waters, we estimate a total of 1,316 fishing trips in federal waters (53% of 2,482).

As noted by Crosson (2018), the MRIP has not been implemented in the USVI. Therefore, long-term recreational landings are generally not available. The actual number of recreational fishermen in St. Thomas and St. John is also unknown. Van Beukering et al. (2011) surveyed recreational fishermen in the USVI and found that 20% of the households on St. Thomas and 10% of the households on St. John are involved in recreational fishing. We assume all of these individuals fished in both territorial and federal waters. For St. Thomas and St. John, based on the most recent U.S. census information, the estimate of recreational fishers would be 10,744, which is 20% of the 51,634 population of St. Thomas (10,327) and 10% of the 4,170 population of St. John (417) (U.S. Census Bureau).

Due to the aforementioned lack of data on the recreational fishery, we will use the known relationship of recreational fishers and recreational fishing trips in Puerto Rico to get an estimate of recreational fishing trips in St. Thomas and St. John, assuming a similar relationship. MRIP data estimates an average of 124,674 recreational fishers in Puerto Rico for the period 2016. In Puerto Rico, 653,614 annual recreational trips via charter/private/rental boat were conducted on average during 2016. Therefore, this yields an average estimate of 5.24 annual fishing trips per recreational fisher in Puerto Rico (653,614 trips per year / 124,674 recreational fishers). Using this estimate, we calculate 56,299 total annual recreational fishing trips via boat in St. Thomas and St. John (10,744 recreational fishers * 5.24 trips/year = 56,299 recreational trips/year). Extrapolating that out to consider vessel activity in federal waters yields an estimated average of 23,083 annual recreational fishing trips in federal waters off St. Thomas and St. John (56,299 * 0.41 = 23,083).

In summary, we estimate a total of 24,399 annual commercial and recreational fishing trips within the federal waters off St. Thomas and St. John (1,316 commercial + 23,083 recreational). We assume these trips are associated with fishing for species managed under the St. Thomas/St. John FMP and are associated with the proposed action. This number of trips falls under the hreshold of 65,642 vessel trips that we expect would result in a single sea turtle mortality due to vessel strike effects. As a result, we believe that the risk of sea turtles being struck by a vessel associated with the action covered under this Opinion is extremely low.

With regard to anchor lines, NMFS expects that they would be taut (not loose and looping), and that entanglement with sea turtles would be extremely unlikely.

Scalloped Hammerhead Shark – Fishing Gear Interactions

Commercial

No scalloped hammerhead sharks were present in the 2012-2018 commercial landings data for federal waters.

Recreational

Recreational data are not collected in the U.S. Virgin Islands. Data is collected for Puerto Rico and did not record this species in federal waters. Puerto Rico provides the closest proxy available to estimate what take may occur in St. Thomas/St. John. The areas are close together and the federally-managed fisheries use similar gears. Relying on information from Puerto Rico is potentially an over-estimate as effort in the Puerto Rico recreational fishery is likely greater. Because there were no interactions with the recreational fishery in Puerto Rico federal waters, NMFS would not expect this species to be caught in St. Thomas/St. John federal waters.

Nassau Grouper- Fishing Gear Interactions

Commercial

The only ESA-listed species with reported commercial landings in St. Thomas/St. John from 2012-2018 from federal waters was Nassau grouper. A total of 220 lbs (23 animals) were reported caught and discarded during the 7-year period. Thirty pounds were not listed as discarded, but we assume that these species were discarded and this is a reporting error because harvest and possession of this species is prohibited in both federal and territorial waters, and commercial fishers are aware of and very likely comply with this prohibition. No number (animals) was reported for the non-discarded animals. However, the information from the discarded animals can help estimate the number of animals not discarded. Based on the discard information, Nassau grouper interacting with the fishery weigh approximately 9.57 lbs (220 lbs/23 animals). Of the 30 lbs that were not listed as discarded, using the estimated weight per Nassau grouper would result in approximately 3.14 animals not discarded (30 lbs/9.57 lbs per animal). This provides an estimate of 27 (23 + 3.14 = 26.14) Nassau grouper caught by the St. Thomas/St. John fishery commercial sector over the 7 year period in federal waters, which is approximately 4 per year (27/7 = 3.86). All Nassau grouper were caught via trap gear.

Recreational

Recreational data are not collected in the U.S. Virgin Islands. Data is collected for Puerto Rico and did not record this species. Puerto Rico provides the closest proxy available to estimate what take may occur in St. Thomas/St. John. The areas are close together and the federally-managed fisheries use similar gears. Relying on information from Puerto Rico is potentially an over-estimate as effort in the Puerto Rico recreational fishery is likely greater. Because there were no interactions with the recreational fishery in Puerto Rico federal waters, NMFS would not expect this species to be caught in St. Thomas/St. John federal waters.

Oceanic Whitetip- Fishing Gear Interactions

Commercial

No oceanic whitetip sharks were present in the 2012-2018 commercial landings data for federal waters.

Recreational

Recreational data are not collected in the U.S. Virgin Islands. NMFS decided to estimate takes for the U.S. Virgin Islands based on data from Puerto Rico. Puerto Rico provides the closest proxy available to estimate what take might occur in St. Thomas/St. John, after adjusting for fishable area (relative to Puerto Rico). The areas are close together and the federally-managed fisheries use similar gears. Relying on information from Puerto Rico is potentially an over-estimate as effort in the Puerto Rico recreational fishery is likely greater.

Therefore, if 11 oceanic whitetip sharks are taken by the Puerto Rico FMP recreational fishery activities annually (number discussed in the Puerto Rico section), then given fishable habitat area in federal waters:

(area St. Thomas and St. John/area of Puerto Rico) x (recreational catch Puerto Rico) = recreational catch St. Thomas and St. John

(567 sq. km EEZ/410 sq km EEZ) x 11 = 15.2, rounded to 16, oceanic whitetip sharks taken by recreational fishing activities associated with the St. Thomas/St. John FMP annually.

Effects of Gear Deployment on Acropora Critical Habitat and Coral

As noted in Section 2, we anticipate effects from the proposed action may occur in both Territorial and federal waters, but we anticipate effects of the St. Thomas/St. John FMP are more likely to occur in the EEZ off the St. Thomas/St. John action area. As noted in Section 2, we anticipate direct effects to ESA-listed corals and *Acropora* critical habitat from fishing associated with the St. Thomas/St. John FMP will occur in the EEZ off the St. Thomas/St. John, where the fishing occurs. Other effects to ESA-listed corals and *Acropora* critical habitat, including effects associated with the harvest of herbivorous reef fish, can occur throughout territorial and federal waters, as harvested herbivorous fish may have otherwise traveled to territorial waters and provided beneficial algae grazing to ESA-listed corals and *Acropora* critical habitat in those waters. As noted at the beginning of Section 5.2, those effects are discussed in the Response section below.

Fishermen in the St. Thomas/St. John fishery use the following gears that may affect corals and *Acropora* critical habitat: trap gear; hook-and-line gear, including handlines and rod and reel; and SCUBA diving methods, including spear fishing for reef fish and snares and hand collection for spiny lobster. Standard hook-and-line fishing practices have the potential to impact ESA-listed coral or the essential feature of *Acropora* critical habitat, consolidated substrate or dead coral skeleton, through hooks snagging colonies or the hardbottom or dead skeleton, through the landing of weights or other pieces of gear on the benthos, or through broken or discarded fishing line entangling coral colonies. However, fishermen in St. Thomas/St. John deploy hook-and-line gear in the water column above the reef, so that the gear is not placed on the coral. Thus, we believe any adverse effects from vertical line fishing on colonies would be extremely unlikely to occur. Additionally, we believe hook-and-line gear used by fishermen in St. Thomas/St. John would not affect consolidated hard substrate or dead coral skeleton (the essential feature of *Acropora* critical habitat) in any manner that would appreciably alter the biological or physical characteristics that make them suitable for larval settlement or coral regeneration.

SCUBA-assisted fishing gear (spears, snares, or hand collection) is not thought to have impacts to the reef since divers generally do not intentionally contact the reef. Additionally, SCUBA-assisted fishing is highly selective, and the listed corals are not a target species; thus, we believe it is likely that divers would be able to avoid directly taking these species. Additionally, SCUBA divers that do encounter these coral or dead coral skeletons when fishing for managed species would likely try to avoid them to reduce any chance of fishing gear becoming fouled. Thus, we believe adverse effects from SCUBA fishing on ESA-listed corals or *Acropora* critical habitat extremely unlikely.

The use of traps is known to cause physical damage to benthic habitats when they are set, hauled, lost, or abandoned (Chiappone et al. 2002, Sheridan et al. 2003, Mangi and Roberts 2006). Traps and/or trap lines can directly affect coral through breakage or abrasion, but traps can also destroy newly settled planulae during setting or hauling. Additionally, any space occupied by a trap temporarily prohibits that area from functioning as habitat because that space has been preempted by the trap making it unavailable for the settlement and growth of corals. The physical impacts of traps on coral and *Acropora* critical habitat in St. Thomas/St. John may be estimated by analyzing the number of traps, the percentage of those traps reported to be used in coral or hardbottom areas, and the total area of coral cover in St. Thomas/St. John.

This Opinion uses the analysis that provides the most conservative estimate (largest number) of traps (fish traps and lobster traps) used in federal waters off St. Thomas/St. John in order to analyze the greatest possible impact that could occur during fishing activities managed under the St. Thomas/St. John FMP. Those fish trap and lobster trap estimates are then used to estimate impacts to ESA-listed corals and *Acropora* critical habitat as summarized below.

Average fish trap dimensions in St. Thomas/St. John are 4 ft (122 cm), by 4 ft (122 cm), by 18 inches (in) (46 cm) (Sheridan et al. 2006). Average wire lobster trap dimensions are 3 ft (91 cm), by 2 ft (61 cm), by 2 ft (61 cm) and average plastic lobster trap dimensions are 2.5 ft (76.2 cm) by 1.8 ft (54.9 cm) by 1.5 ft (45.7 cm) (Sheridan et al. 2006).

Kojis et al. (2017) conducted a census of licensed commercial fishermen in the USVI in 2016. The census reported a total of 119 commercial fishermen on St. Thomas/St. John, fishing in territorial and federal waters. Of those 119 fishermen, an estimated 48 use fish traps, 15 use wire lobster traps, and 13 use plastic lobster traps. Kojis et al. (2017) reported the average number of days that the commercial fishermen soak their traps in territorial or federal waters off St. Thomas/St. John: 7 days for fish traps, 10.6 days for wire lobster traps, and 13.6 days for plastic lobster traps. Kojis et al. (2017) also reported the average number of traps hauled per trip across federal and territorial waters: 50.2 for fish traps, 16.0 for wire lobster traps, and 47.3 for plastic lobster traps. The average soak time was used to calculate the average number of trips per month per fisherman (trips per month = 30/average soak time), which was then multiplied by the estimated number of fishermen and by the average number of traps hauled per trip. That number was then multiplied by 12 to get the estimated number of fish traps and lobster traps hauled each year in federal and territorial waters off St. Thomas/St. John (Table 5.2). While it may be unlikely that every fisherman would fish all 12 months in a year, this approach was chosen to avoid underestimating the potential impacts to the ESA-listed coral species and *Acropora* critical habitat.

Table 5.2. Estimated number of commercial fish traps and lobster traps hauled in St. Thomas/St. John (territorial and federal waters combined) per year based on the average soak time (days).

Trap Type	Soak Time (days)	Trips/ Month	Number of Fishermen	Traps/ Trip	Total Traps Hauled/Month	Total Traps Hauled/Year
Fish	7	4.3	48	50.2	10,361	124,332
Lobster, wire	10.6	2.8	15	16.0	672	8,064
Lobster, plastic	13.6	2.2	13	47.3	1,353	16,236

Estimates of the number of recreational fishermen using trap gear in federal waters off St. Thomas/St. John are not available at this time, but are assumed to be zero or near zero. That assumption is based on the fact that since traps are prohibited gear for recreational fishermen in territorial waters, it is extremely unlikely that recreational fishermen would use trap gear in federal waters. Therefore, the following discussion assumes trap use is in the commercial fisheries.

Effects of Traps on ESA-listed Corals

The following sections calculate exposure of each of the ESA-listed coral species to trap gear associated with the proposed action.

In a survey of the distribution of trap locations, Sheridan et al. (2005) found 54% of all surveyed traps to be within colonized hard bottom dominated by soft coral or reef. Sheridan et al. (2005) also found that about 50% of all traps cause damage to bottom habitats, including coral areas, with instances of damage (scrapes, breakage) most prevalent among gorgonians and sponges, followed by corals. Damage was evaluated within 5 meters of where the trap was set, and thus the study accounted for potential damage from trap movement.

From the above estimates, approximately 124,332 fish traps, 8,064 wire lobster traps, and 16,236 plastic lobster traps are set annually in fishable habitat in St. Thomas/St. John, in territorial and federal waters combined.

Traps used in St. Thomas/St. John have an average area of 16 ft^2 (fish trap), 6 ft^2 (wire lobster trap), and 4.5 ft^2 (plastic lobster trap). However, only 41% of fishable habitat is located in federal waters off St. Thomas/St. John, and therefore we expect that only 41% of the trap fishing, and 41% of the damage from trap use, occurs in federal waters and is related to the proposed action. We apply all of this information to calculate damage to corals from trap gear (Table 5.3). The total estimated amount of damage to corals (from fish traps + wire lobster traps + plastic lobster traps) in federal waters off St. Thomas/St. John would be 233,661 ft^2 (note that this final number represents calculations based on Table 5.3 with one rounding applied on the sum).

Table 5.3

Trap type	Total traps	Traps in coral (total traps x .54)	Traps with damage (traps in coral x 0.50)	Area damaged (traps with damage x area of trap)	Federal area (area damaged x 0.41)
fish	124,332	67,139.280	33,569.640	537,114.240	220,216.838
wire lobster	8,064	4,354.560	2,177.280	13,063.680	5,356.109
plastic lobster	16,236	8,767.440	4,383.720	19,726.740	8,087.963
				sum	**233,661**

The estimates in this opinion are likely very conservative (over-estimates) because we assume that the whole area of the trap causes total damage to the coral when it is more likely that the corals are scraped, not completely lost.

The average coverage for each ESA-listed coral species must be applied to this total damage estimate in order to calculate impacts to each ESA-listed species. The average coverage for the seven ESA-listed species was determined from NMFS unpublished data[45]. Note- NMFS used the largest percent coverage data available for areas surveyed (where multiple numbers were available) to conservatively estimate effects.

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[45] These estimates are derived from benthic surveys conducted in the area by multiple groups that NOAA compiled into a database (not published, so not citable). Surveys were conducted differently by each group, but the values are for the percent of the area surveyed that was occupied by the species. For example, along a 10 m x 1 m

Elkhorn coral

percent cover =0.0164%
233,661 ft^2 x 0.000164 = 38.3 ft^2

Staghorn coral

percent cover =0.1133%
233,661 ft^2 x 0.001133 = 264.7 ft^2

Rough cactus coral

percent cover =0.0098%
233,661 ft^2 x 0.000098 = 22.9ft^2

Pillar Coral

percent cover =0.0520%
233,661 ft^2 x 0.00052 = 121.5 ft^2

Lobed star coral

percent cover =4.6353%
233,661 ft^2 x 0.046353= 10,830.9 ft^2

Mountainous star coral

percent cover =3.7885%
233,661 ft^2 x 0.037885 = 8,852.2 ft^2

Boulder star coral

percent cover =7.375%
233,661 ft^2 x 0. = 17,232.5 ft^2

Effects of Traps on Acropora Critical Habitat

For this analysis, we conservatively calculate the impact on *Acropora* critical habitat from trap use and assume all trap gear set in the colonized coral hard bottom and on corals also is set on designated critical habitat. This overstates the impact of gear on species and critical habitat since gear are set on species or critical habitat, not both simultaneously, but is the best way to conservatively estimate potential effects. The essential feature of *Acropora* critical habitat includes substrate of suitable quality and availability, which is defined as consolidated hardbottom or dead coral skeleton that is free from fleshy macroalgae cover and sediment cover. Since traps do not cause consolidated hardbottom to become unconsolidated, nor do they cause growth of macroalgae or cause sedimentation, we believe it is unlikely that traps would affect this portion of the essential feature. However, traps placed on the consolidated hardbottom could temporarily preclude new settlement of planulae (free-swimming coral larvae), affecting the functionality of the essential feature. In addition, we believe that traps could damage dead coral skeletons. There are currently no data available to determine the number of dead coral skeletons occurring inside critical habitat within the EEZ of the U.S. Caribbean. Therefore, our analysis of trap effects acts conservatively and assumes that the entire area of critical habitat does have dead coral skeletons. Sheridan et al. (2005) indicate 54% of traps were in colonized hard bottom, and that the traps deployed at various habitat sites, including coral areas cause damage at 50% sites, to 13.7% of corals. However, only 41% of fishable habitat is located in federal waters off St. Thomas/St. John, and we assume only 41% of the trap fishing that causes damage to critical habitat occurs in federal waters. For this analysis, we assume our analysis of the effects to live corals is analogous and applicable to effects to coral skeletons (and the essential feature for *Acropora* critical habitat), and the damage to coral skeletons would be similar to damage that occurs when gear interacts with live branching coral. Thus, we apply the analysis just used in the previous section. Therefore, the total estimated amount of damage to coral skeletons from fish traps, wire lobster traps, and plastic lobster traps in federal waters off St. Thomas/St. John would be 32,012 ft^2 (233,661 ft^2 of coral area damaged x 0.137 of corals, and therefore dead coral skeletons, damaged). Therefore, we anticipate direct effects from fishing occurring in the EEZ will occur to 32,012 ft^2 of *Acropora* critical habitat annually.

Effects of Fishing Vessel Anchors on ESA-Listed Coral and *Acropora* Critical Habitat

Effects on ESA-listed corals

Here we examine the number of trips taken by fishing vessels, apply factors for the percentage of those vessels deploying anchors in areas with coral and the percentage of those anchors that do damage, and then analyze those findings based on an anchor footprint appropriate for the average vessel used in the fisheries.

Per our analysis above, an estimated 24,399 recreational and commercial trips occur per year (23,083 recreational vessel trips and 1,316 commercial vessel trips) in federal waters of St. Thomas/St. John. This number is an over-estimation of the number of anchors used since the majority of commercial fishermen (more than 95%) operating in federal waters off St. Thomas/St. John do not use anchors during fishing activities (C. Farchette, Caribbean Fishery Management Council, pers. comm. to S. Stephenson, March 31, 2020). Additionally, more than 50% of surveyed recreational fishermen participate in inshore or offshore trolling methods (Kojis and Tobias 2016), and thus do not use anchors. NMFS assumes that 50% of the trips use anchors (this errs on the side of caution, considering "most" is 50% or more, but does not use the 95% figure which would exclude almost all trips). Therefore, 24,399 x 0.50 = 12,200 commercial and recreational trips occur annually that set anchors.

Rogers et al. (1988) conducted a survey of boats in 1987 that revealed 14% of boats that anchored were anchored in coral communities and that about 40% of the anchors on these vessels caused damage. This study pertained to all boats observed in a particular area off of St. John, not just fishing vessels. In the absence of more specific data, it is a reasonable proxy for the potential number of fishing vessels anchored in coral communities in federal waters off St Thomas/St. John that cause damage. Therefore, if we multiply the total number of fishing trips (1 anchoring event) with anchoring taken in federal waters (12,200) by the percent anchored in coral communities (i.e., 14%), we arrive at a number of anchors in coral communities (1,708). If we then apply the percentage of anchors causing damage (i.e., 40%) to the estimate for the number of anchors in coral habitat (1,708), the result is our estimate for the total number of trips with anchors interacting with corals (684). We then assume most 16- to 27-ft boats use of a 4-lb. aluminum anchor with measurements of 24 in x 19 in (3.16 ft^2). Multiplying the area of each anchor by the total number of anchor interactions yielded an estimate of the total area impacted by anchoring events of 2,162 ft^2 (684 x 3.16 = 2,162 ft^2 per year). In assuming that the area of impact is limited to the size of the anchor, we are not assuming additional effects from anchor dragging. Anchor dragging could cause additional damage to reefs. However, fishing vessels operating in the managed fisheries in federal waters are relatively small, often with no place to take shelter from adverse weather. This makes it less likely that fishermen would choose to fish during weather or sea states that are most conducive to causing dragging. Thus, we believe anchor dragging is unlikely to be an issue of concern.

Next it is necessary to calculate the impact to each listed coral species using the percent of cover of each in federal waters off St. Thomas/St. John.

Staghorn coral	2,162 ft^2 per year x 0.1133% = 2.4 ft^2
Elkhorn coral	2,162 ft^2 per year x 0.0164% = 0.4 ft^2
Rough cactus coral	2,162 ft^2 per year x 0.0098% = 0.2 ft^2
Pillar Coral	2,162 ft^2 per year x 0.0520 % = 1.1 ft^2
Lobed star coral	2,162 ft^2 per year x 4.6353% = 100.2 ft^2
Mountainous star coral	2,162 ft^2 per year x 3.7885% = 81.9 ft
Boulder star coral	2,162 ft^2 per year x 7.375% = 159.4 ft^2

Effects on Acropora Critical Habitat

It is necessary to consider the effects of anchoring on *Acropora* critical habitat. NMFS uses the calculation of number of square feet per year of coral damaged by anchoring and applies it to critical habitat (2,162 ft^2 per year). Based on Rogers et al. (1988), we assume 14% of vessels that anchor do so in coral communities. We assume that the critical habitat is spread throughout that potential anchoring area, and that vessels are not anchored in critical habitat in any other area (i.e., the 14% figure covers anchoring in coral areas, including all *Acropora* critical habitat). Therefore, we assume that anchoring damaged 2,162 ft^2 of *Acropora* critical habitat annually.

Anchor dragging could cause additional damage to critical habitat. However, fishing vessels operating in the managed fisheries in federal waters are relatively small, often with no place to take shelter from adverse weather. This makes it less likely that fishermen would choose to fish during weather or sea states that are most conducive to causing dragging. Thus, we believe anchor dragging is unlikely to be an issue of concern.

5.2.3 St. Croix FMP

Sea Turtles- Fishing Gear Interactions-Commercial and Recreational Combined

As discussed and explained in Section 5.2.1, our analyses of the fishing gear effects on sea turtles focus on the effects of traps and hook-and-line gear on sea turtles. This is because we determined that other fishing methods in the U.S. Caribbean (i.e., by hand and spear[46]) are not likely to adversely affect sea turtles, and there is no new information to indicate otherwise.

Stranding data from for the U.S. Caribbean was provided by the SEFSC to assist with the analysis of effects to species from the St. Croix FMP activities. Actual fishery bycatch data from the Caribbean fisheries was not available for sea turtles. In the U.S. Caribbean, strandings represent the best available information upon which to estimate potential interactions between fishing vessels and sea turtles. We are not confident in our ability to monitor non-lethal effects occurring from these interactions. Thus, our analysis here is conservative and assumes that any interactions will result in mortality. Strandings can be a valuable source of data with the limitations discussed in the previous section 5.2.1, including but not limited to the fact that sea turtle strandings may represent as little as 5-6% of actual at-sea nearshore-mortality events.

SEFSC data from 2012 to 2017 (as noted at the outset, this data reflects the way the fishery currently fishes, and is the best available information) indicate that 1 green, 1 hawksbill, and 0 leatherback sea turtles may have interacted with fishing gear (gear was associated with the stranding) in territorial and federal waters off St. Croix. To conservatively estimate possible effects to the species, NMFS assumes all these animals stranded as a result of fishing interactions. Since strandings only represent as little as 5% of actual at sea events, NMFS calculated the likely total strandings as follows:

1/.05 = 20 possible green sea turtle strandings (all gear, in territorial and federal waters)
1/.05 = 20 possible hawksbill sea turtle strandings (all gear, in territorial and federal waters)
0 possible leatherback sea turtle strandings (all gear, in territorial and federal waters)

That means that over 6 years, the annual average:

20/6 = 3.33 greens are likely taken by fishing gear per year
20/6 = 3.33 hawksbills are likely taken by fishing gear per year
0 leatherbacks are likely taken by fishing gear per year

As discussed earlier in this Opinion, under the St. Croix FMP fishing only occurs in 18% of the fishable area off of St. Croix. Adjusting by fishable area assumes an equal likelihood of interaction across fishable area in territorial and federal waters. In the absence of more specific data, this is reasonable because fishers use similar gears and similar effort across these waters, and turtle abundance is similar in areas where fishing occurs across all fishable habitat.

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[46] Under the current Spiny Lobster FMP, spears are unlawful for the harvest of spiny lobster (50 CFR 622.452(a)), and this prohibition would be retained in the Puerto Rico FMP.

Therefore, the expected take of sea turtles by fishing from the FMP activities must be adjusted as follows:

3.33 x 0.18 = 0.60, rounded to 1, green sea turtles taken annually by FMP activities
3.33 x 0.18 = 0.60, rounded to 1, hawksbill sea turtles taken annually by FMP activities
0 leatherback sea turtles taken annually by FMP activities

Sea Turtles- Vessel Strikes and Anchor Line-Commercial and Recreational Combined

The most recent fisher census survey (Kojis et al. 2017) reported 141 commercial fishermen living in St. Croix. Of those 141 fishermen, 14.6% said they fished primarily in federal waters and 26.4% said the fished equally in federal and territorial waters. The number of commercial fishermen submitting catch reports in 2016 was 74, with a total of 2,489 trips reported during the year. Of those 2016 catch reports, 36% of the total fishing trips were reported in federal waters. Additionally, 10% of the trips taken during the year were reported from an “unknown” location, either because the fishermen were not certain if the fishing location was in territorial or federal waters, or because the location field on the catch report was left blank. It is possible that some of those “unknowns” occurred in federal waters. To err on the side of the species, assuming all of these “unknown” trips occurred in federal waters, we estimate that 1,145 of commercial fishing trips occurred in federal waters in 2016 (46% of 2,489 fishing trips).

As with St. Thomas and St. John, the number of recreational fishers on St. Croix has not been quantified (Goedeke et al. 2016). Van Beukering et al. (2011) surveyed recreational fishermen in the USVI and found that 27% of the households on St. Croix are involved in recreational fishing. We assume all of these individuals fished in both territorial and federal waters. Based on the most recent U.S. census information, the estimate of recreational fishers would be 13,663, which is 27% of the population of St. Croix (50,601) (U.S. Census Bureau).

Employing the same approach to calculate total recreational trips as with St. Thomas and St. John (assuming 5.24 trips per recreational fisher per year based on available information from Puerto Rico), we calculate a total annual estimate of 71,595 recreational fishing trips in federal and territorial waters off St. Croix (1,318 recreational fishers * 5.24 recreational trips/year = 71,594.12). To examine just recreational trips occurring in federal waters, we modify this total with the percentage of fishable habitat in the action area (i.e., 18%), which yields 12,888 annual recreational trips in federal waters off St. Croix.

