Endangered Species Act - Section 7 Consultation
Biological Opinion

Agency:	National Oceanic and Atmospheric Administration (NOAA), National Marine Fisheries Service (NMFS), Southeast Regional Office (SERO), Sustainable Fisheries Division (F/SER2)
Activity:	Continued Authorization of Reef Fish Fishing Managed under the Reef Fish Fishery Management Plan (FMP) of Puerto Rico and the U.S. Virgin Islands (CRFFMP) (Consultation Number F/SER/2010/06680)
Consulting Agency:	NOAA, NMFS, SERO, Protected Resources Division (F/SER3)
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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 ensure any action they authorize, fund, or carry 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. NMFS and the U.S. Fish and Wildlife Service (USFWS) share responsibilities for administering the ESA. When the action of a federal agency may affect an ESA-listed species or its critical habitat, that agency is required to consult with either NMFS or the USFWS, depending upon the protected species that may be affected.

Consultations on most listed species and critical habitat in the marine environment are conducted between the action agency and NMFS. Consultations are concluded after NMFS determines that an action is not likely to adversely affect listed species or critical habitat, or issues a biological opinion (opinion) identifying whether a proposed action is likely to jeopardize the continued existence of a listed species, or destroy or adversely its modify critical habitat. If jeopardy or destruction or adverse modification is found to be likely, NMFS must identify reasonable and prudent alternatives (RPAs) to the action, if any, that would avoid violating section 7(a)(2) of the ESA. The opinion also includes an incidental take statement (ITS) specifying the amount or extent of incidental take of the listed species that may occur. Non-discretionary reasonable and prudent measures (RPMs) to minimize the impact of the incidental taking are included, and conservation recommendations are made. No incidental destruction or adverse modification of critical habitat can be authorized. Therefore, there are no reasonable and prudent measures, only reasonable and prudent alternatives that must avoid destruction or adverse modification.

This document constitutes NMFS' opinion on the effects of its continued authorization of reef fish fishing 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 continued operation of reef fish fishing managed under the CRFFMP including all amendments implemented to date, as well as the actions proposed in Amendments 5 and 6 to the CRFFMP. 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, F/SER2 is the action agency and the consulting agency is F/SER3.

This opinion has been prepared in accordance with section 7 of the ESA and regulations promulgated to implement that section of the ESA. This opinion is based on information provided in Amendment 5 and 6 to the CRFFMP (hereafter Amendment 5 or the ACL Amendment, or Amendment 6), including a Draft Environmental Impact Statement, Biological Assessment, Regulatory Impact Review, Initial Regulatory Flexibility Analysis, and Social Impact Assessment (Caribbean Fishery Management Council [CFMC] and NMFS 2011) and published and unpublished scientific information on the biology and ecology of endangered and threatened sea turtles, corals, and coral reefs as cited herein.

1.0 Consultation History

Summary of Previous Consultations
An informal section 7 consultation was completed on the original CRFFMP. The consultation concluded that based on the best available information, populations of endangered and threatened species and their critical habitat would not be adversely affected by the FMP.

On April 28, 1989, NMFS conducted a section 7 consultation on the effects of all commercial fishing activities in the Southeast Region. The resulting opinion concluded that commercial fishing activities in the Southeast Region were not likely to jeopardize the continued existence of any threatened or endangered species.

Amendment 1 to the CRFFMP, implemented in December 1990, proposed no changes to the manner in which species listed under the Endangered Species Act were affected by fishing managed under the CRFMP. NMFS concluded its proposed actions would have no anticipated impact on threatened or endangered species.

An informal ESA section 7 consultation was conducted on Amendment 2 to the CRFFMP in August 1992. NMFS concluded that neither the directed fisheries nor the management measures proposed in Amendment 2 would not jeopardize endangered or threatened species, or adversely impact their critical habitat.

An informal ESA section 7 consultation was conducted on a regulatory amendment to the CRFFMP in October 1996. NMFS concluded that the regulatory amendment and the operation of the U.S. Caribbean reef fish fishery, in accordance with the amendment, were not likely to adversely affect endangered or threatened species under NMFS' jurisdiction.

An informal ESA section 7 consultation was conducted on the Generic EFH Amendment in 1998. NMFS concluded that the amendment would not adversely affect the recovery of endangered or threatened species, or their critical habitat. NMFS only partially approved the Generic EFH Amendment in March 1999, finding that it did not evaluate all managed species or all fishing gears with the potential to damage fish habitat.

An informal ESA section 7 consultation was conducted on the Comprehensive Amendment to FMPs of the U.S. Caribbean in August 2001. NMFS concluded the management measures proposed in the amendment were not likely to adversely affect any listed species under the ESA. The Comprehensive Amendment was ultimately disapproved by NMFS in May 2002.

Effects of the reef fish fishery managed under the CRFFMP were most recently analyzed as part of a May 19, 2005, opinion [NMFS 2005a; hereafter referred to as the 2005 Caribbean opinion or NMFS (2005a)], which evaluated the effects of all Caribbean fisheries in the EEZ on listed species and designated critical habitat. NMFS (2005a) concluded the continued authorization of all Caribbean fisheries in the EEZ (including the reef fish fishery) was not likely to jeopardize the continued existence of green, hawksbill, leatherback, or loggerhead sea turtles. An incidental take statement was issued specifying the amount and extent of anticipated take of these species, along with reasonable and prudent measures and associated terms and conditions deemed

necessary and appropriate to minimize the impact of these takes; both the reef fish and spiny lobster fisheries were allotted take. Other listed species (olive ridley sea turtles and listed marine mammals) and designated critical habitat for sea turtles in the action area were determined not likely to be adversely affected by the continued authorization of any Caribbean EEZ fisheries.

Cause for Reinitiation and Present Consultation History
As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of taking specified in the incidental take statement is met or exceeded; (2) new information reveals effects of the action that may affect listed species or critical habitat (when designated) in a manner or to an extent not previously considered; (3) the identified action is subsequently modified in a manner that causes an effect to listed species or critical habitat that was not considered in the opinion; or (4) a new species is listed or critical habitat designated that may be affected by the identified action.

On March 30, 2010, F/SER2 requested reinitiation of section 7 consultation on the reef fish fishery managed by the CRFFMP specifically to address adverse effects on threatened elkhorn (*Acropora palmata*) and staghorn (*Acropora cervicornis*) corals and their designated critical habitat, which were all listed subsequent to completion of the 2005 Caribbean opinion. At that time, F/SER2 and the CFMC were in the process of developing a public hearing draft for Amendment 5 to establish ACLs and accountability measures (AMs) for reef fish species classified as undergoing overfishing in the U.S. Caribbean. F/SER2 requested that F/SER3 use that document, when available, as its biological assessment.

The anticipated Amendment 5 and associated rulemaking was expected to potentially change the extent to which elkhorn and staghorn corals and their critical habitat would be affected in the future. Therefore, F/SER3 indicated the consultation and new opinion would need to be coordinated with the rulemaking. F/SER3 notified F/SER2 that the consultation could not be formally initiated until the proposed action was defined by preferred alternatives and associated analysis was complete.

F/SER3 received several early drafts of Amendment 5 via its participation on the interdisciplinary planning team for that amendment. At the April 2010 CFMC meeting, F/SER3 provided a preliminary assessment of Amendment 5 and recommended additional measures be taken to reduce effects on elkhorn and staghorn corals and their critical habitat. Initial preferred alternatives were selected by the CFMC for all actions except for one (i.e., Action 6) at that meeting. Following the meeting, CFMC and F/SER2 staff continued to analyze these alternatives.

In July 2010, the public hearing draft for Amendment 5 was completed by F/SER2 and CFMC staff. Public hearings were held July 19-22, 2010 and the written public comment period on the draft ended on September 7, 2010.

The CFMC met again September 2010. During this meeting they reviewed the public hearing draft of Amendment 5, along with public comments received on it. After revising some of the preferred alternatives selected at the August meeting and selecting a preferred alternative for

Action 6, the CFMC voted to submit Amendment 5 to the Secretary for review. Following the September 2010 meeting, components of the amendment and draft proposed rule needed to be modified to reflect changes to the proposed action made at that meeting. The final draft of Amendment 5 in accord with the September 2010 meeting was completed in December 2010. On January 11, 2011, F/SER2 provided F/SER3 with a copy of it and requested that F/SER3 analyze its preferred alternatives as part of the ongoing consultation. Formal consultation was reinitiated upon receipt of that draft.

The Magnuson-Stevens Act (MSA) requires that ACLs and AMs be established by 2011 for all other managed species not undergoing overfishing. A scoping document to establish ACLs and AMs for all other CFMC-managed species not classified as undergoing overfishing was prepared by NMFS and CFMC staff during fall 2010 and reviewed by the CFMC at their December 9-10, 2010 meeting in Fajardo, Puerto Rico. Scoping hearings were held in Puerto Rico and the USVI in February 2011. Based on the decisions at a March 29-31, 2011 CFMC meeting, a draft public hearing document, including (among other CFMC FMP amendments) Amendment 6 to the CRFFMP, was developed for public input and approved by the CFMC for Secretarial review at the August 2011 meeting.

On April 8, 2011, F/SER2 informed F/SER3 of modifications to its proposed action subject to section 7 consultation resulting from decisions made by the CFMC at their March 29-31, 2011, meeting. Specifically, the CFMC voted to reduce the proposed ACLs for parrotfish on each of the three island groups from the levels recommended by the Scientific and Statistical Committee (SSC) in order to account for uncertainty in the scientific and management process. For St. Croix, further reductions to the proposed ACL were also made to reduce the impacts of parrotfish harvest on *Acropora* in St. Croix waters where parrotfish harvest is particularly intense. The reduction factors were 15% for parrotfish in Puerto Rico and St. Thomas/St. John, and ~20% for parrotfish in St. Croix. These uncertainty reductions had been included when setting proposed ACLs for all other reef fish species included in Amendment 5, but were not initially included for parrotfish. The revised proposed parrotfish ACLs are 240,000 pounds of whole fish for St. Croix, 42,500 pounds of whole fish for St. Thomas and St. John, and 68,000 pounds of whole fish for Puerto Rico. These values are a reduction from the SSC recommendations of 300,000 pounds for St. Croix, 50,000 pounds for St. Thomas and St. John, and 80,000 pounds for Puerto Rico.

During the March 29-31, 2011 meeting, the Council also discussed alternatives for Amendment 6 to the Reef Fish FMP, which would establish ACLs for the remaining reef fish species not considered to be undergoing overfishing, including surgeonfish. The Council chose a preferred alternative for establishing an uncertainty reduction for surgeonfish in Amendment 6. Specifically they chose a 25% reduction from the allowable biological catch (ABC) when setting the ACL for surgeonfish because of their essential role in coral reef communities, and because of the uncertainty surrounding the level of harvest and the impact of that harvest with respect to provisioning of critical habitat. However, at the meeting, the time series to be used to calculate the surgeonfish ABC and ACL had not been determined, so there was no numerical value to which to apply the proposed 25% reduction. During that meeting, the CFMC had received another update from F/SER3 on the status of the draft opinion regarding the relationship between parrotfish, parrotfish harvest, and the availability of critical habitat substrate for *Acropora*

settlement. The intent of the decision was to allow F/SER3 to consider both major herbivore group ACLs in the ongoing consultation on Amendment 5 to (1) help keep the timeline intact for Amendment 6 by avoiding or reducing future consultation time needs, and (2) consider the cumulative effects of the herbivore ACLs.

On May 12, 2011, F/SER2 requested F/SER3 incorporate the surgeonfish ACLs proposed in Amendment 6 into the opinion being developed for the continued authorization of the Caribbean reef fish fishery. On May 27, 2011, F/SER2 provided revised proposed ACLs for surgeonfishes in St. Thomas/St. John and St. Croix, based on the outcome of a May 24-25 SSC meeting. The memorandum also indicated that the surgeonfish ACL for Puerto Rico had not been specified, but was unlikely to be greater than 10,000 lbs.

On August 24, 2011, F/SER2 informed F/SER3 that during a special meeting convened on July 21, 2011, the Council selected preferred alternatives for all remaining actions in Amendment 6. During that meeting the surgeonfish ABCs and ACLs for St. Thomas/St. John and St. Croix specified in the May 12, 2011, F/SER 2 memo were confirmed and did not change. However, the surgeonfish ACL for Puerto Rico was set at 10,768 lbs, after reducing the ABC by 25% to account for uncertainty.

2.0 Description of the Proposed Action

On January 12, 2007, Congress amended the MSA with passage of the Magnuson-Stevens Fishery Conservation and Management Reauthorization Act (MSRA). While maintaining the requirement that "conservation and management measures shall prevent overfishing while achieving, on a continuing basis, the optimum yield from each fishery for the United States fishing industry," the MSRA added new requirements to end and prevent overfishing including the use of Annual Catch Limits (ACLs) and Accountability Measures (AMs). Specifically, the MSRA requires that FMPs "establish a mechanism for specifying ACLs in the plan (including a multiyear plan), implementing regulations, or annual specifications, at a level such that overfishing does not occur in the fishery, including measures to ensure accountability" (MSRA Section 303(a)(15)). The MSRA requires that ACLs and AMs be established in 2010 for those species classified as undergoing overfishing and by 2011 for most other managed species not classified as undergoing overfishing.

F/SER2 is proposing the continued authorization of the federal Caribbean reef fish fishery as managed under the CRFFMP, including proposed Amendment 5 (CFMC and NMFS 2011) and Amendment 6. Amendment 5, which was prepared by the CFMC and F/SER, is designed to bring the Caribbean reef fish fishery into compliance with the 2007 revisions to the MSA for species classified as undergoing overfishing (i.e., snapper, grouper, and parrotfish). The proposed ACLs for the remaining CFMC-managed reef fish species, including surgeonfish, would be implemented with other actions via Amendment 6.

Amendment 5

The purpose of Amendment 5 is to revise management reference points and status determination criteria for snapper, grouper, and parrotfish; specify ACLs and AMs for those species/species groups classified as undergoing overfishing to prevent overfishing of these species/species

groups; establish framework measures to facilitate regulatory modifications; adjust management measures as needed to constrain harvest to specified ACLs; and minimize, to the extent practicable, negative socioeconomic impacts that may result from the amendment actions. Six actions are included in Amendment 5 to achieve its purpose: (1) amend the stock complexes in the reef fish fishery management unit; (2) revise management reference points to transition U.S. Caribbean reef fish management toward those mandated by the MSRA; (3) specify ACLs for the various U.S. Caribbean island groups; (4) establish management measures to prohibit the harvest of three, relatively large and long-lived parrotfish species (midnight, blue, and rainbow) that serve an essential ecological role and recreational bag limits for select reef fish species, including parrotfish; (5) provide guidelines for triggering AMs and for applying those AMs; (6) establish framework provisions separately for reef fish. Collectively, these actions serve to provide a basic foundation for place-based reef fish fisheries management in the U.S. Caribbean.

Amend the Stock Complexes in the Reef Fish Fishery Management Unit

The reef fish complexes that comprise the grouper and snapper components of the Reef Fish FMP for the U.S. Caribbean are composed of five grouper units and four snapper units. Unit composition presently excludes several species of commonly harvested fish and also fails to aggregate species in an ecologically consistent manner. Black grouper (*Mycteroperca bonaci*) is not included in any of the units although this species is frequently caught by recreational anglers. Both misty (*Epinephelus mystacinus*) and yellowedge (*E. flavolimbatus*) grouper are presently included in Grouper Unit 4, but these two species are found at water depths much greater than are the other members of Grouper Unit 4. Creole-fish (*Paranthias furcifer*) are rarely caught by commercial or recreational fishers. The cardinal snapper (*Pristipomoides macropthalmus*) is commonly caught by commercial fishers but is not included as a member of any FMU. Lastly, the wenchman (*P. aquilonaris*) presently is included as a member of Snapper Unit 2 but clusters most closely with members of Snapper Unit 1 (silk (*Lutjanus vivanus*), black (*Apsilus dentatus*), blackfin (*L. buccanella*), and vermilion (*Rhomboplites aurorubens*)), based upon depth and habitat preferences. The proposed action to amend the stock complexes in the reef fish FMU would: (1) add black grouper to Grouper Unit 4 along with other members of that unit with common habitat and depth preferences, (2) create a new Grouper Unit 5 that would contain both misty and yellowedge grouper, (3) move creole-fish from Grouper Unit 3 to the 'data collection only' category, (4) add cardinal snapper to Snapper Unit 2 because of similarities with the queen snapper (*Etelis oculatus*) in landings records and depth distribution, and (5) move the wenchman into Snapper Unit 1.

Revise Management Reference Points and Establish ACLs and AMs and Island-Specific Allocation and Management

The proposed action would revise the MSY, OY and overfishing thresholds and establish ACLs and AMs for all snapper, grouper, and parrotfish in the Caribbean reef fish FMP (Table 2.1). The U.S. Caribbean constitutes a single EEZ, but effective island-specific management of reef fish resources requires that management boundaries be established between islands. This is necessary to enable application of AMs in response to harvesting activities on a single island (Puerto Rico, St. Croix) or island group (St. Thomas/St. John) without affecting activities on the other islands or island groups. The proposed action would establish geographic boundaries between islands/island groups based upon an equidistant approach that uses a mid-point to divide the EEZ among islands. Recreational landings data are available for Puerto Rico, but not the

USVI. Thus, separate sector ACLs (commercial and recreational ACLs) would only be established for species in Puerto Rico. CFMC chose average landings as the best proxy for Maximum Sustainable Yield, from which ACLs would be constructed. The year sequence used for each ACL represents the longest time series of catch data prior to the Comprehensive Sustainable Fisheries Act Amendment (which included provisions that may have substantially altered catch patterns) that is consistently reliable across all U.S. Caribbean islands. The proposed ACLs account for uncertainty in the scientific and management process. The proposed ACLs determined for the Caribbean reef fish species (including parrotfish) include a 15% uncertainty reduction applied to them. For St. Croix only, another 5.88% reduction was applied to the parrotfish ACL value. The reductions to the parrotfish ACL in St Croix were made to address uncertainty regarding the impact parrotfish harvest may be having on *Acropora* because of their role in grazing on macro-algae. In St. Croix, parrotfish harvest is particularly intense and, therefore, the further reduction was applied for this island management area to reduce the impacts of parrotfish harvest on *Acropora* species in St. Croix waters.

Table 2.1 Proposed ACLs by Management Area and Species or Species Groups

ACL	Puerto Rico		St. Croix landings	St. Thomas/St. John landings
	Commercial landings	Recreational landings		
Parrotfishes	52,737 lb (23,915 kg)	15,263 lb (6,921 kg)	240,000 lb (108,863 kg)	42,500 lb (19,278 kg)
Snapper Unit 1	284,685 lb (129,131 kg)	95,526 lb (43,330 kg)	--	--
Snapper Unit 2	145,916 lb (66,186 kg)	34,810 lb (15,790 kg)	--	--
Snapper Unit 3	345,775 lb (156,841 kg)	83,158 lb (37,720 kg)	--	--
Snapper Unit 4	373,295 lb (169,324 kg)	28,509 lb (12,931 kg)	--	--
Snappers	--	--	102,946 lb (46,696 kg)	133,775 lb (60,679 kg)
Groupers	177,513 lb (80,519 kg)	77,213 lb (35,023 kg)	30,435 lb (13,805 kg).	51,849 lb (23,518 kg)

Accountability measures are designed to rectify the potential impacts that may result from harvest exceeding the ACL. Two components are considered, the first identifying the conditions under which AMs are triggered and the second describing the action(s) that would occur if AMs are triggered. Under the proposed action, AMs would be triggered: (1) in the year following a determination that the ACL was exceeded, based upon a single year of landings from 2010, (2) in the year following a determination that the ACL was exceeded, based upon a 2-year average of landings from 2010 and 2011, and (3) in the year following a determination that the ACL is exceeded based upon a 3-year average of landings from 2010, 2011, and 2012. Thereafter, the determination would be based upon the most recent 3-year average of landings. Both commercial and recreational landings of a species, species group, or complex vary substantially from year to year; applying a 3-year average dampens that variability and provides better information with regard to precision of the comparison. When AMs are triggered, NMFS would reduce the length of the fishing season for the affected species or species group the year following the trigger determination by the amount needed to prevent such an overage from occurring again.

Establish General Management Measures

The proposed action would prohibit the EEZ harvest of three largest species of parrotfish that occur on Caribbean coral reefs (blue, midnight, and rainbow) due to the combination of their large body size, high susceptibility to spear gear and fish trap, resultant relatively low resilience, and lack of abundance compared with most parrotfish occupying U.S. Caribbean waters. Amendment 5 would establish bag limits to restrict recreational harvest. This provision is

intended to slow progress in achieving the sector-specific ACL allocated for the recreational fishery and maximize the length of the recreational fishing season. Under the proposed action, Puerto Rico, St Croix, and St Thomas/St John daily recreational harvest would be limited to an aggregate of five fish per person (all snapper, grouper, and parrotfish combined) with no more than two parrotfish per person within the aggregate. NMFS also proposes a daily limit of 15 fish per vessel, including no more than six parrotfish per vessel.

Establish Framework Provisions Specifically for Reef Fish
NMFS is proposing framework measures for the CRFFMP. Under the proposed action, management measures that could be adjusted through framework amendments include quotas, closures, limits, gear rules, and reference point modifications. The purpose of the framework is to allow the CFMC to more expeditiously adjust these reference points and management measures in response to changing fishery conditions.

Table 2.2 Summary of Proposed Action Under Amendment 5

Stock complexes	Separate Grouper Unit 4 into Grouper Unit 4 and Grouper Unit 5
	Move creole-fish from Grouper Unit 3 into the "data collection only" unit
	Add cardinal snapper to Snapper Unit 2
	Move wenchman from Snapper Unit 2 into Snapper Unit 1
Management reference points	Redefine management reference points or proxies for the snapper, grouper, and/or parrotfish complexes based on the longest time series of pre-Comprehensive SFA Amendment catch data that is considered to be consistently reliable across all islands
ACL allocation/management	Define aggregate reference points for snapper and grouper in the USVI and define aggregate reference points for grouper but not snapper in PR
	Specify separate commercial and recreational annual catch limits in Puerto Rico based on the preferred management reference point time series
	Divide and manage annual catch limits by island group (i.e., Puerto Rico, St. Thomas and St. John) based on the preferred reference point time series using a mid-point or equidistant method for dividing the EEZ among islands
General management measure	Prohibit fishing for or possessing in the EEZ of midnight, blue, and rainbow parrotfishes
	Establish an aggregate bag limit for snapper, grouper, and parrotfish FMUs of: 5 per fisher including not more than two parrotfish per fisher or six parrotfish per boat, and 15 aggregate snapper, grouper, and parrotfish per boat on a fishing day
Triggering and applying accountability measures	Trigger AMs if the ACL is exceeded [as determined by a single year of landings effective beginning 2010, (2) a 2-year average of landings effective 2011, or (3) a 3-year running average of landings effective 2012 and thereafter (i.e., 2010, 2010-2011, 2010-2012, 2011-2013, etc.)] and NMFS' SEFSC (in consultation with CFMC and its Scientific and Statistical Committee) determines that the overage occurred because catches increased versus data collection/monitoring improved. If AMs are triggered, then reduce the length of the fishing season for that species or species group the year following the trigger determination by the amount needed to prevent such an overage from occurring again. The needed changes would remain in effect until modified.

Table 2.2 Summary of Proposed Action Under Amendment 5 (cont'd)

Establish framework measures	Amend the framework procedures for the Reef Fish FMP to provide a mechanism to expeditiously adjust the following reference points and management measures through framework action: a. Quota Requirements b. Seasonal Closures c. Area Closures d. Fishing Year e. Trip/Bag Limits f. Size Limits g. Gear Restrictions or Prohibitions h. Fishery Management Units (FMUs) i. Total Allowable Catch (TAC) j. Annual Catch Limits (ACLs) k. Accountability Measures (AMs) l. Annual Catch Targets (ACTs) m. Maximum Sustainable Yield (MSY) n. Optimum Yield (OY) o. Minimum Stock Size Threshold (MSST) p. Maximum Fishing Mortality Threshold (MFMT) q. Overfishing Limit (OFL) r. Acceptable Biological Catch (ABC) control rules s. Actions to Minimize the Interaction of Fishing Gear with Endangered Species or Marine Mammals

Amendment 6

This amendment to the FMPs for reef fish in the U.S. Caribbean is designed to bring those fisheries not classified as undergoing overfishing into compliance with the 2007 revisions to the Magnuson-Stevens Fishery Conservation and Management Act. These alternatives will consider measures to revise management reference points, implement annual catch limits (ACLs) and accountability measures (AMs) to prevent overfishing in both the commercial and recreational sectors, establish recreational fishing bag limits, establish exclusive economic zone sub-boundaries for purposes of applying AMs, adjust management measures as needed to constrain harvest to specified ACLs, and minimize to the extent practicable negative socioeconomic impacts.