In summary, we estimate a total of 14,033 annual commercial and recreational fishing trips within federal waters off St. Croix (1,145 + 12,888). We assume these trips are associated with fishing for species managed under the St. Croix FMP and are associated with the proposed action. This number of trips falls under the threshold of 65,642 vessel trips that we expect would result in a single sea turtle mortality due to vessel strike effects. As a result, we believe that the risk of sea turtles being struck by a vessel associated with the action covered under this Opinion is extremely low.

With regard to anchor lines, NMFS expects that they would be taught (not loose and looping), and that entanglement with sea turtles would be extremely unlikely.

Scalloped Hammerhead Shark- Fishing Gear Interactions

Commercial

The only ESA-listed species with reported commercial landings (territory and federal waters) in St. Croix from 2012-2018 was the scalloped hammerhead shark. A total of 2.78 lbs were reported for federal waters during the 7-year period, for an average of 0.40 lbs per year. Using the weight to count conversion factor of 10.96 lbs/fish (Southeast Fisheries Science Center, Trip Interview Program (TIP) database (SEFSC unpublished data)) this converts to an annual estimate of 0.04 scalloped hammerhead sharks caught by the St. Croix fishery, commercial sector. All scalloped hammerhead sharks were caught via hook and line gear.

Therefore, we expect 0.04, rounded to 1, scalloped hammerhead sharks will be taken by the St. Croix FMP-managed commercial fishery activities annually.

Recreational

Recreational data are not collected in the U.S. Virgin Islands. Data is collected for Puerto Rico and did not record this species. Puerto Rico provides the closest proxy available to estimate what take may occur in St. Croix. The areas are close together and the federally-managed fisheries use similar gears. Relying on information from Puerto Rico is potentially an over-estimate as effort in the Puerto Rico recreational fishery is likely greater. Because there were no interactions with the recreational fishery in Puerto Rico federal waters, NMFS would not expect this species to be caught in St. Croix.

Nassau Grouper- Fishing Gear Interactions

Commercial

No Nassau grouper were present in the commercial landings data from 2012-2018 for federal waters.

Recreational

Recreational data are not collected in the U.S. Virgin Islands. Data is collected for Puerto Rico and did not record this species. Puerto Rico provides the closest proxy available to estimate what take may occur in St. Croix. The areas are close together and the federally-managed fisheries use similar gears. Relying on information from Puerto Rico is potentially an over-estimate as effort in the Puerto Rico recreational fishery is likely greater. Because there were no interactions with the recreational fishery in Puerto Rico federal waters, NMFS would not expect this species to be caught in St. Croix.

Oceanic Whitetip- Fishing Gear Interactions

Commercial

No oceanic whitetip sharks were present in the commercial landings data from 2012-2018 for federal waters.

Recreational

Recreational data are not collected in the U.S. Virgin Islands. NMFS decided to estimate takes for the U.S. Virgin Islands based on data from Puerto Rico. Puerto Rico provides the closest proxy available to estimate what take make occur in St. Croix, after adjusting for fishable area (relative to Puerto Rico). The areas are close together and the federally-managed fisheries use similar gears. Relying on information from Puerto Rico is potentially an over-estimate as effort in the Puerto Rico recreational fishery is likely greater.

Therefore, if 11 oceanic whitetip sharks are taken by the FMP recreational fishery activities annually in federal waters off Puerto Rico (number discussed in the Puerto Rico section), then given fishable habitat area in federal waters:

(area St. Croix/area of Puerto Rico) x (recreational catch Puerto Rico) = recreational catch St. Thomas and St. John (68 sq. km EEZ/410 sq km EEZ) x 11 = 1.8, rounded to 2, oceanic whitetip sharks taken by fishing activities associated with the St. Croix FMP annually

Effects of Gear Deployment on *Acropora* Critical Habitat and ESA-listed Coral

As noted in Section 2, we anticipate direct effects to ESA-listed corals and *Acropora* critical habitat from fishing associated with the St. Croix FMP will occur in the EEZ off the St. Croix, where the fishing occurs. Other effects to ESA-listed corals and *Acropora* critical habitat, including effects associated with the harvest of herbivorous reef fish, can occur throughout territorial and federal waters, as harvested herbivorous fish may have otherwise traveled to territorial waters and provided beneficial algae grazing to ESA-listed corals and *Acropora* critical habitat in those waters. As noted at the beginning of Section 5.2, those effects are discussed in the Response section below.

Fishermen in the St. Croix fishery use the following gears that may affect the corals and *Acropora* critical habitat: hook-and-line gear, including handlines and vertical lines; SCUBA diving methods, including spear fishing for reef fish and snare and hand collection for spiny lobster; and traps. Standard vertical line fishing practices have the potential to impact ESA-listed coral or *Acropora* critical habitat through hooks snagging colonies or the essential feature of *Acropora* critical habitat, consolidated substrate and dead coral skeleton, through the landing of weights or other pieces of gear on the benthos, or through broken or discarded fishing line entangling coral colonies. However, fishermen in St. Croix deploy hook-and-line gear in the water column above the reef, so that the gear is not placed on the coral. Thus, we believe any adverse affects from vertical line fishing on colonies would be extremely unlikely to occur. Additionally, we believe hook-and-line gear used by fishermen in St. Croix would not affect consolidated hard substrate or dead coral skeleton (the essential feature of *Acropora* critical habitat) in any manner that would appreciably alter the biological or physical characteristics that make them suitable for larval settlement or coral regeneration.

SCUBA-assisted fishing gear is not thought to have impacts to the reef since divers generally do not intentionally contact the reef. Additionally, SCUBA-assisted fishing is highly selective, and the listed corals are not a target species; thus, we believe it is likely that divers would be able to avoid directly taking these species. Additionally, SCUBA divers that do encounter these colonies or dead coral skeletons when fishing for managed species would likely try to avoid them to reduce any chance of fishing gear becoming fouled. Thus, we believe adverse effects from SCUBA fishing on ESA-listed corals or *Acropora* critical habitat extremely unlikely.

The use of traps is known to cause physical damage to benthic habitats when they are set, hauled, lost, or abandoned (Chiappone et al. 2002, Sheridan et al. 2003, Mangi and Roberts 2006). Traps and/or trap lines can directly affect coral through breakage or abrasion, but traps can also destroy newly settled planulae during setting or hauling. Additionally, any space occupied by a trap temporarily prohibits that area from functioning as habitat because that space has been preempted by the trap making it unavailable for the settlement and growth of corals. The physical impacts of traps on coral and *Acropora* critical habitat in St. Croix may be estimated by analyzing the number of traps, the percentage of those traps reported to be used in coral or hardbottom areas, and the total area of coral cover in St. Croix.

This Opinion uses the calculation providing the most conservative estimate (largest number) of traps (fish traps and lobster traps) used in federal waters off St. Croix in order to analyze the greatest possible impact that could occur during fishing activities managed under the St. Croix FMP. Those fish trap and lobster trap estimates were then used to estimate impacts to ESA-listed coral and *Acropora* critical habitat as summarized below.

Average fish trap and lobster trap dimensions in St. Croix are 5 ft (152 cm), by 4 ft (122 cm), by 18 in (46 cm) (Sheridan et al. 2006).

Kojis et al. (2017) conducted a census of licensed commercial fishermen in the USVI in 2016. The census reported a total of 141 commercial fishermen on St. Croix, fishing in territorial and federal waters. Of those 141 fishermen, an estimated 31 use fish traps and 1 uses lobster traps. Kojis et al. (2017) reported the average number of days that the commercial fishermen soak their traps in federal and territorial waters off St. Croix: 8.3 days for fish traps and 8 days for lobster traps. The average number of traps hauled per trip was not reported for St. Croix, so the average number of traps owned per fisherman was used: 34.3 for fish traps and 20 for lobster traps (Kojis et al. 2017). The average soak time was used to calculate the average number of trips per month per fisherman (trips per month = 30/average soak time), which was then multiplied by the estimated number of fishermen and by the average number of traps hauled per trip. That number was then multiplied by 12 to get the estimated number of fish traps and lobster traps hauled each year in federal and territorial waters off St. Croix (Table 5.3). While it may be unlikely that every fisherman would fish all 12 months in a year, this approach was chosen to avoid underestimating the potential impacts to the ESA-listed coral species and *Acropora* critical habitat.

Table 5.3. Estimated number of commercial fish traps and lobster traps hauled in St. Croix (federal and territorial waters, combined) per year based on the average soak time (days).

Trap Type	Soak Time (days)	Trips/ Month	Number of Fishermen	Traps/ Trip	Total Traps Hauled/Month	Total Traps Hauled/Year
Fish	8.3	3.6	31	34.3	3,828	45,936
Lobster	8.0	3.8	1	20	76	912

Estimates of the number of recreational fishermen using trap gear in federal waters off St. Croix are not available at this time, but are assumed to be zero or near zero. That assumption is based on the fact that since traps are prohibited gear for recreational fishermen in territorial waters, it is extremely unlikely that recreational fishermen would use trap gear in federal waters. Therefore, the following discussion assumes trap use is in the commercial fisheries.

Effects of Traps on ESA-listed Corals

The following sections calculate exposure of each of the ESA-listed coral species to St. Croix trap gear.

In a survey of the distribution of trap locations, Sheridan et al. (2005) found 54% of all surveyed traps to be within colonized hard bottom dominated by soft coral or reef. Sheridan et al. (2005) also found that about 50% of all traps cause damage to bottom habitats, including coral areas, with instances of damage (scrapes, breakage) most prevalent among gorgonians and sponges, followed by corals. Damage was evaluated within 5 meters of where the trap was set, and thus the study accounted for potential damage from trap movement.

From the above estimates, approximately 45,936 fish traps and 912 lobster traps are set annually in fishable habitat in territorial and federal waters off St. Croix. Since Sheridan et al. (2005) reported 54% of traps were in colonized hard bottom, we estimate that 24,805 (45,936 x 0.54 = 24,805) fish traps and 492 (912 x 0.54 = 492) lobster traps were placed in colonized hard bottom. Since Sheridan et al. (2005) found that 50% of the traps deployed cause damage, we estimate that 12,403 (24,805 x 0.50 = 12,403) fish traps and 246 (492 x 0.50 = 246) lobster traps cause damage to coral.

Both fish and lobster traps used in St. Croix have an average area of 20 ft^2. Applying this information to the estimated number of fish traps and lobster traps expected to cause damage to corals, impacts are expected to be 248,060 ft^2 (12,403 x 20 ft^2) for fish traps and 4,920 ft^2 (246 x 20 ft^2) for lobster traps.

However, only 18% of fishable habitat is located in federal waters off St. Croix and therefore we expect that only 18% of the trap fishing, and 18% of the damage from trap use, occurs in federal waters and is related to the proposed action. Therefore an estimated amount of 44,651 ft^2 (248,060 x 0.18) of damage to colonized coral areas occurs from fish traps annually and 886 ft^2 (4,920 x 0.18) of damage occurs from lobster traps annually in federal waters off St. Croix. The total estimated amount of damage to corals (from fish traps + lobster traps) in federal waters off St. Croix would be 45,537 ft^2 (44,651+886).

The estimates in this opinion are likely very conservative (over-estimates) because we assume that the whole area of the trap causes total damage to the coral when it is more likely that the corals are scraped, not completely lost.

The average coverage for each ESA-listed coral species must be applied to this total damage estimate in order to calculate impacts to each ESA-listed species. The average coverage for the seven ESA-listed species was determined was determined from NMFS unpublished data. Note- NMFS used the largest percent coverage data available for areas surveyed (where multiple numbers were available) to conservatively estimate effects.

Elkhorn coral

percent cover = 0.1397%
45,537 ft^2 x 0.001397 = 63.6 ft^2

Staghorn coral

percent cover =0.0136%
45,537 ft^2 x 0.000136 = 6.2 ft^2

Rough cactus coral

percent cover =0.0085%
45,537 ft^2 x 0.000085 = 3.9 ft^2

Pillar Coral

percent cover =0.0059%
45,537 ft^2 x 0.000059 = 2.7 ft^2

Lobed star coral

percent cover =0.9299%
45,537 ft^2 x 0.009299 = 423.4 ft^2

Mountainous star coral

percent cover = 0.4141%
45,537 ft^2 x 0.004141 = 188.6 ft^2

Boulder star coral

percent cover =0.8863%
45,537 ft^2 x 0.008863 = 403.6 ft^2

Effects of Traps on Acropora Critical Habitat

For this analysis, we conservatively calculate the impact on *Acropora* critical habitat and assume all gear set in colonized coral hard bottom and corals also is set on designated critical habitat. This overstates the impact of gear on species and critical habitat since gear are set on species or critical habitat, not both simultaneously, but is the best way to conservatively estimate potential effects. The essential feature of *Acropora* critical habitat includes substrate of suitable quality and availability, which is defined as consolidated hardbottom or dead coral skeleton that is free from fleshy macroalgae cover and sediment cover. Since traps do not cause consolidated hardbottom to become unconsolidated, nor do they cause growth of macroalgae or cause sedimentation, we believe it is unlikely that traps would affect this portion of the essential feature. However, traps placed on the consolidated hardbottom could temporarily preclude new settlement of planulae (free-swimming coral larvae), affecting the functionality of the essential feature. In addition, we believe that traps could damage dead coral skeletons. There are currently no data available to determine the number of dead coral skeletons occurring inside critical habitat within the EEZ of the U.S. Caribbean. Therefore, our analysis of trap effects acts conservatively and assumes that the entire area of critical habitat does have dead coral skeletons. Sheridan et al. (2005) indicate 54% of traps were in colonized hard bottom, and that the traps deployed at various habitat sites, including coral areas, cause damage at 50% sites, to 13.7% of corals. However, only 18% of fishable habitat is located in federal waters off St. Croix, and we assume only 18% of the trap fishing that causes damage to critical habitat occurs in federal waters. For this analysis, we assume our analysis of the effects to live corals is analogous and applicable to effects to coral skeletons (and the essential feature for *Acropora* critical habitat), and the damage to coral skeletons would be similar to damage that occurs when gear interacts with live branching coral. Thus, we apply the analysis just used in the previous section. The total estimated amount of damage to coral skeletons (from fish traps and lobster traps) in St. Croix would be 6,239 ft^2 (45,537 ft^2 area colonized hard bottom affected x 0.137 of dead skeleton affected). Therefore, we anticipate direct effects from fishing occurring in the EEZ will occur to 6,239 ft^2 of *Acropora* critical habitat annually.

Effects of Fishing Vessel Anchors on ESA-Listed Coral and *Acropora* Critical Habitat

Effects on ESA-listed corals

Here we examine the number of trips taken by fishing vessels, apply factors for the percentage of those vessels deploying anchors in areas with coral and the percentage of those anchors that do damage, and then analyze those findings based on an anchor footprint appropriate for the average vessel used in the fisheries.

Per our analysis above, an estimated 14,033 recreational and commercial fishing trips occur per year (12,888 recreational vessel trips and 1,145 commercial vessel trips) in federal waters off St. Croix. This number is an over-estimation of the number of anchors used since the majority of commercial fishermen (more than 95%) do not use anchors during fishing activities (C. Farchette, Caribbean Fishery Management Council, pers. comm. to S. Stephenson, NMFS SERO, March 31, 2020). Additionally, more than 50% of surveyed recreational fishermen participate in inshore and offshore trolling methods (Kojis and Tobias 2016), and thus do not use anchors. NMFS assumes that 50% of the trips use anchors (this errs on the side of caution, considering "most" is 50% or more, but does not use the 95% figure, which would exclude almost all trips). Therefore, 14,033 x 0.50 = 7,017 commercial and recreational trips occur annually that set anchors.

Rogers et al. (1988) conducted a survey of boats in 1987 that revealed 14% of boats that anchored were anchored in coral communities and that about 40% of the anchors on these vessels caused damage. This study pertained to all boats observed in a particular area off of St. John, not just fishing vessels. In the absence of more specific data, it is a reasonable proxy for the potential number of fishing vessels anchored in coral communities in federal waters off St Croix that cause damage. Therefore, if we multiply the number of fishing trips in federal waters that set anchors annually (7,017) by the percent anchored in coral communities (i.e., 14%), we arrive at a number of anchors in coral communities (983). If we then apply the percentage of anchors causing damage (i.e., 40%) to the estimate for the number of anchors in coral communities (983), the result is our estimate for the total number of trips with interactions between anchors and corals (394). We then assume most 16- to 27-ft boats use of a 4-lb. aluminum anchor with measurements of 24 in x 19 in (3.16 ft^2). Multiplying the area of each anchor by the total number of anchor interactions yielded an estimate of the total area impacted by anchoring events of 1,246 ft^2 (394 x 3.16 = 1,246 ft^2 per year). In assuming that the area of impact is limited to the size of the anchor, we are not assuming additional effects from anchor dragging. Anchor dragging could cause additional damage to reefs. However, fishing vessels operating in the managed fisheries in federal waters are relatively small, often with no place to take shelter from adverse weather. This makes it less likely that fishermen would choose to fish during weather or sea states that are most conducive to causing dragging. Thus, we believe anchor dragging is unlikely to be an issue of concern.

Next it is necessary to calculate the impact to each listed coral species using the percent of cover of each.

Staghorn coral	1,246 ft^2 per year x 0.0136% = 0.2 ft^2
Elkhorn coral	1,246 ft^2 per year x 0.1397% = 1.7 ft^2
Rough cactus coral	1,246 ft^2 per year x 0.0085% = 0.1 ft^2
Pillar Coral	1,246 ft^2 per year x 0.0059% = 0.1 ft^2
Lobed star coral	1,246 ft^2 per year x 0.9299% = 11.6 ft^2
Mountainous star coral	1,246 ft^2 per year x 0.4141% = 5.2 ft^2
Boulder star coral	1,246 ft^2 per year x 0.8863% = 11.0 ft^2

Effects on Acropora Critical Habitat

It is necessary to consider the effects of anchoring on *Acropora* critical habitat. NMFS uses the calculation of number of square feet per year (1,246 ft^2) just calculated for species (which incorporates setting on coral and how much is damaged) and applies it to critical habitat. Based on Rogers et al. (1988), we assume 14% of vessels that anchor do so in coral communities. We assume that the critical habitat is spread throughout that potential anchoring area, and that vessels are not anchored in critical habitat in any other area (i.e., the 14% figure covers anchoring in coral areas, including all *Acropora* critical habitat). Therefore, we assume that anchoring damaged 1,246 ft^2 of *Acropora* critical habitat annually.

Anchor dragging could cause additional damage to critical habitat. However, fishing vessels operating in the managed fisheries in federal waters are relatively small, often with no place to take shelter from adverse weather. This makes it less likely that fishermen would choose to fish during weather or sea states that are most conducive to causing dragging. Thus, we believe anchor dragging is unlikely to be an issue of concern.

<u>Summary of Expected Take-All FMPs (Annually)</u>

	Individuals	**Area (ft^2)**
Sea Turtle- green*		
Puerto Rico FMP	2	n.a.
St. Thomas/St. John FMP	3	n.a.

St. Croix FMP	1	n.a.
Total	6	
Sea Turtle- hawksbill		
Puerto Rico FMP	2	n.a.
St. Thomas/St. John FMP	0	n.a.
St. Croix FMP	1	n.a.
Total	3	
Scalloped Hammerhead Shark		
Puerto Rico FMP	0	n.a.
St. Thomas/St. John FMP	0	n.a.
St. Croix FMP	1	n.a.
Total	1	
Nassau Grouper		
Puerto Rico FMP	0	n.a.
St. Thomas/St. John FMP	4	n.a.
St. Croix FMP	0	n.a.
Total	4	
Oceanic Whitetip Shark		
Puerto Rico FMP	11	n.a.
St. Thomas/St. John FMP	16	n.a.
St. Croix FMP	2	n.a.
Total	29	
Elkhorn Coral		
Puerto Rico FMP	n.a.	206.5
St. Thomas/St. John FMP	n.a.	38.7
St. Croix FMP	n.a.	65.3
Total		310.5
Staghorn Coral		
Puerto Rico FMP	n.a.	188.9
St. Thomas/St. John FMP	n.a.	267.1
St. Croix FMP	n.a.	6.4
Total		462.4
Rough Cactus Coral		
Puerto Rico FMP	n.a.	12.2
St. Thomas/St. John FMP	n.a.	23.1
St. Croix FMP	n.a.	4.0
Total		39.3
Pillar Coral		

Puerto Rico FMP	n.a.	29.7
St. Thomas/St. John FMP	n.a.	122.6
St. Croix FMP	n.a.	2.8
Total		155.1
Lobed Star Coral		
Puerto Rico FMP	n.a.	140.3
St. Thomas/St. John FMP	n.a.	10,931.1
St. Croix FMP	n.a.	435.0
Total		11,506.4
Mountainous Star Coral		
Puerto Rico FMP	n.a.	327.6
St. Thomas/St. John FMP	n.a.	8,934.1
St. Croix FMP	n.a.	193.8
Total		9,455.5
Boulder Star Coral		
Puerto Rico FMP	n.a.	170.2
St. Thomas/St. John FMP	n.a.	17,391.9
St. Croix FMP	n.a.	414.6
Total		17,976.8

*** = green sea turtles could be from the NA or SA DPS; a single total (number) of green sea turtles are taken (some mix of both DPSs), however the jeopardy analysis is conducted for each DPS with the assumption all animals are taken are coming from each DPS (i.e., two jeopardy analyses) (see below).**

Summary of Effects to Critical Habitat

The amount of *Acropora* Critical Habitat affected is 22,222 ft^2 (Puerto Rico) + 34,174 ft^2 (St. Thomas/St. John) + 7,485 ft^2 (St. Croix) = 63,881 ft^2 total annually.

5.3 Response

5.3.1 Effects of Harvest of Herbivorous Fish on Corals and *Acropora* Critical Habitat

Climate change, disease, sedimentation (particularly terrigenous sediments), and nutrients from land-based sources have likely played a significant role in the current status of coral reefs (i.e., generally algal dominated) within the U.S. Caribbean. However, the purpose of this biological opinion is to include evaluation of the effects anticipated from the federal actions – the approval and implementation of the island-based fishery management plans. We acknowledge a number of other factors are likely acting or have acted synergistically with the adverse effects of the proposed actions to create the current state of the coral reefs in the U.S. Caribbean. Our analysis in this section focuses only on the effects of harvest of herbivorous fish that we anticipate will occur with the operation of the fishery components managed under the FMPs that are the subject of this consultation. In particular, we are evaluating the effect of harvesting herbivorous fish on algae cover and the resulting effects to corals and *Acropora* critical habitat.

Mumby et al. (2015), Roff and Mumby (2012), and Connell (1997) identify the geographic differences in coral reefs across the globe, specifically calling out the striking geographic differences between coral reef resilience in the Atlantic and Pacific Oceans. There also appear to be marked differences in the average state (i.e., coral-dominated versus algal-dominated) and underlying ecosystem processes acting on coral reefs in the Atlantic versus the Pacific Oceans (Mumby et al. 2015). For example, reefs of the Atlantic appear to exhibit much higher levels of fleshy macroalgae even for a given biomass of herbivores (Roff and Mumby 2012). Mumby et al. (2015) points out that in the Caribbean, the speed at which macroalgae grows and its overall abundance appears much greater in the region than in the Pacific (Roff and Mumby 2012). Additionally, there is emerging evidence indicating the amount of dissolved organic carbon (DOC) (De Goeij et al. 2013; Kline et al. 2006; Loh et al. 2015; Pawlik et al. 2016), and even dust from the Saharan Africa (Shinn et al. 2000) in the Caribbean Basin may be reducing the resilience of reefs in this region, while having far less impact on Pacific reefs.

Because of these differences, we believe it is important that our analyses, to the extent possible, put the most emphasis on studies collecting data on coral, algae, and herbivorous fish, and the relationship of each to one another, from within the Caribbean Basin, and specifically the U.S. Caribbean action area. We recognize that the competition between corals and algae is universal; however, how those competitive processes play out, and the overall effect of that competition, varies by oceanic basin and even at the reef level. Section 3 of this Opinion detailed the coral reef, algae, and reef fish herbivory dynamics relevant for this Opinion and that inform the discussion of effects to coral and *Acropora* critical habitat discussed here.

As discussed in the last biological opinion on the Caribbean reef fish fishery (NMFS 2011a) and Section 3 of this Opinion, much of the scientific literature indicates that herbivores are important in maintaining coral reefs, especially in mitigating the spread of algal cover. Many studies indicate a numerically abundant, high biomass, intact size structure and diverse herbivorous fish population would likely achieve the highest herbivory rates; some even state that only unfished stocks of herbivores can achieve the maximum mitigative effect (Lubchenco and Gaines 1981, Duffy 2002, Mumby 2006, Burkepile and Hay 2008, Lokrantz et al. 2008, Bonaldo and Bellwood 2008). The harvest of surgeonfish and parrotfish under the proposed actions, even at the reduced levels provided for by the ACLs and other management measures, including bag limits applicable to recreational fishing, is likely to adversely affect *Acropora* designated critical habitat and ESA-listed coral species. However, it is difficult to determine the magnitude of these effects.

Effects of Harvest of Herbivorous Fish on *Acropora* Critical Habitat

When analyzing effects to *Acropora* critical habitat, if herbivorous fish harvest were the primary cause of the current phase-shifting (from coral dominated to algae dominated) in the *Acropora* critical habitat units in the U.S. Caribbean, one would expect certain units to show a greater rate of phase-shift, indicated by significantly more algal cover, when compared to other units, based on the harvest levels NMFS (2011a) explained that the phase-shift was just as severe in all three critical habitat units. This suggested that although harvest of herbivorous fish may be indirectly adversely affecting the critical habitat essential feature, it is most likely not the driver of the phase-shift, but just one component of a larger function.

Our last consultation addressing this issue (NMFS 2011a) concluded that the main reason for the phase-shift that currently affects the availability of the essential habitat feature of *Acropora* critical habitat are effects to corals from diseases, elevated sea surface temperature, damage from hurricanes, combined with the loss of *Diadema* (sea urchins). The anthropogenic effects from herbivorous fish harvest and nutrient input have likely acted synergistically to exacerbate those factors. The action in the 2011 consultation included the harvest of herbivorous species at reduced levels through the ACLs, as do the proposed actions in this consultation. In the 2011 consultation, the extent to which the harvest of herbivorous fish would adversely affect the availability of the essential feature of *Acropora* critical habitat was determined to be uncertain and unquantifiable; however, the information reviewed in the 2011 consultation indicated even unfished populations of herbivores were unlikely to completely reverse the current phase shift due to the magnitude of the other factors affecting reefs in the U.S. Caribbean. Limiting the harvest of herbivorous fish[47] under the ACLs and prohibiting the harvest of the three large-bodied parrotfish (midnight, blue, and rainbow parrotfish) was expected to result in population increases of herbivorous fish, especially the large-bodied parrotfish (believed to be the most efficient algal grazers), throughout all three critical habitat units, which means there would be greater amounts of grazing under the proposed action than there were at the time of designation of *Acropora* critical habitat (2006), when parrotfish and surgeonfish harvests were unrestricted (ACLs were fully implemented in 2011). Therefore, the action analyzed in 2011 was expected to adversely affect *Acropora* critical habitat; however, those adverse affects were likely to be reduced by some amount that was unquantifiable. This expectation remains the same for the analysis in this Opinion.