Revise Management Reference Points and Establish ACLs and AMs and Island-Specific Allocation and Management

The proposed action would revise MSY, OY and overfishing thresholds and establish ACLs and AMs for the Caribbean reef fish FMP species not classified as undergoing overfishing (Table 2.3). The U.S. Caribbean constitutes a single EEZ, but effective island-specific management of reef fish resources requires that management boundaries be established between islands. This is necessary to enable application of AMs in response to harvesting activities on a single island (Puerto Rico, St. Croix) or island group (St. Thomas/St. John) without affecting activities on the other islands or island groups. The proposed action would establish geographic boundaries between islands/island groups based upon an equidistant approach that uses a mid-point to divide the EEZ among islands. Recreational landings data are available for Puerto Rico, but not the USVI. Thus, separate sector ACLs (commercial and recreational ACLs) would only be established for species in Puerto Rico. CFMC chose median landings as the best proxy for Maximum Sustainable Yield (MSY) for Puerto Rico and average landings as the best proxy for

MSY in the USVI. ACLs for each species were based on their respective MSYs. The year sequence used for each ACL represents the longest year sequence of reliable landings data for each island group. The proposed ACLs account for uncertainty in the scientific and management process. The proposed ACLs determined for the Caribbean reef fish species, except surgeonfish, include a 15% uncertainty reduction applied to them. For surgeonfish species, a 25% uncertainty reduction was applied to the ACL value. A higher percent reduction was applied to the surgeonfish ACL because of these species' role in grazing on macroalgae.

Table 2.3 Proposed ACLs by Management Area and Species or Species Groups (Values in Pounds)

ACL	Puerto Rico		St. Croix landings	St. Thomas/St. John landings
	Commercial landings	Recreational landings		
Angelfish	8,983	4,491	304	7,897
Boxfish	95,683	5,129	9,370	30,978
Goatfish	19,517	402	4,184	356
Grunts	202,662	5,587	40,979	41,797
Jacks	95,621	56,668	17,210	58,785
Porgies	27,488	2,863	5,153	24,243
Squirrelfish	18,514	4,323	134	4,712
Surgeonfish	7,179	3,589	33,603	29,249
Tilefish	10,846	5,423	0	0
Triggerfish	64,972	24,365	27,755	82,719
Wrasses	60,163	5,611	8	650

Accountability measures are designed to rectify the potential impacts that may result from harvest exceeding the ACL. Two components were proposed, the first identifying the conditions under which AMs are triggered and the second describing the action(s) that would occur if AMs are triggered. Under the proposed action, the AM triggers change over time. For example, AMs may be triggered in the year following the implementation of the ACL, based upon only the landings in 2011. AMs may be triggered in the second year following implementation of the ACL, based upon a 2-year average of landings from 2011 and 2012. After three years, determination of whether the ACL has been exceeded will be based on a 3-year average of landings from 2011, 2012, and 2013. Thereafter, the determination would be based upon the most recent 3-year average of landings. This approach was selected because commercial and recreational landings of a species, species group, or complex may vary substantially from year to year; applying a 3-year average dampens that variability and provides better information with regard to precision of the comparison. When AMs are triggered, NMFS would reduce the length of the fishing season for the affected species or species group by the amount needed to prevent such an overage from occurring again in the year following the determination that the ACL had been exceeded.

Table 2.4 Summary of Proposed Action under Amendment 6.

Management reference points	Redefine management reference points or proxies for the species within the Reef Fish FMP not undergoing overfishing based on the longest year sequence of reliable landings data.
Geographic Allocation and Management	Divide and manage annual catch limits by island group (i.e., Puerto Rico, St. Thomas and St. John) based on the preferred reference point time series using a mid-point or equidistant method for dividing the EEZ among islands
ACL allocation/management	Specify separate commercial and recreational annual catch limits in Puerto Rico based on the preferred management reference point time series
	Establish an aggregate bag limit of: 5 per fisher including not more than two parrotfish per fisher or six parrotfish per boat, and 15 aggregate snapper, grouper, and parrotfish per boat on a fishing day
Triggering and applying accountability measures	Trigger AMs if the ACL is exceeded [as determined by a single year of landings effective beginning 2011, (2) a 2-year average of landings effective 2012, or (3) a 3-year running average of landings effective 2013 and thereafter (i.e., 2011, 2011-2012, 2011-2013, 2011-2014, etc.)] and NMFS' SEFSC (in consultation with CFMC and its Scientific and Statistical Committee) determines that the overage occurred because catches increased versus data collection/monitoring improved. If AMs are triggered, then reduce the length of the fishing season for that species or species group the year following the trigger determination by the amount needed to prevent such an overage from occurring again. The needed changes would remain in effect until modified.

Creation of a New FMP Specific to Aquarium Trade Species

The proposed action would take the administrative action of moving species currently targeted for the aquarium trade, but managed under the CRFFMP, into their own FMP. The current "Aquarium Trade" category in the CRFFMP contains a total of 121 species or species groups: 58 species in the Reef Fish FMP and 63 in the Coral FMP. Of those 121 species, commercial landings data are available for 8 species or species group and recreational landings data are available for 22 species or species groups. All of those landings data come from Puerto Rico as there are no available landings data for aquarium trade species that are specific to the USVI. Commercial or recreational harvest of aquarium trade species in USVI is prohibited unless a harvest permit is obtained. The authorization of fishing conducted under this new FMP will be subject to its own consultation; therefore, we do not analyze its effects in the present consultation.

2.1 Overview of Management and Regulations

2.1.1. The Federal Fishery Management Process

The U.S. Caribbean reef fish fishery is managed by the CRFFMP, and implementing regulations at 50 CFR Part 622, under the MSA (16 U.S.C. 1801 et seq.), originally enacted in 1976 as the Fishery Conservation and Management Act. The MSA claims sovereign rights and exclusive fishery management authority over most fishery resources within the U.S. EEZ, an area extending 200 nautical miles from the seaward boundary of each of the coastal states, and authority over U.S. anadromous species and continental shelf resources that occur beyond the U.S. EEZ. In the U.S. Caribbean, federal waters subject to management under the CRFFMP extend to 200 nautical miles offshore from the nine-mile seaward boundary of the

Commonwealth of Puerto Rico and the three-mile seaward boundary of the territory of the USVI.

Responsibility for federal fishery management decision-making in the U.S. Caribbean is divided between the Secretary of Commerce (Secretary) and the CFMC. The CFMC is responsible for preparing, monitoring, and revising management plans for fisheries needing management within their jurisdiction. The Secretary is responsible for promulgating regulations to implement proposed plans and amendments after ensuring that management measures are consistent with the MSA, and with other applicable laws[1]. The Secretary has delegated this authority to NMFS.

The CFMC consists of seven voting members: four public members appointed by the Secretary, one each from the fishery agencies of Puerto Rico and the USVI, and one from NMFS. Public interests are also involved in the fishery management process through participation on advisory panels and through CFMC meetings, which, with few exceptions for discussing personnel matters, national security, or litigation, are open to the public.

If approved by NMFS, CFMC management actions are implemented in accordance with the Administrative Procedures Act, in the form of "notice and comment" rulemaking, which provides extensive opportunity for public scrutiny and comment, and requires consideration of and response to those comments.

2.1.2 History of the Fishery, the CRFFMP and Implementing Regulations

Prior to World War II, Puerto Rico and the USVI had a poorly organized fish trap fishery, which was mainly artisanal. Throughout most of the 20th century, the fishery was not heavily capitalized and was mainly limited to nearshore harvest. The influx of military personnel into Puerto Rico and St. Thomas during World War II resulted in a dramatic increase in demand for local fish, and the fishery expanded greatly in the 1950s and 1960s as tourism and the local population grew (CFMC 1985).

The CRFFMP (CFMC 1985; 50 FR 34850) was implemented in September 1985 to establish a management program for shallow-water reef fish resources within the area of authority of the CFMC and NMFS around Puerto Rico and the USVI. The CRFFMP defined the reef fish fishery management unit, described objectives for the shallow water reef fish fishery, and established management measures to achieve those objectives. Of some 350 species of shallow-water reef fish in the Caribbean, the CRFFMP defined the reef fish fishery management unit to include the most commonly landed shallow water species only (i.e., 64 species of groupers, grunts, goatfishes, leatherjackets, squirrelfishes, snappers, wrasses, parrotfishes, jacks, porgies, butterflyfishes, angelfishes, surgeonfishes, and boxfishes) which composed the bulk of the catch from Puerto Rico and the U.S.V.I. At that time, the assemblage of species was exploited by

[1] Administrative Procedures Act, Coastal Zone Management Act, Data Quality Act, Endangered Species Act, Executive Orders 12612 (Federalism) 12866 (Regulatory Planning and Review), 12630 (Takings), 12898 (Federal Actions to Address Environmental Justice in Minority Populations and Low Income Housing), 12962 (Recreational Fisheries), 13084 (Consultation and Coordination with Indian Tribes) 13089 (Coral Reef Protection), 13158 (Marine Protected Areas) 13186 (Responsibilities of Federal Agencies to Protect Migratory Birds); Marine Mammal Protection Act, Migratory Bird Treaty Act, National Environmental Act, National Marine Sanctuaries Act, Paperwork Reduction Act, Regulatory Flexibility Act

approximately 1,500 to 2,000 commercial fishermen using fish traps, hook-and-line, nets, seines, and spears; fish traps, followed by handline, were historically the dominant commercial gear type. There were also an estimated 12,000 recreational boats in the fishery, with the occupants of these boats mainly using hook-and-line or spears. Unpublished data for the years 1979 through 1982 showed a decline in landings as well as catch per trap. Total recreational and commercial shallow-water reef fish landings in 1982 were estimated at 7.5 million lbs with a commercial value of $8.7 million. Conflicts such as trap poaching were detected within the commercial sector (CRFFMP 1985).

The original objectives of the CRFFMP were to: (1) obtain the necessary data for management and monitoring, (2) reverse declining trends [i.e., decrease in landings and catch per unit of effort (CPUE)] in the resource, (3) reduce conflicts among harvesters of the resource, (4) promote compatible, if not uniform, management of the Pan-Caribbean species in the management unit, and (5) help solve a ciguatera problem. Primary management measures included:

- Definition of maximum sustainable yield as equal to 7.7 million lbs.
- Definition of OY as all of the fishes in the management unit that can be harvested by U.S. fishermen under the provisions of the FMP (also estimated at 7.7 million lbs).
- Specification of criteria for the construction of fish traps, including a minimum 1 1/4-inch mesh size requirement and a requirement that fish traps contain a self-destruct panel and/or self-destruct door fastening.
- Requirement to identify and mark gear and boats.
- Prohibition on the use of poisons, drugs, and other chemicals and explosives to take reef fish.
- Prohibition on the take of yellowtail snapper that measure less than 8 inches total length for the first fishing year, to be increased one inch per year until the minimum size limit reached 12 inches.
- Prohibition on the take of Nassau grouper that measure less than 12 inches total length for the first fishing year, to be increased one inch per year until the minimum size limit reached 24 inches.
- Prohibition on the take of Nassau grouper from January 1 to March 31 each year, a period that coincides with the spawning season of this species.

In December 1990, Amendment 1 to the CRFFMP (CFMC 1990; 55 FR 46214) was implemented. The action was taken because information indicated that more stringent management measures were needed to accomplish the objectives of the CRFFMP. Data provided by the local fishery agencies demonstrated that, in spite of the management measures implemented, there was still a declining trend in reef fish fisheries, indicated by a shift in species composition and a decrease in landings. For example, the parrotfish, which was had been considered second and third class in most sectors of the fishery, was being sold as first class and was one of the most frequently landed species, displacing the less abundant snappers and groupers. Primary management measures implemented to address the problem included:

- Increase in the minimum mesh size for traps to 2 inches.
- Prohibition on the take or possession of Nassau grouper; and

- Prohibition on fishing in an area southwest of St. Thomas, USVI, from December 1 through February 28 of each year, a period that coincides with the spawning season for red hind (this seasonal closure would later become a year-round closure with the implementation of the Hind Bank Marine Conservation District through Amendment 1 to the Coral FMP).

Amendment 1 also defined overfished and overfishing for shallow-water reef fish. "Overfished" was defined as a biomass level below 20% of the spawning stock biomass per recruit (SSBR) that would occur in the absence of fishing. For overfished stocks, "overfishing" was defined as a rate of harvest that is not consistent with a program that has been established to rebuild a stock or stock complex to the 20% SSBR level. For stocks not overfished, "overfishing" was defined as "a harvesting rate that if continued would lead to a state of the stock or stock complex that would not at least allow a harvest of OY on a continuing basis."

A regulatory amendment to the CRFFMP (CFMC 1991; 56 FR 48755) was implemented October 1991. The primary management measures contained in this amendment, included:

- Modification to the mesh size increase implemented through Amendment 1 to allow a mesh size of 1.5 inches for hexagonal mesh, and a change in the effective date of the 2-inch minimum mesh size requirement for square mesh to September 13, 1993.
- Change in the specifications for degradable panels for fish traps related to the required number of panels (required two panels per trap), and their size, location, construction, and method of attachment.

Amendment 2 to the CRFFMP (CFMC 1993) was implemented in November 1993. The actions in Amendment 2 addressed continuing and growing concerns by the CFMC over scarce resources, the need to protected important species when they aggregate for spawning, and the need to extend protection to other reef-associated species that were not previously in the management unit. The amendment redefined the reef fish fishery management unit to include the major species of deep-water reef fish and marine aquarium finfish. The deep-water reef fish fishery, ranging from the outer reaches of the shallow-water fishery (40 fathoms) to depths up to about 300 fathoms, generally were targeted with heavy duty traps and by electrically powered reels; bottom longlines are deployed to a limited extent. Reported landings of all demersal fishes in Puerto Rico declined from a peak of 2,402 metric tons (mt) in 1979 to 519 mt in 1990 and during that same time period deep-water snappers declined from 340 to 80 mt (Appledorn et al. 1992 in CRFFMP 1993). Primary management measures implemented through this amendment included:

- Prohibition on the use of any gear other than hand-held dip nets and slurp guns to collect marine aquarium fishes.
- Prohibition on the harvest or possession of Goliath grouper (formerly known as jewfish).
- Prohibition on the harvest, possession, and/or sale of certain species used in the aquarium trade, including seahorses and foureye, banded, and longsnout butterflyfish.
- Prohibition on fishing in an area off the west coast of Puerto Rico (Tourmaline Bank) from December 1 through February 28 each year, a period that coincides with the spawning season for red hind.

- Prohibition on fishing in an area off the east coast of St. Croix, USVI (Lang Bank), from December 1 through February 28 each year, a period that coincides with the spawning season for red hind.
- Prohibition on fishing in an area off the southwest coast of St. Croix, USVI, from March 1 through June 30 each year, a period that coincides with the spawning season for mutton snapper.

Existing definitions of MSY and OY were applied to all reef fish within the revised fishery management unit, with the exception of marine aquarium finfish. The MSY and OY of marine aquarium finfish remained undefined.

A technical amendment to the CRFFMP (59 FR 11560), implemented in April 1994, clarified the minimum mesh size allowed for fish traps.

An additional regulatory amendment to the CRFFMP (CFMC 1996; 61 FR 64485) was implemented in January 1997. That action reduced the size of the Tourmaline Bank closure that was originally implemented in 1993, and prohibited fishing in two areas off the west coast of Puerto Rico (Abrir La Sierra Bank [Buoy 6] and Bajo de Sico) from 1 December to 28 February of each year, a period that coincides with the spawning season of red hind.

Amendment 3 to the CRFFMP (CFMC 2005; 70 FR 62073) was implemented in 2005 with the approval of the Comprehensive SFA Amendment, in which the CFMC redefined the fishery management units and defined rebuilding plans for overfished species. Primary management measures implemented through this amendment are as follows:

- Established new Fishery Management Units (FMU) for reef fish.
- Required that fish traps have an 8-inch by 8-inch panel (with mesh not smaller than the mesh of the trap) on one side of the trap (excluding top, bottom and the side of the door) attached with untreated jute twine (diameter less than 1/8 inch).
- Required that individual traps or pots have at least one buoy attached that floats on the surface.
- Required that traps or pots tied together in a trap line have at least one buoy that floats at the surface at each end of the trap line.
- Prohibited the use of gillnets and trammel nets in the EEZ.
- Established a seasonal area closure in the area known as Grammanik Bank south of St. Thomas.
- Prohibited the use of bottom tending gear (trap), pots, gillnets, trammel nets, bottom longlines) in the seasonally closed areas including Grammanik Bank.
- Required an anchor retrieval system for anyone fishing or possessing Caribbean reef fish species.
- Prohibited the filleting of fish at sea.
- Established seasonal closures (no fishing or possession), every year during the specified months, for Snapper Unit 1 (silk, black, blackfin and vermillion snapper) from October 1 through December 31, Gulf Unit 4 (tiger, yellowfin, yellowedge, red and black) from February 1 through April 30, red hind from December 1 through the last day of February, and lane and mutton snapper from April 1 through June 30.

- Established Maximum Sustainable Yield (MSY), Optimum Yield (OY), Minimum Stock Size Threshold (MSST), and Maximum Fishing Mortality Threshold (MFMT) for the FMUs.

A notice of intent to prepare a draft environmental impact statement (DEIS) for Amendment 4 to the CRFFMP was published in the *Federal Register* on October 9, 2007 (72 FR 57307). The proposed alternatives would consider measures to implement escape vents in the trap fishery sector. However, Amendment 4 was postponed until a pilot study could be conducted on the effective size of escape vents (CFMC 2007).

In November 2010, a regulatory amendment to the CRFFMP (CFMC 2010; 75 FR 67247) was implemented to extend the seasonal closure of Bajo de Sico from a 3-month closure to a 6-month closure, and to prohibit fishing for and possession of Caribbean reef fish in or from the exclusive economic zone (EEZ) portion of Bajo de Sico during the closure. The regulatory amendment also prohibits anchoring in the EEZ portion of Bajo de Sico year-round. In addition to the measures contained in the regulatory amendment, the associated final rule (75 FR 67201) also added spear to the list of allowable gears in the commercial sector of the Caribbean reef fish fishery and revised the title of the FMP in the list of authorized fisheries and gear. The intended effect of the rule is to (1) provide further protection for red hind spawning aggregations and large snappers and groupers, and better protect the essential fish habitat (EFH) where these species reside, and (2) to correct an oversight with respect to the current list of allowable gears for the commercial reef fish fishery. Spear is and has been a historically used gear in the commercial reef fish fishery; CFMC intended for it to be previously included in the list of allowable gears. Table 2.1.2.1 includes a summary of federal regulations implemented to date in the Caribbean reef fish fishery.

Compatible reef fish regulations exist in the U.S. Caribbean for Nassau and goliath grouper; fishing and possession of these species has been prohibited from the shore to the EEZ since 2004 for goliath grouper and since 2006 for Nassau grouper.

Seasonal closures established in the EEZ since 2005 have been also established for some of the same species groups in the territorial and state waters. Fishing for and possession of Grouper Unit 4 (yellowfin, yellowedge, red, tiger) as well as black grouper is prohibited in the territorial waters of the USVI and in the EEZ from February 1st to April 30th each year, in Puerto Rico only one species from this group (yellowfin) is regulated during this period; Snapper Unit 3 from April 1st to June 30th in the EEZ and for two species within this group (lane and mutton) in the USVI, but only for one species within this group (mutton) from April 1st to May 31st in Puerto Rico; one species from Grouper Unit 3 (red hind) from December 1st to last day of February in the EEZ and Puerto Rico but not in the USVI; Snapper Unit 1 from October 1st to December 31st in the EEZ and USVI and only 2 species within this group (silk and blackfin) are regulated during these months in Puerto Rico.

Size regulations for yellowtail snapper have been implemented in the EEZ and Puerto Rico but not in the USVI.

Gear restrictions (e.g., mesh size in traps) also provide additional protection to the reef fish resources in the U.S. Caribbean. The mesh size for traps in the U.S. Caribbean is 2" (5.1 cm) rectangular and 1.5" (3.8 cm) hexagonal mesh; the same requirements apply for escape panels, and tying materials have been specified across the jurisdictions. Trammel and gillnets are prohibited in the EEZ and in the USVI; Puerto Rico has regulated the mesh size and length of the nets.

Table 2.1.2.1 Existing Federal Regulations Affecting Reef Fish Fishing

Multiple Caribbean Stock Complexes
Permanent Area Closures:
Fishing for any species and anchoring is prohibited year-round in the Hind Bank Marine Conservation District off St. Thomas.
Seasonal Area Closures:
From March 1 through June 30 each year, all fishing is prohibited in the Mutton Snapper Spawning Aggregation Area off St. Croix.
From December 1 through February 28 each year, fishing is prohibited in the four Red Hind Spawning Aggregation Areas (Lang Bank east of St. Croix and in Tourmaline Bank, and Abrir La Sierra Bank off western Puerto Rico)
From October 1 through March 31, no person may fish for or possess Caribbean reef fish species in or from those parts of the Bajo de Sico closed area that are in the EEZ (off western Puerto Rico).
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 off St. Thomas.
Gear Prohibitions and/or Restrictions:
Fishing with pots, traps, bottom longlines, gillnets, or trammel nets is prohibited year-round in the Red Hind Spawning Aggregation Areas, Grammanik Bank closed area, Mutton Snapper Spawning Aggregation Area and EEZ portion of Bajo de Sico. Anchoring is prohibited in Bajo de Sico and the Hind Bank Marine Conservation District year-round.
When fishing for Caribbean reef fish species in or from the Caribbean EEZ, the vessel must use only an anchor retrieval system that recovers the anchor by its crown, which prevents the anchor from dragging along the bottom.
An explosive may not be used to fish in the U.S. Caribbean EEZ.
A power assisted tool may not be used in the U.S. Caribbean EEZ to harvest Caribbean reef fish.
A powerhead may not be used in the U.S. Caribbean EEZ to harvest Caribbean reef fish.
A poison, drug, or other chemical may not be used to fish for Caribbean reef fish in the U.S. Caribbean EEZ.
A gillnet or trammel net may not be used in the U.S. Caribbean EEZ to fish for Caribbean reef fish.
A fish trap used or possessed in the U.S. Caribbean EEZ must have an escape mechanism as defined and comply with minimum mesh size regulations.
Snapper Unit 1 (silk, black, vermilion, blackfin)
Seasonal EEZ Closure:
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 U.S. Caribbean EEZ.
Snapper Unit 3 (lane and mutton snapper)
Seasonal EEZ Closure:
From April 1 through June 30 each year, no person may fish for or possess lane or mutton snapper in or from the Caribbean EEZ.
Sale and Purchase Restriction
A live mutton snapper in or from the Caribbean EEZ may not be sold or purchased and used in the marine aquarium trade.

Table 2.1.2.1 Existing Federal Regulations Affecting Reef Fish Fishing (cont'd)

Grouper Unit 1 and 2 (Nassau and goliath grouper)
Permanent EEZ Closure:
No person may fish for or possess Nassau or goliath grouper in or from the U.S. Caribbean EEZ. Such fish caught must be released immediately with a minimum of harm.
Grouper Unit 3 (red hind)
Sale and Purchase Restriction
A live red hind in or from the Caribbean EEZ may not be sold or purchased and used in the marine aquarium trade.
Grouper Unit 4 (red, misty, tiger, yellowedge, and yellowfin)
Seasonal EEZ Closure:
From February 1 through April 30 each year, no person may fish for or possess red, tiger, black, yellowfin, or yellowedge grouper in or from the Caribbean EEZ.

2.1.3 Fishery Data History, Monitoring and Reporting

Commercial and recreational fishery data available for conducting assessments in the U.S. Caribbean are limited. SEDAR (2009) notes among the primary concerns regarding the data are the scarce, missing, or unreliable information on fishing effort, spatial/geographic patterns, and life history parameters. Although some fishery independent data are available, they are spatially and temporally limited and previous assessments have been unable to incorporate a viable time series into the analyses (CFMC and NMFS 2011).

Commercial fisheries landings data have been collected since 1974 from St. Thomas/St. John, since 1975 from St. Croix, and since 1967 from Puerto Rico (although not available in electronic format for Puerto Rico until 1983). However, most of the USVI landings data have not been recorded to species with adequate reliability so species-specific landings information cannot be utilized to document historical trends. Beginning in 1998 (St. Croix) and 2000 (St. Thomas/St. John), finfish landings have been reliably reported to the species group or family level (e.g., snapper, grouper, parrotfish).

There are no federal licenses or permits issued for the commercial harvest of reef fish in the EEZ of the U.S. Caribbean. Instead, NMFS and the CFMC rely on Puerto Rico and USVI monitoring and reporting data. Both Puerto Rico and the USVI require commercial fishing permits and reporting. The Government of Puerto Rico requires commercial fishing licenses for fishing in commonwealth waters. In the USVI, all commercial fishermen are required to have a commercial permit, as well as any person who uses a pot, trap, set-net, or haul seine, even if for personal consumption. Anyone trading or selling any part of his catch, including charter operators, must also have a commercial permit. In the USVI, a moratorium on new commercial fishing licenses has been in place since 2001.

All anglers fishing recreationally in the U.S. Caribbean EEZ are now required to register through the national registry (https://www.countmyfish.noaa.gov/howtoregister/index.htm) if fishing for species other than highly migratory species (HMS) since there are already permits in place for HMS anglers. Fishing licenses and permits are a legal mandate for recreational harvesters in Puerto Rico although a licensing program has not yet been implemented. In the USVI there are

no licenses or permits required for recreationally fishing in territorial waters. The USVI is currently developing regulations for recreational fishing activity.

Management of Exempted Fishing, Scientific Research, and Exempted Educational Activity
Regulations at 50 CFR 600.745 allow the Regional Administrator to authorize the target or incidental harvest of species managed under an FMP or fishery regulations that would otherwise be prohibited for scientific research activity, limited testing, public display, data collection, exploratory, health and safety, environmental cleanup, hazardous waste removal purposes, or for educational activity. Every year, SERO may issue a small number of exempted fishing permits (EFPs), scientific research permits (SRPs), and/or exempted educational activity authorizations (EEAA) exempting the collection of a limited number of specimens from U.S. Caribbean federal waters from regulations implementing the FMPs. These EFPs, SRPs, and EEAAs typically involve fishing by commercial or research vessels, similar or identical to the fishing methods of the commercial and/or recreational fisheries, which are the primary subject of this opinion. In these cases, the types and rates of interactions with listed species from the EFP, SRP, and EEAA activities would be expected to be similar to those analyzed in this opinion. If the fishing type is similar and the associated fishing effort does not represent a significant increase over the effort levels for the overall fishery considered in this opinion, then issuance of some EFPs, SRPs, and EEAAs would be expected to fall within the level of effort and impacts considered in this opinion. For example, issuance of an EFP to an active commercial vessel likely does not add additional effects than would otherwise accrue from the vessel's normal commercial activities. Similarly, issuance of an EFP, SRP, or EEAA to a vessel to conduct a minimal number of fishing trips with a currently allowable gear likely would not add sufficient fishing effort to produce a detectable change in the overall amount of fishing effort in a given year. Therefore, we consider the issuance of most EFPs, SRPs, and EEAAs by SERO to be within the scope of this opinion. The included EFPs, SRPs, and EEAAs would be those involving fishing consistent with the description of fishing in Section 2.3 and not expected to increase fishing effort significantly.

2.2 Action Area

The action area for an opinion is defined as all of the areas affected directly or indirectly by the federal action and not merely the immediate area involved in the action. Therefore, to determine the action area for this opinion, we reviewed the area where fishing is authorized, the area where actual fishing is likely to occur, and the surrounding areas for potential direct or indirect effects from the proposed action.

The U.S. Caribbean is located in the Caribbean archipelago, about 1,100 miles east-southeast of Miami, Florida. It consists of the Commonwealth of Puerto Rico in the Greater Antilles and the Territory of the USVI in the Lesser Antilles island chain, both of which separate the Caribbean Sea from the western central Atlantic Ocean. The rectangular-shaped island of Puerto Rico is the smallest and the most eastern island of the Greater Antilles, and is located between the North Atlantic Ocean and the Caribbean Sea. The Commonwealth includes the islands of Mona, Monito, and various other isolated islands. The Mona Passage, which separates the island from Hispaniola to the west, is about 75 miles (120 km) wide and more than 3,300 ft (1,000 m) deep. Off the northern coast is the 28,000 ft (8,500 m) deep Puerto Rico Trench, and to the south the sea bottom descends to the 16,400 ft (5,000 m) deep Venezuelan Basin of the Caribbean. The

USVI are part of the Virgin Islands chain, which lies about 50 mi east of Puerto Rico and consists of about 80 islands and cays, and includes St. Croix, St. Thomas, and St. John. St. Croix is located about 40 nmi (74 km) south of St. Thomas and St. John and is entirely surrounded by the Caribbean Sea. The islands of St. Thomas and St. John are bordered by the Atlantic Ocean to the north and the Caribbean Sea to the south.

NMFS authorizes reef fish fishing under the Caribbean Reef Fish FMP in waters from nine miles seaward of Puerto Rico, and three miles seaward of the USVI, up to 200 miles from shore (i.e., the U.S. Caribbean EEZ). Fishing activity within the authorized area is determined by a variety of biological (e.g., distribution of reef fish), socio-economic (e.g., market factors, location of ports, operating costs), and regulatory factors (e.g., gear-restricted areas and closed areas).

Due to the steep continental slopes that occur off Puerto Rico and the USVI, fishable habitat off these islands is defined in the CRFFMP as those waters 100 fathoms or shallower. The majority of fish habitat occurs in that area, as does the majority of fishing activity for reef fish species. Beyond 100 fathoms, the sea bed drops off dramatically and is difficult to fish, as it requires larger vessels and more gear (e.g., more line for fish traps, handlines, etc.), both of which are not typical of non-highly migratory species (HMS) U.S. Caribbean fisheries.

The total area of fishable habitat in the U.S. Caribbean is about 2,467 nm^2 (see Figure 2.2.1 on p. 21). Only 355 nm^2 (14.4%) of that area occurs in federal waters where NMFS authorizes fishing: 116 nm^2 (4.7%) off Puerto Rico; 240 nm^2 (9.7%), off the USVI. The vast majority of the fishable habitat in federal waters off Puerto Rico is located off the west coast. The vast majority of the fishable habitat in federal waters off the USVI is located off the north coast of St. Thomas.

Reef fish management actions and regulations in the EEZ affect populations of fishes and invertebrates in commonwealth and territorial waters because these populations are continuous, fluid, and move across imposed jurisdictional boundaries (e.g., the EEZ boundary). Currents transport eggs and larvae, sometimes for long distances, generating interconnections among areas (Boehlert 1996). Strong connectivity among areas implies that local populations may depend on processes occurring elsewhere. Consequently, local management initiatives may be ineffective in providing local benefits and thus an increase in the scale of management may be necessary (Roberts 1997).

Management parameters including targets and limits such as ACLs proposed in Amendments 5 and 6 are established for commonwealth, territory and federal U.S. Caribbean waters because data do not exist to identify and distinguish between harvest that occurs in commonwealth and territorial waters and harvest that occurs in federal waters. The FMP establishes uniform targets and limits for all waters in which the species occur and seeks commonwealth and territorial consistency.

Throughout its range of operation in the EEZ, the federal reef fish fishery may affect one or more listed species directly via their potential exposure to fishing vessels and fishing gear leading to injury or mortality, or indirectly via impacts on the benthic environment associated with the harvest of key herbivorous fishes. By harvesting these species, their ability to maintain robust

populations and to graze algae may be reduced, potentially affecting the resilience of reef species, including threatened *Acropora* spp. In the U.S. Caribbean, because of the connectivity of reef fish stocks and their management between commonwealth, territorial, and EEZ waters, the proposed action may also indirectly affect listed *Acropora* spp. and their critical habitat in commonwealth and territorial waters. Based on this information, the action area for this consultation includes all U.S. Caribbean commonwealth, territorial, and EEZ waters less than 100 fathoms.

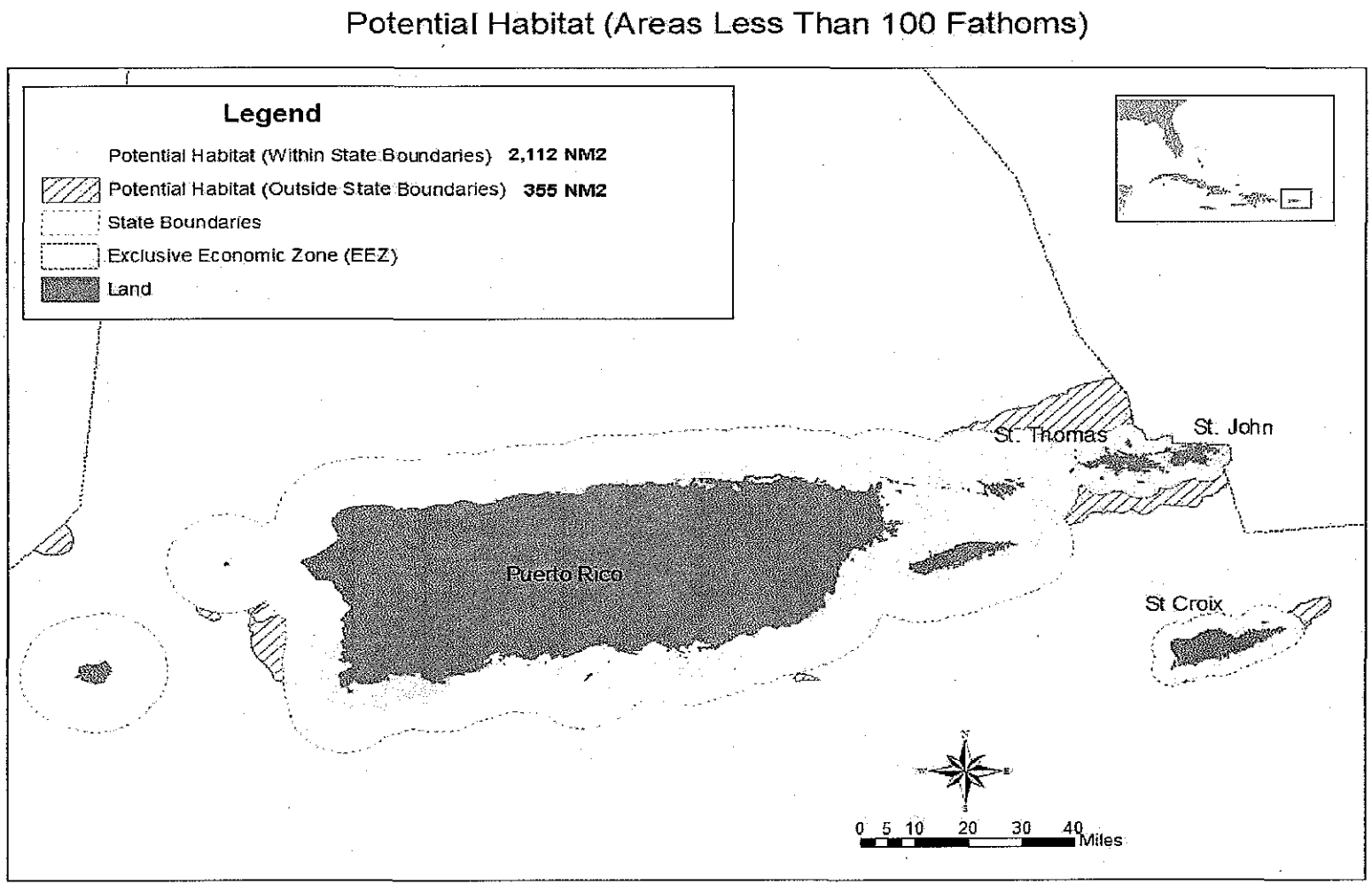


Figure 2.2.1. Potential Habitat Areas Less Than 100 Fathoms

2.3 The Caribbean Reef Fish FMU and Status of its Stocks

At present, over 137 reef fish species comprise the FMU (four groups are listed at the genus level and there are several species within each of these groups, so the exact number of species comprising the FMU is not fully defined.). Of the 137+ species, approximately 58 are associated with the aquarium trade (this includes the four groups listed at the genus level), leaving 80 reef fish species subject to CFMC management (see Table 2.3.1).

Currently, there are four species or species groups in the FMU that are classified as undergoing overfishing in the U.S. Caribbean. These groups are: parrotfish, Grouper Unit (GU) 1 (Nassau grouper), GU4 (tiger, yellowfin, red, misty, and yellowedge grouper), and Snapper Unit (SU) 1 (black, blackfin, silk, and vermilion snapper). GU1, GU2 (goliath grouper), and GU4 are classified as overfished; SU1 and parrotfish are classified as approaching an overfished status. The status of Caribbean reef fish species/species groups has not been assessed since the CFMC and NMFS took action to address overfishing through the 2005 Caribbean SFA Amendment (CFMC 2005).

Table 2.3.1 CRF FMP FMU, Excluding Aquarium Trade Species

Reef Fish Complex	Common Name	Genus/Species
Grouper Unit 1	Nassau	*Epinephelus striatus*
Grouper Unit 2	Goliath	*Epinephelus itajara*
Grouper Unit 3	Red hind Coney Rock hind Graysby, Creole-fish	*Epinephelus guttatus* *Cephalopholis fulva* *Epinephelus adscensionis* *Cephalopholis cruentatus*
Grouper Unit 4	Yellowfin Red Tiger Yellowedge Misty	*Mycteroperca venenosa* *Epinephelus morio* *Mycteroperca tigris* *Mycteroperca bonaci*
Snapper Unit 1	Silk Black Blackfin Vermilion	*Lutjanus vivanus* *Apsilus dentatus* *Lutjanus buccanella* *Rhomboplites aurorubens* *Pristipomoides aquilonaris*
Snapper Unit 2	Queen Wenchman	*Etelis oculatus* *Pristipomoides macrophthalmus*
Snapper Unit 3	Gray Lane Mutton Dog Schoolmaster Mahogany	*Lutjanus griseus* *Lutjanus synagris* *Lutjanus analis* *Lutjanus jocu* *Lutjanus apodus* *Lutjanus mahogoni*
Snapper Unit 4	Yellowtail	*Ocyurus chrysurus*

Reef Fish Complex	Common Name	Genus/Species
Parrotfish	Blue	*Scarus coeruleus*
	Midnight	*Scarus coelestinus*
	Rainbow	*Scarus guacamaia*
	Queen	*Scarus vetula*
	Princess	*Scarus taeniopterus*
	Striped	*Scarus iseri*
	Redband	*Sparisoma aurofrenatum*
	Redfin	*Sparisoma rubripinne*
	Redtail	*Sparisoma chrysopterum*
	Stoplight	*Sparisoma viride*
Grunts	White Grunt	*Haemulon plumieri*
	Margate	*Haemulon album*
	Tomtate	*Haemulon aurolineatum*
	Bluestriped Grunt	*Haemulon sciurus*
	French Grunt	*Haemulon flavolineatum*
	Porkfish	*Anisotremus virginicus*
Goatfishes	Spotted Goatfish	*Psuedupeneus maculatus*
	Yellow Goatfish	*Mulloidichthys martinicus*
Porgies	Jolthead Porgy	*Calamus bajonado*
	Sea Bream	*Archosargus rhomboidalis*
	Sheepshead Porgy	*Calamus penna*
	Pluma	*Calamus pennatula*
Squirrelfishes	Blackbar Soldierfish	*Myripristis jacobus*
	Bigeye	*Priacanthus arenatus*
	Longspined Squirrelfish	*Holocentrus rufus*
	Squirrelfish	*Holocentrus adscensionis*
Tilefish	Blackline Tilefish	*Caulolatilus cyanops*
	Sand Tilefish	*Malacanthus plumieri*
Jacks	Blue Runner	*Caranx crysos*
	Horse-eye Jack	*Caranx latus*
	Black Jack	*Caranx lugubris*
	Almaco Jack	*Seriola rivoliana*
	Bar Jack	*Caranx ruber*
	Greater Amberjack	*Seriola dumerili*
	Yellow Jack	*Caranx bartholomaei*
Surgeonfish	Blue Tang	*Acanthurus coeruleus*
	Ocean Surgeonfish	*Acanthurus bahianus*
	Doctorfish	*Acanthurus chirurgus*
Triggerfish and Filefish	Ocean Triggerfish	*Canthidermis sufflamen*
	Queen Triggerfish	*Balistes vetula*
	Sargassum Triggerfish	*Xanthichthys ringens*
	Scrawled Filefish	*Aluterus scriptus*
	Whitespotted Filefish	*Cantherhines macrocerus*
	Black Durgon	*Melichthys niger*
Boxfish	Honeycomb Cowfish	*Acanthostracion polygonia*
	Scrawled Cowfish	*Acanthostracion quadricornis*
	Trunkfish	*Lactophrys trigonus*
	Spotted Trunkfish	*Lactophrys bicaudalis*
	Smooth Trunkfish	*Lactophrys triqueter*
Wrasses	Hogfish	*Lachnolaimus maximus*
	Puddingwife	*Halichoeres radiatus*
	Spanish Hogfish	*Bodianus rufus*

Reef Fish Complex	Common Name	Genus/Species
Angelfish	Queen Angelfish	*Holocanthus ciliaris*
	Gray Angelfish	*Pomocanthus arcuatus*
	French Angelfish	*Pomocanthus paru*

2.4 Description of the Fishery

It is difficult to describe the specific characteristics of the federal reef fish fishery and its components in the U.S. Caribbean because available data are not specific to EEZ waters. Commercial fishermen in Puerto Rico and USVI use multiple number and types of gears – fish traps, hook-and-line, nets, and SCUBA among others – and are non-specialized harvesters; thus, fishery characteristics by target species are also largely lacking. The best available commercial fishing information stems largely from censuses conducted of all active commercial fishermen in the USVI and Puerto Rico (e.g., Kojis 2004, Matos-Caraballo and Agar 2011). Data on the recreational component of the reef fishery is even scarcer, with MRFSS only collecting data from Puerto Rico, and USVI recreational information mainly related to billfish and other pelagic species not managed by the CFMC.

The following description of the fishery provides a general characterization of Puerto Rico and USVI fisheries of which the federal reef fish fishery subject to this consultation is a component. The commercial fishing activity description is based mainly on the latest censuses (Matos-Caraballo and Agar (2011), and Kojis (2004). The recreational activity description is mainly excerpted from Amendment 5 (CFMC and NMFS 2011). Wherever available, information specific to the reef fish fishery is described. NMFS (2005a) provided a description of the reef fish fishery based on the best available information at that time. Additional information is also provided in Amendment 5, which provides both historic and updated information.

2.4.1 Puerto Rico Commercial Fishing Activity

Between January and October 2008, Puerto Rico's Department of Natural and Environmental Resources (DNER) Commercial Fisheries Statistics Program (CFSP) conducted a census of Puerto Rico fishermen. In all, CFSP personnel conducted 868 in-person, voluntary interviews with commercially active fishermen around 92 fishing centers in 39 coastal communities. Only about 5% of the population of purported active commercial fishermen either declined to participate in the census or could not be reached by CFSP staff. The Commonwealth was partitioned into four coastal regions: north, east, south and west. The northern region extends from the municipalities of Isabella to Luquillo. The eastern region runs from the municipalities of Fajardo to Maunabo, including the islands of Vieques and Culebra, and the southern region stretches from the municipalities of Patillas to Lajas. The western region spans the municipalities of Cabo Rojo to Aguadilla. The following section provides excerpted summary information from that census.

Fishing participants
The 2008 census indicates there are 868 active commercial fishermen in Puerto Rico (Matos-Caraballo and Agar 2011). Earlier censuses of active fishermen, which used the same methods as the 2008 study, show that participation rates have decreased noticeably since the late 1980s (Matos-Caraballo and Torres-Rosado 1989, Matos-Caraballo 1998, Matos-Caraballo et al. 2005).

Of the 868 fishermen interviewed, 557 self-reported to have valid licenses, with 394 full-time licenses, 46 part-time licenses, and 117 apprentice (or beginner) licenses. The majority of respondents were full-time fishermen, with the greatest number of full-time fishermen from the west coast, where local fishing grounds are most productive. Puerto Rican fishermen devoted, on average, 30 hours per week to catching fish.

Fleet Characteristics

The active commercial fleet consisted of 670 vessels. Almost all of the captains owned a single (primary) fishing boat whereas only 15% of the crew owned a single fishing boat. Most of the fishing vessels were small with moderate levels of mechanization. The average length of the primary fishing vessel was 20 feet. About 97% of the primary vessels ranged between 10 and 30 feet in length. Most of the vessels were small (20 ft) and had a single outboard engine. The average propulsion rate of the primary vessel was 80 horsepower (hp), up from 66 hp in 2002 (Matos-Caraballo et al., 2005). Propulsion rates ranged from 65 hp in the south region to 103 hp. Most hulls were built of fiberglass (65%) and, to a lesser extent, of fiberglass and wood (22%). Only 12% of the hulls of the primary vessels were made of wood. Thirty percent of the primary vessels were built after 2000. Most fishing vessels were manned by a captain and helper. Diving operations tended to have a captain and two helpers because the crew need to tend the boat and dive.

Slightly less than one-third of the vessels had depth finders (32%), radios (30%), and fish finders (29%). Less than 5% of the fleet had emergency position indicating radio beacons (EPIRBs). Over 32% of the vessels had electric reels, and 5% had hydraulic reels. The north region had the highest percentage (66%) of vessels with electric reels. Approximately one in five vessels had winches. Regionally, winches were relatively more prevalent in the east (33%) and less common in the south (15%).

Fishing Ground and Target Species

Puerto Rican fishermen have and continue to favor the continental shelf and shelf break as their prime fishing grounds. About 82% of respondents reported fishing on the continental shelf which is about the same as the 83% who reported fishing on the continental shelf in 2002 (Matos-Caraballo et al. 2005). In contrast, the proportion of fishermen who reported fishing in deep-waters dropped from 46% in 1996 and 48% in 2002 to 35% in 2008, which Matos-Caraballo and Agar (2011) attribute to higher fuel costs and tighter fishing regulations such as minimum size limits and closed seasons.

The west coast has consistently been the most productive area in terms of fishing over time (Collazo and Calderón 1987-88, Matos-Caraballo, 2007). Along the west coast is the Mona Passage, which is one of the primary fishing grounds for west coast commercial fishers because it is there where snapper, sea basses, grouper, trunkfish, and pelagic species such as king mackerel and jacks are caught; HMS-species not managed by the CFMC are also caught there. The south coast consistently has accounted for the second most landings.

Across the four coasts, most fishers reported that they target reef fish (Table 2.5.1.1). On the north, east, and west coasts, the second most targeted species are deep-water snapper. Deep-water snapper species include silk snapper (Snapper Unit 1), reef fishes include those species in

Grouper Units 1 (Nassau grouper) and 4 (red, misty, tiger, yellowedge, and yellowfin), and parrotfish.

Table 2.4.1.1 Species Targeted by Interviewed Commercial Fishers, 2008
(Source: Matos-Caraballo and Agar (2011)).

Targeted Species	Percent of Interviewed Fishers				
	North Coast	South Coast	East Coast	West Coast	All Coasts
Reef Fishes	88%	88%	76%	65%	77%
Ornamental	1%	1%	2%	2%	2%
Deep-Water Snapper*	72%	40%	72%	51%	56%
Pelagic Fishes	65%	30%	67%	27%	42%
Lobster	28%	57%	65%	47%	49%
Conch	13%	45%	35%	35%	33%
Octopus	2%	19%	0%	1%	6%
Bait	53%	31%	33%	18%	31%
Land Crab	9%	6%	10%	2%	6%
Sirajo Gobies	8%	1%	0%	0%	2%

*At least one of the percentages for the 4 coasts in this row may be under-reported.

Fishing Gear Use

Hook-and-line gear was the most common and productive gear, accounting for about 49% of the total landings in 2008. Matos-Caraballo and Agar (2011) note fishermen, particularly part-time fishermen, favored this gear because of its efficiency and cost effectiveness for catching reef fish and pelagic species. Handlines comprised about 56% of the gear in this group, followed by bottom lines (16%), troll lines (14%), rods and reels (12%), and longlines (2%). Handlines and troll lines were reportedly used to catch dolphin, skipjack, little and blackfin tunas, and king mackerel; thus, they do not appear to be the dominant gear type used to target reef fish species in Puerto Rico. However, anchored bottom lines reportedly were used to target solely reef fish species (i.e., queen, silk, and lane snappers) and drifting bottom lines were used to target reef fish (i.e., yellowtail and mutton snappers) and other species (dolphin, king and cero (*S. regalis*) mackerels; little, skipjack and blackfin tunas; and sharks). Rods and reels were used to land dolphinfish, yellowtail snapper, king and cero mackerels, and little tunny. Longlines were used to catch sharks and lane, silk, and mutton snappers. Overall, the total amount of hook-and-line gear decreased from 12,314 units in 2002 (Matos-Caraballo et al. 2005) to 10,244 units in 2008.