We continue to find that the approving the Puerto Rico FMP, St. Thomas/St. John FMP, and the St. Croix FMP and allowing the harvest of certain herbivorous fish under certain limits (ACLs and recreational bag limits),[48] is likely to reduce success of coral larvae settlement and fragment reattachment because of increases in algae,[49] leading to a reduction in the essential features of *Acropora* critical habitat compared to not allowing the fishing. However, by limiting the harvest of certain managed parrotfish and surgeonfish species under ACLs and recreational bag limits, and prohibiting the harvest of the large-bodied parrotfish (midnight, blue, and rainbow parrotfish), the proposed actions reduce these adverse effects by some unquantifiable amount.

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[47] The Caribbean Reef Fish FMP, subject to the 2011 consultation, manages the following herbivorous fish: Blue, midnight, princess, queen, rainbow, redfin, redtail, stoplight, redband, and striped parrotfishes, and blue tang, ocean surgeonfish, and doctorfish.

[48] The Puerto Rico FMP would prohibit the harvest of blue, midnight, and rainbow parrotfish, and manage the harvest of queen, princess, redtail, stoplight, redband, and striped parrotfish, and blue tang, ocean surgeonfish, and doctorfish with commercial and recreational sector ACLs and recreational bag limits. Both of the St. Thomas/St. John and the St. Croix FMPs would prohibit the harvest of blue, midnight, and rainbow parrotfish, and manage the harvest of queen, princess, redtail, stoplight, redband, striped, and redfin parrotfish, and blue tang, ocean surgeonfish, and doctorfish with ACLs and recreational bag limits.

[49] Throughout the 2011 Biological Opinion we used the term “macroalgae” and “algae” interchangeably, often times using macroalgae to refer to any number of species of algae. Since macroalgae is a specific type of algae, we now believe our original use of “macroalgae” in instances meant to refer to several species of algae was imprecise. We now use the term “algae” to refer generally to fleshy macroalgae and turf algae.

We believe many factors have worked synergistically to reduce the essential feature of critical habitat necessary for successful *Acropora* recruitment, and believe herbivory has played a secondary role in that process. The incremental impact the harvest of herbivorous fish has on the reduction of the essential feature of critical habitat is uncertain, currently unquantifiable, and largely unobservable outside of a controlled environment.[50]

Effects of Harvest of Herbivorous Fish on ESA-Listed Corals

Lobed star, boulder star, mountainous star, pillar, rough cactus, elkhorn, and staghorn coral species are subject to effects from herbivorous fish removal associated with the proposed actions through the loss or reduction in the mitigative effect herbivorous fishes have on competitively dominant algae. Corals and algae compete for space on reefs and interact through several mechanisms (McCook et al. 2001). Coral planulae cannot settle on algae, and therefore the space occupied by algae reduces the availability of suitable settlement space for corals (Steneck 1988). Algae can trap sediment that smothers coral recruits (Birkeland 1977) and direct contact with algae reduces coral growth rates (Tanner 1995) and fecundity (Tanner 1995, Foster et al. 2008) and may even result in direct overgrowth and coral mortality (Nugues and Bak 2006). It is also feasible that algae can negatively influence corals through allelochemicals (Fearon and Cameron 1996, Rasher and Hay 2010), triggering disease (Nugues et al. 2004), causing tissue and possibly even colony mortality (Rasher and Hay 2010), and enhancing microbial activity driven by algal-derived dissolved organic carbon (Smith et al. 2006). Authorizing the harvest of certain parrotfish and surgeonfish species under proposed actions will decrease the grazing rates of these herbivores compared to not authoring the harvest, with the potential consequences just described. Therefore, the fisheries are likely to adversely affect these corals, by reducing the overall grazing capacity of the reef fish populations, as compared to an unfished assemblage, by reducing the herbivore assemblage's ability to mitigate increasing algal cover, which suppress their reproduction. However, the proposed actions also include new restrictions to prohibit the harvest of *Diadema*, which also potentially will contribute to additional grazing of algae (and thus corals) that would not have occurred but for the harvest prohibition. We believe this information supports the expectations of the proposed actions are likely to adversely affect the ESA-listed corals by reducing herbivore biomass and grazing capacity compared to not allowing harvest of these herbivores, but that the ACLs and recreational bag limits, and the prohibition on the harvest of *Diadema*, will reduce the expected adverse effects to the coral reef-algae dynamics in the action area by some unquantifiable amount.

Summary

The ultimate role the fishery components play in changes from coral- toward algal-dominated systems cannot be determined from the data currently available. A number of non-fishery-related factors affect the growth and spread of algae (e.g., nutrient enrichment) and a reef's ability to resist phase shifts (e.g., global climate change). However, herbivorous fish, particularly parrotfish (please refer to Table 9.2 for managed parrotfish species), are now the

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[50] As discussed in Section 5, there are a variety of factors contributing to algal growth and the reduction in the essential feature of critical habitat, but the precise relative contributions of these factors are unknown. It is not possible to determine the contribution of each factor without being able to identify and control each factor in a controlled environment, such as a laboratory experiment. Such a controlled environment does not exist in the open ocean, and no such controlled experiments have been conducted to inform this opinion.

primary entity providing active control over algae on reefs in the USVI and Puerto Rico and the fishery components under the proposed actions will remove some of these species from those reefs. Herbivory is an important factor in maintaining resilience. For this reason, we anticipate that operation of the island-based fisheries will have indirect adverse effects, albeit currently unquantifiable, to all ESA-listed corals and *Acropora* critical habitat.

5.3.2 Capture and/or Entanglement (Hooks and Line) In Fishing Gear

Sea turtles, Nassau grouper, oceanic whitetip sharks, and scalloped hammerhead sharks may be adversely affected by fishing activity through incidental hooking or entanglement in actively fished or discarded fishing line, as described more fully below. These species have historically been captured in both recreational and commercial fisheries and are known to become entangled in fishing debris. Sea turtles are particularly prone to entanglement as a result of their body morphology and behavior. In Section 5.2, above, we estimated the number of each of these species that might be exposed to hook and line gear based on available data. In this section, we describe how the species are expected to respond to such exposure, including whether the interaction is likely to result in injury or mortality.

<u>Sea Turtles</u>
Hook-and-line gear can adversely affect sea turtles via entanglement, hooking, and trailing line. Records of stranded or entangled sea turtles reveal that fishing gear can wrap around the neck, flipper, or body of a sea turtle and severely restrict swimming or feeding. Sea turtles released alive may later succumb to injuries sustained at the time of capture or from exacerbated trauma from fishing hooks or lines that were ingested, entangled, or otherwise still attached when they were released. Of the sea turtles hooked or entangled that do not die from their wounds, some may suffer impaired swimming or foraging abilities, altered migratory behavior, and altered breeding or reproductive patterns.

The current understanding of the effects of hook-and-line gear on sea turtles is related primarily to the effects observed in association with commercial fisheries (particularly longline fisheries); less data exist on the effects of recreational fishing on sea turtles. Dead sea turtles found stranded with hooks in their digestive tract have been reported, though it is assumed that most sea turtles hooked by recreational fishers are released alive (Thompson 1991). Little information exists on the frequency of recreational fishing captures and the status of the sea turtles after they are caught. Regardless, effects sea turtles are likely to experience as a result of interactions with recreational hook-and-line gear (i.e., entanglement, hooking, and trailing line) are expected to be the same as those that might occur in commercial fisheries. The following discussion summarizes in greater detail the available information on how individual sea turtles may be affected by interactions with hook-and-line gear.

Hooking
In addition to being entangled in hook-and-line gear, sea turtles are also injured and killed by being hooked. Hooking can occur as a result of a variety of scenarios, some depending on the foraging strategies and diving and swimming behavior of the various species of sea turtles. Sea turtles are either hooked externally in the flippers, head, shoulders, armpits, or beak, or internally inside the mouth or when the animal has swallowed the bait (Balazs et al. 1995).

Swallowed hooks are of the greatest concern. A sea turtle's esophagus (throat) is lined with strong conical papillae directed towards the stomach (White 1994). The presence of these papillae in combination with an S-shaped bend in the esophagus make it difficult to see hooks when looking through a sea turtle's mouth, especially if the hooks have been deeply ingested. Because of a sea turtle's digestive structure, deeply ingested hooks are also very difficult to remove without seriously injuring the turtle. A sea turtle's esophagus is also firmly attached to underlying tissue; thus if a sea turtle swallows a hook and tries to free itself or is hauled on board a vessel, the hook can pierce the sea turtle's esophagus or stomach and can pull organs from its connective tissue. These injuries can cause the sea turtle to bleed internally or can result in infections, both of which can kill the sea turtle.

If a hook does not lodge into, or pierce, a sea turtle's digestive organs, it can pass through the sea turtle entirely (Aguilar et al. 1995; Balazs et al. 1995) with little damage (Work 2000). For example, a study of loggerheads deeply hooked by the Spanish Mediterranean pelagic longline fleet found ingested hooks could be expelled after 53 to 285 days (average 118 days) (Aguilar et al. 1995). If a hook passes through a sea turtle's digestive tract without getting lodged, the hook probably has not harmed the turtle.

Trailing Line

Trailing line (i.e., line left on a sea turtle after it has been captured and released), particularly line trailing from a swallowed hook, poses a serious risk to sea turtles. Line trailing from a swallowed hook is also likely to be swallowed, which may irritate the lining of the digestive system. The line may cause the intestine to twist upon itself until it twists closed, creating a blockage ("torsion") or may cause a part of the intestine to slide into another part of intestine like a telescopic rod ("intussusception") which also leads to blockage. In both cases, death is a likely outcome (Watson et al. 2005). The line may also prevent or hamper foraging, eventually leading to death. Trailing line may also become snagged on a floating or fixed object, further entangling a turtle and potentially slicing its appendages and affecting its ability to swim, feed, avoid predators, or reproduce. Sea turtles have been found trailing gear that has been snagged on the sea floor, or has the potential to snag, thus anchoring them in place (Balazs 1985). Long lengths of trailing gear are likely to entangle the sea turtle, eventually, leading to impaired movement, constriction wounds, and potentially death. If an individual sea turtle is entangled when young, the fishing line can become tighter and more constricting as the individual grows, cutting off blood flow and causing deep gashes, some severe enough to remove an appendage.

Expected Mortalities

Sea Turtles

The injury to sea turtles from hook-and-line captures and ultimately the post-release mortality (PRM) will depend on numerous factors including how deeply the hook is embedded, whether it was swallowed or was an external hooking, whether the sea turtle was released with trailing line, and how soon and how effectively the hooked sea turtle was de-hooked or otherwise cut loose and released.

In the U.S. Caribbean, strandings represent the best available information upon which to estimate potential interactions between fishing gear and sea turtles. The stranding data does not have specific information used in other contexts to evaluate immediate and post-release mortality, such as location where the individual was hooked or the amount of trailing line. Therefore, we are not confident in our ability to monitor non-lethal effects occurring from these interactions. Thus, our analysis here is conservative and assumes that any interactions will result in mortality, all 3 FMPs combined.

Up to 5 mortalities of the NA DPS of green sea turtles a year
Up to 5 mortalities of the SA DPS of green sea turtles a year
Up to 2 mortalities of hawksbill sea turtles a year

These animals will be lost to their respective populations.

<u>Nassau Grouper, Scalloped Hammerhead Shark, Oceanic Whitetip Shark</u>

FMP gear can adversely affect fish via entanglement, hooking, and trailing line. External and internal damage from gear, injury from handling, and negative effects of stress can all result in severe impacts to scalloped hammerhead sharks, Nassau grouper, and oceanic whitetip sharks. Hooking and entanglement can lead to cuts, puncture wounds, mouth or other tissue damage, and animals can suffer from the stress of the capture.

Based on catch report data, NMFS expects 4 Nassau grouper will interact with the fisheries for all 3 FMPs combined (interactions were expected from fishing managed under the St. Thomas/St. John FMP only). No information exists on injury or post-release mortality for Nassau grouper captured through fishing managed under the FMPs, and we do not have any reasonable proxies from other fisheries, given the different gears used. Because we are not confident in our ability to monitor non-lethal effects occurring from these interactions, our analysis here is conservative and assumes that any interactions will result in mortality.

In addition, based on catch report data, NMFS expects 1 scalloped hammerhead shark (St. Croix FMP only) and 29 oceanic whitetip shark (all FMPs) will be interact with the fisheries, for all 3 FMPs combined. Since there is no prohibition against keeping scalloped hammerhead or oceanic whitetip sharks, it is possible that 100% of the animals could be retained by fishermen. Therefore, to conduct a conservative analysis, we expect all of these interactions to result in mortality.

Thus, we expect the following annual mortalities, for all three FMPs, combined.

Nassau Grouper = 4 mortalities annually
Scalloped Hammerhead = 1 mortality annually
Oceanic Whitetip Shark = 29 mortalities annually

These animals will be lost to their respective populations.

5.3.3 Effects to Sea Turtles from Vessel Strike

Vessel strikes may result in direct injury or death through collision (concussive) impacts or propeller wounds. Although sea turtles, with the exception of leatherback sea turtles, have hard carapaces, they are unable to withstand the strike of a rapidly moving vessel or the cut of a propeller. A sea turtle's spine and ribs are fused to the shell, which is a living part of their body that grows, sheds, and bleeds. Rapidly moving vessels may strike the head or carapace and result in fractures. Injuries to the carapace can involve fractures to the spinal column and buoyancy problems. A propeller can easily cut through the shell and sever or damage the spine and internal organs. Propeller injuries may range from mild to severe and include head lacerations, eye injury, injury to limbs, and carapace lacerations and fractures. Chronic and/or partially healed propeller wounds also may be associated with secondary problems such as emaciation and increased buoyancy (Jacobson et al. 1989). Abnormally buoyant sea turtles are unable to dive for food or escape predators or future vessel strikes. Seriously injured or dead turtles may be struck multiple times by vessels before they drift ashore.

The proportion of vessel-struck sea turtles that survive or die is unknown. In many cases, it is not possible to determine whether documented injuries on stranded animals resulted in death or were post-mortem injuries. Sea turtles found alive with concussive or propeller injuries are frequently brought to rehabilitation facilities; some are later released and others are deemed unfit to return to the wild and remain in captivity. Sea turtles in the wild are documented with healed injuries; thus, we know at least some sea turtles survive without human intervention.

Expected Mortalities
The injury to sea turtles from boat strikes will depend on numerous factors including what part of the body was struck and the speed of the vessel. While some turtles potentially survive interactions, we are not confident in our ability to monitor non-lethal effects occurring from these interactions. Thus, our analysis here is conservative and assumes that any interactions will result in mortality, all 3 FMPs combined (interactions are expected from the fisheries managed under the Puerto Rico FMP only).

Up to 1 mortality of the NA DPS of green sea turtles a year
Up to 1 mortality of the SA DPS of green sea turtles a year
Up to 1 mortality of hawksbill sea turtles a year

These animals will be lost to their respective populations.

5.3.4 Effects to Coral and *Acropora* Critical Habitat from Fishing Gear and Anchors

The gear and anchors associated with the FMP-managed fishing can adversely affect coral by direct contact through breakage or abrasion, but can also destroy newly settled planulae during setting or hauling. Additionally, any space occupied by gear and anchors temporarily prohibits that area from functioning as *Acropora* critical habitat because that space has been preempted by the gear or anchor, making it unavailable for the settlement and growth of corals. Thus, the use of gear and anchors may affect *Acropora* critical habitat and coral through breakage, other physical damage, or interference in reproduction. Interference with settlement and damage to critical habitat essential features (e.g., coral skeletons) resulting from gear and anchors would negatively affect important area for the reproduction of corals and growth of colonies.

In Section 5.2, the Exposure section, NMFS estimated the amount of ESA-listed corals and *Acropora* critical habitat that would be exposed to adverse effects associated with the proposed action from trap use and vessel anchoring. In this section, NMFS explains the response to that exposure.

NMFS expects that the interactions between the ESA-listed corals and the traps and anchors will result in mortality to the species, and therefore that the proposed actions will result in loss (mortality) of up to 310.5 ft^2 of elkhorn coral, 462.4 ft^2 of staghorn coral, 39.3 ft^2 of rough cactus coral, 155.1 ft^2 of pillar coral, 11,506.4 ft^2 of lobed star coral, 9,455.5 ft^2 of mountainous star coral, and 17,976.8 ft^2 of boulder star coral annually. Mortality means the coral will be lost to their respective populations (including lost reproduction from that lost coral).

Additionally, 63,881 ft^2 of *Acropora* critical habitat will be affected annually, by destroying the essential features.

6.0 Cumulative Effects

Cumulative effects include the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action areas of 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. Cumulative effects from unrelated, non-federal actions occurring in the action area may affect sea turtles, the Central and Southwest Atlantic DPS of scalloped hammerhead sharks, Nassau grouper, oceanic whitetip sharks, coral, and *Acropora* critical habitat.

The activities discussed in Sections 3 and 4 of this Opinion described as occurring within the action area are expected to continue as described into the foreseeable future, concurrent with the proposed action. The primary non-federal actions likely to continue to affect species and critical habitat in the action area include vessel operations, pollution and marine debris, non-federal fisheries, and climate change.

Watercraft are the greatest contributors to overall noise in the sea and have the potential to interact with sea turtle species though direct impacts or propellers. The effects of fishing vessels, recreational vessels, or other types of commercial vessels on listed sea turtle species may involve disturbance or injury and potential mortality due to collisions or entanglement in anchor lines. Commercial traffic and recreational pursuits can also adversely affect through propeller strikes. Although minor vessel collisions may not kill an animal directly, they may weaken or otherwise affect an animal, which makes it more likely to become vulnerable to effects such as entanglements.

Human activities in the action areas causing pollution are reasonably certain to continue in the future, as are impacts from them to species in this Opinion, however, the level of impacts cannot be projected. Marine debris (e.g., debris, discarded fishing line or lines from boats) can entangle sea turtles in the water and drown them, or impair their normal movement and behavior. For example, sea turtles have been documented stranded in the U.S. entangled in plastics, monofilament, discarded netting, and many other waste items. Entanglement can lead to death, injury, mutilation, starvation, and increased susceptibility to predation. Ingestion of plastic, rubber, fishing line and hooks, tar, string, Styrofoam, epoxy, and aluminum has been documented in marine species, potentially resulting in digestive tract impaction or toxic absorption.

Activities affecting corals are highly regulated federally; therefore, any future activities within the action areas will likely require ESA section 7 consultation. However, much of the coastal development occurring on the USVI and Puerto Rico, and the upland development in Puerto Rico, has been shown to affect water quality, in particular through increases in sedimentation rates, and projects without a federal nexus are considered in this analysis. Depending on the number and location of the coastal and upland developments, sediment and nutrient loading to nearshore waters could become a chronic stressor. Indeed, results of water quality monitoring from sites around USVI by Department of Planning and Environmental Resources indicate that this is becoming the case with the number of impaired sites increasing each year (Rothenberger et al. 2008). In Puerto Rico, both coastal development and development at upland watersheds that have outlets to nearshore waters likely have contributed to nearshore sedimentation. As the rate of development continues to accelerate in the USVI and Puerto Rico, sedimentation is likely to increase, leading to continued impacts to coral colonies that result in decreases in growth and percent cover, as well as decreases in the amount of suitable habitat for coral larvae and fragments to settle. Continued increases in the number of vessels transiting and anchoring in the areas and concomitant increases in accidental spills of petroleum products, leaching of chemicals from anti-fouling paints, marine debris, and accidental groundings, will also affect colonies of listed corals.

Global climate change is likely adversely affecting scalloped hammerhead sharks, Nassau grouper, oceanic whitetip sharks, sea turtles, coral, and *Acropora* critical habitat. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The effects on ESA-listed species are unknown at this time. There are multiple hypothesized effects to ESA-listed species including changes in their range and distribution, as well as prey distribution and abundance due to water temperature changes. Ocean acidification may also negatively affect marine life, particularly organisms with calcium carbonate shells that serve as important prey items for many species. Global climate change may also affect reproductive behavior in animals, including earlier onset of nesting, shorter intervals between nesting, and a decrease in the length of nesting season for sea turtles. A decline in reproductive fitness as a result of global climate change could have effects on the abundance and distribution of animals in the Atlantic.

NMFS is not aware of any proposed or anticipated changes in these factors that would substantially change the impacts each has on the ESA-listed Central and Southwest Atlantic DPS of scalloped hammerhead sharks, Nassau grouper, oceanic whitetip sharks, sea turtles, coral, and *Acropora* critical habitat covered by this Opinion. Therefore, NMFS expects that the levels of effects described for each of the factors will continue at similar levels.

7.0 Jeopardy Analyses/ Destruction or Adverse Modification

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 elkhorn, staghorn, boulder star, mountainous star, lobed star, rough cactus, and pillar corals, the NA and SA DPSs of green sea turtles, hawksbill sea turtles, Nassau grouper, the Central and Southwest Atlantic DPS of scalloped hammerhead shark, and the oceanic whitetip shark. This section will also determine if the proposed actions will result in destruction or adverse modification of *Acropora* critical habitat. In Section 5, we outlined how the proposed action would affect species at the individual level and the extent of those effects in terms of the number of associated interactions, captures, and mortalities of each species to the extent possible with the best available data. Now we assess each of these species' response to this impact, 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), are likely to jeopardize their continued existence in the wild. We also consider the extent to which the proposed actions are likely to result in destruction or adverse modification of coral critical habitat.

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

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

The status of each listed species likely to be adversely affected by the proposed action is reviewed in Section 3. For any species listed globally, our jeopardy determination must find the proposed action will appreciably reduce the likelihood of survival and recovery at the global species range. For any species listed as a DPS, a jeopardy determination must find the proposed action will appreciably reduce the likelihood of survival and recovery of that DPS.

7.1 Green Sea Turtles (NA DPS and SA DPS)

As discussed in the Exposure section, within U.S. waters individuals from both the NA and SA DPSs can be found on foraging grounds, and we expect individuals from both DPSs to be found in waters in the action areas for the proposed actions. To analyze effects in a precautionary manner, we will conduct two jeopardy analyses, one for each DPS (i.e., assuming animals would be taken from both DPSs). Since we have no information specific to U.S. Caribbean waters regarding relative percentages of animals likely to occur in any area, this analysis will be done for the NA and SA DPSs as if all of the takes would occur to each of the DPSs.

7.1.1 Green Sea Turtle NA DPS

The proposed actions, combined, could result in up to 6 mortalities of NA DPS green sea turtles per year. The potential lethal take of individuals from the NA DPS of green sea turtles per year would reduce the number of NA DPS green sea turtles, compared to their numbers in the absence of the proposed actions, assuming all other variables remained the same. Lethal interactions would also result in a potential reduction in future reproduction, assuming some individuals would be females and would have survived otherwise to reproduce. For example, as discussed in this Opinion, an adult green sea turtle can lay up to 7 clutches (usually 3-4) of eggs every 2-4 years, with up to an average of 110-115 eggs per nest, of which a small percentage is expected to survive to sexual maturity. The anticipated lethal interactions are expected to occur anywhere in the action areas and only affect a small portion of the DPS, and sea turtles generally have large ranges in which they disperse; thus, no reduction in the distribution of green sea turtles within the NA DPS is expected from these captures.

Whether the reductions in numbers and reproduction of this species would appreciably reduce the species' likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends. In the Status of Species of this Opinion, we presented the status of the DPS, outlined threats, and discussed information on estimates of the number of nesting females and nesting trends at primary nesting beaches. In the Environmental Baseline, this Opinion outlined the past and present impacts of all state, federal, or private actions and other human activities in or having effects in the action area that have impacted and continue to impact this DPS. The Cumulative Effects section of this Opinion discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action area.

Seminoff et al. (2015) estimated that there are greater than 167,000 nesting females in the NA DPS. The nesting at Tortuguero, Costa Rica, accounts for approximately 79% of that estimate (approximately 131,000 nesters), with Quintana Roo, Mexico, (approximately 18,250 nesters; 11%), and Florida, USA (approximately 8,400 nesters; 5%) also accounting for a large portion of the overall nesting (Seminoff et al. 2015).

At Tortuguero, Costa Rica, the number of nests laid per year from 1999 to 2010 increased, despite substantial human impacts to the population at the nesting beach and at foraging areas (Campell and Lagueux 2005; Troëng 1998; Troëng and Rankin 2005).

Nesting locations in Mexico along the Yucatan Peninsula also indicate the number of nests laid each year has increased (Seminoff et al. 2015). In the early 1980s, approximately 875 nests/year were deposited, but by 2000 this increased to over 1,500 nests/year (NMFS and USFWS 2007a) (NMFS and USFWS 2007a). By 2012, more than 26,000 nests were counted in Quintana Roo (J. Zurita, CIQROO, unpubl. data, 2013, in Seminoff et al. 2015)

In Florida, most nesting occurs along the Atlantic coast of eastern central Florida, where a mean of 5,055 nests were deposited each year from 2001 to 2005 (Meylan et al. 2006) and 10,377 each year from 2008 to 2012 (B. Witherington, Florida Fish and Wildlife Conservation Commission, pers. comm., 2013). As described in the Section 3.2, according to data collected from Florida's index nesting beach survey from 1989-2019, green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high in 2019 with almost 41,000 nests on the Index Nesting Beaches (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/). In-water studies conducted over 24 years in the Indian River Lagoon, Florida, suggest similar increasing trends, with green sea turtle captures up 661% (Ehrhart et al. 2007). Similar in-water work at the St. Lucie Power Plant site revealed a significant increase in the annual rate of capture of immature green sea turtles over 26 years (Witherington et al. 2006).

In summary, nesting at the primary nesting beaches has been increasing over the course of decades, against the background of the past and ongoing human and natural factors (environmental baseline) that have contributed to the current status of the species. We believe these nesting trends are indicative of a species with a high number of sexually mature individuals. Since the abundance trend information for NA DPS green sea turtles is clearly increasing, we believe the potential lethal take of 6 green sea turtles from the NA DPS per year attributed to the proposed actions will not have any measurable effect on that trend. After analyzing the magnitude of the effects of the proposed actions, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe the proposed actions are not reasonably expected to cause an appreciable reduction in the likelihood of survival of the green sea turtle NA DPS in the wild.

Recovery

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

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

Green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high in 2019 with almost 41,000 nests on the Index Nesting Beaches (https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/). (Please refer to section 3 for more details on the dynamics of the trend increase.) There are currently no estimates available specifically addressing changes in abundance of individuals on all foraging grounds. Given the clear increases in nesting, however, it is likely that numbers on foraging grounds have increased.