SCUBA and skin diving were the second most productive gears, together they were responsible for approximately 29% of the total landings in 2008. Divers made up about 44% of the population of active fishermen, down from 53% in 2002 (Matos-Caraballo et al. 2005). SCUBA gear was used primarily to harvest species not managed in the CRFFMP (i.e., queen conch and spiny lobster) but to a lesser extent was used to target reef fish species including hogfish, parrotfish, boxfish (*Ostraciidae spp.*), and queen triggerfish (*B. vetula*). Skin diving mainly caught only queen conch and spiny lobster.

Traps or pots were the third most productive gears and accounted for almost 13% of the total commercial landings in 2008. Fish traps accounted for 48% of the trap units, followed by lobster traps (40%) and deep-water snapper traps (12%). Fish pots landed spiny lobsters, grunts, boxfishes, queen triggerfishes, and parrotfishes, whereas lobster traps landed mainly spiny

lobsters. Deep-water snapper traps caught silk, queen, vermilion, and blackfin snappers. The total number of traps dropped from 13,146 units in 2002 (Matos-Caraballo et al., 2005) to 9,597 units in 2008.

Nets were the fourth most productive gears. Over 9% of the island's total yield was derived from this gear in 2008. Nets totaled 1,712 units in 2008, down from 2,798 units in 2002 (Matos-Caraballo et al. 2005). Since November 2005, federal regulations have prohibited the use of gill and trammel nets in the U.S. Caribbean EEZ to harvest Caribbean reef fish, which may partially explain the decreased use of net gear, although depth of water in the EEZ does not favor the use of these nets (i.e., nets are more typically used in commonwealth and territorial waters). Among the nets, gillnets accounted for 33% of the units, trammel nets for 19%, and bait cast nets for 47%. Gillnets were used to catch bar jacks, porgies, ballyhoos, grunts, parrotfishes, and various snappers, whereas trammel nets catch spiny lobsters, grunts, parrotfishes, and boxfishes. Puerto Rico regulates the mesh size and length of the nets that can be used.

2.4.2 USVI Commercial Fishing Activity

Kojis (2004) conducted a census of the commercial fishers of the USVI. From July 2003 to January 2004, a total of 323 commercial fishermen were interviewed; 116 in St. Thomas/St. John and 217 in St. Croix. Over 70% of licensed commercial fishers in St. Thomas/St. John and all the licensed commercial fishers in St. Croix were interviewed. The following section provides a summary of that census.

Fishing Participants
Two-thirds of commercial fishers in the USVI considered themselves full-time commercial fishers based on how much time they spent fishing and carrying out fishing related activities each week (defined as >36 hrs per week). A higher percentage of commercial fishers in St. Croix (39%) considered themselves part-time or opportunistic fishers than in St. Thomas/St. John (21.8%) (Kojis 2004).

Fleet Characteristics
Most commercial fishers owned a boat with a single outboard motor with an average boat length of 21 feet. Most fishers owned boats ranging in length from 16 to 25 feet. Boats are constructed primarily of fiberglass and wood. Over half of fishers in the USVI carried a cell phone when they fished and a quarter owned a GPS. Echo sounders were installed on over a third of the fishing boats in the USVI. Approximately, 25% of vessels in St. Thomas/St. John were equipped with winches, 10% of vessels were equipped with electric reels in St. Croix, and 5% used electric reels in St. Thomas/St. John (Kojis 2004).

Target Species
The most commonly targeted categories of fish were reef fish and coastal pelagics. Approximately 82% of fishers in St. Thomas/St. John and St. Croix target reef fish. In St. Croix, deepwater snappers were the second most commonly targeted group of species with approximately 42% of fishers targeting them, while in St. Thomas/St. John only 4.5% of fishers targeted these species (Kojis 2004). Coastal pelagics were also more commonly targeted in St. Croix (33.0%) than in St. Thomas/St. John (9.8%) (Kojis 2004).

Fishing Gear Use
Most fishing in the USVI occurred during the day, with approximately 80% of fishing trips taking less than nine hours. Fishers in St. Croix fished an average of 3.3 trips per week compared with an average of only 2.6 trips per week for St. Thomas/St. John fishers, for an average of 3.1 times a week. Trip duration was on average 1.5 hours longer in St. Thomas/St. John than in St. Croix while the number of trips per week was higher in St. Croix than in St. Thomas/St. John (Kojis 2004). In essence, St. Thomas/St. John fishers took fewer but longer trips than St. Croix fishers.

Fishers in the USVI used a variety of fishing gear that included traps, lines, nets, and SCUBA. There were distinct differences in the gears used in each district. About 50 St. Thomas/St. John commercial fishers used more than 7,500 fish traps, modified lobster traps, and plastic lobster traps to target fish and lobster. In St. Croix, traps were not as commonly used. Instead fishers diversified into other gears such as multi-hook vertical setlines, gill and trammel nets, and SCUBA. These gears were used by fishers in St. Thomas/St. John as well, but not as commonly. Most fishers in the USVI used handlines. Rod-and-reel gear was less commonly used (Kojis 2004).

Modified fish pots (traps) and plastic lobster pots were only reported in St. Thomas/St. John. Similarly, umbrella nets and deepwater shrimp pots were only reported in St. Croix. However, even there, only few fishers used these gears (Kojis 2004). Some gears were used primarily in one district. For example, gill nets, trammel nets, and SCUBA diving were predominately used only in St. Croix. Conversely, haul seines, surface longlines, and bottom longlines were not commonly used in either district (Kojis 2004).

Pots (traps) have been an important fishing gear since 1930 and in 2003 they were still an important component of the fishery, especially in St. Thomas/St. John. However, other gears such as nets and vertical setlines were more commonly reported in 2003 than in 1930 or 1968 (Kojis 2004).

Presently, the use of all gill and trammel nets (single or multiple wall entanglement) is prohibited, with the exception of single-wall surface gillnets for the baitfish ballyhoo, gar, and flying fish (V.I.C., Title 12, Chapter 9A, §321-1). Surface gillnets must be tended at all times, may not be more than 1,800 feet long as measured by the float line, and may not be used within 20 feet from the bottom.

2.4.3 Puerto Rico Recreational Fishing Activity

The National Angler Registry, which began in 2010 as part of the MRIP program, has in its database, for 2010, 1419 anglers registered as fishing in the EEZ of which there were 1382 registered in Puerto Rico (F. Darby, NMFS/OSF, pers. comm.). Although recreational and sportfishing activities in Puerto Rico are prominent, 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 from the recreational fishing sector dates to 2000, when the MRFSS was implemented in Puerto Rico, and which has

continued to collect data to date. However, the survey does not target SCUBA divers, a potential major activity in the U.S. Caribbean (Garcia-Moliner et al. 2001).

Expanded estimates of the recreational catch (in numbers) and effort (number of trips and participants) are always accompanied by a calculation of the proportional standard error (PSE). In 2008 the total number of participants was estimated at 149,544 (with 127,863 resident participants and 21,681 out-of-state participants) with a PSE of 11%. These 149,544 participants in the recreational fishery made a total of 798,551 trips (all included: shore, private and charter) with a PSE of 9% for all modes combined. Landings for 2008 were estimated at 1,910,542 pounds for all finfish species (CFMC and NMFS 2011).

A summary of all available information for Puerto Rico from the recreational sector, including number of participants, number of trips taken by mode (shore, charter and private boat), and the total catch (all species reported) from 2000 to 2008 is presented in Table 2.4.3.1. A relatively flat trend in number of fishing trips and pounds landed is present from 2000-2008, except for an as-yet unexplained anomaly in 2006. The percent of trips taken to the shore (53-61%) is always higher than the percent of trips taken in private boats (36-45%), which in turn is always higher than the number of charter trips (1-3%). However, the private boats account for a greater proportion of the landings (45-94% of the total) followed by shore landings (4-53%) and finally (as expected from much catch and release in the area) by the charters (1-2%). Between the years 2000 and 2008, the total landings from the recreational sector ranged from 955,123 to 4,601,741 pounds (an average of 2,607,640 pounds per year from all finfish species). The number of participants has also varied annually from a low of 141,743 in 2005 to a maximum of 249,868 in 2000 (CFMC and NMFS 2011).

Table 2.4.3.1 Recreational Landings Statistics Generated from MRFSS Intercept Program in Puerto Rico from 2000-2008 (Source: CFMC and NMFS 2011)

Year	Mode	Total	Charter	Private	Shore	Participants
2000	Pounds	4,601,741	48,173	4,195,832	357,736	249,868
	Trips	1,332,703	16,899	522,914	792,890	
2001	Pounds	3,301,922	23,281	2,752,165	526,476	222,128
	Trips	1,411,943	10,919	504,349	896,675	
2002	Pounds	2,452,048	22,438	2,236,507	193,103	237,995
	Trips	1,301,059	34,277	572,844	693,938	
2003	Pounds	3,754,963	28,254	3,320,974	405,735	219,910
	Trips	1,111,405	21,764	471,741	617,900	
2004	Pounds	2,145,475	40,435	1,940,892	164,148	163,833
	Trips	1,050,299	22,028	389,469	638,802	
2005	Pounds	1,971,263	41,689	1,835,863	93,711	141,743
	Trips	866,722	17,969	379,910	468,843	
2006	Pounds	955,123	16,823	431,274	507,026	213,005
	Trips	896,582	16,906	386,111	493,565	
2007	Pounds	2,375,687	43,063	2,197,800	134,824	185,429
	Trips	1,080,096	10,734	453,907	615,455	
2008	Pounds	1,910,542	39,974	1,793,360	77,208	149,544
	Trips	798,552	12,623	362,739	423,190	

The MRFSS program also offers information on the total number of trips by mode and area (≤10 miles being roughly equivalent to state waters and ≥10 miles being roughly equivalent to EEZ waters) from 2000 to 2008 (Figure 2.5.3.1). Twenty percent of the trips taken to EEZ waters were by recreational fishers in private boats, but most recreational trips occur within state waters. The narrowest PSEs are from the private and shore fishing sectors, ranging from 10 to 16%, while for the charter mode PSEs range from 40 to 91% (CFMC and NMFS 2011). The MRFSS sampling was based mostly on the shoreline mode, with limited sampling of private vessels. A specific reporting protocol is being developed for the for-hire sector (G. Rodríguez, PRDNER, pers. comm. in CFMC and NMFS 2011).

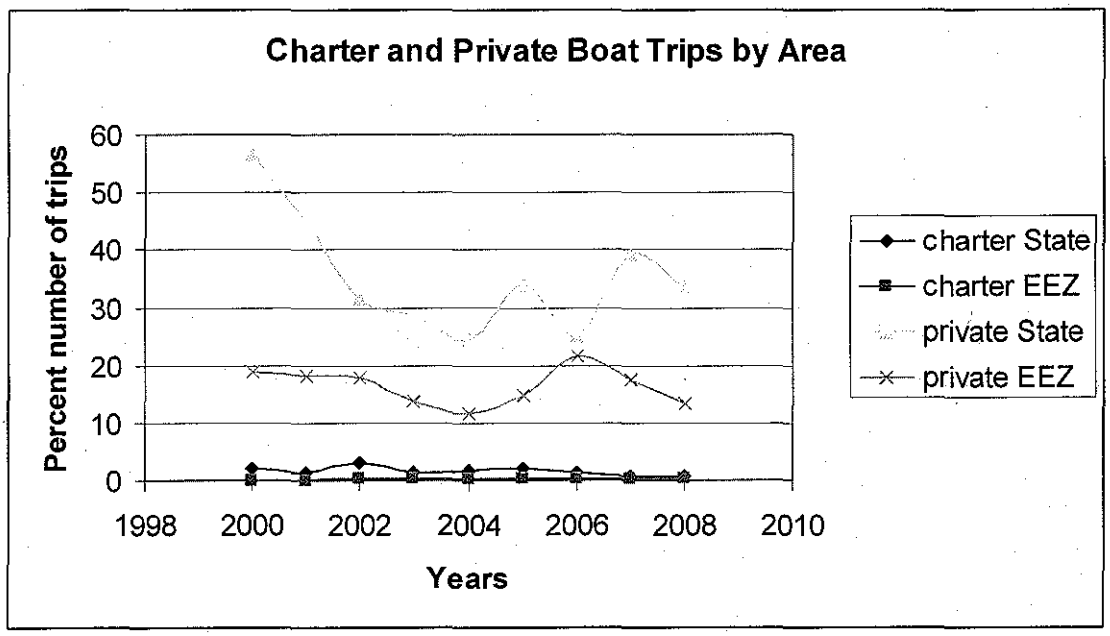


Figure 2.4.3.1. Puerto Rico Charter and Private Vessel Trips in Commonwealth (State) Waters and the EEZ, 2000-2008. (Source: CFMC and NMFS 2011)

The number of vessels registered in Puerto Rico with the United States Coast Guard peaked at 60,640 (Figure 2.5.3.2) including 1,125 boats registered as commercial fishing vessels (A. Cruz, PRDNER, pers. comm. in CFMC and NMFS 2011). This boat registry can be used as an indicator of the potential number of recreational fishers in Puerto Rico. Furthermore, "saltwater recreational fishing in Puerto Rico is an important industry generating $754.8 million in trip and durable goods expenditures" (B. Gentner and J. Agar, SEFSC, pers. comm. in CFMC and NMFS 2011). Regardless of its limitations, MRFSS provides useful information on the potential impact of recreational harvest on the finfish species considered in Amendments 5 and 6 (Table 2.5.3.1).

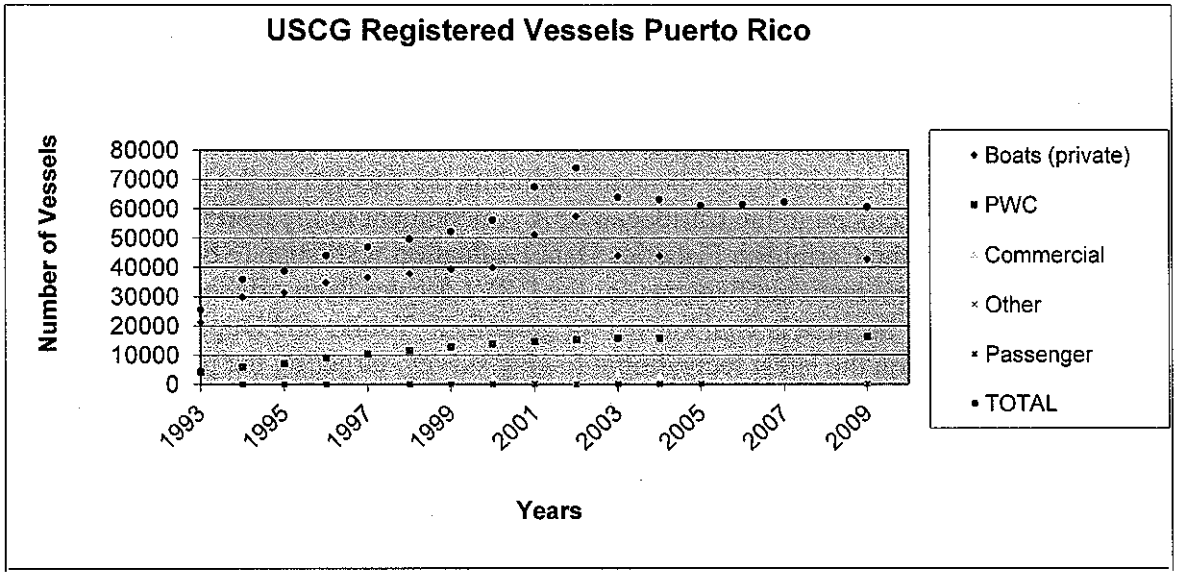


Figure 2.4.3.2 Puerto Rico Recreational and Commercial Vessel Registrations PWC = personal watercraft (Source: CFMC and NMFS 2011)

Species specific data from the recreational harvest is limited but Figure 2.5.3.3 shows the total number of fish estimated per year since 2000 for Snapper Unit (SU) 1, Grouper Unit (GU) 1, GU4 and parrotfish. MRFSS data indicates that large numbers of fish are being landed by the recreational sector, most significantly within SU1 (silk, blackfin, black, and vermillion snapper). However, the total number of fish reported from the intercepts is limited to 172 silk snapper measured between the years 2000 and 2008; and only 52, 36 and 3 vermillion, blackfin, and black snapper sampled during the same time period. A total of 43 individual parrotfish were sampled between 2000 and 2008 (CFMC and NMFS 2011).

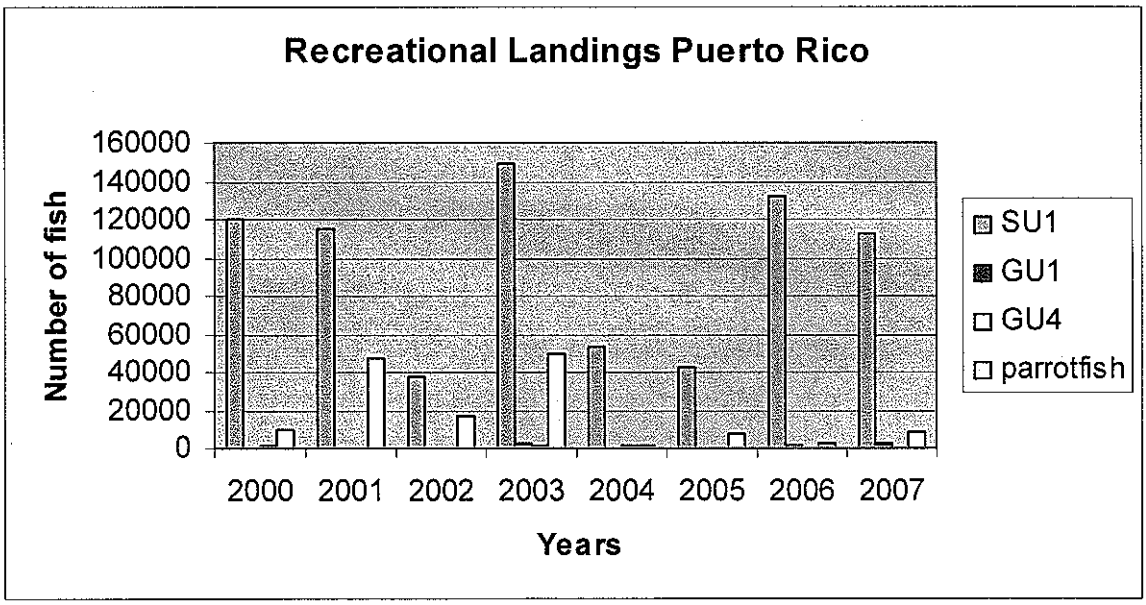


Figure 2.4.3.3. Recreational Landings in Puerto Rico by Management Unit, 2000-2008 (SU 1 = Snapper Unit1, GU 1 = Grouper Unit 1, GU 4 = Grouper Unit 4) (Source: CFMC and NMFS 2011)

2.4.4 USVI Recreational Fishing Activity

The most recent report on recreational fishing activity in USVI waters (Tobias and Dupigny 2009) reviews the information available for the area, including the surveys on the recreational fishing activity in general included in the Caribbean SFA Amendment to the FMPs (CFMC 2005) and most recently in the Caribbean Fisheries Data Evaluation workshop (SEDAR 2009). Most of the information on recreational fisheries for the USVI derives from offshore billfish and other pelagic fisheries since the area is well known for gamefish. Tobias and Dupigny (2009) summarize the information on the latest recreational fishing survey, targeting the pelagic fleet. None of the reports on the recreational fishing activity in the USVI target the fleet harvesting reef fish.

Logbook data identify snapper (yellowtail, mutton, and dog), jacks, and grouper (red hind, coney) as being harvested but accounting for less than 2% of the total fish reported in the logbooks (Tobias and Dupigny 2009).

Telephone surveys targeting boat-based and shore fishers provide an estimate of 10% of the USVI population participating as recreational fishers (Jennings 1992, Mateo 1999). In all cases, pelagic species are the most commonly targeted (Tobias and Dupigny 2009). In St. Thomas/St. John, 7,000 vessels were registered in 2005-2006 and 250 were registered in St. Croix (Tobias and Dupigny 2009). The National Angler Registry, which began in 2010 as part of the MRIP program, has in its database, for 2010, 1,389 anglers registered as fishing in the U.S. Caribbean EEZ (1,352 in Puerto Rico and 37 in the USVI) (F. Darby, NMFS/OSF, pers. comm.). There is no additional information on the fishing fleet of the USVI targeting reef fish.

3.0 Status of Listed Species and Critical Habitat

The following endangered and threatened species and critical habitat under the jurisdiction of NMFS may occur in the action area:

Invertebrates	**Status**
Staghorn coral (*Acropora cervicornis*)	Threatened
Elkhorn coral (*Acropora palmata*)	Threatened

Marine Mammals	**Status**
Sei whale (*Balaenoptera borealis*)	Endangered
Fin whale (*Balaenoptera physalus*)	Endangered
Humpback whale (*Megaptera novaeangliae*)	Endangered
Sperm whale (*Physeter macrocephalus*)	Endangered

Sea Turtles	
Loggerhead sea turtle (*Caretta caretta*)	Threatened
Green sea turtle (*Chelonia mydas*)	Endangered/Threatened[2]
Leatherback sea turtle (*Dermochelys coriacea*)	Endangered
Hawksbill sea turtle (*Eretmochelys imbricata*)	Endangered

Designated Critical Habitat for	**Region Where Designated**
Elkhorn and staghorn coral ("*Acropora*")	South Atlantic/Caribbean
Green sea turtle	Caribbean
Hawksbill sea turtle	Caribbean
Leatherback sea turtle	Caribbean

3.1 Species and Critical Habitat Not Likely to be Adversely Affected

Endangered Marine Mammals

At least seventeen species of whales and dolphins have been reported in or near U.S. waters in the northeastern Caribbean (Mignucci-Giannoni 1998). ESA-listed species known to occur in this area include the humpback, fin, sei, and sperm whale. The area provides feeding grounds for some of these species, and reproductive grounds for others. Most cetacean species in this area are sighted during the winter and early spring, with the increase in sightings beginning in December, peaking in February, and gradually decreasing in March and April; there are few sightings from May through November. Additionally, some species do not migrate, utilizing these waters for feeding and reproduction throughout the year (Mignucci-Giannoni 1998). Except for the humpback whale, which occurs in specific areas during winter to breed and calf, abundances and distributions of most marine mammals in the northeastern Caribbean are poorly known (Mignucci-Giannoni 1998).

[2] Green sea turtles in U.S. waters are listed as threatened except for the Florida breeding population, which is listed as endangered.

Under section 118 of the Marine Mammal Protection Act (MMPA), NMFS must publish, at least annually, a "List of Fisheries" that places all U.S. commercial fisheries into one of three categories based on the level of incidental serious injury and mortality of marine mammals that occurs in each fishery. The final rule for the 2010 List of Fisheries classifies all U.S. Caribbean commercial fisheries under the Caribbean Fishery Management Council's jurisdiction 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 (74 FR 58859).

Mignucci-Giannoni et al. (1999) conducted an assessment of cetacean strandings in waters of Puerto Rico and the Virgin Islands (both the USVI and the British Virgin Islands) to identify, document, and analyze factors associated with 129 (159 individuals) reported mortality events recorded between 1867 and 1995. The majority of these stranding events were reported for Puerto Rico (74.0%), with 15.7% of the events specifically reported for Puerto Rico's west coast. Overall, causes were not determined in 62.8% of the cases, though due to a cooperative effort there has been a recent reduction in the number of strandings attributed to undetermined causes. Of the 16.3% of the total 159 strandings that were attributed to human-related causes (n=26), 28.6% of those incidents were due to entanglement (n=7). The study documented 9 humpback whale strandings and 13 sperm whale strandings over the 128-year time period. Using the same percentages above specifically to the ESA-listed species that are the subject of this opinion, during the 128-year time period of the study, approximately 4 humpback and sperm whale strandings would be attributed to human-related causes, and only 1 of those strandings would be due to entanglement (e.g., by fishing gear). Based on the this information, the chance of the proposed action affecting ESA-listed species of large whales is discountable. NMFS concludes that the proposed action is not likely to adversely affect listed whales.

Loggerhead Sea Turtles

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 two 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 two loggerhead sea turtles were identified on St. Croix. 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.