The potential lethal take of up to 6 NA DPS green sea turtles per year will result in a reduction in numbers when captures occur, but it is unlikely to have any detectable influence on the recovery objectives and trends noted above, even when considered in the context of the of the Status of the Species, the Environmental Baseline, and Cumulative Effects discussed in this Opinion. Thus, the proposed actions will not impede achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of NA DPS green sea turtles' recovery in the wild.

Conclusion

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

7.1.2 Green Sea Turtle SA DPS

The proposed actions, combined, may result in up to 6 green sea turtle mortalities from the SA DPS each year. The potential lethal take would reduce the number of green sea turtles, compared to their numbers in the absence of the proposed actions, assuming all other variables remained the same. Lethal interactions would also result in a potential reduction in future reproduction, assuming the individuals caught would at least in some years be female and would have survived otherwise to reproduce. For example, as discussed in this Opinion, an adult green sea turtle can lay up to 7 clutches (usually 3-4) of eggs every 2-4 years, with up to an average of 110-115 eggs per nest, of which a small percentage is expected to survive to sexual maturity. The anticipated lethal interactions are expected to occur anywhere in the action areas and sea turtles generally have large ranges in which they disperse; thus, no reduction in the distribution of green sea turtles within the SA DPS is expected from these mortalities.

Whether the reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends. In the Status of Species of this Opinion, we presented the status of the DPS, outlined threats, and discussed information on estimates of the number of nesting females and nesting trends at primary nesting beaches. In the Environmental Baseline, this Opinion considered the past and present impacts of all state, federal, or private actions and other human activities in or having effects in, the action areas that have impacted and continue to impact this DPS. The Cumulative Effects section of this Opinion considered the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action areas.

In Section 3, we summarized available information on number of nesters and nesting trends at SA DPS beaches. Seminoff et al. (2015) estimated that there are greater than 63,000 nesting females in the SA DPS, though they noted the adult female nesting abundance from 37 beaches could not be quantified. The nesting at Poilão, Guinea-Bissau, accounted for approximately 46% of that estimate (approximately 30,000 nesters), with Ascension Island, United Kingdom, (approximately 13,400 nesters; 21%), and the Galibi Reserve, Suriname (approximately 9,400 nesters; 15%) also accounting for a large portion of the overall nesting (Seminoff et al. 2015).

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

Seminoff et al. (2015) reported that the nesting concentration at Galibi Reserve and Matapica in Suriname was stable from the 1970s through the 1980s. From 1975–1979, 1,657 females were counted (Schulz 1982), a number that increased to a mean of 1,740 females from 1983–1987 (Ogren 1989), and to 1,803 females in 1995 (Weijerman et al. 1998). Since 2000, there appears to be a rapid increase in nest numbers (Seminoff et al. 2015).

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

Nesting at the primary nesting beaches has been increasing over the course of decades, against the background of the past and ongoing human and natural factors (environmental baseline) that have contributed to the current status of the species. We believe these nesting trends are indicative of a species with a high number of sexually mature individuals. Since the abundance trend information for green sea turtles is clearly increasing, we believe the potential lethal take of 6 green sea turtle from the SA DPS every year attributed to the proposed actions will not have any measurable effect on that trend. After analyzing the magnitude of the effects of the proposed actions, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe the proposed actions are not reasonably expected to cause an appreciable reduction in the likelihood of survival of the green sea turtle SA DPS in the wild.

Recovery

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

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

The nesting recovery objective is specific to the NA DPS, because it concerns nesting in Florida, but demonstrates the importance of increases in nesting to recovery. As previously stated, nesting at the primary SA DPS nesting beaches has been increasing over the course of decades. There are currently no estimates available specifically addressing changes in abundance of individuals on foraging grounds. Given the clear increases in nesting and in-water abundance, however, it is likely that numbers on foraging grounds have increased.

The potential lethal take of up to 6 SA DPS green sea turtles every year will result in a reduction in numbers when capture occurs, but it is unlikely to have any detectable influence on the trends noted above, even when considered in context with the Status of the Species, the Environmental Baseline, and Cumulative Effects discussed in this Opinion. Thus, the proposed actions will not impede achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of the SA DPS of green sea turtles' recovery in the wild.

Conclusion
The lethal takes of green sea turtles associated with the proposed actions are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the SA DPS of green sea turtle in the wild.

7.2 Hawksbill Sea Turtle

The proposed actions, combined, may result in 3 hawksbill sea turtle mortalities each year. The lethal take of 3 hawksbill sea turtles per year associated with the proposed actions represents a reduction in numbers. These lethal takes could also result in a future reduction in reproduction as a result of lost reproductive potential, as some of these individuals may be females who would have survived other threats and reproduced in the future, thus eliminating each female individual's contribution to future generations. For example, an adult female hawksbill sea turtle can nest every 2-3 years and generally lay 3-5 nests per season with clutches of as many as 250 eggs per nest, though nests in the U.S. Caribbean more typically contain approximately 140 eggs. Thus the loss of adult female sea turtles could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. A reduction in the distribution of hawksbill sea turtles is not expected from lethal takes attributed to the proposed actions. Because all the potential interactions are expected to occur at random throughout the proposed action areas, which accounts for a tiny fraction of the species' overall range, the distribution of hawksbill sea turtles is expected to be unaffected.

Whether the reductions in hawksbill sea turtle numbers and reproduction attributed to the proposed actions would appreciably reduce the likelihood of survival for hawksbills depends on what effect these reductions in numbers and reproduction would have on overall population sizes and trends, i.e., whether the estimated reductions, when viewed within the context of the environmental baseline, status of the species, and cumulative effects are of such an extent that adverse effects on population dynamics are appreciable. In the Status of Species of this Opinion, we considered the status of the DPS, outlined threats, and discussed information on estimates of the number of nesting females and nesting trends at primary nesting beaches. In the Environmental Baseline, this Opinion considered the past and present impacts of all state, federal, or private actions and other human activities in or having effects in, the action area that have impacted and continue to impact this DPS. The Cumulative Effects section of this Opinion considered the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the action areas.

In the absence of any total population estimates for hawksbill sea turtles, nesting trends are the best proxy we have for estimating population changes. The 5-year status review estimated between 22,000 and 29,000 adult females nested each year at the time of its writing in 2007 (NMFS 2013); this estimate does not include juveniles of either sex or mature males. As discussed Section 3, overall nesting at key beaches in the action areas have shown increasing trends. We believe the potential loss of up to 3 hawksbill sea turtles every year will not have any detectable effect on the population, distribution, or reproduction of hawksbills. Therefore, we do not believe the proposed actions will cause an appreciable reduction in the likelihood of survival.

Recovery

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

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

Nesting populations are increasing at the Puerto Rico (Mona Island) and U.S. Virgin Islands (Buck Island Reef National Monument) index beaches. Also in the U.S. Caribbean, additional nesting beaches are now being more systematically monitored to allow for future population trend assessments.

The status of adults, subadults, and juveniles on foraging grounds is being monitored via in-water research. An in-water research project at Mona Island, Puerto Rico, has been ongoing for 15 years. However, abundance indices have not yet been incorporated into a rigorous analysis or a published trends assessment, as of yet. In addition, standardized in-water surveys have been initiated within the wider Caribbean (e.g., Pearl Cays, Nicaragua), but the time series is not long enough to detect a trend. In Florida, 2 in-water projects have been ongoing in Key West and Marquesas Keys conducted by the In-Water Research Group and Palm Beach County (NMFS 2013).

The proposed actions could cause the loss of up to 3 hawksbill sea turtles every year and the animals may or may not be an adult and may or may not be a female. We determined the potential lethal captures associated with the proposed actions would not have any detectable influence on the magnitude of nesting population trends. While information on trends for adults, subadults, and juveniles at key foraging areas is not yet available, we also believe it is unlikely the potential removal of 3 hawksbills every year will have any detectable influence over the numbers of adults, subadults, and juveniles occurring at foraging areas. Thus, we believe the proposed actions are not likely to impede the recovery objectives above and will not result in an appreciable reduction in the likelihood of hawksbill sea turtles’ recovery in the wild.

Conclusion

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

7.3 Nassau Grouper

As discussed in section 5, as a result of the proposed actions, we expect a total of 4 Nassau grouper to be killed annually.

The potential lethal capture of 4 Nassau grouper annually would reduce the number of fish in the population. The species consists of a single population over a broad geographic range. As discussed in this Opinion, insufficient stock assessment or population estimates exist for the Nassau grouper, therefore there is insufficient total population abundance data or trend data to which the 4 annual mortalities can be compared. Instead, the population at spawning aggregation sites will be used as a proxy for total population abundance. Additionally, data does not exist regarding the size class, age, sex ratio, or reproductive status of the 4 annual expected mortalities, therefore it is not possible to incorporate these variables into this analysis. As mentioned in the *Status of the Species*, during the first U.S. survey of the fishery resources of Puerto Rico, the Nassau grouper was noted as a common and very important food fish. By 1970, Nassau grouper was still the fourth most common shallow-water species landed in Puerto Rico, and it was common in the reef fish fishery of the Virgin Islands, where an aggregation in the 1970s contained an estimated 2,000-3,000 individuals. During the 1980s, port sampling in the USVI showed that Nassau grouper accounted for 22% of grouper landings with 85% of the Nassau grouper catch coming from spawning aggregations. By 1981, "the Nassau grouper ha(d) practically disappeared from the local catches and the ones that d(id) appear (were) small compared with previous years" and by 1986, the Nassau grouper was considered commercially extinct in the U.S. Virgin Islands and Puerto Rico region. Although there are few data on historic abundance of Nassau grouper off the U.S. mainland, it appears that abundance was once high in southern Florida and anecdotal reports from spearfishers noted large daily catches in the 1950s. It appears that Nassau grouper were once caught in much greater numbers from the upper Florida Keys and the Bahamas and the species was reported frequently at Alligator Reef in the Florida Keys. Additionally, historically, Nassau grouper was a component of the grouper fishery in Florida, suggesting once healthy (sub)population(s) in southeastern U.S. mainland waters that does not exist today. While we do not have exact population numbers for this species, and while we recognize that numbers are reduced, the best proxy of population numbers available to assess the impacts of the proposed actions is the number of fish at spawning aggregation sites. This data is not exact, but the number of fish at these locations is expected to be in the thousands of animals (NMFS 2013b). Estimates of the population at spawning aggregation sites pertain to the number of reproductive adults, which are a subset of the entire population. The removal of 4 fish annually would result in a future reduction in reproduction as a result of lost reproductive potential, as some of these individuals could potentially have reproduced in the future. However, the animals taken would represent a small fraction of the entire population (which ranges throughout the Caribbean). We do not believe the reductions in numbers and reproduction resulting from the proposed actions are likely to reduce the population's ability to persist into the future.

Farmer (2016) summarized the factors leading to listing the Nassau grouper and discussed two "high risk" threats that are key to the status of the species and will continue to affect it over the foreseeable future—fishing at spawning aggregations and inadequate law enforcement (especially to control fishing on aggregations). Farmer (2016) stated that existing regulatory mechanisms and law enforcement have not been effective in preventing fishing at many spawning aggregation sites. Many countries have few, if any, specific Nassau grouper regulations. Instead they rely on general fisheries regulations (e.g., Anguilla, Antigua-Barbuda, Colombia, and Cuba all rely only on size limits, while Guadeloupe and Martinique, Honduras, Jamaica, Mexico, St. Lucia, and the Turks and Caicos rely on a variety of general fishing regulations). Additionally, where Nassau grouper-specific regulations do exist, law enforcement in many foreign countries is less than adequate, thus rendering the regulations ineffective.

Harvest of the Nassau grouper in the United States (Florida and the U.S. Caribbean) has been illegal since the early 1990s and is not authorized under the proposed actions. Since 1993, a ban on fishing for and possessing Nassau grouper was implemented for the state of Florida. Additional bans have been enacted in all U.S. state waters within the species' confirmed range, namely Puerto Rico and the USVI. Within federal waters, no person may fish for or possess Nassau grouper in or from the Puerto Rico or the USVI EEZ, and this prohibition is continued under the proposed actions. Such fish caught must be released immediately with a minimum of harm. Therefore, no animals are legally targeted in federal waters in Puerto Rico or the USVI. While incidental capture occurs in the fishery, the proposed actions do not authorize fishing during the times when certain grouper species are expected to form spawning aggregations, and these seasonal and area closures (e.g., Grammanik Bank closed season) prevent fishing on known Nassau grouper spawning aggregations. In addition, the Hind Bank Marine Conservation District, which is closed to fishing year round and will remain so under the proposed actions, protects a historic Nassau grouper spawning site. Thus, the proposed actions will protect against one of the main reasons the Nassau grouper was listed. Furthermore, mortalities due to fishing in federally managed fisheries in the U.S. Caribbean have been significantly reduced since fishing for Nassau grouper was prohibited in U.S. federal waters. The prohibition will continue under the proposed actions and we expect it to be protective.

The abundance of Nassau grouper has been dramatically reduced in relation to historical records, however, the information obtained during the status review indicates the species is not currently in danger of extinction (though reduced in number, the species maintains its historical range and still forms spawning aggregations at some sites including Mexico, Bahamas, Cuba, and Belize). Additionally, recent evidence suggests that spawning is occurring at what may be reconstituted or novel spawning sites in both Puerto Rico and the USVI (Hill and Sadovy de Mitcheson 2013). Suspected or anecdotal evidence also identifies spawning aggregations in Los Roques, Venezuela (Boomhower et al. 2010) and Old Providence in Colombia's San Andrés Archipelago (Prada et al. 2004). Although the proposed actions would reduce abundance of Nassau grouper, spawning is still occurring where adequate protections are effectively being implemented. Additionally, conservation efforts in some nations (U.S., including Puerto Rico and the USVI, and Belize) have almost certainly prevented further declines. The proposed actions are not affecting or contributing to the key threats facing the species because the proposed actions continue to prohibit fishing on spawning aggregations, and enforcement of these provisions is considered adequate. We do not believe the reductions in numbers and reproduction resulting from the proposed actions are likely to reduce the population's ability to persist into the future, and the proposed actions will not appreciably reduce the likelihood of the Nassau grouper's survival within its range.

Recovery

We next considered whether the proposed actions are likely to impede the recovery of Nassau grouper. Because the Nassau grouper has only recently been listed, no recovery plan yet exists for the species. However, a recovery outline exists for this species. The outline and the information in the listing rule for this species analyzed the threats facing this species and those that are key to recovery[51] include:

- fishing at spawning aggregations: and
- inadequate law enforcement (particularly at spawning aggregations)

The proposed actions prohibit fishing for Nassau grouper, and prohibit fishing at known spawning aggregation sites (e.g., Grammanik Bank seasonal closure and the Hind Bank Marine Conservation District year-round closure). While incidental take does occur in the fisheries, these measures afford protections to the species and its spawning aggregations. Therefore, the fisheries managed under the Puerto Rico, St. Thomas/St. John, and St. Croix FMPs are protecting against this major threat on recovery. Law enforcement is and will continue to be important to enforce these protections, and is expected to be adequate. Therefore, the fisheries do not negatively contribute to the impact of this threat as it relates to recovery. We conclude the proposed actions will not appreciably diminish the likelihood of recovery for the Nassau grouper.

Conclusion

In conclusion, we believe that the effects associated with the proposed actions are not expected to cause an appreciable reduction in the likelihood of both the survival and recovery of the Nassau grouper.

7.4 Scalloped Hammerhead– Central and Southwest Atlantic DPS

The proposed actions may result in 1 lethal scalloped hammerhead shark take annually. The loss of 1 scalloped hammerhead will reduce the number of scalloped hammerheads as compared to the number of scalloped hammerheads that would have been present in the absence of the proposed actions assuming all other variables remained the same. This lethal take could also result in the loss of reproduction value as compared to the reproductive value in the absence of the proposed actions, if a female is taken. The death of a female eliminates an individual's contribution to future generations, and the proposed actions would result in a reduction in future scalloped hammerhead reproduction. While scalloped hammerhead sharks are less migratory than other sharks, they are still wide-ranging. We believe the potential loss of animals would not affect the distribution of the species.

There is currently no accurate population estimate for the Central and Southwest Atlantic DPS of scalloped hammerhead sharks. However, Miller et al. (2014) concluded that abundance numbers for this DPS are likely similar to, and probably worse than, those found in the Northwest Atlantic

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[51] The recovery outline also discusses that loss of nearshore habitat and changes in water quality could negatively impact survival and growth of local recruits. However, the information needed to accurately identify these settlement and growth areas for local recruitment of Nassau grouper is not currently available. The proposed actions are not expected to affect nearshore habitat or water quality.

and Gulf of Mexico DPS. The virgin population estimates for the Northwest Atlantic and Gulf of Mexico DPS ranged from 142,000 and 169,000 individuals (range 116,000-260,000) (Hayes et al. 2009). The population estimates for the most recent time period (2005) estimate a much smaller population: 24,850-27,900 individuals (Hayes et al. 2009). Since Miller et al. (2014) concluded that abundance numbers for this DPS are likely similar to, and probably worse than, those found in the Northwest Atlantic and Gulf of Mexico DPS, we will conservatively base our analysis on the 24,850 population number. The annual loss of 1 scalloped hammerhead will not significantly decrease the populations within the Central and Southwest Atlantic DPS as this is a limited amount of loss relative to the estimated population size, nor will it change their distribution. Thus, we believe the proposed actions are not likely to appreciably reduce the likelihood of survival of the Central and Southwest Atlantic DPS of scalloped hammerhead sharks in the wild.

<u>Recovery</u>

The following analysis considers the effects of expected take on the likelihood of recovery in the wild. Since scalloped hammerhead sharks have just recently been listed, a recovery plan for them is not yet available. However, recovery is the process by which the ecosystems of a species are restored and the threats to the species are removed. Restoring ecosystems and eliminating threats will help support self-populating and self-regulating populations so they can become persistent members of the native biological communities (USFWS and NMFS 1998). Thus, the first step in recovering a species is to reduce identified threats; only by alleviating threats can lasting recovery be achieved. The Final Listing Rule (79 FR 38213, July 3, 2014) noted the following potential threats to the Central and Southwest Atlantic DPS of scalloped hammerhead sharks:

1) Overutilization in artisanal fisheries, north of Brazil, that operate in nearshore and inshore environments that are likely nursery areas, and overutilization in artisanal and commercial fisheries within Brazil that target scalloped hammerhead sharks.
2) Operation of domestic artisanal fisheries and foreign commercial fisheries in areas without adequate fisheries regulations and operation of domestic and foreign fisheries in areas without capacity to enforce existing fishery regulations.
3) Scalloped hammerhead sharks' physiology makes them very susceptible to mortality in fishing gear. They often suffer very high at-vessel fishing mortality (e.g., Morgan and Burgess, 2007; Macbeth et al., 2009), and their schooling behavior increases their likelihood of being caught in large numbers.

The proposed actions will not contribute to the overutilization in Brazil, and the FMPs will manage and regulate fishing that may affect the species. Additionally, as discussed previously, the lethal take from proposed actions is not likely to impede the Central and Southwest Atlantic DPS of scalloped hammerhead sharks from continuing to survive. Therefore, we believe the proposed actions are not likely to impede the recovery of the species, and will not result in an appreciable reduction in the likelihood of the Central and Southwest Atlantic DPS of scalloped hammerhead shark's recovery in the wild.

Conclusion

The effects from proposed actions are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the Central and Southwest Atlantic DPS of scalloped hammerhead sharks in the wild.

7.5 Oceanic Whitetip Shark

The proposed actions may result in 29 oceanic whitetip shark lethal takes annually. There is currently no accurate population estimate for oceanic whitetip sharks. Oceanic whitetip sharks are wide-ranging and can be found worldwide, with no present indication of a range contraction. While a global population size estimate or trend for the oceanic whitetip shark is currently unavailable, numerous sources of information, including the results of a recent stock assessment and several other abundance indices are available to infer and assess current regional abundance trends of the species. Relative abundance of oceanic whitetip sharks may have stabilized in the North Atlantic since 2000 and in the Gulf of Mexico and Caribbean since the late 1990s at a significantly diminished abundance (Cortés et al. 2007; Young et al. 2016). Notably, the subject fisheries are not a new impact to the species. The proposed actions continue to manage fisheries that have been ongoing since the 1980s, before the population stabilized, so the past and present impacts are part of the baseline.

If females are taken, the loss of 29 oceanic whitetip sharks annually could result in the loss of reproduction value as compared to the reproductive value in the absence of the proposed actions. While we have no reason to believe the proposed actions will disproportionately affect females or adults, the loss of an adult female oceanic whitetip shark could preclude the production of future progeny. The death of a female eliminates an individual’s contribution to future generations, and the proposed actions would result in a reduction in future oceanic whitetip shark reproduction.

Likewise, the loss of those individuals would represent a reduction in numbers compared to the number of oceanic whitetip sharks that would have been present in the absence of the proposed actions assuming all other variables remained the same. However, we believe that the loss in numbers and reproduction are likely small relative to the species’ size and reproductive potential. Additionally, the populations within the action area considered in this Opinion are thought to have stabilized since the late 1990s, during which time the fisheries that will continue to be managed under the proposed actions, and its impacts, were already occurring. There is no basis to believe that the loss of 29 individuals annually will reduce the distribution of the species. The species is widespread and wide-ranging, and the takes occur throughout the action area.

Therefore, we conclude that the proposed actions are not expected to have a population-level impact on the reproduction, numbers, or distribution of oceanic whitetip shark and we believe the proposed actions are not likely to appreciably reduce the likelihood of survival of the oceanic whitetip shark in the wild.

Recovery

Since oceanic whitetip sharks were recently listed, a recovery plan for them is not yet available. However, recovery is the process by which the ecosystems of a species are restored and the threats to the species are removed. Restoring ecosystems and eliminating threats will support self-populating and self-regulating populations so they can become persistent members of the native biological communities (USFWS and NMFS 1998). Thus, the first step in recovering a species is to reduce identified threats. The Final Listing Rule (83 FR 4153, January 30, 2018) noted the following potential threats to the oceanic whitetip shark: In the Northwest Atlantic, the oceanic whitetip is caught incidentally as bycatch by a number of fisheries, including (but not limited to) the pelagic longline fishery for Atlantic HMS, the Cuban "sport" fishery (i.e., private artisanal and commercial), and the Colombian oceanic industrial longline fishery operating in the Caribbean. Oceanic whitetip sharks are also a preferred species for their large, morphologically distinct fins, as they obtain a high price in the Asian fin market, and thus they are valuable as incidental catch for the international shark fin trade. Oceanic whitetip sharks possess life history characteristics that increase their vulnerability to harvest, including slow growth, relatively late age of maturity, and low fecundity. The species' low genetic diversity in concert with steep global abundance declines and ongoing threats of overutilization may pose a viable risk to the species in the foreseeable future. While incidental take does occur in the fisheries associated with the proposed actions, we conclude that the proposed actions are not expected to have a population-level impact on the reproduction, numbers, or distribution of oceanic whitetip shark and we believe the proposed actions are not likely to appreciably reduce the likelihood of survival of the oceanic whitetip shark in the wild. NMFS does not believe that the level of take associated with the proposed actions will significantly affect population dynamics such that reproduction and population numbers will be impacted at a level that will impede recovery efforts for this species. The proposed actions will not impede the process of restoring the ecosystems that affect oceanic whitetip sharks.

In addition, the proposed actions are not likely to interfere with the preliminary recovery strategy for the species. While there is no recovery plan at this time, NMFS has developed a recovery outline, which guides initial recovery actions while ensuring that future recovery options are not precluded due to a lack of interim planning. It presents a preliminary strategy for recovery of the species, as well as recommended high priority actions to stabilize and recover the species. In advance of an approved recovery plan, the initial focus of the interim recovery program will be two-fold: 1) to stabilize population trends through reduction of threats, such that the species is no longer declining throughout a majority of its range and 2) to gather additional information through research and monitoring on the species' current distribution and abundance; reproductive periodicity and seasonality; location of breeding and nursery grounds; and mortality rates in commercial fisheries (including at-vessel and post-release mortality). Because the oceanic whitetip shark largely occurs in waters outside of U.S. jurisdiction, international coordination will be critical to ensuring recovery of the species. Therefore, to be effective, recovery actions would need to be undertaken throughout the species' range, both domestically and internationally. The proposed action is not expected to have a population-level impact on oceanic whitetip sharks and thus would not impede the first goal of the recovery outline. The second goal, to gather additional information, would not be impeded by the proposed action.

For all of these reasons, we believe the proposed actions are not likely to impede the recovery of the species, and will not result in an appreciable reduction in the likelihood of the oceanic whitetip shark's recovery in the wild.

Conclusion
The effects from proposed actions are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of oceanic whitetip sharks in the wild.

7.6 Staghorn Coral

As calculated in the Effects Section of this Opinion, approximately 462.4 ft^2 annually of this coral will be lethally taken by fishing gear deployment and anchoring associated with the proposed actions. Staghorn coral is most commonly found between 0-30 m. Based on staghorn coral cover per reef site in each of the island areas (introduced in the Effects Section), and the amount of coral reef off Puerto Rico, St. Thomas/St. John, and St. Croix,[52] we calculate the amount of staghorn coral off all three island areas (25,572,759.1 ft^2). Therefore, approximately 0.0018% of the staghorn coral in all waters off Puerto Rico, St. Thomas/St. John, and St. Croix, combined, is taken annually by the proposed actions (462.4 ft^2 staghorn coral taken under the proposed actions annually / 25,572,759.1 ft^2 staghorn coral in all waters off Puerto Rico, St. Thomas/St. John, and St. Croix). This amount is expected to have a negligible impact on coral cover (numbers) and reproduction in the U.S. Caribbean waters, and this percentage has even less impact within the context of the greater amount of this species found throughout its range. This minimal reduction in the amount of staghorn coral in the U.S. Caribbean is unlikely to affect the species' distribution.

Additionally, since asexual reproduction via fragmentation is a key mode of reproduction for staghorn coral, contact with traps and anchors themselves does not necessarily mean a reduction in reproduction would occur. It is unlikely that contact with a trap or anchor will cause damage significant enough to remove the potential for asexual reproduction (i.e., traps pulverizing corals). Therefore, given the manner in which traps and fishing vessel anchors are used in the action areas, we do not expect a reduction in reproduction from direct interactions.

Taken together, we do not believe effects of trap use and fishing vessel anchoring from the proposed actions will reduce the distribution of the species or have more than a minimal (negligible) impact on the amount of coral (numbers) or coral reproduction.

Harvest of herbivorous fish is allowed under the proposed actions, reducing the number of herbivorous fish available to graze on macroalgae. It is unclear whether the effects from the reduction of macroalgae mediation by herbivorous fish is directly reducing the areal coverage (numbers) of staghorn coral. Nugues and Bak (2006) point out that many of the studies addressing the adverse effects of algae on corals are either observational, (i.e., describing the death of the coral tissue overgrown by algae), or correlative, showing opposite trends in coral and algal abundance over time (McCook et al. 2001). Other studies that specifically investigated

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[52] The amount of coral reef off Puerto Rico is 14,001,731,651 ft^2, the amount of coral reef off St. Thomas/St. John is 2,643,221,773 ft^2, and the amount of coral reef off St. Croix is 3,450,401,472 ft^2 (OCM Partners 2020).

the correlation between macroalgae and coral mortality have produced mixed results, with some macroalgae apparently having little to no effect on coral colony mortality (Jompa and McCook 1998, River and Edmunds 2001), while others appear to actually overgrow and kill corals (i.e., Jompa and McCook 2002a, Nugues and Bak 2006). The effects of macroalgae on corals also appear to be dependent upon which macroalgae and corals species are interacting (Nugues and Bak 2006). Nugues and Bak (2006) reported that algae appeared to cause mortality in a very limited number of coral species. Of the six coral species they tested (one was the ESA-listed *Montastraea* (now *Orbicella*) *franksi*), only one species (*Agaricia agaricites*) experienced greater tissue mortality after exposure to brown macroalgae *Lobophora variegata*, which they indicate is a "relatively potent competitor against corals" (Nugues and Bak 2006). There is currently no specific data on the potential effects of macroalgae on staghorn coral mortality, but it appears unlikely that macroalgae is causing direct morality of this species. Thus, we believe a reduction in number of staghorn colonies is not likely as a result of effects to corals from the harvest of herbivorous fish associated with the proposed actions.

With respect to effects on the species' reproduction, it appears that a phase shift toward high macroalgae and low coral cover is occurring, or has occurred in the U.S. Caribbean. High macroalgae reduces the available substrate for coral settlement (reproduction). We believe the major stressors (i.e., global climate change, increasing sea surface temperatures, ocean acidification, impacts to water quality (principally nutrient input) coral diseases, damage from hurricanes), in combination with the *Diadema* die-off are the primary causes of this phase shift. The data indicate that Caribbean-wide changes in benthic assemblages were likely precipitated by massive coral mortality events, namely the loss of elkhorn and staghorn coral from white band disease (WBD) (as macroalgae are generally unable to actively overgrow and kill live corals) (Aronson and Precht 2001). Coral mortality and the die-off of *Diadema* created conditions conducive to macroalgae growth and spread, and macroalgae was able to out-compete other corals to re-occupy substrate space left barren by coral mortalities. Over several decades, the re-occupation of substrate previously suitable for coral asexual or sexual reproduction by macroalgae established a feedback loop resistant to coral re-colonization (Hughes and Connell 1999).

The harvest of herbivorous fish under the proposed actions is likely exacerbating the phase-shift that is occurring or has occurred, however, based on available research noted above, it is unlikely to be a primary factor causing the degradation of features necessary for reproduction. In the last NMFS analysis of these effects (NMFS 2011a) of the proposed actions, NMFS concluded that the existing data appeared to indicate that the levels of herbivorous fish harvest were unlikely to be a primary contributor to growth and spread of macroalgae. The proposed actions include the harvest of herbivorous species under specific limits (e.g., ACLs, recreational bag limits, and size limits). The extent to which the harvest of herbivorous fish is adversely affecting the availability of substrate for reproduction remains unquantifiable. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful coral reproduction. We also acknowledge that this reduction in suitable substrate negatively affects the species' chances of surviving and recovering in the wild. Because of these potential impacts, through this Opinion, we will continue to require monitoring actions be taken to help better understand the harvest of herbivorous. However, we believe herbivorous fish harvest is only one of many factors that have led to high macroalgal levels. Thus, the proposed actions appear to have only a small, incremental role in what is believed to be only a moderate threat (NMFS 2005

Status Review) to the species. Since we believe the effects of the proposed actions on habitat are not currently a primary cause of macroalgal growth, we believe the proposed actions are causing a very minimal impact on the species' reproduction. In addition, as we explained above, macroalgae appear unlikely to cause direct mortality to adult staghorn coral. Thus, the number of colonies available for asexual or sexual reproduction appears unlikely to change as result of harvest of herbivorous fish. Unlike lethal takes of other species that may forever remove a sexually mature individual for the population, macroalgae appears to affect sexual and asexual reproduction success, but not preclude it from occurring eventually when favorable conditions return, as removal of reproductive, adult colonies from the population would. Based on the above information we believe that the proposed actions are likely adversely affecting staghorn coral from the harvest of herbivorous reef fish, but those effects are minimally contributing to reductions in reproduction.

With respect to the species' distribution, we do not anticipate that any colonies will actually be lost due to macroalgal growth associated with the harvest of herbivorous fish associated with the proposed actions. Therefore, we do not anticipate any change in the overall distribution of corals as a result of loss of the coral.

Macroalgal growth can also affect coral larval settlement, which could affect distribution. The proposed actions are the continued harvest of herbivorous species under specific limits, including ACLs, through the FMPs. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful settlement. Since we believe the effects of the proposed actions are not currently a primary cause in macroalgal growth, we believe the incremental impacts caused by the proposed actions are causing an extremely small reduction in the amount of substrate available for settlement, and this could affect the species' distribution. However, we do not believe the proposed actions will cause a reduction in the species' overall distribution as the species are widely distributed beyond the action area, which is only a small portion of its overall range.

Overall, we expect a very small reduction reproduction from the effects of the proposed actions. We expect no reductions in numbers or distribution. Because we anticipate very small impacts only to species' reproduction within the action areas, and because the species is widely distributed, we do not expect the proposed action will alter the trends and trajectory of the species, or appreciably reduce the likelihood of staghorn coral's survival in the wild.

Recovery

The Recovery Plan for elkhorn and staghorn corals (NMFS 2015) lists the following major actions:

- Improve understanding of population abundance, trends, and structure through monitoring and experimental research.
- Develop and implement appropriate strategies for population enhancement through restocking and active management.
- Implement ecosystem-level actions to improve habitat quality and restore keystone species (e.g., sea urchins) and functional processes such as herbivory to sustain adult colonies and promote successful natural recruitment in the long term.

- Curb ocean warming and acidification impacts to health, reproduction, and growth, and possibly curb disease threats, by reducing atmospheric greenhouse gas concentrations.
- Reduce locally-manageable stress and mortality threats (e.g., predation, anthropogenic physical damage, acute sedimentation, nutrients, contaminants).
- Determine coral health risk factors and their inter-relationships and implement mitigation or control strategies to minimize or prevent impacts to coral health.

The proposed actions would not negatively affect (and may actually help) efforts to understand or monitor this coral, will not impact efforts relating to strategies for population enhancement, will not affect efforts to curb ocean warming and acidification impacts, or affect efforts to determine coral health risk factors and their inter-relationships and implement mitigation or control strategies to minimize or prevent impacts to coral health. Restrictions on harvest of herbivore species under the proposed actions, including the limits on the catch of herbivorous fish and the prohibition on the harvest of species of sea urchins, would contribute to the third recovery objective noted above, efforts to implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes such as herbivory to sustain adult colonies and promote successful natural recruitment in the long term. Regarding the fifth recovery objective noted above, the proposed actions could cause the coral mortality. However, while the proposed actions would not reduce locally-manageable stress and mortality on the whole, though they do employ some protective measures, like anchoring protections for certain areas, we determined the potential lethal take associated with trap use and anchoring, and the effect of the harvest of herbivorous fish on reproduction, associated with the proposed actions would not have any detectable influence on the population. We believe the proposed actions are not likely to impede the recovery objectives above and will not result in an appreciable reduction in the likelihood of this coral's recovery in the wild.

Conclusion

The effects associated with the proposed actions are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the staghorn coral in the wild.

7.7 Elkhorn Coral

As calculated in the Effects Section of this Opinion, approximately 310.5 ft^2 annually of this coral will be lethally taken by fishing gear deployment and anchoring associated with the proposed actions. Elkhorn coral is most commonly found between 0-30 m. Based on elkhorn coral cover per reef site in each of the island areas (introduced in the Effects Section), and the amount of coral reef off Puerto Rico, St. Thomas/St. John, and St. Croix,[53] we calculate the amount of elkhorn coral off all three island areas (29,420,688.1 ft^2). Therefore, approximately 0.0011% of the elkhorn coral in all waters off Puerto Rico, St. Thomas/St. John, and St. Croix, combined, is taken annually by the proposed actions (310.5 ft^2 elkhorn coral taken under the proposed actions annually / 29,420,688.1 ft^2 elkhorn coral in all waters off Puerto Rico, St. Thomas/St. John, and St. Croix). This amount is expected to have a negligible impact on coral

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[53] The amount of coral reef off Puerto Rico is 14,001,731,651 ft^2, the amount of coral reef off St. Thomas/St. John is 2,643,221,773 ft^2, and the amount of coral reef off St. Croix is 3,450,401,472 ft^2 (OCM Partners 2020).

cover (numbers) and reproduction in the U.S. Caribbean waters, and this percentage has even less impact within the context of the greater amount of this species found throughout its range. This minimal reduction in the amount of elkhorn coral in the U.S. Caribbean is unlikely to affect the species' distribution.

Additionally, since asexual reproduction via fragmentation is a key mode of reproduction for elkhorn coral, contact with traps and anchors themselves does not necessarily mean a reduction in reproduction would occur. It is unlikely that contact with a trap or anchor will cause damage significant enough to remove the potential for asexual reproduction (i.e., traps pulverizing corals). Therefore, given the manner in which traps and fishing vessel anchors are used in the action areas, we do not expect a reduction in reproduction from direct interactions.

Taken together, we do not believe effects of trap use and fishing vessel anchoring from the proposed actions will reduce the distribution of the species or have more than a minimal (negligible) impact on the amount of coral (numbers) or coral reproduction.

Harvest of herbivorous fish is allowed under the proposed actions, reducing the number of herbivorous fish available to graze on macroalgae. It is unclear whether the effects from the reduction of macroalgae mediation by herbivorous fish is directly reducing the areal coverage (numbers) of elkhorn coral. Nugues and Bak (2006) point out that many of the studies addressing the adverse effects of algae on corals are either observational, (i.e., describing the death of the coral tissue overgrown by algae), or correlative, showing opposite trends in coral and algal abundance over time (McCook et al. 2001). Other studies that specifically investigated the correlation between macroalgae and coral mortality have produced mixed results, with some macroalgae apparently having little to no effect on coral colony mortality (Jompa and McCook 1998, River and Edmunds 2001), while others appear to actually overgrow and kill corals (i.e., Jompa and McCook 2002a, Nugues and Bak 2006). The effects of macroalgae on corals also appear to be dependent upon which macroalgae and corals species are interacting (Nugues and Bak 2006). Nugues and Bak (2006) reported that algae appeared to cause mortality in a very limited number of coral species. Of the six coral species they tested (one was the ESA-listed *Montastraea* (now *Orbicella*) *franksi*), only one species (*Agaricia agaricites*) experienced greater tissue mortality after exposure to brown macroalgae *Lobophora variegata,* which they indicate is a "relatively potent competitor against corals" (Nugues and Bak 2006). There is currently no specific data on the potential effects of macroalgae on elkhorn coral mortality, but it appears unlikely that macroalgae is causing direct morality of this species. Thus, we believe a reduction in number of elkhorn colonies is not likely as a result of effects to corals from the harvest of herbivorous fish associated with the proposed actions.

With respect to effects on the species' reproduction, it appears that a phase shift toward high macroalgae and low coral cover is occurring, or has occurred in the U.S. Caribbean. High macroalgae reduces the available substrate for coral settlement (reproduction). We believe it is the major stressors (i.e., global climate change, increasing sea surface temperatures, ocean acidification, impacts to water quality (principally nutrient input) coral diseases, damage from hurricanes), in combination with the *Diadema* die-off that are the primary causes of this phase shift. The data indicate that Caribbean-wide changes in benthic assemblages were likely precipitated by massive coral mortality events, namely the loss of elkhorn and staghorn coral from white band disease (WBD) (as macroalgae are generally unable to actively overgrow and kill live corals) (Aronson and Precht 2001). Coral mortality and the die-off of *Diadema* created conditions conducive to macroalgae growth and spread, and macroalgae was able to out-compete other corals to re-occupy substrate space left barren by coral mortalities. Over several decades, the re-occupation of substrate previously suitable for coral asexual or sexual reproduction by macroalgae established a feedback loop resistant to coral re-colonization (Hughes and Connell 1999).

The harvest of herbivorous fish under the proposed actions is likely exacerbating the phase-shift that is occurring or has occurred, however, based on available research noted above, it is unlikely to be a primary factor causing the degradation of features necessary for reproduction. In the last NMFS analysis of these effects (NMFS 2011a) of the proposed actions, NMFS concluded that the existing data appeared to indicate that the levels of herbivorous fish harvest were unlikely to be a primary contributor to growth and spread of macroalgae. The proposed actions include the harvest of herbivorous species under specific limits (e.g., ACLs, recreational bag limits, and size limits). The extent to which the harvest of herbivorous fish is adversely affecting the availability of substrate for reproduction remains unquantifiable. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful coral reproduction. We also acknowledge that this reduction in suitable substrate negatively affects the species' chances of surviving and recovering in the wild. Because of these potential impacts, through this Opinion, we will continue to require monitoring actions be taken to help better understand the harvest of herbivorous fish. However, we believe herbivorous fish harvest is only one of many factors that have led to high macroalgal levels. Thus, the proposed actions appear to have only a small, incremental role in what is believed to be only a moderate threat (NMFS 2005 Status Review) to the species. Since we believe the effects of the proposed actions on habitat are not currently a primary cause of macroalgal growth, we believe the proposed actions are causing a very minimal impact on the species' reproduction. In addition, as we explained above, macroalgae appear unlikely to cause direct mortality to adult elkhorn coral. Thus, the number of colonies available for asexual or sexual reproduction appears unlikely to change as result of harvest of herbivorous fish. Unlike lethal takes of other species that may forever remove a sexually mature individual for the population, macroalgae appears to affect sexual and asexual reproduction success, but not preclude it from occurring eventually when favorable conditions return, as removal of reproductive, adult colonies from the population would. Based on the above information we believe that the proposed actions are likely adversely affecting elkhorn coral from the harvest of herbivorous reef fish, but those effects are minimally contributing to reductions in reproduction.

With respect to the species' distribution, we do not anticipate that any colonies will actually be lost due to macroalgal growth associated with the harvest of herbivorous fish associated with the proposed actions. Therefore, we do not anticipate any change in the overall distribution of corals as a result of loss of the coral.

Macroalgal growth can also affect coral larval settlement, which could affect distribution. The proposed actions are the continued harvest of herbivorous species under specific limits, including ACLs, through the FMPs. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful settlement. Since we believe the effects of the proposed actions are not currently a primary cause in macroalgal growth, we believe the incremental impacts caused by the proposed actions are causing an extremely small reduction in the amount of substrate available for settlement, and this could affect the species' distribution. However, we do not believe the proposed actions will cause a reduction in the species' overall distribution as the species are widely distributed beyond the action area, which is only a small portion of its overall range.

Overall, we expect a very small reduction reproduction from the effects of the proposed actions. We expect no reductions in numbers or distribution. Because we anticipate very small impacts only to species' reproduction within the action areas, and because the species is widely distributed, we do not expect the proposed action will alter the trends and trajectory of the species, or appreciably reduce the likelihood of elkhorn coral's survival in the wild.

Recovery

The Recovery Plan for elkhorn and staghorn corals (NMFS 2015) lists the following major actions:

- Improve understanding of population abundance, trends, and structure through monitoring and experimental research.
- Develop and implement appropriate strategies for population enhancement through restocking and active management.
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes such as herbivory to sustain adult colonies and promote successful natural recruitment in the long term.
- Curb ocean warming and acidification impacts to health, reproduction, and growth, and possibly curb disease threats, by reducing atmospheric greenhouse gas concentrations.
- Reduce locally-manageable stress and mortality threats (e.g., predation, anthropogenic physical damage, acute sedimentation, nutrients, contaminants).
- Determine coral health risk factors and their inter-relationships and implement mitigation or control strategies to minimize or prevent impacts to coral health.

The proposed actions would not negatively affect (and may actually help) efforts to understand or monitor this coral, will not impact efforts relating to strategies for population enhancement, will not affect efforts to curb ocean warming and acidification impacts, or affect efforts to determine coral health risk factors and their inter-relationships and implement mitigation or control strategies to minimize or prevent impacts to coral health. Restrictions on harvest of herbivore species under the proposed actions, including the limits on the catch of herbivorous fish and the prohibition on the harvest of species of sea urchins, would contribute to the third recovery objective noted above, efforts to implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes such as herbivory to sustain adult colonies and promote successful natural recruitment in the long term. Regarding the fifth recovery objective noted above, the proposed actions could cause coral mortality. However, while the proposed actions would not reduce locally-manageable stress and mortality, on the whole, though they do employ some protective measures, like anchoring protections for certain areas, we determined the potential lethal take associated with trap use and anchoring, and the effect of the harvest of herbivorous fish on reproduction, associated with the proposed actions would not have any detectable influence on the population. We believe the proposed actions are not likely to impede the recovery objectives above and will not result in an appreciable reduction in the likelihood of this coral's recovery in the wild.

Conclusion

The effects associated with the proposed actions are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the elkhorn coral in the wild.

7.8 Rough Cactus Coral

As calculated in the Effects Section of this Opinion, approximately 39.3 ft^2 annually of this coral will be lethally taken by fishing gear deployment and anchoring associated with the proposed actions. Rough cactus coral is most commonly found between 0-30 m. Based on rough cactus coral cover per reef site in each of the island areas (introduced in the Effects Section), and the amount of coral reef off Puerto Rico, St. Thomas/St. John, and St. Croix,[54] we calculate the amount of rough cactus coral off all three island areas (1,980,496.5 ft^2). Therefore, approximately 0.0020% of the rough cactus coral in all waters off Puerto Rico, St. Thomas/St. John, and St. Croix, combined, is taken annually by the proposed actions (39.3 ft^2 rough cactus coral taken under the proposed actions annually / 1,980,496.5 ft^2 rough cactus coral in all waters off Puerto Rico, St. Thomas/St. John, and St. Croix). This amount is expected to have a negligible impact on coral cover (numbers) and reproduction in the U.S. Caribbean waters, and this percentage has even less impact within the context of the greater amount of this species found throughout its range. This minimal reduction in the amount of rough cactus coral in the U.S. Caribbean is unlikely to affect the species' distribution.

Additionally, since asexual reproduction via fragmentation is a key mode of reproduction for rough cactus coral, contact with traps and anchors themselves does not necessarily mean a reduction in reproduction would occur. It is unlikely that contact with a trap or anchor will cause damage significant enough to remove the potential for asexual reproduction (i.e., traps pulverizing corals). Therefore, given the manner in which traps and fishing vessel anchors are used in the action areas, we do not expect a reduction in reproduction from direct interactions.

Taken together, we do not believe effects of trap use and fishing vessel anchoring from the proposed actions will reduce the distribution of the species or have more than a minimal (negligible) impact on the amount of coral (numbers) or coral reproduction.

Harvest of herbivorous fish is allowed under the proposed actions, reducing the number of herbivorous fish available to graze on macroalgae. It is unclear whether the effects from the reduction of macroalgae mediation by herbivorous fish is directly reducing the areal coverage (numbers) of rough cactus coral. Nugues and Bak (2006) point out that many of the studies addressing the adverse effects of algae on corals are either observational, (i.e., describing the

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[54] The amount of coral reef off Puerto Rico is 14,001,731,651 ft^2, the amount of coral reef off St. Thomas/St. John is 2,643,221,773 ft^2, and the amount of coral reef off St. Croix is 3,450,401,472 ft^2 (OCM Partners 2020).

death of the coral tissue overgrown by algae), or correlative, showing opposite trends in coral and algal abundance over time (McCook et al. 2001). Other studies that specifically investigated the correlation between macroalgae and coral mortality have produced mixed results, with some macroalgae apparently having little to no effect on coral colony mortality (Jompa and McCook 1998, River and Edmunds 2001), while others appear to actually overgrow and kill corals (i.e., Jompa and McCook 2002a, Nugues and Bak 2006). The effects of macroalgae on corals also appear to be dependent upon which macroalgae and corals species are interacting (Nugues and Bak 2006). Nugues and Bak (2006) reported that algae appeared to cause mortality in a very limited number of coral species. Of the six coral species they tested (one was the ESA-listed *Montastraea* (now *Orbicella*) *franksi*), only one species (*Agaricia agaricites*) experienced greater tissue mortality after exposure to brown macroalgae *Lobophora variegata*, which they indicate is a "relatively potent competitor against corals" (Nugues and Bak 2006). There is currently no specific data on the potential effects of macroalgae on rough cactus coral mortality, but it appears unlikely that macroalgae is causing direct morality of this species. Thus, we believe a reduction in number of rough cactus colonies is not likely as a result of effects to corals from the harvest of herbivorous fish associated with the proposed actions.

With respect to effects on the species' reproduction, it appears that a phase shift toward high macroalgae and low coral cover is occurring, or has occurred in the U.S. Caribbean. High macroalgae reduces the available substrate for coral settlement (reproduction). We believe it is the major stressors (i.e., global climate change, increasing sea surface temperatures, ocean acidification, impacts to water quality (principally nutrient input) coral diseases, damage from hurricanes), in combination with the *Diadema* die-off that are the primary causes of this phase shift. The data indicate that Caribbean-wide changes in benthic assemblages were likely precipitated by massive coral mortality events (as macroalgae are generally unable to actively overgrow and kill live corals) (Aronson and Precht 2001). Coral mortality and the die-off of *Diadema* created conditions conducive to macroalgae growth and spread, and macroalgae was able to out-compete other corals to re-occupy substrate space left barren by coral mortalities. Over several decades, the re-occupation of substrate previously suitable for coral asexual or sexual reproduction by macroalgae established a feedback loop resistant to coral re-colonization (Hughes and Connell 1999).

The harvest of herbivorous fish under the proposed actions is likely exacerbating the phase-shift that is occurring or has occurred, however, based on available research noted above, it is unlikely to be a primary factor causing the degradation of features necessary for reproduction. In the last NMFS analysis of these effects (NMFS 2011a) of the proposed actions, NMFS concluded that the existing data appeared to indicate that the levels of herbivorous fish harvest were unlikely to be a primary contributor to growth and spread of macroalgae. The proposed actions include the harvest of herbivorous species under specific limits (e.g., ACLs, recreational bag limits, and size limits). The extent to which the harvest of herbivorous fish is adversely affecting the availability of substrate for reproduction remains unquantifiable. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful coral reproduction. We also acknowledge that this reduction in suitable substrate negatively affects the species' chances of surviving and recovering in the wild. Because of these potential impacts, through this Opinion, we will continue to require monitoring actions be taken to help better understand the harvest of herbivorous fish. However, we believe herbivorous fish harvest is only one of many factors that have led to high macroalgal levels. Thus, the proposed actions appear to have only a small, incremental role in what is believed to be only a moderate threat (NMFS 2005 Status Review) to the species. Since we believe the effects of the proposed actions on habitat are not currently a primary cause of macroalgal growth, we believe the proposed actions are causing a very minimal impact on the species' reproduction. In addition, as we explained above, macroalgae appear unlikely to cause direct mortality to adult rough cactus coral. Thus, the number of colonies available for asexual or sexual reproduction appears unlikely to change as result of harvest of herbivorous fish. Unlike lethal takes of other species that may forever remove a sexually mature individual for the population, macroalgae appears to affect sexual and asexual reproduction success, but not preclude it from occurring eventually when favorable conditions return, as removal of reproductive, adult colonies from the population would. Based on the above information we believe that the proposed actions are likely adversely affecting rough cactus coral from the harvest of herbivorous reef fish, but those effects are minimally contributing to reductions in reproduction.

With respect to the species' distribution, we do not anticipate that any colonies will actually be lost due to macroalgal growth associated with the harvest of herbivorous fish associated with the proposed actions. Therefore, we do not anticipate any change in the overall distribution of corals as a result of loss of the coral.

Macroalgal growth can also affect coral larval settlement, which could affect distribution. The proposed actions are the continued harvest of herbivorous species under specific limits, including ACLs, through the FMPs. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful settlement. Since we believe the effects of the proposed actions are not currently a primary cause in macroalgal growth, we believe the incremental impacts caused by the proposed actions are causing an extremely small reduction in the amount of substrate available for settlement, and this could affect the species' distribution. However, we do not believe the proposed actions will cause a reduction in the species' overall distribution as the species are widely distributed beyond the action area, which is only a small portion of its overall range.

Overall, we expect a very small reduction reproduction from the effects of the proposed actions. We expect no reductions in numbers or distribution. Because we anticipate very small impacts only to species' reproduction within the action areas, and because the species is widely distributed, we do not expect the proposed action will alter the trends and trajectory of the species, or appreciably reduce the likelihood of rough cactus coral's survival in the wild.

Recovery

There is no recovery plan for this species, however a recovery outline exists and lists the following major actions:

- Improve understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling
- Through research, improve understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility

- Reduce locally-manageable stress and mortality sources (e.g., acute sedimentation, nutrients, contaminants, over-fishing).
- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit D. cylindrus, M. ferox, and Orbicella spp.
- Develop and implement U.S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment

The proposed actions would not negatively affect efforts to understand, monitor, research, or model this coral. They will not prohibit any efforts to prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit this species. Nor will they affect efforts to develop and implement measures to reduce atmospheric carbon dioxide. Restrictions on harvest of herbivore species would benefit efforts to reduce mortality sources (management measures are intended to prevent overfishing of key herbivores). This may also help with efforts to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment. The proposed actions could cause the loss of coral. Thus, while the proposed actions would not reduce locally-manageable stress and mortality of coral on the whole, though they do employ some protective measures, like anchoring protections for certain areas, we determined the potential take associated with the proposed actions would not have any detectable influence on the population. We believe the proposed actions are not likely to impede the recovery objectives above and will not result in an appreciable reduction in the likelihood of this coral's recovery in the wild.

Conclusion
The effects associated with the proposed actions are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the rough cactus coral in the wild.

7.9 Pillar Coral

As calculated in the Effects Section of this Opinion, approximately 155.1 ft^2 annually of this coral will be lethally taken by fishing gear deployment and anchoring associated with the proposed actions. Pillar coral is most commonly found between 0-30 m. Based on pillar coral cover per reef site in each of the island areas (introduced in the Effects Section), and the amount of coral reef off Puerto Rico, St. Thomas/St. John, and St. Croix,[55] we calculate the amount of pillar coral off all three island areas (5,050,478.5 ft^2). Therefore, approximately 0.0031% of the pillar coral in all waters off Puerto Rico, St. Thomas/St. John, and St. Croix, combined, is taken annually by the proposed actions (155.1 ft^2 pillar coral taken under the proposed actions annually

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[55] The amount of coral reef off Puerto Rico is 14,001,731,651 ft^2, the amount of coral reef off St. Thomas/St. John is 2,643,221,773 ft^2, and the amount of coral reef off St. Croix is 3,450,401,472 ft^2 (OCM Partners 2020).