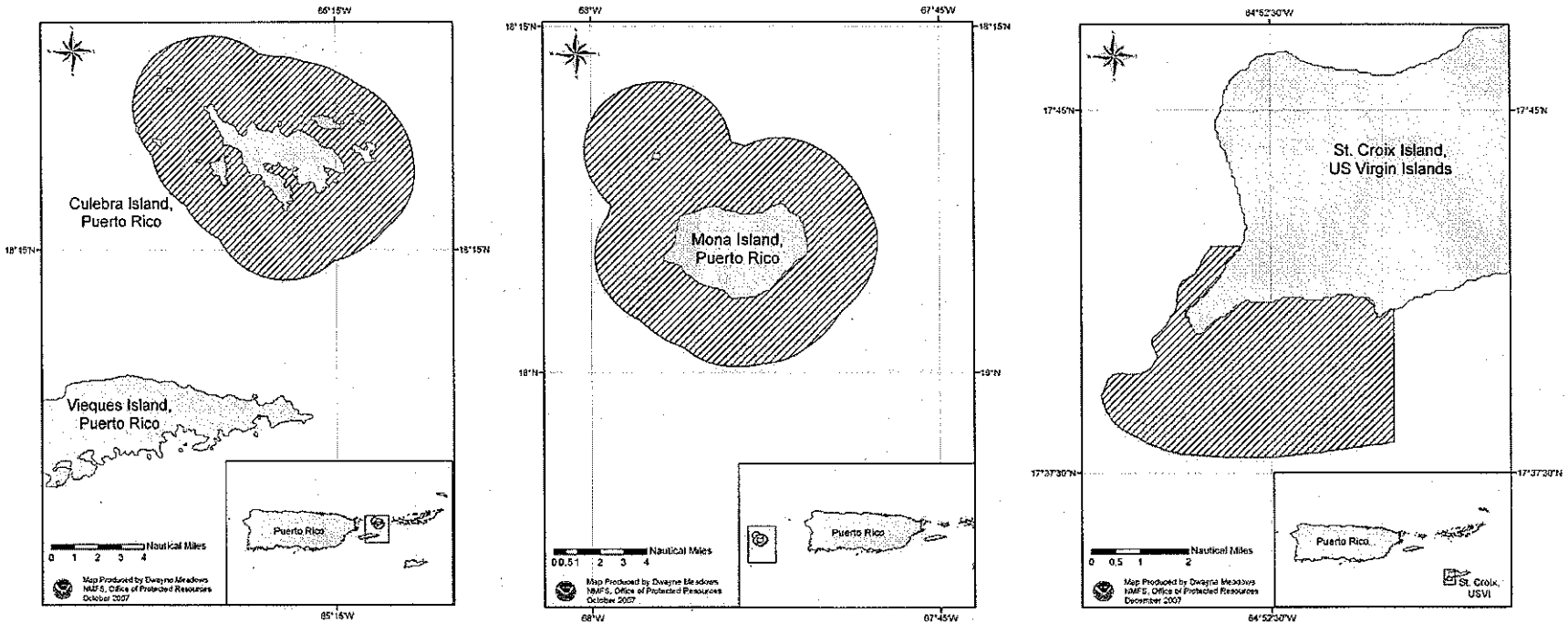
In areas where loggerhead sea turtles are abundant (e.g. Gulf of Mexico), they are subject to hook-and-line capture via their attraction to baited hooks (e.g., NMFS 2009). Loggerhead sea turtles are also typically vulnerable to entanglement in trap lines because of their attraction to, or

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 reef fish gear authorized by NMFS in the U.S. Caribbean are extremely unlikely and therefore, discountable. Therefore, loggerhead sea turtles are not likely to be adversely affected by the proposed action.

Sea Turtle Critical Habitat

Critical habitat for green, hawksbill, and leatherback sea turtles occurs in the 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 3.1.1A). 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 3.1.1B). 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, 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 (Figure 3.1.1C). The critical habitats for green and hawksbill sea turtles within the action area 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.

Critical habitat for green, hawksbill or leatherback sea turtles is not likely to be adversely affected by the proposed action. 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, authorized fishing activities under the proposed action would not overlap with the critical habitat areas and the proposed action is unlikely to have any measurable effect on sea turtles use these areas. Puerto Rico and USVI waters were included as part of the action area, but only because the NMFS-authorized EEZ fishery and its management indirectly affect local herbivorous fish populations and *Acropora*. Green and hawksbill sea turtles do 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 indirect effects on sea turtles from reef fish harvest are not a factor. The proposed action is not expected to result in changes in abaundance 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 extremely unlikely from the proposed action.



(A) Green Sea Turtle Critical Habitat (B) Hawksbill Sea Turtle Critical Habitat (C) Leatherback Sea Turtle Critical Habitat

Figure 3.1.1. Sea Turtle Critical Habitat in the Caribbean

3.2 The Deepwater Horizon MC252 Oil Release Event

On April 20, 2010, while working on an exploratory well approximately 50 miles offshore Louisiana, the semi-submersible drilling rig Deepwater Horizon (DWH) experienced an explosion and fire. The rig subsequently sank and oil and natural gas began leaking into the Gulf of Mexico. Oil flowed for 86 days, until finally being capped on July 15, 2010. Official estimates are that just under 5 million barrels of oil were released into the Gulf, with some experts estimating even higher volumes. Additionally, approximately 1.84 million gallons of chemical dispersant was applied both subsurface and on the surface to attempt to break down the oil. There is no question that the unprecedented Deepwater Horizon event and associated response activities (e.g., skimming, burning, and application of dispersants) have resulted in adverse effects on listed sea turtles. Elkhorn and staghorn corals can also be adversely affected by oil, but at this time there is no evidence documenting effects on elkhorn and staghorn coral from this particular oil spill.

At this time, the effects of the oil spill on species found throughout the Gulf of Mexico, including ESA-listed sea turtles, are not known. There is currently an ongoing investigation and analysis being conducted under the National Resource Damage Assessment (NRDA) program, but the final outcome of that investigation may not be known for many months to years from the time of this biological opinion. Additionally, the NRDA evaluation focuses primarily on attempting to quantify injuries in order to determine how those injuries can be compensated, and does not necessarily result in an understanding of the population-level impacts to a species. Ultimately restoration efforts that occur as part of the legal requirement stemming from the spill will help to offset at least some of the losses experienced by the species, but just as the impacts from the spill are not yet known, the success of any future restoration efforts is also unknowable at this time. However, despite the lack of solid information on the population level impacts to sea turtles, we must attempt a reasonable assessment of what those impacts may be based upon the limited available information, knowledge of the species involved, and best professional scientific judgment. This is needed in order to analyze how the continuation of the Caribbean Reef Fish Fishery would impact sea turtle species in light of the environmental baseline effects from the DWH event.

As of February 15, 2011, a total of 1,146 sea turtles have been documented as stranded or collected during response efforts in the spill area. Up through October 20, 2010, all stranded or distressed sea turtles found in the area were included on the list of DWH-related strandings or collections, regardless of evidence of oil exposure. Subsequent to that, only confirmed visibly oiled animals were added to the list. The available data on sea turtle strandings and response collections during the time of the spill are expected to represent an unknown fraction of the actual losses to the species, as most individuals likely were never recovered. It also does not provide insights into potential sub-lethal impacts that could reduce long-term survival or fecundity of individuals affected. However, it does provide some insight into the potential relative scope of the impact among the sea turtle species in the area. It appears that Kemp's ridley sea turtles may have been the hardest- hit species, as they accounted for almost 71 percent of all

stranded/collected turtles, and 79 percent of all dead turtles. Green turtles represented the second highest number of total individuals found, at 17.5 percent, but only 4.8 percent of the dead individuals. Loggerheads comprised only 7.7 percent of the total individuals, and 11 percent of the total dead. The remaining turtles were hawksbills and dead hardshell turtles that were not identified. No leatherbacks were counted among the stranded/collected turtles in the spill area (Table 3.2.1). However, we only anticipate green, hawksbill, and leatherback sea turtles will occur in the action area.

Table 3.2.1 Sea Turtles Documented in the DWH Spill Area.

Sea Turtle Species	Alive	Dead	Total
Green sea turtle (*Chelonia mydas*)	172	29	201
Hawksbill sea turtle (*Eretmochelys imbricata*)	16	0	16
Kemp's ridley sea turtle (*Lepidochelys kempii*)	328	481	809
Loggerhead sea turtle (*Caretta caretta*)	21	67	88
Unknown sea turtle species	0	32	32
Total	**537**	**609**	**1146**

(Source: http://www.nmfs.noaa.gov/pr/health/oilspill/turtles.htm)

Although extraordinarily high numbers of threatened and endangered sea turtle strandings have been documented since the start of the Deepwater Horizon MC252 oil spill (primarily within Mississippi Sound, outside of the action area), the vast majority of sea turtles documented have shown no visible signs of oil. Due to the oil spill there has been an increase in awareness and human presence in the northern Gulf of Mexico, which likely has resulted in some of the increased documentation of stranded turtles. However, we do not believe this factor fully explains the increase. Investigations, including necropsies, were undertaken by NMFS to attempt to determine the cause of those strandings. Based on the findings, the two primary considerations for the cause of death of the sea turtles that were necropsied are forced submergence or acute toxicosis. The only known plausible cause of forced submergence that could explain this event is incidental capture in fishing gear. NMFS has assembled information regarding fisheries operating in the area during and just prior to these strandings. While there is some indication that lack of compliance with existing TED regulations or the operations of other trawl fisheries that do not require TEDs may have occurred in the area at the time of the strandings, concrete evidence that those events caused the unusual level of strandings is not available. NOAA scientists tested sea turtle tissue samples for biotoxins of concern in the northern Gulf of Mexico, which is a standard measure in marine animal mortality investigations. Current environmental information does not indicate a harmful algal bloom of threat to marine animal health was present in the area. More information on the stranding event, including number of strandings, can be found at http://www.nmfs.noaa.gov/pr/species/turtles/gulfofmexico.htm.

In addition to effects on subadult and adult sea turtles, the May through September sea turtle nesting season in the northern Gulf may also have been adversely affected by the Deepwater Horizon MC252 oil spill. Setting booms to protect beaches may have had

unintended effects, such as preventing females from reaching nesting beaches and thereby reducing nesting. However, there is almost no sea turtle nesting in Louisiana, and limited nesting in Mississippi, which is where most of the booming of the coastline in response to the oils spill occurred, thus such effects were likely very minimal.
The oil spill may also have adversely affected hatchling success. In the northern Gulf area, approximately 700 nests are laid annually in the Florida Panhandle and up to 80 nests are laid annually in Alabama. Most nests are made by loggerhead sea turtles; however, a few Kemp's ridley and green turtle nests were also documented in 2010. Hatchlings begin emerging from nests in early to mid-July, with approximately 50,000 hatchlings anticipated to be produced from northern Gulf sea turtle nests in 2010. To avoid the loss of most, if not all, of that year's northern Gulf of Mexico hatchling cohort, all sea turtle nests laid along the northern Gulf coast were visibly marked to ensure that nests were not harmed during oil spill cleanup operations that are undertaken on beaches. In addition, a sea turtle late-term nest collection and hatchling release plan was implemented to provide the best possible protection for sea turtle hatchlings emerging from nests in Alabama and the Florida Panhandle. Starting in June, northern Gulf Coast nests were relocated to the Atlantic to provide the highest probability of reducing the anticipated risks to hatchlings as a result of the Deepwater Horizon MC252 oil spill. Of the species of sea turtles affected by the oil spill that occur in the Caribbean, a total of four green sea turtle nests were translocated from the northern Gulf of Mexico to the east coast of Florida so that the hatchlings could be released in areas not affected by the oil spill. Ultimately, 455 green sea turtle hatchlings were released as part of this translocation process. In mid-August, 2010, it was determined that the risks to hatchlings emerging from beaches and entering waters off the coast of Franklin and Gulf counties had diminished significantly and nest translocations were ceased on August 19, 2010.

The survivorship and future nesting success of individuals from one nesting beach being transported to and released at another nesting beach is unknown. Green sea turtles comprised the second-most common species collected as part of the DWH response, with 201 individuals. However, only 29 of those were found dead or later died during attempts at rehabilitation. While green sea turtles regularly utilize the northern Gulf of Mexico, they have a widespread distribution throughout the entire Gulf of Mexico, Caribbean, and Atlantic. As described in the Status of the Species section, nesting is also relatively rare in the northern Gulf beaches. Therefore, while it is expected that adverse impacts occurred, a relatively small proportion of the population is believed to have been exposed to and impacted by the DWH event.

Presently available information indicates hawksbill and leatherback sea turtles were least affected by the oil spill. No leatherbacks and only 16 hawksbills (all alive) were counted among the stranded and response-collected sea turtles. Hawksbills do not typically utilize the northern Gulf of Mexico in large numbers, and thus population-level effects from the spill are expected to be negligible. Leatherbacks rarely nest along the Gulf coast, but do utilize the offshore waters. Potential DWH-related impacts to leatherback sea turtles could include ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources. There is no information available to determine the extent of those impacts, if they occurred. However, leatherback prey

species are typically jellyfish and other cnidarians, salps, and tunicates, which occur in great abundance throughout much of the Gulf of Mexico, and tend to be fast-reproducing taxa.

3.3 Analysis of the Species Likely to be Adversely Affected

Elkhorn and staghorn corals and green, hawksbill, and leatherback sea turtles may be adversely affected by the proposed action. All of these sea turtle species are vulnerable to one or more of the gear types used in the Caribbean reef fish fishery based on strandings records and their capture in other southeast fisheries using similar gear. Elkhorn and staghorn corals may be affected by direct injury from fishing gear and the fishery's effects on populations of reef fish, particularly algae-grazing parrotfish and surgeon fish. The remaining sections of this opinion will focus solely on these species.

The following subsections are synopses of the best available information on the life history, distribution, population trends, and current status of the three species of sea turtles and two species of *Acropora* that are likely to be adversely affected by one or more components of the proposed action. Much of the information for this section, as well as additional detailed information relating to species' biology, habitat requirements, status, threats, and recovery objectives, can be found in the status review and recovery plan for each species (see www.nmfs.noaa.gov/prot_res/PR3/recovery.html). Additional background information on the status of sea turtle species can be found in a number of published documents, including: recovery plans for the Atlantic green sea turtle (NMFS and USFWS 1991), hawksbill sea turtle (NMFS and USFWS 1993), and leatherback sea turtle (NMFS and USFWS 1992); Pacific sea turtle recovery plans (NMFS and USFWS, 1998a-c) and sea turtle status reviews, stock assessments and other biological reports (NMFS and USFWS 1995, NMFS and USFWS 2007a-d, ; NMFS SEFSC 2001, and Marine Turtle Expert Working Group (TEWG) (2007). Information on life history and threats to *Acropora* corals comes primarily from the *Acropora* status review document (*Acropora* BRT 2005) and ESA listing, section 4(d), and critical habitat rules (71 FR 26852, 73 FR 64264, 73 FR 72210).

The sea turtle subsections focus primarily on the Atlantic Ocean populations of these species because these are the populations that may be directly affected by the proposed action. However, these species are listed as global populations (with the exception of Florida greens, whose distribution is entirely in the Atlantic, including the Gulf of Mexico). The global status and trends of these species, therefore, are included as well, to provide a basis and frame of reference for our final determination of the effects of the proposed action on the species as listed under the ESA.

3.3.1 Elkhorn and Staghorn Corals

Elkhorn and staghorn corals (collectively "*Acropora*"), the only two species of acroporids in the Caribbean, are two of the major reef-building corals in the wider Caribbean. Elkhorn colonies form flattened to near-round branches that typically radiate outward from a central trunk that is firmly attached to the sea floor. Staghorn colonies are stag antler-like, with cylindrical, straight, or slightly curved branches. The branching morphology of

these species provides important habitat for other reef organisms. Historically, both acroporid species formed dense thickets at shallow (<5 m) and intermediate (10 to 15 m) depths in many reef systems, including locations in the Florida Keys, western Caribbean (e.g., Jamaica, Cayman Islands, Caribbean Mexico, Belize), and eastern Caribbean. In the 1960s and 1970s in the USVI, elkhorn coral was the main reef-building coral at depths less than 10 meters (Rogers et al. 2002). Elkhorn coral grew in nearly monospecific stands on the reef crest and in the upper and lower forereef zones of well-developed fringing and bank barrier reefs, as well as on isolated patch reefs (Rogers et al. 2002). In terms of accretion rates and the formation of structurally complex reefs, the structural and ecological roles of Atlantic *Acropora* in the wider Caribbean are unique and cannot be filled by other reef-building corals (Bruckner 2002).

Life History and Distribution

The maximum range in depth reported for elkhorn coral is <1 m to 30 m, but historic data for this coral in the USVI indicate that it was common at depths from 1 to 15 m (Bacle 2002, Rogers et al. 2008). The preferred habitat of elkhorn coral is the seaward face of a reef (turbulent shallow water), including the reef crest, and shallow spur-and-groove zone (Shinn 1963, Cairns 1982, Rogers et al. 1982). At low tide, colonies are sometimes exposed. Colonies of elkhorn coral often grow in nearly monospecific, dense stands and form an interlocking framework known as a thicket in fringing and barrier reefs (Jaap 1984, Tomascik and Sander 1987, Wheaton and Jaap 1988). Storm-generated fragments are often found occupying back reef areas immediately landward of the reef flat/reef crest, while colonies are rare on lagoonal patch reefs (Dunne and Brown 1979).

Historically, staghorn coral was reported from depths ranging from <1 to 60 m (Goreau and Goreau 1973). It is suspected that 60 m is an extreme situation and that the coral is relatively rare below 20 m depth. The common depth range at which staghorn coral is currently observed is 5 to 17 m. In the USVI, this species was abundant, but not often found in dense thickets or well-defined zones (Rogers et al. 2002); unlike in areas in the western Caribbean where this species was historically the primary constructor of mid-depth (10 to 15 m) reef terraces (Adey 1978).

Atlantic *Acropora* are considered to be environmentally sensitive, requiring relatively clear, well-circulated water (Jaap et al. 1989). Atlantic *Acropora* are almost entirely dependent upon sunlight for nourishment compared to massive, boulder-shaped species in the region (Porter 1976, Lewis 1977), which are more dependent on zooplankton. Therefore, *Acropora* may not be able to compensate for bleaching or reduced photosynthesis with an alternate food source, such as zooplankton or suspended particulate matter, like other corals. Subsequently, Atlantic *Acropora* are much more susceptible to increases in water turbidity than some other coral species. Reductions in long-term water clarity can also reduce the coral photosynthesis to respiration ratio (P/R ratio). Telescnicki and Goldberg (1995) and Yentsch et al. (2002) found that elevated turbidity levels did not affect gross photosynthetic oxygen production, but did lead to increased respiration that consumed the products of photosynthesis with little remaining for coral growth.

Optimal water temperatures for elkhorn and staghorn corals range from 25° to 29°C, although colonies in the USVI have been known to tolerate short-term temperatures around 30°C without obvious bleaching (loss of zooxanthellae) (Rothenberger et al. 2008). All *Acropora* require near oceanic salinities (34 to 37 ppt). All Atlantic acroporids are susceptible to bleaching due to adverse environmental conditions (Ghiold and Smith 1990, Williams and Bunkley-Williams 1990). The major El Niño/La Niña Southern Oscillation cycle in 1997-1998 resulted in a large bleaching event in the Caribbean and the Atlantic, as well as massive losses of corals in the Indian Ocean and Western Pacific (Wilkinson and Souter 2008). Elevated temperatures in the fall of 1998 led to a loss of coral cover in study sites in the USVI (Rogers et al. 2008). However, the most significant bleaching event to date in the USVI and other areas of the Caribbean occurred in 2005 when sea surface temperatures exceeded the 29.5°C coral bleaching threshold for twelve weeks, and maximum temperatures exceeded 30°C (Woody et al. 2008). Bleaching occurred in twenty-two species, including *Acropora*, over a wide range of depths and affected more than 90% of the coral cover, on average, between July and November in the USVI (Woody et al. 2008).

Atlantic *Acropora*, like many stony coral species, propagate sexually and asexually through fragmentation. Atlantic *Acropora* reproduce sexually by broadcast spawning, meaning that coral larvae develop externally to the parental colonies (Szmant 1986), and both species are simultaneous hermaphrodites, meaning that a given colony will contain both female and male reproductive parts during the spawning season. Despite being simultaneous hermaphrodites, *Acropora* are obligate out-crossers, which means two non-clonal colonies must be present for sexual reproduction to occur. Gametes (eggs and sperm) are located in different layers of the same polyp (Soong 1991). The spawning season for elkhorn and staghorn corals is relatively short, with gametes released only a few nights during July, August, and/or September. Observations in the USVI and Puerto Rico indicate that spawning of elkhorn populations occurs up to ten days following the full moon during August and/or September (Williams 2006, Anlauf Toller 2007), and staghorn corals spawn within a week of the full moon in July and/or August (Lirman 2002). Annual egg production in elkhorn and staghorn populations studied in Puerto Rico was estimated to be 600 to 800 eggs per cm^2 of living coral tissue (Szmant 1986).

In *Acropora*, fertilization and development are exclusively external. Embryonic development culminates with the development of planktonic larvae called planulae. Little is known concerning larval settlement patterns (Bak 1977, Sammarco 1980, Rylaarsdam 1983). In general, upon proper stimulation, coral larvae, whether released from parental colonies or developed in the water column external to the parental colonies, settle and metamorphose on appropriate substrates, in this case preferably coralline algae. Initial calcification ensues with the forming of the basal plate. Buds that form on the initial corallite develop into daughter corallites.

Studies of elkhorn and staghorn corals on the Caribbean coast of Panama indicated that larger colonies of both species (as measured by surface area of the live colony) have higher fertility rates (Soong and Lang 1992). For elkhorn coral, the larger the colony, the higher the fecundity rate; over 80% of the colonies larger than 4000 cm^2 were fertile. The

estimated size at puberty for elkhorn coral was 1600 cm^2 and the smallest reproductive colony observed was 128 cm^2. Only colonies of staghorn coral with a branch length larger than 9 cm were fertile and over 80% of colonies with branches longer than 17 cm (n=18) were fertile. The estimated size at puberty for staghorn coral was 17 cm in branch length and the smallest reproductive colony observed was 9 cm in branch length (Soong and Lang 1992).

Spatial and temporal patterns of coral recruitment have been intensively studied on wider Caribbean reefs (Birkeland 1977, Bak and Engel 1979, Rogers et al. 1984, Baggett and Bright 1985, Chiappone and Sullivan 1996). Biological and physical factors that have been shown to affect spatial and temporal patterns of coral recruitment include substrate availability and community structure (e.g., Birkeland 1977), grazing pressure (e.g., Rogers et al. 1984, Sammarco 1985), fecundity, mode and timing of reproduction (e.g., Harriot 1985, Richmond and Hunter 1990), behavior of larvae (e.g., Lewis 1974, Goreau et al. 1981), hurricane disturbance (e.g., Hughes and Jackson 1985), physical oceanography (e.g., Baggett and Bright 1985, Fisk and Harriot 1990), the structure of established coral assemblages (e.g., Lewis 1974, Harriot 1985), and chemical cues (e.g., Morse et al. 1988). Studies of *Acropora* from across the wider Caribbean confirm two overall patterns of sexual recruitment: (1) Low juvenile densities relative to other coral species and (2) low juvenile densities relative to the commonness of adults (Porter 1987). This pattern suggests that the composition of the adult population is dependent upon variable recruitment.

The growth rate of elkhorn coral, expressed as the linear extension of branches, is reported to range from 2 to 11 cm annually (Vaughan 1915, Jaap 1974, Gladfelter et al. 1978, Becker and Mueller 2001) with values from 4.7 to 9.9 cm/year reported for the USVI (Gladfelter et al. 1978). The growth rate for staghorn coral has been reported to range from 3 to 11.5 cm/yr (Vaughan 1915, Shinn 1966, Jaap 1974, Shinn 1976, Gladfelter et al. 1978, Becker and Mueller 2001) with a value of 7.1 cm/year for the USVI (Gladfelter et al. 1978). These growth rates are relatively fast compared to other corals and historically enabled the species to construct significant reefs in locations throughout the wider Caribbean (Adey 1978). Growth of elkhorn and staghorn corals is also expressed in expansion, occurring as a result of fragmenting and forming new centers of growth (Bak and Criens 1982, Tunnicliffe 1981). A broken branch may be carried by waves and currents to a distant location or may land in close proximity to the original colony. If the location is favorable, branches grow into a new colony, expanding and occupying additional area. Fragmenting and expansion, coupled with a relatively fast growth rate, facilitates potential spatial competitive superiority for elkhorn and staghorn corals relative to other corals and other benthic organisms (Shinn 1976, Neigel and Avise 1983, Jaap et al. 1989). Because growth rates decline with increasing colony size, fragmentation may help maintain high growth rates by pruning colonies to create new, smaller units. However, severe fragmentation, such as caused by hurricanes, can limit sexual reproduction by breaking colonies to such a degree that energy is shifted from reproduction to stabilization and regeneration (Lirman 2002).

In the decades of the 1960s and 1970s, many Caribbean reefs were described as having an elkhorn (*A. palmata*) zone and a staghorn (*A. cervicornis*) zone, based upon high coverage and colony density and in some cases near exclusiveness of these species at particular depths. Figure 3.3.1.1 illustrates these zones based on zonation-descriptive studies (Goreau 1959, Kinzie 1973, Bak 1977).

Typically, the elkhorn zone extended from the surface to about 5 m depth and the staghorn zone extended from about 7 to 15 m depth. Throughout much of the wider Caribbean, elkhorn coral dominated 1 to 8 m depth (reef flat, wave zone, reef crest) in diverse areas including Jamaica (Goreau 1959), Alacran Reef, Yucatan peninsula (Kornicker and Boyd 1962), Abaco Island, Bahamas (Storr 1964), the southwestern Gulf of Mexico, Bonaire (Scatterday 1974), the Florida Keys (Jaap 1984, Dustan and Halas 1987), and reef systems at depths of 0 to 5m around St. John, St. Thomas, and St. Croix (Bacle 2002). The predominance of elkhorn coral in shallow reef zones is related to the degree of wave energy; in areas with strong wave energy conditions, only isolated colonies may occur, while thickets may develop at intermediate wave energy conditions (Geister 1977). Staghorn coral was documented in several reef systems such as the north coast of Jamaica (Goreau 1959) and the leeward coast of Bonaire (Scatteryday 1974). In many other reef systems in the wider Caribbean, most notably the western Caribbean areas of Jamaica, Cayman Islands, Belize, and eastern Yucatan (Adey 1977), staghorn coral was a major mid-depth (10 to 25 m) reef-builder. As noted earlier, staghorn coral was abundant historically in the USVI, but did not commonly form dense thickets and was not confined to well-defined zones (Rogers et al. 2002).

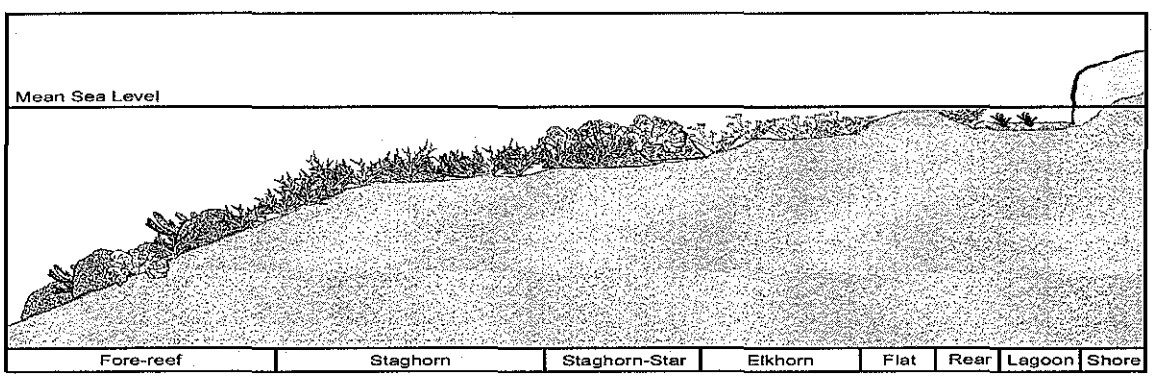


Figure 3.3.1.1. Reef Zone Schematic

Population Dynamics and Status

Recent information is available on the status of Atlantic *Acropora* from 60 to 75% of all the reefs where these species are known to occur (Buckner 2002). Both elkhorn and staghorn corals still occupy their historic range, but localized range reductions and extirpations have occurred with most populations experiencing losses from 80-98% of their 1970s baseline (Bruckner 2002). The 1970s were established as a baseline for stable, healthy populations through the historic range of Atlantic *Acropora* and the 1980s were established as the baseline for the regional decline due to mortality events associated with white band disease outbreaks and subsequent hurricane damage (Richards Kramer 2002, Rogers et al. 2002). For this reason, available information on the historical distribution and abundance patterns focus on percent coverage, density, and relative size

of the corals during three periods: pre-1980, the 1980 – 1990 decades, and recent (since 2000). During 2001 through 2002, elkhorn coral populations in a number of locations were considered stable, although these populations were at only 5% of their historic abundance (Bruckner 2002).

Both acroporid species underwent precipitous declines in the early 1980s throughout their ranges and this decline has generally continued. Although quantitative data on former distribution and abundance are scarce, in the few locations where quantitative data are available (e.g., Florida Keys, Dry Tortugas, Belize, Jamaica, and the USVI), declines in abundance (coverage and colony numbers) are estimated at >97% (*Acropora* BRT 2005). Although this downward trend has been documented as continuing in the late 1990s, and up to the time of listing the species in some locations, local extirpations (i.e., at the island or country scale) have not been definitively documented. In addition to declines in numbers of colonies and percent cover, the total surface area of live tissue is now much less than historically because colonies are small and sometimes encrusting rather than complex, three-dimensional structures. Historically, colonies stood meters above the substrate with live tissue from the branches down to the base of the colony. Thus, overall tissue reduction in reef zones formerly dominated by elkhorn is two orders of magnitude or more (i.e., >99%) based on studies of populations around St. Croix (Rogers et al. 2002).

Figure 3.3.1.2 summarizes the abundance trends of specific locations throughout the wider Caribbean where quantitative data exist (eight locations) illustrating the overall trends of decline of elkhorn and staghorn corals from the 1970s and 1980s through 2004. The green squares in Figure 3.3.1.2 represent the percent loss of staghorn coral and the yellow triangles represent elkhorn coral percent loss. Shaded areas on map illustrate the general range of elkhorn and staghorn corals (*Acropora* BRT 2005). It is important to note that the data are from the same geographic area, not repeated measures at an exact reef/site that would indicate more general trends.

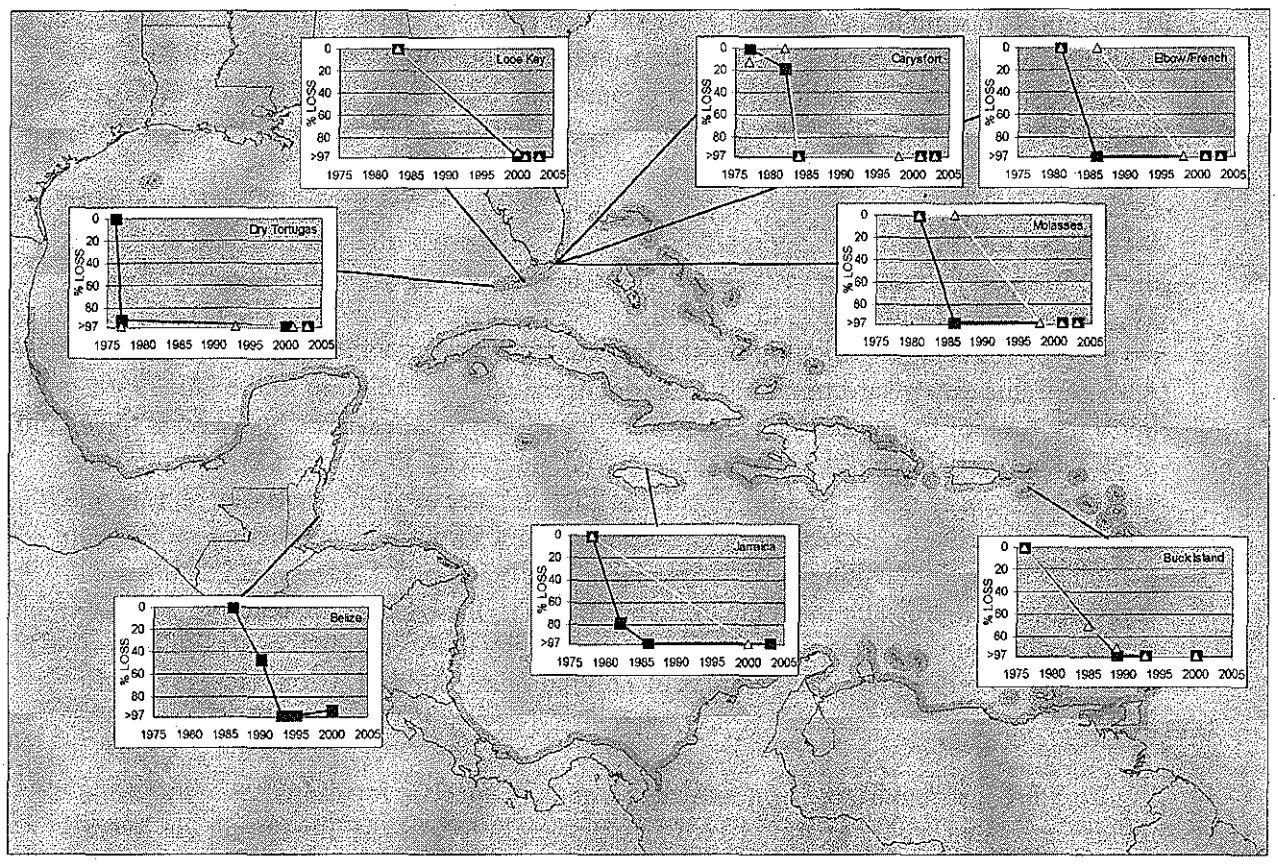


Figure 3.3.1.2. Percent Loss of Staghorn and Elkhorn Coral Throughout the Caribbean

In 2005 a major bleaching event was recorded in the Caribbean that affected coral throughout the basin, including the U.S. Caribbean. This bleaching event led to the direct mortality of some coral colonies, while other colonies succumbed to a disease outbreak that attacked the weakened, bleached colonies.

Species at reduced abundance are at a greater risk of extinction due to stochastic environmental and demographic factors (e.g., episodic recruitment factors). Both elkhorn and staghorn coral have persisted at extremely reduced abundance levels (in most areas with quantitative data available, less than 3% of prior abundance) for at least two decades. In addition, appropriate substrate availability for fragments to attach has been reduced due to changes in benthic community structure on many Caribbean reefs related to algal growth attributed to the mass die-off of *Diadema* and the harvest of herbivorous fishes, which in some cases have been overfished[3] for decades (Jackson et al. 2001), and changes in sediment deposition patterns associated with coastal development. Because algal turfs can trap and retain sediments, the combined impacts of these factors on larval settlement can exceed impacts of algae or sedimentation separately (Birrell et al. 2005).

[3] The term overfished is frequently used in publications by the author(s) to describe an exploitation level of fishes that in the author(s)' opinion is excessive. This term is not necessarily used as defined by the Magnuson-Stevens Fishery Conservation and Management Act.

These factors are expected to further reduce successful larval recruitment below the level that can compensate for observed rates of ongoing mortality (i.e., mortality outpacing growth and recruitment).

In many locations, populations of Atlantic *Acropora* have been reduced to such an extent that the potential for recovery through re-growth of fragments is limited and recovery is dependent on sexual reproduction. Unfortunately, since elkhorn and staghorn are broadcast spawners, once colonies become rare, the distance between colonies may limit fertilization success and there is substantial evidence to suggest that sexual recruitment of both elkhorn and staghorn corals is currently compromised. Reduced colony density in some areas is compounded by low genotypic diversity, indicating that fertilization success and consequently, larval availability, is likely reduced. This can have long-term implications for genetic variability of remaining colonies due to the reduced potential for exchange of genetic material between populations that are spatially further apart (Bruckner 2002).

Data on levels of genetic diversity and population structure suggest that there is a population structure among islands, and even over spatial scales of no more than 20 km, as well as varying degrees of genetic diversity within local populations (Lirman 2002, Vollmer 2002). For instance, one clone of staghorn coral may dominate areas up to 10 m^2 in size and the clones are generally spatially discrete with larval exchange between staghorn populations as close as 2 to 15 km being extremely limited, suggesting that larval sources need to be conserved on a very small spatial scale (Baums et al. 2005, Vollmer and Palumbi 2007).

A study of genetic exchange and clonal population structure in elkhorn corals from eleven locations throughout its geographic range indicates that the eastern Caribbean (St. Vincent and the Grenadines, the USVI, Curacao, and Bonaire) has experienced little or no recent genetic exchange with populations in the western Caribbean (Bahamas, Florida, Mexico, Panama, Navassa, and Mona Island). Clonal structural analysis found that populations of elkhorn corals in the eastern Caribbean were denser and more genotypically diverse suggesting a greater contribution from sexual recruitment (Baums et al. 2005). Within each region, some locations were found to be entirely self-recruiting while others were found to receive larvae from other locations (Baums et al. 2005) suggesting that the conservation of local populations is critical to recovery of both elkhorn populations. Baums et al. (2005) also identified Mona Island and Puerto Rico as an area of mixing between the two subregions. Their results revealed a significant genetic discontinuity that indicates populations from the eastern Caribbean and from the western Caribbean have experienced little if any gene flow between them in the recent past. Genets from Puerto Rico and Mona Island have mixed ancestry, which suggests this area may be particularly important in maintaining or expanding the genetic diversity of the species.

Threats

Elkhorn and staghorn corals face myriad stressors that in some cases act synergistically. Diseases, temperature-induced bleaching, and physical damage from hurricanes are deemed to be the greatest threats to elkhorn and staghorn corals' survival and recovery. The impact of disease, though clearly severe, is poorly understood in terms of etiology and possible links to anthropogenic stressors. Impacts from anthropogenic physical damage (e.g., vessel groundings, anchors, and divers/snorkelers), coastal development, competition, and predation are deemed to be moderate. The major threats (e.g., disease, elevated sea surface temperature, and hurricanes) to elkhorn and staghorn corals' persistence are severe, unpredictable, likely to increase in the foreseeable future, and, at current levels of knowledge, unmanageable. However, managing some of the stressors identified as less severe (e.g., nutrients, sedimentation, macroalgae) may assist in decreasing the rate of elkhorn and staghorn corals' decline by enhancing coral condition and decreasing synergistic stress effects. Table 3.2.1.1 summarizes the factors affecting the status of elkhorn and staghorn corals and the identified sources of those stressors.

Table 3.2.1.1. Stressors Affecting Elkhorn and Staghorn Coral

Major Stressors	
Natural abrasion and breakage Source: storm events	**Disease** Source: undetermined/understudied
Temperature Source: hypothermal events global climate change power plant effluents El Niño-Southern Oscillation events	
Moderate Stressors	
Sedimentation Source: land development/run-off dredging/disposal sea level rise major storm events	**Anthropogenic abrasion and breakage** Source: divers vessel groundings anchor impact fishing debris
Loss of genetic diversity Source: population decline/bottleneck	**Predation** Source: fishing natural trophic reef interactions
Nutrients Source: point-source non-point-source	**Contaminants** Source: point-source non-point-source
Competition Source: fishing	**CO_2** Source: fossil fuel consumption
Sea level rise Source: global climate change	**Sponge boring** Source: undetermined/understudied

Virtually all of the threats impacting coral reef ecosystems, including land-based and marine pollution, fishing, global climate change, and ocean acidification, have been suggested as drivers or facilitators of infectious disease. Infectious disease in corals has increased in frequency and distribution since the 1970s when white band disease was first reported in Atlantic *Acropora*. There has since been an exponential increase in the numbers of reported diseases, host species, and locations where infections have been observed (Raymundo et al. 2008). Current research suggests that human activities that lead to point and non-point source discharges of nutrients, sediments, and other

substances from land and discharges of ballast water and vessel waste, among others, may exacerbate existing opportunistic infections in combination with stressors such as poor water quality, macroalgal allelopathic metabolites, and sea surface temperature increases. It may be that increased temperatures enhance the virulence of pathogens, or that the ability of corals to fight infections at higher temperatures is lessened.

White band disease (WBD), which affects acroporid corals, was first observed on reefs around St. Croix in 1977 (Gladfelter et al. 1978). In the Caribbean, the incidence of WBD ranges from <1 to 64% of the colonies in a single area. WBD is thought to be the major factor responsible for the rapid loss of Atlantic *Acropora* due to mass mortalities. WBD 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). Land-based pollution, in particular human waste streams that enter coastal waters, has been implicated in the search for causal agents of coral disease. Isolates from diseased tissues of elkhorn coral infected with another coral disease known as white pox, were found to match *S. marcescens*, a fecal enteric bacterium in humans (Patterson et al. 2002). Enteric bacteria associated with human fecal material have been found in surface mucus layers of corals in the Florida Keys, but the study by Patterson et al. (2002) is one of the first to isolate a specific bacterium from diseased tissue that implicates human fecal contamination as the causal agent for white pox. In 2011, Sutherland et al (2011) were able to definitively identify human waste as a cause for white pox disease in elkhorn corals. Data from the study by Patterson et al. (2002) also indicate that the rate of tissue loss due to white pox correlates with seasonal conditions of elevated temperature. This supports work by other scientists indicating that elevated temperatures lead to accelerated growth of pathogens and reduce the capacity of the coral's immune system to combat the disease.

Disease has also been linked to sunscreen use in areas containing corals based on a study of tourist destinations in Indonesia; Akumal, Mexico (Caribbean); Thailand; and the Red Sea (Danovaro et al. 2008). Nubbins from *Acropora* spp., as well as samples from two other corals were collected from various colonies, washed with virus-free seawater, and incubated in situ. In all replicates and sampling sites, sunscreen additions even at very low concentrations resulted in the release of large amounts of mucus by the corals within 18 to 48 hours, and complete bleaching of hard corals within 96 hours (Danovaro et al. 2008). Different sunscreen brands, protective factors, and concentrations were compared, and all were found to cause bleaching, although bleaching rates were faster the more sunscreen was used and under conditions of elevated temperatures. Viral abundance in seawater surrounding coral branches also increased significantly when sunscreens were added. Because the corals were washed and incubated in virus-free seawater prior to any treatments, Danovaro et al. (2008) concluded that sunscreen caused coral bleaching by inducing the lytic cycle in zooxanthellae with latent viral infections. Based on their results, Danovaro et al. (2008) concluded that, because at least 25% of the amount of sunscreen applied washes off during a 20-minute swim and based on the annual production of UV filters and the estimated number of tourists per year in tropical reef areas, a potential level of 4,000 to 6,000 tons/year of sunscreen is released in coral areas. They further concluded that, because 90% of tourists are expected to be concentrated in

approximately 10% of all reef areas, up to 10% globally of coral reefs are potentially threatened by sunscreen-induced coral bleaching.

One of the stressors with the greatest effect on corals is the increase in sea surface temperatures, which causes increased stress to corals and results in coral bleaching and, often, mortality, due in part to associated reductions in the ability of corals to combat infections and their increased susceptibility to other stressors, such as macroalgal allelopathic metabolites. Bleaching results in a loss of zooxanthellae and a reduction in the energy producing systems of corals; this can lead to severe stress and mortality. Coupled with increasing CO_2 concentrations, which lower the pH of seawater, reducing the capacity of corals and other organisms to produce calcium carbonate skeletons, and local stressors such as declining water quality and fishing, these stressors reduce the resiliency of coral reefs and reef-building organisms such as Atlantic *Acropora*. Sea surface temperatures rose by an average of 0.3°C between the 1950s and 1990s making it likely that corals are now 1°-1.5°C closer to their upper thermal limit and explaining why sustained temperatures as little as 1°-2°C above the normal summer maximum are sufficient to cause coral bleaching (Kleypass and Hoegh-Guildberg 2008).

Hurricanes are acute physical factors that have immediate and long-term effects on corals. Damage to reefs occurs from the physical force of hurricane induced waves, sand-blasting of live tissue, abrasion impact with dislodged coral and rubble fragments, smothering or burial of organisms, increases in turbidity, salinity reduction, and increases in nutrient concentrations after heavy runoff or from the breakdown of moribund tissue (Rogers et al. 1982, Brown 1997). Recovery of hard coral populations following a hurricane takes place through (1) settlement, survival, and growth of sexually produced coral recruits, (2) healing and regeneration of damaged colonies, and (3) growth of coral fragments (Rogers et al. 1991). Recovery is presumably faster if the dominant coral are fast growing branching species (e.g. Acroporids). However, recovery will be impeded if (1) the substrate for settlement (by larvae and fragments) has been altered, (2) grazing by herbivorous fish of substrate suitable for settlement has been reduced, or (3) disturbances recur or continue (Rogers et al. 1991).

Many natural disturbances are discrete, periodic events and often occur with enough time between events to allow for recovery between impacts by larval and fragment-mediated recruitment and growth. In other words, the effects from a hurricane on a coral reef can often depend on the length of time between storms (e.g., Connell 1978, Hughes 1989, Witman 1992, Connell et al. 1997). While hurricanes are an important part of the disturbance regime, the spatial and temporal variability in effects to reefs and between corals with different susceptibilities means that even frequent hurricane disturbance may be 'intermediate' in its effects in promoting system-wide diversity (Bythell et al. 2000).

Human activity in coral reef areas is another stressor of elkhorn and staghorn coral, particularly boating/anchoring, fishing, SCUBA diving, and snorkeling (*Acropora* BRT 2005). Ships/boats can dislodge and fracture corals, pulverize coral skeletons into small debris-rubble, displace sediment deposits, flatten the topography, and destroy or fracture the reef platform. Salvage operations often result in additional damage due to

inappropriate methods and poor control of operations. In some cases, the ship's hull is ruptured, and cargo and fuel are spilled on the reef (*Acropora* BRT 2005).

The shallow habitat of elkhorn coral makes this species especially vulnerable to vessel groundings (particularly recreational boat traffic). Evidence indicates certain populations near high boat traffic areas suffer chronic damage from repeated groundings. For example, two boats grounded on elkhorn coral reefs inside Virgin Islands National Park, St. John, in the same year. In April 2002, an 85-foot ferry boat struck Johnson's Reef, damaging approximately 920 m^2 of coral, mostly elkhorn coral. In August 2002, another vessel (a 60-foot sportfishing boat) struck Johnson's Reef from the other side, damaging approximately 1,650 m^2 of coral reef; 35 corals, mostly elkhorn, were injured or destroyed (*Acropora* BRT 2005).

Anchor (and chain) damages are also stressors. The size of the anchor, weather, and frequency of anchoring are directly related to the magnitude of the damages. In areas with chronic anchor damage to coral reefs, those effects can be mitigated by installing special mooring buoys, eliminating the need to anchor (Halas 1985, 1997). Multiple vessels anchoring in the same area for relief from adverse weather can also cause major damage (Davis 1977). In areas where large ships anchor on coral reefs, the damage can be significant; especially if the area is a designated anchorage or are frequently visited by large ships. Anchors from large vessels may weigh several tons and are usually attached to the ship by a heavy chain. Heavy chains can drag across the reef as the ship responds to any change in the wind, tides, and currents, thus resulting in dislodged and fractured corals for hundreds of meters (Smith 1988).

Fishing can also affect corals. Fishing is the most widespread exploitative activity on coral reefs and poses significant threats to the biodiversity and condition of marine ecosystems (Jennings and Polunin 1996). Fishing can influence the population structure of species by not only affecting their abundance, size, growth, and mortality, but can also modify species interactions such as competition and predation by altering structural complexity (Russ 1991, Auster and Langton 1999). Various ecological effects occur when traps and bottom trawls are deployed, but impacts may also occur when large numbers of anglers use hook-and-line gear to fish (Jennings and Lock 1996, Jones and Syms 1998). Derelict fishing gear can destroy benthic organisms and entangle both benthic and mobile fauna, (Donohue et al. 2001), especially elkhorn and staghorn corals, due to their branching morphology.

Additional anthropogenic impacts can be caused by SCUBA diving/snorkeling. The effects of divers/snorkelers are dose-dependent and difficult to quantify. Novice snorkelers/divers may stand on or kick elkhorn and staghorn coral causing breakage, although there are no studies that document the frequency of this damage. The Cayman Islands Department of the Environment studied diver impact at mooring buoy sites off of Grand Cayman Island and concluded that sites with visitation greater than 5,000 divers per year (14 divers a day) showed coral injuries. Sites that had 15,000 divers in a year experienced a major loss in coral diversity and cover, suggesting areas experiencing

heavy usage by divers/snorkelers may degrade coral reefs, and that limiting diver usage may enhance reef condition (*Acropora* BRT 2005).

Effects from sedimentation are yet another stressor of corals. Elkhorn and staghorn corals are sessile, light-dependent animals that cannot move away from stressful situations (Marshall and Orr 1931, Cortes 1985, Rogers 1990). Early reports noted that shallow-water tropical reef corals require highly transparent, sediment-free water (Wells 1957, Stoddart 1969). Other studies indicated that some corals can tolerate episodic turbidity (Hubbard and Pocock 1972) and, in some settings, chronic sediment loading. There is clear variation among coral species in the mechanisms and degree of sediment tolerance. The ability to remove sediments from the colony surface is an important behavioral characteristic influencing the growth, survival, and distribution of corals such as elkhorn and staghorn coral. Corals reject sediment from their tissues using gravity, beating cilia, and trapping the sediment in their mucus and discarding the sheet of mucus/sediment.