/ 5,050,478.5 ft^2 pillar coral in all waters off Puerto Rico, St. Thomas/St. John, and St. Croix). This amount is expected to have a negligible impact on coral cover (numbers) and reproduction in the U.S. Caribbean waters, and this percentage has even less impact within the context of the greater amount of this species found throughout its range. This minimal reduction in the amount of pillar coral in the U.S. Caribbean is unlikely to affect the species' distribution.

Additionally, since asexual reproduction via fragmentation is a key mode of reproduction for pillar coral, contact with traps and anchors themselves does not necessarily mean a reduction in reproduction would occur. It is unlikely that contact with a trap or anchor will cause damage significant enough to remove the potential for asexual reproduction (i.e., traps pulverizing corals). Therefore, given the manner in which traps and fishing vessel anchors are used in the action areas, we do not expect a reduction in reproduction from direct interactions.

Taken together, we do not believe effects of trap use and fishing vessel anchoring from the proposed actions will reduce the distribution of the species or have more than a minimal (negligible) impact on the amount of coral (numbers) or coral reproduction.

Harvest of herbivorous fish is allowed under the proposed actions, reducing the number of herbivorous fish available to graze on macroalgae. It is unclear whether the effects from the reduction of macroalgae mediation by herbivorous fish is directly reducing the areal coverage (numbers) of pillar coral. Nugues and Bak (2006) point out that many of the studies addressing the adverse effects of algae on corals are either observational, (i.e., describing the death of the coral tissue overgrown by algae), or correlative, showing opposite trends in coral and algal abundance over time (McCook et al. 2001). Other studies that specifically investigated the correlation between macroalgae and coral mortality have produced mixed results, with some macroalgae apparently having little to no effect on coral colony mortality (Jompa and McCook 1998, River and Edmunds 2001), while others appear to actually overgrow and kill corals (i.e., Jompa and McCook 2002a, Nugues and Bak 2006). The effects of macroalgae on corals also appear to be dependent upon which macroalgae and corals species are interacting (Nugues and Bak 2006). Nugues and Bak (2006) reported that algae appeared to cause mortality in a very limited number of coral species. Of the six coral species they tested (one was the ESA-listed *Montastraea* (now *Orbicella*) *franksi*), only one species (*Agaricia agaricites*) experienced greater tissue mortality after exposure to brown macroalgae *Lobophora variegata,* which they indicate is a "relatively potent competitor against corals" (Nugues and Bak 2006). There is currently no specific data on the potential effects of macroalgae on rough cactus coral mortality, but it appears unlikely that macroalgae is causing direct morality of this species. Thus, we believe a reduction in number of pillar coral colonies is not likely as a result of effects to corals from the harvest of herbivorous fish associated with the proposed actions.

With respect to effects on the species' reproduction, it appears that a phase shift toward high macroalgae and low coral cover is occurring, or has occurred in the U.S. Caribbean. High macroalgae reduces the available substrate for coral settlement (reproduction). We believe it is the major stressors (i.e., global climate change, increasing sea surface temperatures, ocean acidification, impacts to water quality (principally nutrient input) coral diseases, damage from hurricanes), in combination with the *Diadema* die-off that are the primary causes of this phase shift. The data indicate that Caribbean-wide changes in benthic assemblages were likely precipitated by massive coral mortality events (as macroalgae are generally unable to actively overgrow and kill live corals) (Aronson and Precht 2001). Coral mortality and the die-off of *Diadema* created conditions conducive to macroalgae growth and spread, and macroalgae was able to out-compete other corals to re-occupy substrate space left barren by coral mortalities. Over several decades, the re-occupation of substrate previously suitable for coral asexual or sexual reproduction by macroalgae established a feedback loop resistant to coral re-colonization (Hughes and Connell 1999).

The harvest of herbivorous fish under the proposed actions is likely exacerbating the phase-shift that is occurring or has occurred, however, based on available research noted above, it is unlikely to be a primary factor causing the degradation of features necessary for reproduction. In the last NMFS analysis of these effects (NMFS 2011a) of the proposed actions, NMFS concluded that the existing data appeared to indicate that the levels of herbivorous fish harvest were unlikely to be a primary contributor to growth and spread of macroalgae. The proposed actions include the harvest of herbivorous species under specific limits (e.g., ACLs, recreational bag limits, and size limits). The extent to which the harvest of herbivorous fish is adversely affecting the availability of substrate for reproduction remains unquantifiable. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful coral reproduction. We also acknowledge that this reduction in suitable substrate negatively affects the species' chances of surviving and recovering in the wild. Because of these potential impacts, through this Opinion, we will continue to require monitoring actions be taken to help better understand the harvest of herbivorous. However, we believe herbivorous fish harvest is only one of many factors that have led to high macroalgal levels. Thus, the proposed actions appear to have only a small, incremental role in what is believed to be only a moderate threat (NMFS 2005 Status Review) to the species. Since we believe the effects of the proposed actions on habitat are not currently a primary cause of macroalgal growth, we believe the proposed actions are causing a very minimal impact on the species' reproduction. In addition, as we explained above, macroalgae appear unlikely to cause direct mortality to adult pillar coral. Thus, the number of colonies available for asexual or sexual reproduction appears unlikely to change as result of harvest of herbivorous fish. Unlike lethal takes of other species that may forever remove a sexually mature individual for the population, macroalgae appears to affect sexual and asexual reproduction success, but not preclude it from occurring eventually when favorable conditions return, as removal of reproductive, adult colonies from the population would. Based on the above information, we believe that the proposed actions are likely adversely affecting pillar coral from the harvest of herbivorous reef fish, but those effects are minimally contributing to reductions in reproduction.

With respect to the species' distribution, we do not anticipate that any colonies will actually be lost due to macroalgal growth associated with the harvest of herbivorous fish associated with the proposed actions. Therefore, we do not anticipate any change in the overall distribution of corals as a result of loss of the coral.

Macroalgal growth can also affect coral larval settlement, which could affect distribution. The proposed actions are the continued harvest of herbivorous species under specific limits, including ACLs, through the FMPs. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful settlement. Since we believe the effects of the proposed actions are not currently a primary cause in macroalgal growth, we believe the incremental impacts caused by the proposed actions are causing an extremely small reduction in the amount of substrate available for settlement, and this could affect the species' distribution. However, we do not believe the proposed actions will cause a reduction in the species' overall distribution as the species are widely distributed beyond the action area, which is only a small portion of its overall range.

Overall, we expect a very small reduction reproduction from the effects of the proposed actions. We expect no reductions in numbers or distribution. Because we anticipate very small impacts only to species' reproduction within the action areas, and because the species is widely distributed, we do not expect the proposed action will alter the trends and trajectory of the species, or appreciably reduce the likelihood of pillar coral's survival in the wild.

Recovery

There is no recovery plan for this species, however a recovery outline exists and lists the following major actions:

- Improve understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling
- Through research, improve understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility
- Reduce locally-manageable stress and mortality sources (e.g., acute sedimentation, nutrients, contaminants, over-fishing).
- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit D. cylindrus, M. ferox, and Orbicella spp.
- Develop and implement U.S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment

The proposed actions would not negatively affect efforts to understand, monitor, research, or model this coral. They will not prohibit any efforts to prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit this species. Nor will they affect efforts to develop and implement measures to reduce atmospheric carbon dioxide. Restrictions on harvest of herbivore species would benefit efforts to reduce mortality sources (management measures are intended to prevent overfishing of key herbivores). This may also help with efforts to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment. The proposed actions could cause the loss of coral. Thus, while the proposed actions would not reduce locally-manageable stress and mortality of coral on the whole, though they do employ some protective measures, like anchoring protections for certain areas, we determined the potential take associated with the proposed actions would not have any detectable influence on the population. We believe the proposed actions are not likely to impede the recovery objectives above and will not result in an appreciable reduction in the likelihood of this coral's recovery in the wild.

Conclusion
The effects associated with the proposed actions are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the pillar coral in the wild.

7.10 Lobed Star Coral

As calculated in the Effects Section of this Opinion, approximately 11,506.4 ft^2 annually of this coral will be lethally taken by fishing gear deployment and anchoring associated with the proposed actions. Lobed star coral is most commonly found between 0-30 m. Based on lobed star coral cover per reef site in each of the island areas (introduced in the Effects Section), and the amount of coral reef off Puerto Rico, St. Thomas/St. John, and St. Croix,[56] we calculate the amount of lobed star coral off all three island areas (171,030,573.4 ft^2). Therefore, approximately 0.0067% of the lobed star coral in all waters off Puerto Rico, St. Thomas/St. John, and St. Croix, combined, is taken annually by the proposed actions (11,506.4 ft^2 lobed star coral taken under the proposed actions annually / 171,030,573.4 ft^2 lobed star coral in all waters off Puerto Rico, St. Thomas/St. John, and St. Croix). This amount is expected to have a negligible impact on coral cover (numbers) and reproduction in the U.S. Caribbean waters, and this percentage has even less impact within the context of the greater amount of this species found throughout its range. This minimal reduction in the amount of lobed star coral in the U.S. Caribbean is unlikely to affect the species' distribution.

Additionally, since asexual reproduction via fragmentation is a key mode of reproduction for lobed star coral, contact with traps and anchors themselves does not necessarily mean a reduction in reproduction would occur. It is unlikely that contact with a trap or anchor will cause damage significant enough to remove the potential for asexual reproduction (i.e., traps pulverizing corals). Therefore, given the manner in which traps and fishing vessel anchors are used in the action areas, we do not expect a reduction in reproduction from direct interactions.

Taken together, we do not believe effects of trap use and fishing vessel anchoring from the proposed actions will reduce the distribution of the species or have more than a minimal (negligible) impact on the amount of coral (numbers) or coral reproduction.

Harvest of herbivorous fish is allowed under the proposed actions, reducing the number of herbivorous fish available to graze on macroalgae. It is unclear whether the effects from the reduction of macroalgae mediation by herbivorous fish is directly reducing the areal coverage (numbers) of lobed star coral. Nugues and Bak (2006) point out that many of the studies addressing the adverse effects of algae on corals are either observational, (i.e., describing the death of the coral tissue overgrown by algae), or correlative, showing opposite trends in coral

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[56] The amount of coral reef off Puerto Rico is 14,001,731,651 ft^2, the amount of coral reef off St. Thomas/St. John is 2,643,221,773 ft^2, and the amount of coral reef off St. Croix is 3,450,401,472 ft^2 (OCM Partners 2020).

and algal abundance over time (McCook et al. 2001). Other studies that specifically investigated the correlation between macroalgae and coral mortality have produced mixed results, with some macroalgae apparently having little to no effect on coral colony mortality (Jompa and McCook 1998, River and Edmunds 2001), while others appear to actually overgrow and kill corals (i.e., Jompa and McCook 2002a, Nugues and Bak 2006). The effects of macroalgae on corals also appear to be dependent upon which macroalgae and corals species are interacting (Nugues and Bak 2006). Nugues and Bak (2006) reported that algae appeared to cause mortality in a very limited number of coral species. Of the six coral species they tested (one was the ESA-listed *Montastraea* (now *Orbicella*) *franksi*), only one species (*Agaricia agaricites*) experienced greater tissue mortality after exposure to brown macroalgae *Lobophora variegata*, which they indicate is a "relatively potent competitor against corals" (Nugues and Bak 2006). There is currently no specific data on the potential effects of macroalgae on lobed star coral mortality, but it appears unlikely that macroalgae is causing direct morality of this species. Thus, we believe a reduction in number of lobed star colonies is not likely as a result of effects to corals from the harvest of herbivorous fish associated with the proposed actions.

With respect to effects on the species' reproduction, it appears that a phase shift toward high macroalgae and low coral cover is occurring, or has occurred in the U.S. Caribbean. High macroalgae reduces the available substrate for coral settlement (reproduction). We believe it is the major stressors (i.e., global climate change, increasing sea surface temperatures, ocean acidification, impacts to water quality (principally nutrient input) coral diseases, damage from hurricanes), in combination with the *Diadema* die-off that are the primary causes of this phase shift. The data indicate that Caribbean-wide changes in benthic assemblages were likely precipitated by massive coral mortality events (as macroalgae are generally unable to actively overgrow and kill live corals) (Aronson and Precht 2001). Coral mortality and the die-off of *Diadema* created conditions conducive to macroalgae growth and spread, and macroalgae was able to out-compete other corals to re-occupy substrate space left barren by coral mortalities. Over several decades, the re-occupation of substrate previously suitable for coral asexual or sexual reproduction by macroalgae established a feedback loop resistant to coral re-colonization (Hughes and Connell 1999).

The harvest of herbivorous fish under the proposed actions is likely exacerbating the phase-shift that is occurring or has occurred, however, based on available research noted above, it is unlikely to be a primary factor causing the degradation of features necessary for reproduction. In the last NMFS analysis of these effects (NMFS 2011a) of the proposed actions, NMFS concluded that the existing data appeared to indicate that the levels of herbivorous fish harvest were unlikely to be a primary contributor to growth and spread of macroalgae. The proposed actions include the harvest of herbivorous species under specific limits (e.g., ACLs, recreational bag limits, and size limits). The extent to which the harvest of herbivorous fish is adversely affecting the availability of substrate for reproduction remains unquantifiable. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful coral reproduction. We also acknowledge that this reduction in suitable substrate negatively affects the species' chances of surviving and recovering in the wild. Because of these potential impacts, through this Opinion, we will continue to require monitoring actions be taken to help better understand the harvest of herbivorous fish. However, we believe herbivorous fish harvest is only one of many factors that have led to high macroalgal levels. Thus, the proposed actions appear to have only a small, incremental role in what is believed to be only a moderate threat (NMFS

2005 Status Review) to the species. Since we believe the effects of the proposed actions on habitat are not currently a primary cause of macroalgal growth, we believe the proposed actions are causing a very minimal impact on the species' reproduction. In addition, as we explained above, macroalgae appear unlikely to cause direct mortality to adult lobed star coral. Thus, the number of colonies available for asexual or sexual reproduction appears unlikely to change as result of harvest of herbivorous fish. Unlike lethal takes of other species that may forever remove a sexually mature individual for the population, macroalgae appears to affect sexual and asexual reproduction success, but not preclude it from occurring eventually when favorable conditions return, as removal of reproductive, adult colonies from the population would. Based on the above information, we believe that the proposed actions are likely adversely affecting lobed star coral from the harvest of herbivorous reef fish, but those effects are minimally contributing to reductions in reproduction.

With respect to the species' distribution, we do not anticipate that any colonies will actually be lost due to macroalgal growth associated with the harvest of herbivorous fish associated with the proposed actions. Therefore, we do not anticipate any change in the overall distribution of corals as a result of loss of the coral.

Macroalgal growth can also affect coral larval settlement, which could affect distribution. The proposed actions are the continued harvest of herbivorous species under specific limits, including ACLs, through the FMPs. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful settlement. Since we believe the effects of the proposed actions are not currently a primary cause in macroalgal growth, we believe the incremental impacts caused by the proposed actions are causing an extremely small reduction in the amount of substrate available for settlement, and this could affect the species' distribution. However, we do not believe the proposed actions will cause a reduction in the species' overall distribution as the species are widely distributed beyond the action area, which is only a small portion of its overall range.

Overall, we expect a very small reduction reproduction from the effects of the proposed actions. We expect no reductions in numbers or distribution. Because we anticipate very small impacts only to species' reproduction within the action areas, and because the species is widely distributed, we do not expect the proposed action will alter the trends and trajectory of the species, or appreciably reduce the likelihood of lobed star coral's survival in the wild.

Recovery

There is no recovery plan for this species, however a recovery outline exists and lists the following major actions:

- Improve understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling

- Through research, improve understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility

- Reduce locally-manageable stress and mortality sources (e.g., acute sedimentation, nutrients, contaminants, over-fishing).

- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit D. cylindrus, M. ferox, and Orbicella spp.
- Develop and implement U.S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment

The proposed actions would not negatively affect efforts to understand, monitor, research, or model this coral. They will not prohibit any efforts to prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit this species. Nor will they affect efforts to develop and implement measures to reduce atmospheric carbon dioxide. Restrictions on harvest of herbivore species would benefit efforts to reduce mortality sources (management measures are intended to prevent overfishing of key herbivores). This may also help with efforts to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment. The proposed actions could cause the loss of coral. Thus, while the proposed actions would not reduce locally-manageable stress and mortality of coral on the whole, though they do employ some protective measures, like anchoring protections for certain areas, we determined the potential take associated with the proposed actions would not have any detectable influence on the population. We believe the proposed actions are not likely to impede the recovery objectives above and will not result in an appreciable reduction in the likelihood of this coral's recovery in the wild.

Conclusion

The effects associated with the proposed actions are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the lobed star coral in the wild.

7.11 Mountainous Star Coral

As calculated in the Effects Section of this Opinion, approximately 9,455.5 ft^2 annually of this coral will be lethally taken by fishing gear deployment and anchoring associated with the proposed actions. Mountainous star coral is most commonly found between 0-30 m. Based on mountainous star coral cover per reef site in each of the island areas (introduced in the Effects Section), and the amount of coral reef off Puerto Rico, St. Thomas/St. John, and St. Croix,[57] we calculate the amount of mountainous star coral off all three island areas (152,763,310.6 ft^2). Therefore, approximately 0.0062% of the mountainous star coral in all waters off Puerto Rico, St. Thomas/St. John, and St. Croix, combined, is taken annually by the proposed actions (9,455.5 ft^2 mountainous star coral taken under the proposed actions annually / 152,763,310.6 ft^2 mountainous star coral in all waters off Puerto Rico, St. Thomas/St. John, and St. Croix). This amount is expected to have a negligible impact on coral cover (numbers) and reproduction in the

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[57] The amount of coral reef off Puerto Rico is 14,001,731,651 ft^2, the amount of coral reef off St. Thomas/St. John is 2,643,221,773 ft^2, and the amount of coral reef off St. Croix is 3,450,401,472 ft^2 (OCM Partners 2020).

U.S. Caribbean waters, and this percentage has even less impact within the context of the greater amount of this species found throughout its range. This minimal reduction in the amount of mountainous star coral in the U.S. Caribbean is unlikely to affect the species' distribution.

Additionally, since asexual reproduction via fragmentation is a key mode of reproduction for mountainous star coral, contact with traps and anchors themselves does not necessarily mean a reduction in reproduction would occur. It is unlikely that contact with a trap or anchor will cause damage significant enough to remove the potential for asexual reproduction (i.e., traps pulverizing corals). Therefore, given the manner in which traps and fishing vessel anchors are used in the action areas, we do not expect a reduction in reproduction from direct interactions.

Taken together, we do not believe effects of trap use and fishing vessel anchoring from the proposed actions will reduce the distribution of the species or have more than a minimal (negligible) impact on the amount of coral (numbers) or coral reproduction.

Harvest of herbivorous fish is allowed under the proposed actions, reducing the number of herbivorous fish available to graze on macroalgae. It is unclear whether the effects from the reduction of macroalgae mediation by herbivorous fish is directly reducing the areal coverage (numbers) of mountainous star coral. Nugues and Bak (2006) point out that many of the studies addressing the adverse effects of algae on corals are either observational, (i.e., describing the death of the coral tissue overgrown by algae), or correlative, showing opposite trends in coral and algal abundance over time (McCook et al. 2001). Other studies that specifically investigated the correlation between macroalgae and coral mortality have produced mixed results, with some macroalgae apparently having little to no effect on coral colony mortality (Jompa and McCook 1998, River and Edmunds 2001), while others appear to actually overgrow and kill corals (i.e., Jompa and McCook 2002a, Nugues and Bak 2006). The effects of macroalgae on corals also appear to be dependent upon which macroalgae and corals species are interacting (Nugues and Bak 2006). Nugues and Bak (2006) reported that algae appeared to cause mortality in a very limited number of coral species. Of the six coral species they tested (one was the ESA-listed *Montastraea* (now *Orbicella*) *franksi*), only one species (*Agaricia agaricites*) experienced greater tissue mortality after exposure to brown macroalgae *Lobophora variegata*, which they indicate is a "relatively potent competitor against corals" (Nugues and Bak 2006). There is currently no specific data on the potential effects of macroalgae on rough cactus coral mortality, but it appears unlikely that macroalgae is causing direct morality of this species. Thus, we believe a reduction in number of mountainous star colonies is not likely as a result of effects to corals from the harvest of herbivorous fish associated with the proposed actions.

With respect to effects on the species' reproduction, it appears that a phase shift toward high macroalgae and low coral cover is occurring, or has occurred in the U.S. Caribbean. High macroalgae reduces the available substrate for coral settlement (reproduction). We believe it is the major stressors (i.e., global climate change, increasing sea surface temperatures, ocean acidification, impacts to water quality (principally nutrient input) coral diseases, damage from hurricanes), in combination with the *Diadema* die-off that are the primary causes of this phase shift. The data indicate that Caribbean-wide changes in benthic assemblages were likely precipitated by massive coral mortality events (as macroalgae are generally unable to actively overgrow and kill live corals) (Aronson and Precht 2001). Coral mortality and the die-off of *Diadema* created conditions conducive to macroalgae growth and spread, and macroalgae was able to out-compete other corals to re-occupy substrate space left barren by coral mortalities. Over several decades, the re-occupation of substrate previously suitable for coral asexual or sexual reproduction by macroalgae established a feedback loop resistant to coral re-colonization (Hughes and Connell 1999).

The harvest of herbivorous fish under the proposed actions is likely exacerbating the phase-shift that is occurring or has occurred, however, based on available research noted above, it is unlikely to be a primary factor causing the degradation of features necessary for reproduction. In the last NMFS analysis of these effects (NMFS 2011a) of the proposed actions, NMFS concluded that the existing data appeared to indicate that the levels of herbivorous fish harvest were unlikely to be a primary contributor to growth and spread of macroalgae. The proposed actions include the harvest of herbivorous species under specific limits (e.g., ACLs, recreational bag limits, and size limits). The extent to which the harvest of herbivorous fish is adversely affecting the availability of substrate for reproduction remains unquantifiable. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful coral reproduction. We also acknowledge that this reduction in suitable substrate negatively affects the species' chances of surviving and recovering in the wild. Because of these potential impacts, through this Opinion, we will continue to require certain research and monitoring actions be taken to help better understand the harvest of herbivorous fish. However, we believe herbivorous fish harvest is only one of many factors that have led to high macroalgal levels. Thus, the proposed actions appear to have only a small, incremental role in what is believed to be only a moderate threat (NMFS 2005 Status Review) to the species. Since we believe the effects of the proposed actions on habitat are not currently a primary cause of macroalgal growth, we believe the proposed actions are causing a very minimal impact on the species' reproduction. In addition, as we explained above, macroalgae appear unlikely to cause direct mortality to adult mountainous star coral. Thus, the number of colonies available for asexual or sexual reproduction appears unlikely to change as result of harvest of herbivorous fish. Unlike lethal takes of other species that may forever remove a sexually mature individual for the population, macroalgae appears to affect sexual and asexual reproduction success, but not preclude it from occurring eventually when favorable conditions return, as removal of reproductive, adult colonies from the population would. Based on the above information we believe that the proposed actions are likely adversely affecting mountainous star coral from the harvest of herbivorous reef fish, but those effects are minimally contributing to reductions in reproduction.

With respect to the species' distribution, we do not anticipate that any colonies will actually be lost due to macroalgal growth associated with the harvest of herbivorous fish associated with the proposed actions. Therefore, we do not anticipate any change in the overall distribution of corals as a result of loss of the coral.

Macroalgal growth can also affect coral larval settlement, which could affect distribution. The proposed actions are the continued harvest of herbivorous species under specific limits, including ACLs, through the FMPs. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful settlement. Since we believe the effects of the proposed actions are not currently a primary cause in macroalgal growth, we believe the incremental impacts caused by the proposed actions are causing an extremely small reduction in the amount of substrate available for settlement, and this could affect the species' distribution.

However, we do not believe the proposed actions will cause a reduction in the species' overall distribution as the species are widely distributed beyond the action area, which is only a small portion of its overall range.

Overall, we expect a very small reduction reproduction from the effects of the proposed actions. We expect no reductions in numbers or distribution. Because we anticipate very small impacts only to species' reproduction within the action areas, and because the species is widely distributed, we do not expect the proposed action will alter the trends and trajectory of the species, or appreciably reduce the likelihood of mountainous star coral's survival in the wild.

Recovery

There is no recovery plan for this species, however a recovery outline exists and lists the following major actions:

- Improve understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling
- Through research, improve understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility
- Reduce locally-manageable stress and mortality sources (e.g., acute sedimentation, nutrients, contaminants, over-fishing).
- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit D. cylindrus, M. ferox, and Orbicella spp.
- Develop and implement U.S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment

The proposed actions would not negatively affect efforts to understand, monitor, research, or model this coral. They will not prohibit any efforts to prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit this species. Nor will they affect efforts to develop and implement measures to reduce atmospheric carbon dioxide. Restrictions on harvest of herbivore species would benefit efforts to reduce mortality sources (management measures are intended to prevent overfishing of key herbivores). This may also help with efforts to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment. The proposed actions could cause the loss of coral. Thus, while the proposed actions would not reduce locally-manageable stress and mortality of coral on the whole, though they do employ some protective measures, like anchoring protections for certain areas, we determined the potential take associated with the proposed actions would not have any detectable influence on the population. We believe the proposed actions are not likely to impede the recovery objectives above and will not result in an appreciable reduction in the likelihood of this coral's recovery in the wild.

Conclusion
The effects associated with the proposed actions are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the mountainous star coral in the wild.

7.12 Boulder Star Coral

As calculated in the Effects Section of this Opinion, approximately 17,976.8 ft^2 annually of this coral will be lethally taken by fishing gear deployment and anchoring associated with the proposed actions. Boulder star coral is most commonly found between 0-30 m. Based on boulder star coral cover per reef site in each of the island areas (introduced in the Effects Section), and the amount of coral reef off Puerto Rico, St. Thomas/St. John, and St. Croix,[58] we calculate the amount of boulder star coral off all three island areas (245,442,978.1 ft^2). Therefore, approximately 0.0073% of the boulder star coral in all waters off Puerto Rico, St. Thomas/St. John, and St. Croix, combined, is taken annually by the proposed actions (17,976.8 ft^2 boulder star coral taken under the proposed actions annually / 245,442,978.1 ft^2 boulder star coral in all waters off Puerto Rico, St. Thomas/St. John, and St. Croix). This amount is expected to have a negligible impact on coral cover (numbers) and reproduction in the U.S. Caribbean waters, and this percentage has even less impact within the context of the greater amount of this species found throughout its range. This minimal reduction in the amount of boulder star coral in the U.S. Caribbean is unlikely to affect the species' distribution.