Elkhorn and staghorn coral appear to be particularly sensitive to sediment rain and shading effects from increased sediment regimes. Because these corals are almost entirely dependent upon sunlight for nourishment compared to massive, boulder-shaped species (Porter 1976, Lewis 1977), they are much more susceptible to increases in water turbidity and sedimentation than other species. Activities or other pollution that reduces long-term water clarity can reduce the ratio of production to respiration below unity. If this occurs, elkhorn and staghorn coral may not be able to compensate with an alternate food source such as zooplankton (Porter 1987).

Rogers (1983) investigated the effects of sedimentation on elkhorn and staghorn coral (among others). Elkhorn coral was the least tolerant of sediment exposure, as single applications of 200 mg/cm^2 to colonies caused coral tissue death as sediments accumulated on the flattened (horizontal) portions of the colonies. The widely spaced, cylindrical branches of staghorn coral facilitated passive sediment removal, making this species more tolerant of sediment accumulation. In another experiment, Rogers (1979) shaded a 20 m^2 area of reef and found that staghorn coral (the most abundant species in this area; 45% of the total living corals) was the first to respond to shading. Three weeks after shading was initiated, most colonies of staghorn corals had bleached. Shading was terminated after 5 weeks. After six weeks, the growth tips of the staghorn colonies were deteriorating or had been grazed away. A few branches recovered; most were dead and covered with algae. After seven weeks, there were more algae on the branches and further disintegration of branch tips.

Nutrients are delivered to coral reefs from both point source and non-point sources. Anthropogenic sources of nutrients include sewage, stormwater and agricultural runoff, river discharge, and groundwater. These source routes may also bring other stressors (e.g., sediments, turbidity, contaminants). As human activities in coastal regions have increased, nutrient discharge has increased as well. However, natural oceanographic sources like internal waves and upwelling also distribute nutrients on coral reefs, and these natural sources may account for more material (nitrogen and phosphorus) than

anthropogenic sources in highly developed areas such as the Florida Keys (Leichter et al. 2003). Notably, the reefs in the Florida Keys are exceptional in that they are located relatively far from land compared to most other reefs in the Caribbean (*Acropora* BRT 2005).

Coral reefs have been generally considered to be nutrient-limited systems, meaning that levels of accessible nitrogen and phosphorus limit the rates of plant growth. When nutrients levels are raised in such a system, plant growth can be expected to increase and this can yield imbalance and changes in community structure. Because corals contain small symbiotic algae within their tissues (zooxanthellae), nutrient enrichment can disrupt the symbiosis (Dubinsky and Stambler 1996), thereby affecting metabolic processes, coral growth, and reproductive success. For example, field experiments have shown decreased fecundity and fertilization success in Pacific *Acropora* spp. subjected to slight increases in nitrogen concentrations in the water column (or phosphorus for fertilization) (Ward and Harrison 2000, Harrison and Ward 2001). Increased growth rates of macroalgae (e.g., turfs and seaweeds) might be expected to yield higher abundances and overgrowth of reef substrates. Indeed, the widespread increase in seaweed abundance on coral reefs has been attributed to nutrient enrichment (e.g., Bell 1991, Lapointe 1997). However, seaweed abundance on coral reefs is also regulated by herbivores and recent experimental evidence suggests that seaweed proliferation is more directly linked with reduced herbivory (e.g., Diaz-Pulido and McCook 2003, McClanahan et al. 2003). The role of nutrient enrichment in reef community shifts remains controversial (Hughes et al. 1999, Lapointe 1999, McClanahan et al. 2004, Szmant 2002).

Competition is another threat posed to elkhorn and staghorn coral. Coral reefs are described as space-limited systems and thus it is believed that competition for space is an important structuring factor. Because of their fast growth rates and canopy-forming morphology, elkhorn and staghorn coral are known to be competitive dominants within coral communities, in terms of their ability to overgrow other stony and soft corals. However, other types of reef benthic organisms (i.e., macroalgae) have higher growth rates and, hence, expected greater competitive ability than elkhorn and staghorn coral. Since the 1980s, many Caribbean reef areas have undergone a shift in benthic community structure involving reduced cover by stony corals and increased coverage by macroalgae. This shift is generally attributed to the greater persistence of macroalgae under reduced grazing regimes due to human overexploitation of herbivorous fishes (Hughes 1994) and the regional mass mortality of the long-spined sea urchin in 1983-84. Impacts to water quality (principally nutrient input) are also believed to enhance macroalgal productivity (*Acropora* BRT 2005).

Aronson and Precht (2001) emphasize, however, that these Caribbean-wide changes in benthic assemblages were 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. In other words, the coral-dominated Caribbean reef system was resistant to reduced herbivory regimes for a period of time as long as corals maintained their occupation of space. However, when coral mortality occurred,

macroalgae were able to pre-empt that space (especially following the loss of grazing by *Diadema*) and were subsequently resistant to coral re-colonization (Hughes and Connell 1999). Thus, the described shifts have been persistent on a decadal scale. The noted exception is in areas where the grazing sea urchins (*Diadema antillarum*) have recently recovered and removed the macroalgal dominants, thereby clearing space to allow enhanced coral recruitment (Edmunds and Carpenter 2001).

Macroalgae are now the major space-occupiers on many Caribbean reefs. Their dominant occupation of reef surfaces impedes the recruitment of new corals (McCook et al. 2001) and hence, recovery by sexual recruits of elkhorn and staghorn coral. It is unlikely, however, that macroalgae have major impacts as direct competitors with healthy adult colonies. Other encrusting invertebrates may also pose a direct overgrowth threat to small colonies or bases of elkhorn and staghorn coral, but the extent of such interactions is not well documented (*Acropora* BRT 2005).

Genetic diversity is important for populations to adapt to environmental changes. Reduced genetic diversity often results when species undergo rapid decline like elkhorn and staghorn coral have in recent decades. This expectation is heightened when the decline is due to a potentially selective factor such as disease, in contrast to a less selective factor such as hurricane damage, which will likely cause disturbance independent of genotype. A species may preserve 90% or more of its original genetic diversity after a severe bottleneck if subsequent recovery is exponential. If the species remains at low densities for prolonged periods of time, genetic diversity may be significantly reduced. Thus, given the dominance of asexual reproduction, the rapid decline (largely from a selective factor), and the lack of rapid recovery that have characterized elkhorn and staghorn coral, it is plausible that these populations have suffered a loss of genetic diversity that could compromise their ability to adapt to future changes in environmental conditions (*Acropora* BRT 2005).

3.3.2 Elkhorn and Staghorn Coral Critical Habitat

On November 26, 2008, a final rule designating *Acropora* critical habitat was published in the Federal Register. Within the geographical area occupied by a listed species, critical habitat consists of specific areas on which are found those physical or biological features essential to the conservation of the species. The feature essential to the conservation of *Acropora* species (also known as essential feature) is substrate of suitable quality and availability, in water depths from the mean high water line to 30 m, to support successful larval settlement, recruitment, and reattachment of fragments. Substrate of suitable quality and availability means consolidated hardbottom or dead coral skeletons free from fleshy macroalgae or turf algae and sediment cover. Areas containing these features have been identified in four locations within the jurisdiction of the United States: Florida, Puerto Rico, St. Thomas/St. John, and St. Croix (Figure 3.3.2.1 A and B).

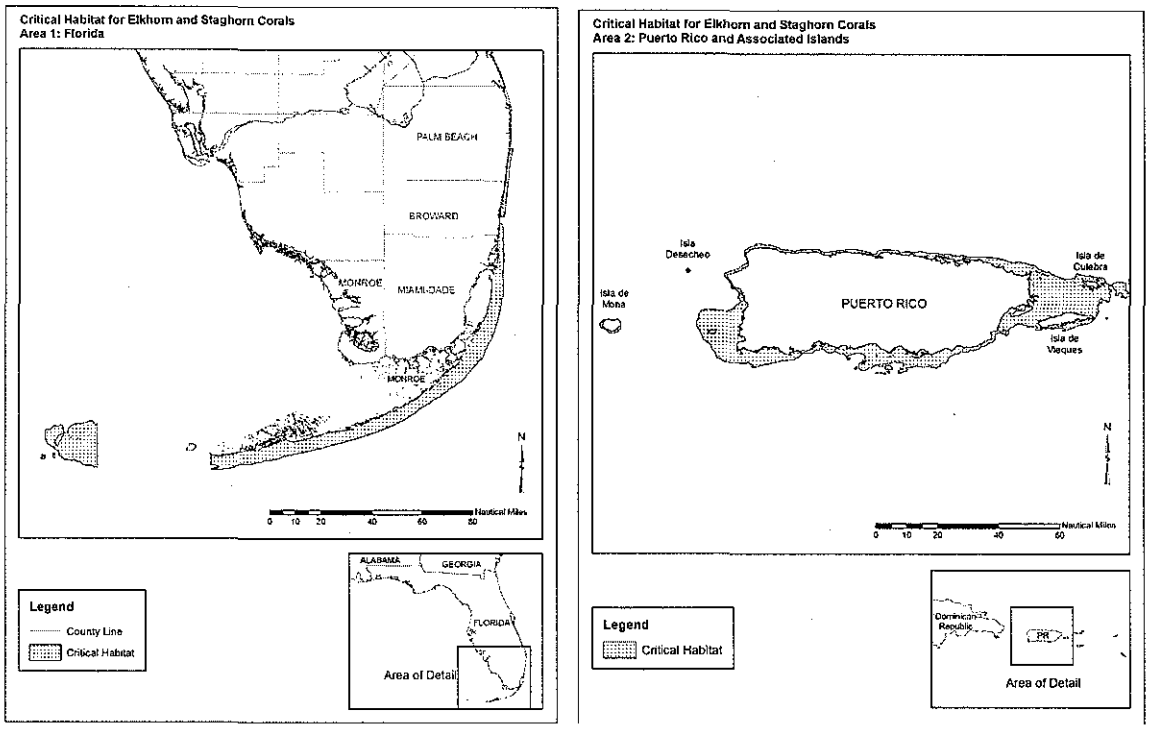


Figure 3.3.2.1 A. Designated Critical Habitat Area 1 and 2 for Elkhorn and Staghorn Corals

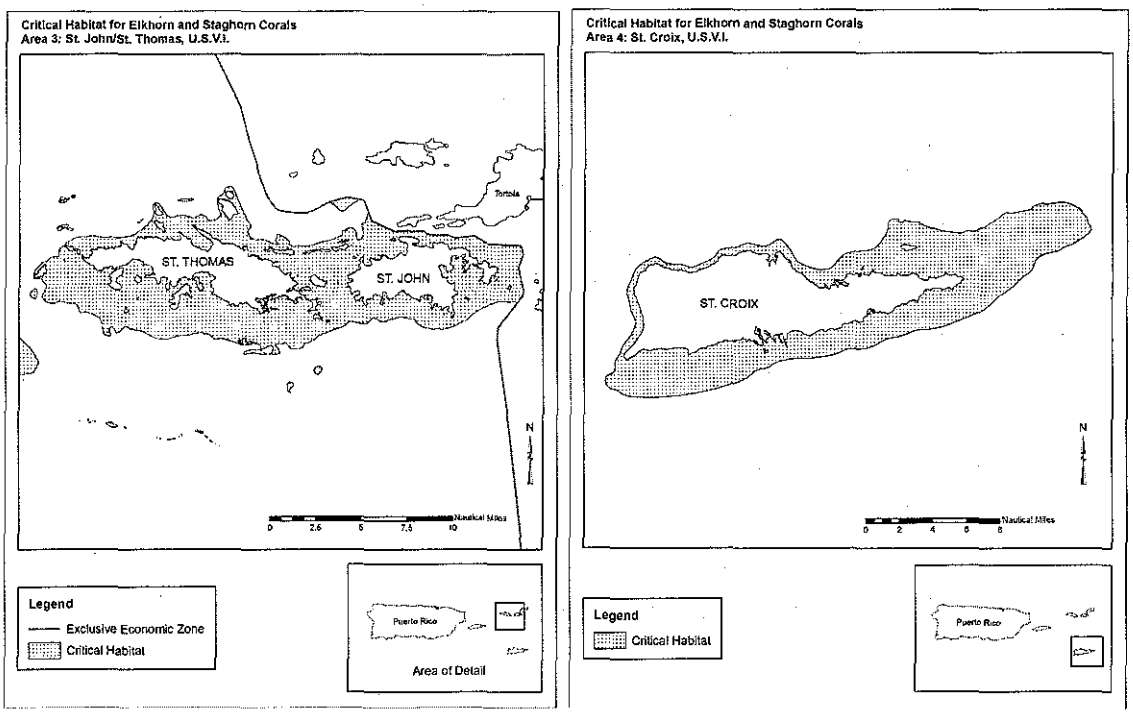


Figure 3.3.2.1 B. Designated Critical Habitat Area 3 and 4 for Elkhorn and Staghorn Corals

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 (Lang 2003).

Benthic maps produced by NOAA's National Ocean Service Biogeography Program for the USVI in 2000 included mapping of 298 km^2 of coral reef and hardbottom in the USVI (Kendall et al. 2001). However, the shift in benthic community structure from coral-dominated to algae-dominated that has been documented since the 1980s reduce the likelihood that larvae settlement or fragment re-attachment is successful (Hughes and Connell 1999). Sediment accumulation on suitable substrate also impedes sexual and asexual reproductive success by preempting available substrate and smothering coral recruits.

While algae, including crustose coralline algae and fleshy macroalgae, are natural components of healthy reef ecosystems, increases in the dominance of algae since the 1980s impedes coral recruitment. Impacts to water quality, in particular nutrient inputs, associated with coastal development and the harvest of macroalgal grazing herbivorous fish are thought to enhance the growth of fleshy macroalgae. 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); additionally, macroalgal species produce lipid-soluble allelopathic metabolites, which commonly cause bleaching, lowered photosynthetic efficiency, and often times, death of coral tissue (Rasher and Hay 2010).

Sediment from natural and anthropogenic sources can also affect reef distribution, structure, growth, and recruitment. Sediments can accumulate on dead and living corals and exposed hardbottom, 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 three sites in Puerto Rico, Torres (2001) found that low-density coral skeleton growth was correlated with increased resuspended 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 resuspension 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 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 two 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.

3.3.3 Green Sea Turtle

Green turtles are distributed circumglobally and can be found in the Pacific, Indian, and Atlantic Oceans as well as the Mediterranean Sea (NMFS and USFWS 91, Seminoff 2004, NMFS and USFWS 2007a). In 1978, the Atlantic population of the green sea turtle was listed as threatened under the ESA, except for the breeding populations in Florida and on the Pacific coast of Mexico, which were listed as endangered.

3.3.3.1 Pacific Ocean

Green turtles occur in the eastern, central, and western Pacific. Foraging areas are also found throughout the Pacific and along the southwestern United States coast (NMFS and

USFWS 1998a). Nesting is known to occur in the Hawaiian archipelago, American Samoa, Guam, and various other sites in the Pacific. The only major population (>2,000 nesting females) of green turtles in the western Pacific occurs in Australia and Malaysia, with smaller colonies throughout the area. Green turtles have generally been thought to be declining throughout the Pacific Ocean, with the exception of Hawaii, from a combination of overexploitation and habitat loss (Seminoff 2002). Indonesia has a widespread distribution of green turtles, but has experienced large declines over the past 50 years. Historically, green turtles were used in many areas of the Pacific for food. They were also commercially exploited and this, coupled with habitat degradation, led to their decline in the Pacific (NMFS and USFWS 1998a). Green turtles in the Pacific continue to be affected by poaching, habitat loss or degradation, fishing gear interactions, and fibropapillomatosis (NMFS and USFWS 1998a, NMFS 2004a).

Hawaiian green turtles are genetically distinct and geographically isolated, and the population appears to be increasing in size despite the prevalence of fibropapilloma and spirochidiasis (Aguirre et al. 1998 in Balazs and Chaloupka 2003). The East Island nesting beach in Hawaii is showing a 5.7% annual growth rate over 25 plus years (Chaloupka et al. 2007). In the Eastern Pacific, mitochondrial DNA analysis has indicated that there are three key nesting populations: Michoacán, Mexico; Galapagos Islands, Ecuador; and Islas Revillagigedos, Mexico (Dutton 2003). The number of nesting females per year exceeds 1,000 females at each site (NMFS and USFWS 2007a). However, historically, greater than 20,000 females per year are believed to have nested in Michoacán alone (Cliffton et al. 1982, NMFS and USFWS 2007a). Thus, the current number of nesting females is still far below what has historically occurred. There is also sporadic green turtle nesting along the Pacific coast of Costa Rica. At least a few of the non-Hawaiian nesting stocks in the Pacific have recently been found to be undergoing long-term increases. Datasets over 25 years in Chichi-jima, Japan; Heron Island, Australia; and Raine Island, Australia show increases (Chaloupka et al. 2007). These increases are thought to be the direct result of long-term conservation measures.

3.3.3.2 Indian Ocean

There are numerous nesting sites for green sea turtles in the Indian Ocean. One of the largest nesting sites for green sea turtles worldwide occurs on the beaches of Oman where an estimated 20,000 green sea turtles nest annually (Hirth 1997, Ferreira et al. 2003). Based on a review of the 32 index sites used to monitor green sea turtle nesting worldwide, Seminoff (2004) concluded that declines in green turtle nesting were evident for many of the Indian Ocean index sites. While several of these had not demonstrated further declines in the more recent past, only the Comoros Island index site in the western Indian Ocean showed evidence of increased nesting (Seminoff 2004).

3.3.3.3. Mediterranean Sea

There are four nesting concentrations of green sea turtles in the Mediterranean from which data are available, including those in Turkey, Cyprus, Israel/Palestine, and Syria. Currently, approximately 300-400 females nest each year among these four sites. On

average, Turkey is visited by the greatest number of nesting females annually (200-230), followed by Cyprus (120-130 nesting females) and Israel/Palestine (1-3 nesting females) (Broderick et al 2002, NMFS and USFWS 2007a). Information on the number of females nesting in Syria is not available; however, Rees et al (2005) indicate approximately 100 green sea turtle nests are recorded annually (NMFS and USFWS 2007a). The 5-year status review noted that no nesting trends could be estimated for the Mediterranean (NMFS and USFWS 2007a). However, a declining trend is apparent along the coast of Palestine/Israel, where 300-350 nests were deposited each year in the 1950s (Sella 1982) compared to a mean of 6 nests per year from 1993-2004 (Kuller 1999, Y. Levy, Israeli Sea Turtle Rescue Center, unpublished data). The discovery of green sea turtle nesting in Syria was important because the realization that such a major nesting concentration could have gone unnoticed until relatively recently (the Syria coast was surveyed in 1991, but nesting activity was attributed to loggerheads) bodes well for the ongoing speculation that the unsurveyed coast of Libya may also host substantial nesting.

3.3.3.4 Atlantic Ocean

Life History and Distribution

The estimated age at sexual maturity for green sea turtles is between 20-50 years (Balazs 1982, Frazer and Ehrhart 1985). Green sea turtle mating occurs in the waters off the nesting beaches. Each female deposits 1-7 clutches (usually 2-3) during the breeding season at 12-14 day intervals. Mean clutch size is highly variable among populations, but averages 110-115 eggs/nest. Females usually have 2 to 4 or more years between breeding seasons, whereas males may mate every year (Balazs 1983). After hatching, green sea turtles go through a post-hatchling pelagic stage during which they are associated with drift lines of algae and other debris. At approximately 20- to 25-cm carapace length, juveniles leave pelagic habitats and enter benthic foraging areas (Bjorndal 1997). Green sea turtles are primarily herbivorous, feeding on algae and sea grasses, but also occasionally consume jellyfish and sponges. The post-hatchling, pelagic-stage individuals are assumed to be omnivorous, but little data are available.

Green sea turtle foraging areas in the southeastern United States include any coastal shallow waters having macroalgae or seagrasses. This includes areas near mainland coastlines, islands, reefs, or shelves, as well as open-ocean surface waters, especially where advection from wind and currents concentrates pelagic organisms (Hirth 1997, NMFS and USFWS 1991). 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, Carr 1984), Florida Bay and the Florida Keys (Schroeder and Foley 1995), the Indian River Lagoon system, Florida (Ehrhart 1983), and the Atlantic Ocean off Florida from Brevard through Broward Counties (Wershoven and Wershoven 1992, Guseman and Ehrhart 1992). Adults of both sexes are presumed to migrate between nesting and foraging habitats along corridors adjacent to coastlines and reefs.

Some of the principal feeding pastures in the western Atlantic Ocean include the upper west coast of Florida and the northwestern coast of the Yucatán Peninsula. Additional important foraging areas in the western Atlantic include the Indian River Lagoon system (including Mosquito Lagoon) and nearshore wormrock reefs between Sebastian and Ft. Pierce Inlets in Florida, Florida Bay, the Culebra archipelago and other Puerto Rico coastal waters, the south coast of Cuba, the Caribbean coast of Panama, the Miskito Coast in Nicaragua, and scattered areas along Colombia and Brazil (Hirth 1997). The summer developmental habitat for green turtles also encompasses estuarine and coastal waters from North Carolina to as far north as Long Island Sound (Musick and Limpus 1997).

Population Dynamics and Status

Nest counts can also be used to estimate the number of reproductively mature females nesting annually. The 5-year status review for the species identified eight geographic areas considered to be primary sites for green sea turtle nesting in the Atlantic/Caribbean and reviewed the trend in nest count data for each (NMFS and USFWS 2007a). These sites include: (1) Yucatán Peninsula, Mexico; (2) Tortuguero, Costa Rica; (3) Aves Island, Venezuela; (4) Galibi Reserve, Suriname; (5) Isla Trindade, Brazil; (6) Ascension Island, United Kingdom; (7) Bioko Island, Equatorial Guinea; and (8) Bijagos Achipelago (Guinea-Bissau) (NMFS and USFWS 2007a). Nesting at all of these sites was considered to be stable or increasing with the exception of Bioko Island and the Bijagos Archipelago where the lack of sufficient data precluded a meaningful trend assessment for either site (NMFS and USFWS 2007a). Seminoff (2004) likewise reviewed green sea turtle nesting data for eight sites in the western, eastern, and central Atlantic, including all of the above with the exception that nesting in Florida was reviewed in place of Isla Trindade, Brazil. Seminoff (2004) concluded that all sites in the central and western Atlantic showed increased nesting with the exception of nesting at Aves Island, Venezuela, while both sites in the eastern Atlantic demonstrated decreased nesting. These sites are not inclusive of all green sea turtle nesting in the Atlantic. However, other sites are not believed to support nesting levels high enough that would change the overall status of the species in the Atlantic (NMFS and USFWS 2007a).

By far, the most important nesting concentration for green turtles in the western Atlantic is in Tortuguero, Costa Rica (NMFS and USFWS 2007a). Nesting in the area has increased considerably since the 1970s, and nest count data from 1999-2003 suggest nesting by 17,402-37,290 females per year (NMFS and USFWS 2007a). The number of females nesting per year on beaches in the Yucatán, Aves Island, Galibi Reserve, and Isla Trindade number in the hundreds to low thousands, depending on the site (NMFS and USFWS 2007a). The vast majority of green sea turtle nesting within the southeastern United States occurs in Florida (Meylan et al. 1995, Johnson and Ehrhart 1994). Green sea turtle nesting in Florida has been increasing since 1989 (Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute Index Nesting Beach Survey Database). Certain Florida nesting beaches have been designated index beaches. 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 turtle nesting shows biennial peaks in abundance with a generally positive trend during

the ten years of regular monitoring. This is perhaps due to increased protective legislation throughout the Caribbean (Meylan et al. 1995). An average of 5,039 green turtle nests were laid annually in Florida between 2001 and 2006, with a low of 581 in 2001 and a high of 9,644 in 2005 (NMFS and USFWS 2007a). Data from the index nesting beaches program in Florida substantiate the dramatic increase in nesting. In 2007, there were 9,455 green turtle nests found just on index nesting beaches, the highest since index beach monitoring began in 1989. The number fell back to 6,385 in 2008, further dropping under 3,000 in 2009, but that consecutive drop may be a temporary deviation from the normal biennial nesting cycle for green turtles, as 2010 saw an increase back to 8,426 nests on the index beaches (FWC Index Nesting Beach Survey Database). Occasional nesting has been documented along the Gulf coast of Florida, at southwest Florida beaches, as well as the beaches on the Florida Panhandle (Meylan et al. 1995). More recently, green turtle nesting occurred on Bald Head Island, North Carolina; just east of the mouth of the Cape Fear River; on Onslow Island; and on Cape Hatteras National Seashore. In 2010, a total of 18 nests were found in North Carolina, 6 nests in South Carolina, and 6 nests in Georgia (nesting databases maintained on www.seaturtle.org). Increased nesting has also been observed along the Atlantic coast of Florida, on beaches where only loggerhead nesting was observed in the past (Pritchard 1997). Recent modeling by Chaloupka et al. (2007) 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%, and the Tortuguero, Costa Rica, population growing at 4.9% annually.