Additionally, since asexual reproduction via fragmentation is a key mode of reproduction for boulder star coral, contact with traps and anchors themselves does not necessarily mean a reduction in reproduction would occur. It is unlikely that contact with a trap or anchor will cause damage significant enough to remove the potential for asexual reproduction (i.e., traps pulverizing corals). Therefore, given the manner in which traps and fishing vessel anchors are used in the action areas, we do not expect a reduction in reproduction from direct interactions.

Taken together, we do not believe effects of trap use and fishing vessel anchoring from the proposed actions will reduce the distribution of the species or have more than a minimal (negligible) impact on the amount of coral (numbers) or coral reproduction.

Harvest of herbivorous fish is allowed under the proposed actions, reducing the number of herbivorous fish available to graze on macroalgae. It is unclear whether the effects from the reduction of macroalgae mediation by herbivorous fish is directly reducing the areal coverage (numbers) of boulder star coral. Nugues and Bak (2006) point out that many of the studies addressing the adverse effects of algae on corals are either observational, (i.e., describing the death of the coral tissue overgrown by algae), or correlative, showing opposite trends in coral and algal abundance over time (McCook et al. 2001). Other studies that specifically investigated

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[58] The amount of coral reef off Puerto Rico is 14,001,731,651 ft^2, the amount of coral reef off St. Thomas/St. John is 2,643,221,773 ft^2, and the amount of coral reef off St. Croix is 3,450,401,472 ft^2 (OCM Partners 2020).

the correlation between macroalgae and coral mortality have produced mixed results, with some macroalgae apparently having little to no effect on coral colony mortality (Jompa and McCook 1998, River and Edmunds 2001), while others appear to actually overgrow and kill corals (i.e., Jompa and McCook 2002a, Nugues and Bak 2006). The effects of macroalgae on corals also appear to be dependent upon which macroalgae and corals species are interacting (Nugues and Bak 2006). Nugues and Bak (2006) reported that algae appeared to cause mortality in a very limited number of coral species. Of the six coral species they tested (one was the ESA-listed *Montastraea* (now *Orbicella*) *franksi*) boulder star coral, only one species (*Agaricia agaricites*) experienced greater tissue mortality after exposure to brown macroalgae *Lobophora variegata*, which they indicate is a “relatively potent competitor against corals” (Nugues and Bak 2006). The limited data on the potential effects of macroalgae on boulder star coral mortality suggests it unlikely that macroalgae is causing direct morality of this species. Thus, we believe a reduction in number of boulder star colonies is not likely as a result of effects to corals from the harvest of herbivorous fish associated with the proposed actions.

With respect to effects on the species’ reproduction, it appears that a phase shift toward high macroalgae and low coral cover is occurring, or has occurred in the U.S. Caribbean. High macroalgae reduces the available substrate for coral settlement (reproduction). We believe it is the major stressors (i.e., global climate change, increasing sea surface temperatures, ocean acidification, impacts to water quality (principally nutrient input) coral diseases, damage from hurricanes), in combination with the *Diadema* die-off that are the primary causes of this phase shift. The data indicate that Caribbean-wide changes in benthic assemblages were likely precipitated by massive coral mortality events (as macroalgae are generally unable to actively overgrow and kill live corals) (Aronson and Precht 2001). Coral mortality and the die-off of *Diadema* created conditions conducive to macroalgae growth and spread, and macroalgae was able to out-compete other corals to re-occupy substrate space left barren by coral mortalities. Over several decades, the re-occupation of substrate previously suitable for coral asexual or sexual reproduction by macroalgae established a feedback loop resistant to coral re-colonization (Hughes and Connell 1999).

The harvest of herbivorous fish under the proposed actions is likely exacerbating the phase-shift that is occurring or has occurred, however, based on available research noted above, it is unlikely to be a primary factor causing the degradation of features necessary for reproduction. In the last NMFS analysis of these effects (NMFS 2011a) of the proposed actions, NMFS concluded that the existing data appeared to indicate that the levels of herbivorous fish harvest were unlikely to be a primary contributor to growth and spread of macroalgae. The proposed actions include the harvest of herbivorous species under specific limits (e.g., ACLs, recreational bag limits, and size limits). The extent to which the harvest of herbivorous fish is adversely affecting the availability of substrate for reproduction remains unquantifiable. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful coral reproduction. We also acknowledge that this reduction in suitable substrate negatively affects the species’ chances of surviving and recovering in the wild. Because of these potential impacts, through this Opinion, we will continue to require monitoring actions be taken to help better understand the harvest of herbivorous fish. However, we believe herbivorous fish harvest is only one of many factors that have led to high macroalgal levels. Thus, the proposed actions appear to have only a small, incremental role in what is believed to be only a moderate threat (NMFS 2005 Status Review) to the species. Since we believe the effects of the proposed actions on habitat are not currently a primary cause of macroalgal growth, we believe the proposed actions are causing a very minimal impact on the species' reproduction. In addition, as we explained above, macroalgae appear unlikely to cause direct mortality to boulder star coral. Thus, the number of colonies available for asexual or sexual reproduction appears unlikely to change as result of harvest of herbivorous fish. Unlike lethal takes of other species that may forever remove a sexually mature individual for the population, macroalgae appears to affect sexual and asexual reproduction success, but not preclude it from occurring eventually when favorable conditions return, as removal of reproductive, adult colonies from the population would. Based on the above information we believe that the proposed actions are likely adversely affecting boulder star coral from the harvest of herbivorous reef fish, but those effects are minimally contributing to reductions in reproduction.

With respect to the species' distribution, we do not anticipate that any colonies will actually be lost due to macroalgal growth associated with the harvest of herbivorous fish associated with the proposed actions. Therefore, we do not anticipate any change in the overall distribution of corals as a result of macroalgal growth on coral colonies directly as a result of loss of the coral.

Macroalgal growth can also affect coral larval settlement, which could affect distribution. The proposed actions are the continued harvest of herbivorous species under specific limits, including ACLs, through the FMPs. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful settlement. Since we believe the effects of the proposed actions are not currently a primary cause in macroalgal growth, we believe the incremental impacts caused by the proposed actions are causing an extremely small reduction in the amount of substrate available for settlement, and this could affect the species' distribution. However, we do not believe the proposed actions will cause a reduction in the species' overall distribution as the species are widely distributed beyond the action area, which is only a small portion of its overall range.

Overall, we expect a very small reduction reproduction from the effects of the proposed actions. We expect no reductions in numbers or distribution. Because we anticipate very small impacts only to species' reproduction within the action areas, and because the species is widely distributed, we do not expect the proposed action will alter the trends and trajectory of the species, or appreciably reduce the likelihood of boulder star coral's survival in the wild.

Recovery

There is no recovery plan for this species, however a recovery outline exists and lists the following major actions:

- Improve understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling
- Through research, improve understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility
- Reduce locally-manageable stress and mortality sources (e.g., acute sedimentation, nutrients, contaminants, over-fishing).

- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit D. cylindrus, M. ferox, and Orbicella spp.

- Develop and implement U.S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats

- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment

The proposed actions would not negatively affect efforts to understand, monitor, research, or model this coral. They will not prohibit any efforts to prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit this species. Nor will they affect efforts to develop and implement measures to reduce atmospheric carbon dioxide. Restrictions on harvest of herbivore species would benefit efforts to reduce mortality sources (management measures are intended to prevent overfishing of key herbivores). This may also help with efforts to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment. The proposed actions could cause the loss of coral. Thus, while the proposed actions would not reduce locally-manageable stress and mortality of coral on the whole, though they do employ some protective measures, like anchoring protections for certain areas, we determined the potential take associated with the proposed actions would not have any detectable influence on the population. We believe the proposed actions are not likely to impede the recovery objectives above and will not result in an appreciable reduction in the likelihood of this coral's recovery in the wild.

Conclusion
The effects associated with the proposed actions are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the boulder star coral in the wild.

7.13 *Acropora* Critical Habitat

This section considers the issue of destruction or adverse modification of *Acropora* critical habitat, specifically whether a direct or indirect alteration of *Acropora* critical habitat associated with the proposed actions appreciably diminishes the value of critical habitat as a whole for the conservation of *Acropora* listed corals.

Elkhorn and staghorn corals are listed under the ESA, 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 and lower effective density (of genetically distinct adults required for sexual reproduction), and a reduced source of fragments for asexual reproduction and recruitment. In view of these threats, the key conservation objective of designated critical habitat for these species is to facilitate increased incidence of successful sexual and asexual reproduction (i.e., increase the potential for sexual and asexual reproduction to be successful), which in turn facilitates increases in the species' abundances, distributions, and genetic diversity. The feature essential to the conservation of the species, which supports the identified conservation objective, is substrate of suitable quality and availability to support larval settlement and recruitment, and reattachment and recruitment of asexual fragments. For purposes of this definition, "substrate of suitable quality and availability" means natural consolidated hard substrate or dead coral skeleton that is free from fleshy or turf macroalgae cover and sediment cover. To this end, our analysis of whether the proposed actions are likely to destroy or adversely modify designated critical habitat seeks to determine if the adverse effects of proposed actions on the essential feature of designated *Acropora* critical habitat will appreciably reduce the capability of the critical habitat as a whole to facilitate an increased incidence of successful sexual and asexual reproduction. This analysis takes into account the current status of each species; for example, the level of increased incidence of successful reproduction that needs to be facilitated may be different depending on the recovery status of elkhorn and staghorn corals. This analysis also takes into account the geographic and temporal scope of the proposed actions, recognizing that functionality of critical habitat necessarily means that it is and will continue to support the conservation of the species and progress toward recovery.

On November 26, 2008, critical habitat containing the features was designated in four areas (73 FR 72210). The action area contains three of the four designated critical habitat units. The Puerto Rico unit includes approximately 1,383 mi^2, the St. John/St. Thomas unit includes approximately 121 mi^2, and the St. Croix unit encompasses approximately 126 mi^2.

As we noted in the effects section, we anticipate effects to the essential feature of *Acropora* critical habitat from gear deployment and anchoring associated with the proposed actions will occur only in the federal waters in the action areas. We believe these effects from fishing are likely to occur in federal waters in all three action areas. The results of our analysis of the effect of gear deployment and anchoring in Section 5 indicate that 63,881 ft^2, or 0.0023 mi^2, of *Acropora* critical habitat is likely to be adversely affected via contact with traps and anchors in the federal waters in the action areas annually. We assumed contact with the traps and anchors would break dead coral skeleton, and reduce the available essential feature. We do not believe this level of impact indicates that the proposed action is destroying or adversely modifying critical habitat. The three designated critical habitat units in the U.S. Caribbean equal 1,630 mi^2. Thus, the potential effects from gear deployment and anchoring associated with the proposed actions would affect less than 0.000142% of the designated critical habitat in the U.S. Caribbean annually (0.0023/1,630 = 0.00000142), and an even smaller percentage of designated critical habitat as a whole (the other area designated as critical habitat is off Florida, and it comprises 1,329 mi^2). The area impacted has no greater importance to the critical habitat as a whole than other areas within or outside of the action areas, and effects will not diminish the conservation of the critical habitat as a whole. In addition, traps placed on the consolidated hardbottom could temporarily preclude new settlement of planulae (free-swimming coral larvae), affecting the functionality of the essential feature. Once the trap is retrieved, the function will be restored. Since the function is likely to be restored as soon as a trap is removed, we do not expect any cumulative effects from trap deployment year after year, or for this route of effect to destroy or adversely modify critical habitat.

Now our analysis turns to the effects of the proposed actions on *Acropora* critical habitat from the harvest of herbivorous fish. When critical habitat for *Acropora* was designated in 2008, the impacts of coral loss and increasing algae had already become severe. Whether those impacts attributed to the reef fish fisheries under the proposed actions would destroy or adversely modify critical habitat depends on what role the proposed actions play overall in the condition of critical habitat and whether the proposed actions appreciably diminish the value of critical habitat as a whole.

The *Acropora* Biological Review Team (BRT) (2005) indicate there are three "major" stressors or threats to *Acropora* that are having the greatest effect on the species as a whole: disease (e.g., white band disease (WBD), white pox), increasing sea surface temperatures, and natural abrasion and breakage (i.e., hurricanes). The BRT also indicates several other "moderate" threats are also affecting *Acropora*. Competition with macroalgae is considered one of those threats. By rank order of threats to elkhorn and staghorn coral with respect to recovery, the BRT places competition with macroalgae in a tie for 5th out of 13 total stressors or threats (including the three major stressors noted here). It appears that a phase shift toward high macroalgae and low coral cover is occurring, or has occurred in the U.S. Caribbean. We believe these major stressors (i.e., global climate change, increasing sea surface temperatures, ocean acidification, impacts to water quality (principally nutrient input) coral diseases, damage from hurricanes), in combination with the *Diadema* die-off are the primary causes of these phase shifts. The data indicate that Caribbean-wide changes in benthic assemblages were likely precipitated by massive coral mortality events, namely the loss of elkhorn and staghorn coral from WBD (as macroalgae are generally unable to actively overgrow and kill live corals) (Aronson and Precht 2001). Coral mortality and the die-off of *Diadema* created conditions conducive to macroalgae growth and spread, and macroalgae was able to out compete other corals to re-occupy substrate space left barren by coral mortalities. Over several decades, the re-occupation of substrate previously suitable for coral asexual and sexual reproduction by macroalgae established a feedback loop resistant to coral re-colonization (Hughes and Connell 1999).

The ACLs and the prohibition of harvest of certain parrotfish under the proposed actions are not expected to alter the populations of herbivorous fish in federal waters of the U.S. Caribbean compared to the status quo. The proposed actions continue to manage harvest of parrotfish under harvest prohibitions and ACLs, which, combined with accountability measures, limit overall harvest. The measures originally put in place under amendments to the prior Caribbean-wide FMPs, and continued under the proposed actions, likely leading to a greater amount of grazing than there has been at any time in the last decade. Additionally, the new FMPs include prohibitions that no person may fish for or possess species of sea urchins, which is expected to improve the algae grazing dynamics occurring in critical habitat. The extent to which the harvest of herbivorous fish is adversely affecting the availability of the essential feature is uncertain and currently unquantifiable. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful *Acropora* reproduction. We also acknowledge that this reduction in suitable substrate has degraded the species' chances of surviving and recovering in the wild. Because of these potential impacts, this Opinion will require monitoring actions be taken to help NMFS better understand the harvest of herbivorous fish. However, we believe the available information on the growth and spread of macroalgae in the U.S. Caribbean indicates that herbivorous fish harvest is contributing very minimally to its growth.

Because we believe the effects of the proposed actions on designated critical habitat are very minimal and are not currently a primary cause in macroalgal growth, we believe the incremental impacts caused by the proposed actions are not likely to appreciably reduce the capacity of critical habitat as a whole to provide an increased incidence of successful sexual and asexual reproduction. The shift towards high macroalgae and low coral cover likely will continue at the same rate with or without the proposed action. Critical habitat is expected to remain functional and the proposed actions are not expected to diminish the value of critical habitat as a whole in supporting sexual and asexual reproduction. Additionally, the effects of gear deployment and vessel anchoring associated with the proposed actions will affect approximately 0.000142% of all critical habitat in the U.S. Caribbean annually, and an even lesser percentage of total designated critical habitat. The area impacted has no greater importance to the critical habitat designation than other areas within or outside of the action areas, and effects will not diminish the conservation of the critical habitat as a whole. Therefore, we believe the proposed actions are likely adversely affecting, but not destroying or adversely modifying *Acropora* critical habitat.

8.0 Conclusion

After reviewing the current status of the species, the environmental baseline, the effects of the proposed actions, and cumulative effects using the best available data, it is NMFS's biological opinion that the proposed actions are not likely to jeopardize the continued existence of the NA DPS green sea turtle, SA DPS green sea turtle, hawksbill sea turtle, Nassau grouper, oceanic whitetip shark, the Central and Southwest Atlantic DPS of scalloped hammerhead shark, elkhorn coral, staghorn coral, rough cactus coral, pillar coral, lobed star coral, mountainous star coral, or boulder star coral, or result in the destruction or adverse modification of designated *Acropora* critical habitat.

9.0 Incidental Take Statement

Section 9 of the ESA and protective regulations issued pursuant to Section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption.

Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or attempt to engage in any such conduct. *Incidental take* is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of Section 7(b)(4) and Section 7(o)(2), taking that would otherwise be considered prohibited under Section 9 or Section 4(d), but which is incidental to and not intended as part of the agency action, is not considered to be prohibited taking under the ESA, provided that such taking is in compliance with the reasonable and prudent measures and the terms and conditions of the incidental take statement (ITS) of the Opinion.

9.1 Anticipated Amount of Incidental Take

We anticipate the following incidental takes may occur in the future as a result of the proposed actions. Our analyses of effects in Section 5 estimated annual takes. However, annual take estimates can have high variability because of natural and anthropogenic variation. Based on our experience monitoring fisheries, we believe a 3-year time period is more appropriate for the meaningful monitoring of our anticipated take. This approach will allow us to reduce the likelihood of requiring reinitiation unnecessarily because of inherent variability in take levels, but it will still allow for an accurate assessment of the managed fisheries' performance versus our expectations. Thus, NMFS anticipates the following incidental takes may occur over consecutive 3-year periods as a result of the proposed actions. An exemption for the take of Nassau grouper, the Central and Southwest Atlantic DPS of scalloped hammerhead shark, oceanic whitetip shark, and lobed star, mountainous star, boulder star, pillar, and rough cactus coral in connection with this Opinion is not needed because take of these species is not prohibited; NMFS has not promulgated a Section 4(d) rule for this species. However, a circuit court has held that non-prohibited incidental take must be included in the ITS.[59] 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.

We chose to quantify coral takes by area of live tissue. We chose this metric because gear deployment and fishing vessel anchoring (the direct routes of effect from fishing activities) affect an area of the seafloor, and using this parameter made quantification of adverse effects more meaningful and it expresses the impacts in a metric that is more easily identified and monitored. The morphology of the species also makes using an area-based metric necessary. Since the polyps that make up coral colonies are so small, monitoring impacts to a single polyp would be exceptionally difficult. There can also be thousands of polyps in a single colony, an additional reason issuing take in polyps is largely impractical. Likewise, because some species (like *Acropora*) are branching, colonial species that use asexual reproduction in addition to sexual reproduction to propagate, determining discrete individuals is impossible without individual genetic identification, which is also impractical. Finally, since colonies can be of any size, issuing an ITS based on colonies would not accurately capture the potential effects to the species. For example, authorizing the take of a single colony could refer to a young, small, sexually immature colony, or it could refer to much older, much larger, sexually mature colony with far greater importance to the species. Therefore, our incidental take statement quantifies coral takes by area. Table 9.1 summarizes these estimates. Effects from the harvest of herbivorous fish are discussed after Table 9.1.

Though not authorized in the take statement, the amount of *Acropora* Critical Habitat affected is 22,222 ft^2 (Puerto Rico) + 34,174 ft^2 (St. Thomas/St. John) + 7,485 ft^2 (St. Croix) = 63,881 ft^2

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[59] *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.

total annually. This value was used in Section 7 to consider the issue of destruction or adverse modification of *Acropora* critical habitat.

Table 9.1 Summary of Anticipated Take Estimates for 3 Year Periods

a. Puerto Rico FMP

Species	Lethal Take	Non-Lethal Take
Sea Turtle- green*	6 individuals	0
Sea Turtle- hawksbill	6 individuals	0
Scalloped Hammerhead Shark	0	0
Nassau Grouper	0	0
Oceanic Whitetip Shark	33 individuals	0
Elkhorn Coral	619.5 ft^2	**
Staghorn Coral	566.7 ft^2	**
Rough Cactus Coral	36.6 ft^2	**
Pillar Coral	89.1 ft^2	**
Lobed Star Coral	420.9 ft^2	**
Mountainous Star Coral	982.8 ft^2	**
Boulder Star Coral	510.6 ft^2	**

*Up to 6 takes of green sea turtles, total, from any combination of the NA and SA DPSs

**We anticipate additional non-lethal take of these species as a result of the effect to corals from the harvest of the herbivorous fish (loss of grazing capacity). We use a take proxy, described below, for this anticipated take.

b. St. Thomas/St. John FMP

Species	Lethal Take	Non-Lethal Take
Sea Turtle- green*	9 individuals	0
Sea Turtle- hawksbill	0	0
Scalloped Hammerhead Shark	0	0
Nassau Grouper	12 individuals	0
Oceanic Whitetip Shark	48 individuals	0
Elkhorn Coral	116.1ft^2	**
Staghorn Coral	801.3 ft^2	**
Rough Cactus Coral	69.3 ft^2	**
Pillar Coral	367.8 ft^2	**
Lobed Star Coral	32,793.3 ft^2	**
Mountainous Star Coral	26,802.3 ft^2	**
Boulder Star Coral	52,175.7 ft^2	**

*Up to 9 takes of green sea turtles, total, from any combination of the NA and SA DPSs

**We anticipate additional non-lethal take of these species as a result of the effect to corals from the harvest of the herbivorous fish (loss of grazing capacity). We use a take proxy, described below, for this anticipated take.

c. St. Croix FMP

Species	Lethal Take	Non-Lethal Take
Sea Turtle- green*	3 individuals	0
Sea Turtle- hawksbill	3 individuals	0
Scalloped Hammerhead Shark	3 individuals	0
Nassau Grouper	0	0
Oceanic Whitetip Shark	6 individuals	0
Elkhorn Coral	195.9 ft^2	**
Staghorn Coral	19.2 ft^2	**
Rough Cactus Coral	12 ft^2	**
Pillar Coral	8.4 ft^2	**
Lobed Star Coral	1,305 ft^2	**
Mountainous Star Coral	581.4 ft^2	**
Boulder Star Coral	1,243.8 ft^2	**

*Up to 3 takes of green sea turtles, total, from any combination of the NA and SA DPSs

**We anticipate additional non-lethal take of these species as a result of the effect to corals from the harvest of the herbivorous fish (loss of grazing capacity). We use a take proxy, described below, for this anticipated take.

In addition to the take of coral species from gear deployment and fishing vessel anchoring, we also anticipate that effects to corals from harvest of herbivorous fishing, which can negatively impact habitat and reduce sexual and asexual recruitment success. Unlike the incidental take of corals resulting from the gear deployment (placement of traps) and anchoring anticipated and listed in Table 9.1, we cannot quantitatively estimate the amount or extent of the area of corals affected by the harvest of herbivorous fish associated with the actions. While we cannot quantitatively determine how much coral will be affected by the increased algal growth likely to occur with the harvest of herbivores, we are able to do the following: monitor those adverse effects via a take proxy (i.e., the biomass of the species and functional groups listed in Table 9.2)[60]; prescribe reasonable and prudent measures that are necessary or appropriate to minimize such impact to the species and habitat; and set forth terms and conditions (including, but not limited to, reporting requirements).

The effects we anticipate from harvest of herbivorous fish associated with the proposed actions are primarily the reduced success of coral larvae settlement and fragment reattachment (e.g., from negative consequences to habitat) because of increases in algae.[61] We require certain Reasonable and Prudent Measures (RPMs) that will help NMFS better understand the harvest of herbivorous fish.

Because impacts to reproduction cannot be monitored directly (i.e., visual detection of coral sexual recruitment is not possible [Miller et al. 2007]), monitoring the factors that affect the features of habitat necessary for coral larvae settlement and fragment reattachment is appropriate. As we believe herbivorous fish harvest is not a primary factor affecting algal cover and expansion of algal cover, and it is possible algae will remain at, or expand from, current levels of abundance even as herbivorous fish populations increase, we believe it is imprudent to monitor algal cover for purposes of assessing the impacts of harvesting herbivorous fish associated with the proposed actions. As explained in Section 3, because of the fundamentally different way that parrotfish interact with macroalgae on coral reefs compared to other herbivorous fish species (i.e., surgeonfish), we assume parrotfish are of utmost importance and focus on interactions between harvest of those species of parrotfish that would be managed by the proposed actions with ESA-listed coral.

The information learned in conjunction with the 2011 reef fish biological opinion (NMFS 2011a) indicated that, in general, intense grazing on algae by certain parrotfish species may prevent the establishment of algae, thereby maintaining algae at levels suitable to enable coral recruitment (i.e., excavating/bioeroding grazers and scraping grazers). By contrast, other species of parrotfish are important for controlling algae and preventing its expansion once it becomes established (i.e., macroalgal browsers) (Burkepile and Hay 2008, 2010). Consequently, the

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[60] For the purpose of the ITS, our reference to biomass is more accurately defined as "relative biomass." Relative biomass refers to a more general measure of the numbers of fish relative to numbers observed previously, and does not expressly take into account size in estimating the total mass of fish in a given area. This should not be confused with "absolute biomass," which is the actual biomass or amount of fish expressed in weight estimated within a given area. The relative biomass values will be estimated using the visual census data, and subsequent values can be used to infer a relative increase or decrease in biomass detected via those surveys over time. The rigorous sampling technique of the visual censuses makes comparison of relative biomass across the surveys appropriate.

[61] We use the term "algae" to refer generally to fleshy macroalgae and turf algae.

parrotfish species whose grazing can expose bare substrate is believed to be especially important for creating and maintaining micro-habitats suitable for coral recruitment (i.e., excavating/bioeroding grazers and scraping grazers) (Green and Bellwood 2009).

The parrotfish most common on hardbottom or reef habitats of the U.S. Caribbean belong to 2 genera, *Sparisoma* and *Scarus*. These species appear to play complementary roles in algal grazing (Burkepile and Hay 2008, 2010). The species within those genera that would be managed under the proposed actions are listed in Table 9.2. In combination, these species graze a wider range of algae found on reefs than any one species or genus would alone. These species can be classified into functional grazing groups ("grazing groups") according to their broadly defined diets and feeding methods. For example, *excavating/bioerodering grazers* take relatively deep, infrequent bites and usually remove portions of the hardbottom/reef substrate along with the algae attached, exposing bare substrate. *Scraping grazers* graze relatively frequently, cropping down existing algal filaments; they may or may not remove calcium carbonate. *Macroalgae browsers* graze on fleshy, established macroalgae (Bellwood and Choat 1990; Bellwood et al. 2004).

Table 9.2. Parrotfish Species Managed under the Proposed Actions and Their Functional Grazing Groups

Scientific Name	*Common Name*	*Grazing Group*
Scarus coelestinus	Midnight parrotfish	Excavating/bioeroding grazers
Scarus coeruleus	Blue parrotfish	Scraping grazer
Scarus guacamaia	Rainbow parrotfish	Excavating/bioeroding grazers
Scarus iseri	Striped parrotfish	Scraping grazer
Scarus taeniopterus	Princess parrotfish	Scraping grazer
Scarus vetula	Queen parrotfish	Scraping grazer
Sparisoma aurofrenatum	Redband parrotfish	Macroalgal browser
Sparisoma chrysopterum	Redtail parrotfish	Macroalgal browser
Sparisoma rubripinne	Redfin parrotfish*	Macroalgal browser
Sparisoma viride	Stoplight parrotfish	Excavating/bioeroding grazers

*Redfin parrotfish would not be managed under the proposed Puerto Rico FMP

We believe the biomass of parrotfish grazing groups (e.g., macroalgal browsers, excavating/bioeroding grazers, and scraping grazers), as well as the biomass of individual parrotfish species listed in Table 9.2, are the best metrics for monitoring whether the proposed actions are ultimately leading to an increase in grazing. We believe an increase in the biomass of these species and/or grazing groups will lead to increased grazing on the reefs, potentially reducing the impacts of algae on corals. If the biomass of these species and/or grazing groups stays the same, grazing capacity analyzed in this Opinion has likely remained the same, as the proposed the actions are not expected to substantially alter fishing effort. If biomass decreases in the future, we believe the overall grazing rate will be lower than what is analyzed in the Opinion and this reduction could contribute to even greater detriment to coral recruitment habitat. As we believe a number of factors may be influencing algal growth, we may or may not observe changes in algal coverage as parrotfish densities change (increase or decrease); however, we can readily observe increases or decreases in parrotfish fish biomass. Therefore, we will use the biomass of these species and functional groups as a proxy for monitoring this incidental take of coral.

The scientific literature also mentions that because the diets and grazing capacity of a parrotfish species may change as the animal grows (Bonaldo and Bellwood 2008), having an intact size structure is important to ensure maximum grazing capacity. We believe the role of parrotfish in mediating the adverse effects of algae on coral recruitment habitat is based on their ability to graze a wide variety of algae types (Burkepile and Hay 2008). We anticipate the greatest grazing capacity will be achieved when each grazing group is adequately represented (see Lokrantz et al. 2008). While we will be collecting information for specific parrotfish species, we believe our monitoring of the take proxy will be most effective by focusing on the biomass of specific grazing groups.

Because we anticipate the amount of fishing for managed parrotfish species will remain largely unchanged under the proposed actions, grazing capacity will likely remain unchanged, and the proposed actions would be expected to maintain the previously occurring level of adverse effects to ESA-listed corals from herbivorous fish harvest. To monitor that assumption we propose to monitor species-specific biomass and parrotfish grazing group biomass, as specified in Table 9.2.

Future sampling will occur each year alternating between the St. Croix and St. Thomas/St. John platform one year and Puerto Rico platform the next year. We will base our estimates of changes in parrotfish grazing (and ultimately the impacts to coral) on biomass changes occurring over time as documented via the monitoring efforts described below.

The Terms and Conditions of this Opinion will require that the biomass of parrotfish species and parrotfish functional groups be monitored. If the biomass of parrotfish species and parrotfish grazing groups are not stable or increasing, reinitiation of consultation is required. We recognize that many factors could affect parrotfish biomass, and will evaluate the potential causes of any changes and their relationship to fishing associated with the proposed actions. Decreases in parrotfish biomass could mean that parrotfish removals from the managed fisheries are greater than currently believed, though the decreases could be from other causes (e.g., parrotfish disease or increases in predation). Regardless of the cause, lower parrotfish biomass than anticipated in this Opinion would result in less grazing of algae on reefs, and potentially greater adverse effects on potential recruitment of coral than anticipated.

9.2 Effect of the Take

NMFS has determined the level of anticipated take associated with the proposed actions and specified in Section 9.1 is not likely to jeopardize the continued existence of the NA of DPS green sea turtle, SA DPS of green sea turtle, hawksbill sea turtle, Nassau grouper, oceanic whitetip shark, the Central and Southwest Atlantic DPS of scalloped hammerhead shark, elkhorn coral, staghorn coral, rough cactus coral, pillar coral, lobed star coral, mountainous star coral, or boulder star coral.

9.3 Reasonable and Prudent Measures (RPMs)

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

The RPMs and terms and conditions are required, per 50 CFR 402.14(i)(1)(ii) and (iv), to minimize the impact of that take on ESA-listed species. These measures and terms and conditions are non-discretionary, and must be implemented by the SFD for the protection of Section 7(o)(2) to apply. The SFD has a continuing duty to regulate the activity covered by this incidental take statement. If it fails to adhere to or require the applicant to adhere to the terms and conditions of the incidental take statement through enforceable terms of permits or other documents, and/or fails to retain oversight to ensure compliance with these terms and conditions, the protective coverage of Section 7(o)(2) may lapse for prohibited take. To monitor the impact of the incidental take, the SFD must report the progress of the action and its impact on the species to PRD, as specified in the incidental take statement [50 CFR 402.14(i)(3)].

We have determined that the following RPMs are necessary or appropriate to minimize the impacts of future takes or to limit adverse effects to sea turtles (the North Atlantic and South Atlantic DPSs of green sea turtles, and hawksbill sea turtles), fish (Nassau grouper, the Central and Southwest DPS of scalloped hammerhead shark, and oceanic whitetip shark), and corals (elkhorn coral, staghorn coral, rough cactus coral, pillar coral, lobed star coral, mountainous star coral, or boulder star coral) to predictable levels, and to monitor levels of incidental take during the proposed actions, and they apply to all 3 proposed actions (fishing under the Puerto Rico FMP, St. Thomas/St. John FMP, and the St. Croix FMP):

1. <u>Minimizing Sea Turtle and Fish Take and Mortality through Outreach and Education</u>

In Section 5, we described how fishing gear can adversely affect sea turtles and fish. Most, if not all, sea turtles and fish released after interaction with in hook-and-line or trap gear have experienced some degree of physiological injury caused by hooking, entanglement, or barotrauma (for barotrauma, Nassau grouper only). Experience with other hook-and-line fisheries has shown that the ultimate severity of these events is dependent not only upon the actual capture circumstances, but the amount of gear remaining on the animal at the time of release. The handling of an animal, including how it is released, also greatly affects its chance of recovery. Therefore, the experience, knowledge, ability, and willingness of fishers to remove gear and use descending devices is crucial to the survival of sea turtles and/or fish and following release. Certain behavior by fishers may also help to reduce the likelihood of takes. For these reasons, NMFS shall conduct outreach and education to ensure that sea turtle and fish takes and mortalities are minimized to the greatest extent practicable.

2. Monitoring and Assessing Parrotfish and Parrotfish Functional Grazing Group Biomass

The proposed actions together are not expected to increase adverse effects on coral from the removal of herbivorous fish, as removal of these fish is expected to remain the same or decrease. If this prediction is incorrect, however, there are several risks: (1) underestimation of the potential adverse effects to coral, and (2) possibly having reached the incorrect conclusion in our jeopardy and destruction or adverse modification analyses. It is imperative that NMFS monitor and track the biomass of the parrotfish species and parrotfish grazing groups listed in Table 9.2 on in all three action areas so NMFS can evaluate any response following the implementation of the proposed actions.

3. Monitoring the Frequency, Magnitude, and Impact of Incidental Take

The jeopardy analyses for sea turtles, fish, and coral are based, in part, on the assumption that the frequency, magnitude, and impact of incidental take estimated in this Opinion are accurate (for corals, from trap and anchor interactions). While the take estimates and associated effects on listed species are both based on the best available information, many assumptions were made to overcome poor or missing data. If our assumptions regarding the frequency and magnitude of incidental take of ESA-listed species prove to be underestimates, or are inaccurate, we risk having misjudged the potential adverse effects to these species. Thus, it is imperative that we monitor and track both the level of take of ESA-listed species under each of the proposed actions and the status of listed species. NMFS must ensure that monitoring and reporting related to sea turtle, fish, and coral take and effects associated with the proposed actions: (1) detect any adverse effects resulting from the proposed actions; (2) assess the actual level of incidental take in comparison with the anticipated incidental take documented in that Opinion; and (3) detect when the level of anticipated take is exceeded.

4. Reducing the Frequency and Amount of Trap and Vessel Anchor Damage to Corals

The proposed actions may result in physical damage to corals via trap use or anchoring. In addition, the effects analysis describes how the use of traps could destroy new growth of planulae (free-swimming coral larvae). Given these expected impacts and the importance of increasing coral recruitment in the action area, NMFS must reduce the frequency of trap and anchor damage to the extent practicable.

5. Prevent Fishing on Nassau Grouper Spawning Sites During Spawning Periods

The proposed actions must maintain existing protections for Nassau grouper spawning aggregations, including applicable seasonal and area closures.

9.4 Terms and Conditions

To be exempt from take prohibitions established by Section 9 of the ESA, the SFD must comply with or ensure compliance with the following terms and conditions for all proposed actions, which implement the RPMs described above. These terms and conditions are non-discretionary.

The following terms and conditions implement RPM No. 1.

1. NMFS, in cooperation with the Caribbean Fishery Management Council (CFMC), must coordinate with the USVI Department of Planning and Natural Resources (DPNR) and Puerto Rico Department of Natural and Environmental Resources (DNER) to develop and implement an outreach program to educate commercial and recreational fishers on the following: (1) the benefits to sea turtles and fish of using circle hooks, (2) the use of available sea turtle release equipment (e.g., dehooking gear), (3) the use of available fish release equipment (e.g., descending devices), and (4) sea turtle and fish handling protocols and guidelines.

The following terms and conditions implement RPM No. 2.

2. NMFS, in conjunction with NOAA's Coral Reef Conservation Program (CRCP), must continue to maintain the fishery-independent visual census surveys on coral reefs, which will allow NMFS to monitor the parrotfish species and grazing groups described in Table 9.2. NMFS, in cooperation with the Southeast Fisheries Science Center (SEFSC), must prepare an annual report on the biomass of the parrotfish species and grazing groups listed in Table 9.2 and other relevant information such as length-frequency of those parrotfish for each of Puerto Rico, St. Thomas/St. John, and St. Croix based on information from these fishery-independent surveys. The annual report should also include percent cover of habitat strata (including macroalgae) and coral cover and data on benthic communities (including densities of *Diadema* sea urchins) by island.
3. NMFS, in cooperation with the SEFSC, must monitor the Trip Interview Program (TIP) for length-frequency data of parrotfish landings in Puerto Rico, St. Thomas/St. John, and St. Croix. NMFS, in cooperation with the SEFSC, must prepare an annual report of the species-specific length-frequency for all parrotfish species listed in Table 9.2 by island. This data is important to compare the species and length of parrotfish targeted by fishers with those observed during the fishery-independent census.
4. NMFS, in cooperation with the SEFSC, must coordinate with the Puerto Rico DNER and USVI DPNR to maintain improvements to reporting of fisheries landings (e.g., species-specific landings, additional bycatch information). Because of the likely importance of herbivorous species diversity and their complementary grazing preferences, species-specific landings will be obtained annually for species listed in Table 9.2.

The following terms and conditions implement RPM No. 3.

5. NMFS, in coordination with the SEFSC, must coordinate with the CFMC, Puerto Rico DNER, and USVI DPNR, to ensure that data collection methods are in place to record the number of all ESA-species (sea turtles and fish) taken by the commercial and recreational fishery components, and their disposition (e.g., landed, released, released with gear) and condition (e.g., dead, alive, injured). Currently no information is collected on recreational fishing in the U.S. Caribbean, and this information is needed. To ensure the quality of the ESA-listed sea turtle and fish data reported, NMFS, in cooperation with the

CFMC, must distribute educational outreach materials regarding the specific information to be reported and ESA-listed sea turtle and fish identification to commercial fishermen.

6. NMFS, in coordination with the SEFSC, must coordinate with the CFMC, Puerto Rico DNER, and USVI DPNR, to collect data on the trap use (number deployed, size, location of use, including habitat features and depth) and anchoring (frequency, size of anchor and length of line, location, including habitat features and depth) by commercial and recreational fishers in federal waters off Puerto Rico, St. Thomas/St. John, and St. Croix.
7. NMFS, in coordination with the Puerto Rico and USVI sea turtle stranding coordinators, must improve collection and reporting of incidental capture and strandings data on all ESA-listed sea turtles from the USVI and Puerto Rico. As a way to do this, a workshop is advised as a mechanism to initiate improved data and coordination.
8. NMFS, in coordination with the Puerto Rico and USVI sea turtle stranding coordinators, must ensure that sea turtle stranding data from Puerto Rico and USVI is reported to the Sea Turtle Stranding and Salvage Network on a regular basis (at least annually).
9. NMFS, in collaboration with the SEFSC, must develop a proposal to conduct a survey on interactions between ESA-listed sea turtles and fish and commercial and recreational fishers, similar to Lewis et al. (2007). In preparing the proposal, NMFS should coordinate with the CFMC, Puerto Rico DNER, and USVI DPNR.
10. NMFS must submit an annual report on the take of sea turtles and fish associated with the proposed actions. With respect to corals, NMFS must estimate the area of coral impacted by traps and anchors associated with the proposed actions.

The following terms and condition implement RPM No. 4.

11. NMFS must assist fishers and the USVI government with efforts already underway to control fishing effort through a trap certificate program. NMFS must consider whether a similar effort is appropriate and feasible in Puerto Rico.
12. NMFS, in cooperation with the CFMC, must coordinate with the USVI DPNR and Puerto Rico DNER to collect information on the amount of anchoring in coral areas in federal waters, including the specific locations.
13. NMFS, in cooperation with the CFMC, must coordinate with the USVI DPNR and Puerto Rico DNER to develop and implement an outreach program to educate commercial and recreational fishers on the impacts of gear use and anchoring in coral areas. A goal of this education would be to teach fishers to avoid trap use and anchoring in coral areas.

The following terms and condition implement RPM No. 5.

14. NMFS must submit an annual report on compliance with federal regulations that prohibit fishing at sites where Nassau grouper form spawning aggregations (i.e., Grammanik Bank seasonal closure (St. Thomas), Hind Bank Marine Conservation District year-round closure (St. Thomas), and Bajo de Sico seasonal closure (Puerto Rico)). These reports should summarize any pertinent information from law enforcement.

10.0 Conservation Recommendations

Section 7(a)(1) of the ESA directs federal agencies to utilize 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.

The following additional measures are recommended. For PRD to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, PRD requests notification of the implementation of any conservation recommendations.

Sea Turtles:

1. To better understand populations and the impacts of incidental take in the fisheries, NMFS should support in-water abundance estimates of affected species to achieve more accurate status assessments for these species and improve our ability to monitor them.
2. Once reasonable in-water estimates are obtained, NMFS should support risk analyses of the populations affected by the fisheries. This will help improve the accuracy of future assessments of the effects of different levels of take on populations.

Coral:

3. NMFS should conduct or fund efforts to increase the assessment, monitoring, and modeling of coral reefs in the U.S. Caribbean to allow for a better understanding of abundance and distribution within the area.
4. NMFS should conduct or fund research into identifying and quantifying the impacts of fishing related marine debris, particularly trap rope, on coral.
5. NMFS should conduct or fund coral restoration efforts in the U.S. Caribbean.
6. NMFS should conduct or fund research designed to better understand the role of *Diadema* and herbivorous fish on a healthy reef ecosystem in the U.S. Caribbean, and whether the loss of *Diadema* can be compensated for by increase in herbivorous fish populations at an ecosystem scale.
7. NMFS should conduct or fund research designed to determine if any actions could or should be taken to enhance and accelerate the recovery of *Diadema*.

Fish

8. NMFS should evaluate whether there are additional Nassau grouper spawning sites or spawning aggregation sites that require protection, and whether existing seasonal closures encompass the entirety of the spawning period.

9. NMFS should conduct or fund research designed to better understand the potential for interactions with the proposed actions and giant manta ray, Central and Southwest Atlantic DPS of scalloped hammerhead shark, and oceanic whitetip shark.

All Species:

10. NMFS should encourage the USVI and Puerto Rico to develop and implement programs aimed at helping conserve the species considered in this opinion and occurring in waters off Puerto Rico and the USVI.
11. NMFS should conduct or fund research into the efficacy of marine debris removal programs, for the purpose of identifying potential ways to improve the efficiency of such programs.
12. NMFS should encourage the USVI and Puerto Rico to apply for funds available under Section 6 of the ESA, to conduct research into the impacts of trap fisheries on species occurring in state waters.
13. The majority of fishing in the U.S. Caribbean is most likely to occur in commonwealth or territorial waters. As such, the greatest conservation value to the species analyzed in this Opinion will come from minimizing adverse impacts from fishing occurring in commonwealth and territorial waters. Therefore, NMFS must engage the Commonwealth of Puerto Rico and the USVI to seek to develop changes in those fisheries that reduce impacts to ESA-listed species. Specifically, NMFS should encourage the Commonwealth of Puerto Rico and the USVI to pursue an ESA section 10(a)(1)(B) Incidental Take Permit and develop a Conservation Plan for their fisheries.

11.0 Reinitiation of Consultation

This concludes formal consultation on the proposed actions. As provided in 50 CFR 402.16, reinitiation of formal consultation is required if discretionary federal action agency involvement or control over the action has been retained, or is authorized by law, and if (1) the amount or extent of the taking specified in the incidental take statement is exceeded; (2) new information reveals effects of the agency action that may affect listed species or designated critical habitat in a manner or to an extent not previously considered in this Opinion; (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 Opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action.

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Appendix A. Boundary Coordinates for Managed Areas in the U.S. Caribbean

Puerto Rico Management Area

Table A.1. Under the Puerto Rico fishery management plan (FMP), the Puerto Rico management area is bounded by rhumb lines connecting, in order, the following points.

Point	North Latitude	West Longitude
A (intersects with the International/EEZ boundary; from Point A proceed southerly to Point B)	19°37′29″	65°20′57″
B (from Point B, proceed southerly along the 3-nautical mile Territorial boundary of the St. Thomas/St. John management area to Point C)	18°25′46.3015″	65°06′31.866″
C (from Point C proceed southeasterly to Point D)	18°13′59.0606″	65°05′33.058″
D (from Point D proceed southwesterly to Point E)	18°01′16.9636″	64°57′38.817″
E (from Point E proceed southerly to Point F)	17°30′00.000″	65°20′00.1716″
F (from Point F, proceed southwesterly, then northerly, then easterly, and finally southerly along the International/EEZ boundary to Point A)	16°02′53.5812″	65°20′00.1716″

Table A.2. Boundary coordinates of the seasonal area closure locations off western Puerto Rico. Management measures included in the Puerto Rico FMP specific to these areas are only applicable in federal waters.

Location	Bounding Point	North Latitude	West Longitude
Tourmaline Bank	A	18°11.2′	67°22.4′
	B	18°11.2′	67°19.2′
	C	18°08.2′	67°19.2′
	D	18°08.2′	67°22.4′
Abrir La Sierra Bank	A	18°06.5′	67°26.9′
	B	18°06.5′	67°23.9′
	C	18°03.5′	67°23.9′
	D	18°03.5′	67°26.9′
Bajo de Sico	A	18°15.7′	67°26.4′
	B	18°15.7′	67°23.2′
	C	18°12.7′	67°23.2′
	D	18°12.7′	67°26.4′

St. Thomas/St. John Management Area

Table A.3. Under the St. Thomas/St. John FMP, the St. Thomas/St. John management area is bounded by rhumb lines connecting, in order, the following points.

Point	North Latitude	West Longitude
A (intersects with the International/EEZ boundary; from Point A, proceed southeasterly along the International/EEZ boundary to Point G)	19°37′29″	65°20′57″
G (from Point G proceed westerly to Point D)	18°03′03″	64°38′03″
D (from Point D proceed northwesterly to Point C)	18°01′16.9636″	64°57′38.817″
C (from Point C proceed northerly along the 3-nautical mile Territorial boundary of the St. Thomas/St. John island group to Point B)	18°13′59.0606″	65°05′33.058″
B (from Point B proceed northerly to Point A)	18°25′46.3015″	65°06′31.866″

Table A.4. Boundary coordinates of the seasonal area closure locations off southern St. Thomas/St. John, U.S. Virgin Islands. Management measures included in the St. Thomas/St. John FMP specific to these areas are only applicable in federal waters.

Location	Bounding Point	North Latitude	West Longitude
Hind Bank Marine Conservation District	A	18°13.2′	65°06.0′
	B	18°13.2′	64°59.0′
	C	18°11.8′	64°59.0′
	D	18°10.7′	65°06.0′
Grammanik Bank	A	18°11.898′	64°56.328′
	B	18°11.645′	64°56.225′
	C	18°11.058′	64°57.810′
	D	18°11.311′	64°57.913′

St. Croix Management Area

Table A.5. Under the St. Croix FMP, the St. Croix management area is bounded by rhumb lines connecting, in order, the following points.

Point	North Latitude	West Longitude
G (from Point G, proceed easterly, then southerly, then southwesterly along the International/EEZ boundary to Point F)	18°03′03″	64°38′03″
F (from Point F proceed northerly to point E)	16°02′53.5812″	65°20′00.1716″
E (from Point E proceed northeasterly to point D	17°30′00.000″	65°20′00.1716″
D (from Point D proceed easterly to Point G)	18°01′16.9636″	64°57′38.817″

Table A.6. Boundary coordinates of the seasonal area closure locations off southwestern and eastern St. Croix, U.S. Virgin Islands. Management measures included in the St. Croix FMP specific to these areas are only applicable in federal waters.

Location	**Bounding Point**	**North Latitude**	**West Longitude**
Mutton Snapper Spawning Aggregation Area	A	17°37.8′	64°53.0′
	B	17°39.0′	64°53.0′
	C	17°39.0′	64°50.5′
	D	17°38.1′	64°50.5′
	E	17°37.8′	64°52.5′
Red Hind Spawning Aggregation Area (Lang Bank)	A	17°50.2′	64°27.9′
	B	17°50.1′	64°26.1′
	C	17°49.2′	64°25.8′
	D	17°48.6′	64°25.8′
	E	17°48.1′	64°26.1′
	F	17°47.5′	64°26.9′

Appendix B. Anticipated Incidental Take of ESA-Listed Sea Turtle Species in Federal Fisheries

Anticipated Incidental Takes of Sea Turtles in Federal Fisheries (Note: loggerhead, leatherback, and Kemp's ridley sea turtles are not likely to be adversely affected by the proposed actions considered in this opinion)

Fishery	ITS Authorization Period	Sea Turtle Species				
		Loggerhead	**Leatherback**	**Kemp's ridley**	**Green**	**Hawksbill**
American Lobster [NER]	1 Year	1-Lethal or nonlethal	7 Lethal or nonlethal	None	None	None
Batched Consultation* (gillnet) [NER]	1 Year	269-No more than 167 lethal (Takes based on a 5-yr average)	4-No more than 3 lethal	4-No more than 3 lethal	4-No more than 3 lethal	None
Batched Consultation* (bottom trawl) [NER]	1 Year	213-No more than 71 lethal (Takes based on a 4-yr average)	4-No more than 2 lethal	3-No more than 2 lethal	3-No more than 2 lethal	None
Batched Consultation* (trap/pot) [NER]	1 Year	1-Lethal or nonlethal	4-Lethal or nonlethal	None	None	None
Caribbean Reef Fish [SER]	3 Years	None	18-All lethal	None	75-All lethal	51-No more than 3 lethal
Coastal Migratory Pelagics [SER]	3 Years	27 Total, 7 lethal	1- Lethal	8- Total, 2 lethal	31-Total, 9 lethal	1- Lethal
Dolphin-Wahoo [SER]	1 Year	12-No more than 2 lethal	12-No more than 1 lethal	3 for all species in combination-no more than 1 lethal take		
Gulf of Mexico Reef Fish [SER]	3 Years	1,044-No more than 572 lethal	11-All lethal	108-No more than 41 lethal	116-No more than 75 lethal	9-No more than 8 lethal

* Batched consultation includes the Northeast Multispecies, Monkfish, Spiny Dogfish, Atlantic Bluefish, Northeast Skate Complex, Mackerel/Squid/Butterfish, and Summer Flounder/Scup/Black Sea Bass Fisheries

Anticipated Incidental Takes of Sea Turtles in Federal Fisheries, continued

Fishery	ITS Authorization Period	Sea Turtle Species				
		Loggerhead	**Leatherback**	**Kemp's ridley**	**Green**	**Hawksbill**
HMS-Non-Pelagic Longline [SER]	3 Years	91-No more than 51 lethal	7-No more than 4 lethal	22-No more than 11 lethal	NA DPS 46-No more than 25 lethal SA DPS 3-No more than 2 lethal	2-No more than 1 lethal
HMS-Pelagic Longline [SER]	3 Years	1,080-No more than 280 lethal	996-No more than 275 lethal	**	**	**
Red Crab [NER]	1 Year	1-Lethal or nonlethal	1-Lethal or nonlethal	None	None	None
Caribbean Spiny Lobster	3 Years	None	9 – Lethal or non-lethal	None	12- Lethal or non-lethal	12 – Lethal or non-lethal take
Gulf of Mexico/South Atlantic Spiny Lobster Fishery [SER]	3 Years	3-Lethal or Nonlethal Take	1 –Lethal or Nonlethal take for Leatherbacks, Hawksbill, and Kemp's ridley		3-Lethal or Nonlethal Take	1 –Lethal or Nonlethal take for Leatherbacks, Hawksbill, and Kemp's ridley
South Atlantic Snapper-Grouper [SER]	3 Years	629-No more than 208 lethal	6-No more than 5 lethal	180-No more than 59 lethal	NA DPS – 111-No more than 42 lethal SA DPS - 6-No more than 3 lethal	6-No more than 4 lethal
Southeastern U.S. Shrimp [SER]	1 Year	Anticipated shrimp trawl effort (i.e., 132,900 days fished in the Gulf of Mexico and 14,560 trips in the south Atlantic) and fleet TED compliance (i.e., compliance resulting in overall average sea turtle catch rates in the shrimp otter trawl fleet at or below 12%) are used as surrogates for numerical sea turtle take levels.				

Table Anticipated Incidental Takes of Sea Turtles in Federal Fisheries, continued

Fishery	ITS Authorization Period	Sea Turtle Species				
		Loggerhead	Leatherback	Kemp's ridley	Green	Hawksbill
Atlantic Sea Scallop – Dredge [NER]	1 Year	161 – No more than 46 lethal	2 –Lethal Takes (gears combined)	3 – No more than 2 Lethal (gears combined)	2 - Lethal takes (gears combined)	None
Atlantic Sea Scallop – Trawl [NER]	1 Year	140 – No more than 66 lethal				None

* Batched consultation includes the Northeast Multispecies, Monkfish, Spiny Dogfish, Atlantic Bluefish, Northeast Skate Complex, Mackerel/Squid/Butterfish, and Summer Flounder/Scup/Black Sea Bass Fisheries

** 21 (8 of which are mortalities) "Other Hardshell" Sea Turtle (any combination of NA green, SA green, hawksbill, Kemp's ridley, or olive ridley sea turtles)