There are no reliable estimates of the number of immature green sea turtles that inhabit coastal areas of the southeastern United States, where they come to forage. However, information on incidental captures of immature green sea turtles at the St. Lucie Power Plant in St. Lucie County, Florida, show that the annual number of immature green sea turtles captured has increased significantly over the years. Green sea turtle annual captures averaged 19 for 1977-1986, 178 for 1987-1996, and 262 for 1997-2001 (FPL 2002). In the five years from 2002-2006, green sea turtles captured averaged 333 per year, with a high of 427 and a low of 267 (FPL and Quantum Resources 2007). More recent unpublished data shows 101 captures in 2007, 299 in 2008, 38 in 2009 (power output was cut for part of that year) and 413 in 2010. Ehrhart et al. (2007) has also documented a significant increase in in-water abundance of green turtles in the Indian River Lagoon area. It is likely that immature green sea turtles foraging in the southeastern United States come from multiple genetic stocks; therefore, the status of immature green sea turtles in the southeastern United States might also be assessed from trends at all of the main regional nesting beaches, principally Florida, Yucatán, and Tortuguero.

Threats

The principal cause of past declines and extirpations of green sea turtle assemblages has been the overexploitation of green sea turtles for food and other products. Although intentional harvest 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. However, there are still significant and ongoing threats to green sea turtles from human-related causes in the United States. These threats include beach armoring, erosion control, artificial lighting, beach disturbance (e.g., driving on the beach), pollution, foraging habitat loss as a result of direct destruction by dredging, siltation, boat damage, other human activities, and interactions with fishing gear.

Additionally, the long-term impacts to sea turtles as a result of habitat impacts, prey loss, and subsurface oil particles and oil components broken down through physical, chemical, and biological processes are not known. Sea sampling coverage in the pelagic driftnet, pelagic longline, Southeast shrimp trawl, and summer flounder bottom trawl fisheries has recorded green turtle captures. There is also the increasing threat from green sea turtle fibropapillomatosis disease. Presently, this disease is cosmopolitan and has been found to affect large numbers of animals in some areas, including Hawaii and Florida (Herbst 1994, Jacobson 1990, Jacobson et al. 1991). Other sources of natural mortality include cold-stunning and biotoxin exposure. Cold-stunning is not considered a major source of mortality in most cases. As temperatures fall below 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 1989). 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, with hundreds found dead, or dying after they were gathered. Another cold-stunning event occurred in the western Gulf of Mexico in February 2011, resulting in approximately 1,500 green turtles found cold-stunned off Texas, and another 300 or so off Mexico, with an as yet undetermined number found dead or dying.

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).

Impacts on sea turtles currently cannot, for the most part, be predicted with any degree of certainty, however significant impacts to the hatchling sex ratios of green turtles may result (NMFS and USFWS 2007a). In marine turtles, sex is determined by temperature in the middle third of incubation, with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25°-35°C (Ackerman 1997). Increases in global temperature could potentially skew future sex ratios toward higher numbers of females (NMFS and USFWS 2007a). Green sea turtle hatchling size also appears to be influenced by incubation temperatures, with smaller hatchlings produced at higher temperatures (Glen et al. 2003).

The effects from increased temperatures may be exacerbated on developed nesting beaches where shoreline armoring and construction has denuded vegetation. 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 (Daniels et al. 1993, Fish et al. 2005, Baker et al. 2006). 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 increased 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., salinity, oceanic currents, dissolved oxygen levels, nutrient distribution, etc.) could influence the distribution and abundance of phytoplankton, zooplankton, submerged aquatic vegetation, forage fish, etc., which could ultimately affect the primary foraging areas of green sea turtles.

3.3.3.5 Summary of Status for Atlantic Green Sea Turtles

Green turtles range in the western Atlantic from Massachusetts to Argentina, including the Gulf of Mexico and the Caribbean Sea, but are considered rare in benthic areas north of Cape Hatteras (Wynne and Schwartz 1999). Green turtles face many of the anthropogenic threats for other sea turtles described herein. In addition, green turtles are also susceptible to fibropapillomatosis, which can result in death. In the continental United States, green turtle nesting occurs on the Atlantic coast of Florida (Ehrhart 1979). Recent population estimates for the western Atlantic area are not available. The pattern of green turtle nesting shows biennial peaks in abundance, with a generally positive trend during more than 20 years of regular monitoring since establishment of index beaches in Florida in 1989.

3.3.4 Leatherback Sea Turtle

The leatherback sea turtle was listed as endangered throughout its global range on June 2, 1970. Leatherbacks are widely distributed throughout the oceans of the world and are found in waters of the Atlantic, Pacific, and Indian Oceans (Ernst and Barbour 1972). Leatherback sea turtles are the largest living turtles and range farther than any other sea turtle species. The large size of adult leatherbacks and their tolerance to relatively low temperatures allows them to occur in northern waters such as off Labrador and in the Barents Sea (NMFS and USFWS 1995). Adult leatherbacks forage in temperate and subpolar regions from 71°N to 47°S latitude in all oceans and undergo extensive migrations to and from their tropical nesting beaches. In 1980, the leatherback population was estimated at approximately 115,000 adult females globally (Pritchard 1982); that number, however, is probably an overestimation as it was based on a particularly good nesting year in 1980 (Pritchard 1996). By 1995, the global population of adult females had declined to 34,500 (Spotila et al. 1996). Pritchard (1996) also called into question the population estimates from Spotila et al. (1996) and felt they may be somewhat low because it ended the modeling on data from a particularly bad nesting year

(1994) while excluding nesting data from 1995, which was a good nesting year. The most recent population estimate for leatherback sea turtles from just the North Atlantic breeding groups is a range of 34,000-90,000 adult individuals (20,000-56,000 adult females) (TEWG 2007).

3.3.4.1 Pacific Ocean

Based on published estimates of nesting female abundance, leatherback populations have collapsed or have been declining at all major Pacific basin nesting beaches for the last two decades (Spotila et al. 1996, NMFS and USFWS 1998b, Sarti et al. 2000, Spotila et al. 2000). For example, the nesting assemblage on Terengganu, Malaysia–which was one of the most significant nesting sites in the western Pacific Ocean–has declined severely from an estimated 3,103 females in 1968 to 2 nesting females in 1994 (Chan and Liew 1996). Nesting assemblages of leatherback turtles are in decline along the coasts of the Solomon Islands, a historically important nesting area (D. Broderick, pers. comm., in Dutton et al. 1999). In Fiji, Thailand, Australia, and Papua New Guinea (East Papua), leatherback turtles have only been known to nest in low densities and scattered colonies.

Only an Indonesian nesting assemblage has remained relatively abundant in the Pacific basin. The largest extant leatherback nesting assemblage in the Indo-Pacific lies on the north Vogelkop coast of Irian Jaya (West Papua), Indonesia, with over 3,000 nests recorded annually (Putrawidjaja 2000, Suárez et al. 2000). During the early-to-mid 1980s, the number of female leatherback turtles nesting on the two primary beaches of Irian Jaya appeared to be stable. More recently, this population has come under increasing threats that could cause this population to experience a collapse that is similar to what occurred at Terengganu, Malaysia. In 1999, for example, local Indonesian villagers started reporting dramatic declines in sea turtle populations near their villages (Suárez 1999). Unless hatchling and adult turtles on nesting beaches receive more protection, this population will continue to decline. Declines in nesting assemblages of leatherback turtles have been reported throughout the western Pacific region, with nesting assemblages well below abundance levels observed several decades ago (e.g., Suárez 1999).

In the western Pacific Ocean and South China Seas, leatherback turtles are captured, injured, or killed in numerous fisheries, including Japanese longline fisheries. The poaching of eggs, killing of nesting females, human encroachment on nesting beaches, beach erosion, and egg predation by animals also threaten leatherback turtles in the western Pacific.

In the eastern Pacific Ocean, nesting populations of leatherback turtles are declining along the Pacific coast of Mexico and Costa Rica. According to reports from the late 1970s and early 1980s, three beaches on the Pacific coast of Mexico supported as many as half of all leatherback turtle nests for the eastern Pacific. Since the early 1980s, the eastern Pacific Mexican population of adult female leatherback turtles has declined to slightly more than 200 individuals during 1998-1999 and 1999-2000 (Sarti et al. 2000). Spotila et al. (2000) reported the decline of the leatherback turtle population at Playa

Grande, Costa Rica, which had been the fourth largest nesting colony in the world. Between 1988 and 1999, the nesting colony declined from 1,367 to 117 female leatherback turtles. Based on their models, Spotila et al. (2000) estimated that the colony could fall to less than 50 females by 2003-2004. Leatherback turtles in the eastern Pacific Ocean are captured, injured, or killed in commercial and artisanal swordfish fisheries off Chile, Columbia, Ecuador, and Peru, and purse seine fisheries for tuna in the eastern tropical Pacific Ocean, and California/Oregon drift gillnet fisheries. Because of the limited data, we cannot provide high-certainty estimates of the number of leatherback turtles captured, injured, or killed through interactions with these fisheries. However, between 8-17 leatherback turtles were estimated to have died annually between 1990 and 2000 in interactions with the California/Oregon drift gillnet fishery; 500 leatherback turtles are estimated to die annually in Chilean and Peruvian fisheries; 200 leatherback turtles are estimated to die in direct harvests in Indonesia; and before 1992 the North Pacific driftnet fisheries for squid, tuna, and billfish captured an estimated 1,000 leatherback turtles each year, killing about 111 of them each year.

Although all causes of the declines in leatherback turtle colonies in the eastern Pacific have not been documented, Sarti et al. (1998) suggest that the declines result from egg poaching, adult and subadult mortalities incidental to high seas fisheries, and natural fluctuations due to changing environmental conditions. Some published reports support this suggestion. Sarti et al. (2000) reported that female leatherback turtles have been killed for meat on nesting beaches like Píedra de Tiacoyunque, Guerrero, Mexico. Eckert (1997) reported that swordfish gillnet fisheries in Peru and Chile contributed to the decline of leatherback turtles in the eastern Pacific. The decline in the nesting population at Mexiquillo, Mexico, occurred at the same time that effort doubled in the Chilean driftnet fishery. In response to these effects, the eastern Pacific population has continued to decline, leading some researchers to conclude that the leatherback is on the verge of extinction in the Pacific Ocean (e.g., Spotila et al. 1996, Spotila et al. 2000). The NMFS assessment of three nesting aggregations in its February 23, 2004, opinion supports this conclusion: If no action is taken to reverse their decline, leatherback sea turtles nesting in the Pacific Ocean either have high risks of extinction in a single human generation (for example, nesting aggregations at Terrenganu and Costa Rica) or they have a high risk of declining to levels where more precipitous declines become almost certain (e.g., Irian Jaya) (NMFS 2004a).

3.3.4.2 Atlantic Ocean

In the Atlantic Ocean, leatherbacks have been recorded as far north as Newfoundland, Canada, and Norway, and as far south as Uruguay, Argentina, and South Africa (NMFS 2001). Female leatherbacks nest from the southeastern United States to southern Brazil in the western Atlantic and from Mauritania to Angola in the eastern Atlantic. The most significant nesting beaches in the Atlantic, and perhaps in the world, are in French Guiana and Suriname (NMFS 2001). Previous genetic analyses of leatherbacks using only mitochondrial DNA (mtDNA) resulted in an earlier determination that within the Atlantic basin there are at least three genetically different nesting populations: the St. Croix nesting population (U.S. Virgin Islands), the mainland nesting Caribbean

population (Florida, Costa Rica, Suriname/French Guiana), and the Trinidad nesting population (Dutton et al. 1999). Further genetic analyses using microsatellite markers in nuclear DNA along with the mtDNA data and tagging data has resulted in Atlantic Ocean leatherbacks now being divided into seven groups or breeding populations: Florida, Northern Caribbean, Western Caribbean, Southern Caribbean/Guianas, West Africa, South Africa, and Brazil (TEWG 2007). When the hatchlings leave the nesting beaches, they move offshore but eventually utilize both coastal and pelagic waters. Very little is known about the pelagic habits of the hatchlings and juveniles, and they have not been documented to be associated with the *Sargassum* areas as are other species. Leatherbacks are deep divers, with recorded dives to depths in excess of 1,000 m (Eckert et al. 1989, Hays et al. 2004).

Life History and Distribution

Leatherbacks are a long-lived species, living for well over 30 years. It has been thought that they reach sexual maturity somewhat faster than other sea turtles (except Kemp's ridley), with an estimated range from 3-6 years (Rhodin 1985) to 13-14 years (Zug and Parham 1996). However, some recent research using sophisticated methods of analyzing leatherback ossicles has cast doubt on the previously accepted age to maturity figures, with leatherbacks in the western North Atlantic possibly not reaching sexual maturity until as late as 29 years of age (Avens and Goshe 2007). Continued research in this area is vitally important to understanding the life history of leatherbacks and has important implications in management of the species.

Female leatherbacks nest frequently (up to 10 nests per year) during a nesting season and nest about every 2-3 years. During each nesting, they produce 100 eggs or more in each clutch and, thus, can produce 700 eggs or more per nesting season (Schultz 1975). However, a significant portion (up to approximately 30%) of the eggs can be infertile. Thus, the actual proportion of eggs that can result in hatchlings is less than this seasonal estimate. The eggs incubate for 55-75 days before hatching. Based on a review of all sightings of leatherback sea turtles of <145 cm curved carapace length (ccl), Eckert (1999) found that leatherback juveniles remain in waters warmer than 26°C until they exceed 100 ccl.

Although leatherbacks are the most pelagic of the sea turtles, they enter coastal waters on an irregular basis to feed in areas where jellyfish are concentrated. Leatherback sea turtles feed primarily on cnidarians (medusae, siphonophores) and tunicates.

Evidence from tag returns and strandings in the western Atlantic suggests that adult leatherback sea turtles engage in routine migrations between boreal, temperate, and tropical waters (NMFS and USFWS 1992). A 1979 aerial survey of the outer continental shelf from Cape Hatteras, North Carolina, to Cape Sable, Nova Scotia, showed leatherbacks to be present throughout the area with the most numerous sightings made from the Gulf of Maine south to Long Island. Leatherbacks were sighted in waters where depths ranged from 1 to 4,151 m, but 84.4% of sightings were in areas where the water was less than 180 m deep (Shoop and Kenney 1992). Leatherbacks were sighted in waters of a similar sea surface temperature as loggerheads from 7°C to 27.2°C (Shoop

and Kenney 1992). However, this species appears to have a greater tolerance for colder waters because more leatherbacks were found at the lower temperatures (Shoop and Kenney 1992). This aerial survey estimated the in-water leatherback population from near Nova Scotia, Canada, to Cape Hatteras, North Carolina, at approximately 300-600 animals.

General differences in migration patterns and foraging grounds may occur between the seven nesting assemblages identified by the TEWG in 2007, but data is limited: Marked or satellite tracked turtles from the Florida and North Caribbean assemblages have been re-sighted off North America, in the Gulf of Mexico and along the Atlantic coast and a few have moved to western Africa, north of the equator. In contrast, Western Caribbean and Southern Caribbean/Guianas animals have been found more commonly in the eastern Atlantic, off Europe and northern Africa, as well as along the North American coast. There are no reports of marked animals from the Western North Atlantic assemblages entering the Mediterranean Sea or the South Atlantic Ocean, though in the case of the Mediterranean this may be due more to a lack of data rather than failure of Western North Atlantic turtles moving into the Sea. The tagging data coupled with the satellite telemetry data indicate that animals from the western North Atlantic nesting subpopulations use virtually the entire North Atlantic Ocean. In the South Atlantic Ocean, tracking and tag return data follow three primary patterns. Although telemetry data from the West African nesting assemblage showed that all but one remained on the shallow continental shelf, there clearly is movement to foraging areas of the south coast of Brazil and Argentina. There is also a small nesting aggregation of leatherbacks in Brazil, and while data are limited to a few satellite tracks, these turtles seem to remain in the southwest Atlantic foraging along the continental shelf margin as far south as Argentina. South African nesting turtles apparently forage primarily south, around the tip of the continent.

Population Dynamics and Status

The status of the Atlantic leatherback population has been less clear than the Pacific population. This uncertainty has been a result of inconsistent beach and aerial surveys, cycles of erosion and reformation of nesting beaches in the Guianas (representing the largest nesting area), a lesser degree of nest-site fidelity than occurs with the hardshell sea turtle species, and inconsistencies in the availability and analyses of data. However, recent coordinated efforts at data collection and analyses by the Leatherback Turtle Expert Working Group have helped to clarify the understanding of the Atlantic population status (TEWG 2007).

The Southern Caribbean/Guianas stock is the largest known Atlantic leatherback nesting aggregation (TEWG 2007). This area includes the Guianas (Guyana, Suriname, and French Guiana), Trinidad, Dominica, and Venezuela, with the vast majority of the nesting occurring in the Guianas and Trinidad. Past analyses had shown that the nesting aggregation in French Guiana had been declining at about 15% per year since 1987 (NMFS 2001). However, from 1979-1986, the number of nests was increasing at about 15% annually, which could mean that the current decline could be part of a nesting cycle that coincides with the erosion cycle of Guiana beaches described by Schultz (1975). It is

thought that the cycle of erosion and reformation of beaches has resulted in shifting nesting beaches throughout this region. This was supported by the increased nesting seen in Suriname, where leatherback nest numbers have shown large recent increases concurrent with declines elsewhere (with more than 10,000 nests per year since 1999 and a peak of 30,000 nests in 2001), and the long-term trend for the overall Suriname and French Guiana population was thought to possibly show an increase (Girondot 2002 in Hilterman and Goverse 2003). In the past, many sea turtle scientists have agreed that the Guianas (and some would include Trinidad) should be viewed as one population and that a synoptic evaluation of nesting at all beaches in the region is necessary to develop a true picture of population status (Reichart et al. 2001). Genetics studies have added support to this notion and have resulted in the designation of the Southern Caribbean/Guianas stock. Using both Bayesian modeling and regression analyses, the TEWG (2007) determined that the Southern Caribbean/Guianas stock had demonstrated a long-term, positive population growth rate (using nesting females as a proxy for population). This positive growth was seen within major nesting areas for the stock, including Trinidad, Guyana, and the combined beaches of Suriname and French Guiana (TEWG 2007).

The Western Caribbean stock includes nesting beaches from Honduras to Colombia. The most intense nesting in that area occurs in Costa Rica, Panama, and the Gulf of Uraba in Colombia (Duque et al. 2000). The Caribbean coast of Costa Rica and extending through Chiriquí Beach, Panama, represents the fourth largest known leatherback rookery in the world (Troëng et al. 2004). Examination of data from three index nesting beaches in the region (Tortuguero, Gandoca, and Pacuare in Costa Rica) using various Bayesian and regression analyses indicated that the nesting population likely was not growing over the 1995-2005 time series of available data (TEWG 2007). Other modeling of the nesting data for Tortuguero indicates a possible 67.8% decline between 1995 and 2006 (Troëng et al. 2007).

Nesting data for the Northern Caribbean stock is available from Puerto Rico, the U.S. Virgin Islands (St. Croix), and the British Virgin Islands (Tortola). In Puerto Rico, the primary nesting beaches are at Fajardo and on the island of Culebra. Nesting between 1978 and 2005 has ranged between 469-882 nests, and the population has been growing since 1978, with an overall annual growth rate of 1.1% (TEWG 2007). At the primary nesting beach on St. Croix, the Sandy Point National Wildlife Refuge, nesting has fluctuated from a few hundred nests to a high of 1,008 in 2001, and the average annual growth rate has been approximately 1.1% from 1986-2004 (TEWG 2007). Nesting in Tortola is limited, but has been increasing from 0-6 nests per year in the late 1980s to 35-65 per year in the 2000s, with an annual growth rate of approximately 1.2% between 1994 and 2004 (TEWG 2007).

The Florida nesting stock nests primarily along the east coast of Florida. This stock is of growing importance, with total nests between 800-900 per year in the 2000s following nesting totals fewer than 100 nests per year in the 1980s (Florida Fish and Wildlife Conservation Commission, unpublished data). Using data from the index nesting beach surveys, the TEWG (2007) estimated a significant annual nesting growth rate of 1.17% between 1989 and 2005. In 2007, a record 517 leatherback nests were observed on the

index beaches in Florida, with 265 in 2008, and then an increase to a new record of 615 nests in 2009, and a slight decline in 2010 back to 552 nests (FWC Index Nesting Beach database). This up-and-down pattern is thought to be a result of the cyclical nature of leatherback nesting, similar to the biennial cycle of green turtle nesting, but overall the trend shows rapid growth on Florida's east coast beaches.

The West African nesting stock of leatherbacks is a large, important, but mostly unstudied aggregation. Nesting occurs in various countries along Africa's Atlantic coast, but much of the nesting is undocumented and the data are inconsistent. However, it is known that Gabon has a very large amount of leatherback nesting, with at least 30,000 nests laid along its coast in one season (Fretey et al. 2007). Fretey et al. (2007) also provide detailed information about other known nesting beaches and survey efforts along the Atlantic African coast. Because of the lack of consistent effort and minimal available data, trend analyses were not possible for this stock (TEWG 2007).

Two other small but growing nesting stocks utilize the beaches of Brazil and South Africa. For the Brazilian stock, the TEWG (2007) analyzed the available data and determined that between 1988 and 2003 there was a positive annual average growth rate of 1.07% using regression analyses and 1.08% using Bayesian modeling. The South African stock has an annual average growth rate of 1.06 based on regression modeling and 1.04% using the Bayesian approach (TEWG 2007).

Estimates of total population size for Atlantic leatherbacks are difficult to ascertain due to the inconsistent nature of the available nesting data. In 1996, the entire Western Atlantic population was characterized as stable at best (Spotila et al. 1996), with numbers of nesting females reported to be on the order of 18,800. A subsequent analysis by Spotila (pers. comm.) indicated that by 2000, the Western Atlantic nesting population had decreased to about 15,000 nesting females. Spotila et al. (1996) estimated that the leatherback population for the entire Atlantic basin, including all nesting beaches in the Americas, the Caribbean, and West Africa, totaled approximately 27,600 nesting females, with an estimated range of 20,082-35,133. This is consistent with the estimate of 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) determined by the TEWG (2007).

Threats

Zug and Parham (1996) pointed out that the main threat to leatherback populations in the Atlantic is the combination of fishery-related mortality (especially entanglement in gear and drowning in trawls) and the intense egg harvesting on the main nesting beaches. Other important ongoing threats to the population include pollution, loss of nesting habitat, and boat strikes.

Of sea turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear. This susceptibility may be the result of their body type (large size, long pectoral flippers, and lack of a hard shell), their attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface, possibly their method of locomotion, and perhaps their attraction to the lightsticks used to attract target species in

longline fisheries. They are also susceptible to entanglement in gillnets and pot/trap lines (used in various fisheries) and capture in trawl gear (e.g., shrimp trawls).

Leatherbacks are exposed to pelagic longline fisheries in many areas of their range. Unlike loggerhead turtle interactions with longline gear, leatherback turtles do not usually ingest longline bait. Instead, leatherbacks are typically foul-hooked by longline gear (e.g., on the flipper or shoulder area) rather than getting mouth-hooked or swallowing the hook (NMFS 2001). A total of 24 nations, including the United States (accounting for 5-8% of the hooks fished), have fleets participating in pelagic longline fisheries in the area. Basin-wide, Lewison et al. (2004) estimated that 30,000-60,000 leatherback sea turtle captures occurred in Atlantic pelagic longline fisheries in the year 2000 alone (note that multiple captures of the same individual are known to occur, so the actual number of individuals captured may not be as high). Genetic studies performed within the Northeast Distant Fishery Experiment indicate that the leatherbacks captured in the Atlantic highly migratory species pelagic longline fishery were primarily from the French Guiana and Trinidad nesting stocks (over 95%); individuals from West African stocks were surprisingly absent (Roden et al. in press).

Leatherbacks are also susceptible to entanglement in the lines associated with trap/pot gear used in several fisheries. From 1990-2000, 92 entangled leatherbacks were reported from New York through Maine (Dwyer et al. 2002). Additional leatherbacks stranded wrapped in line of unknown origin or with evidence of a past entanglement (Dwyer et al. 2002). More recently, from 2002 to 2007, NMFS received 144 reports of entangled sea turtles in vertical lines from Maine to Virginia, with 96 events confirmed (verified by photo documentation or response by a trained responder; NMFS 2008a). Of the 96 confirmed events during this period, 87 events involved leatherbacks. NMFS identified the gear type and fishery for 42 of the 96 confirmed events, which included lobster, whelk, sea bass, crab, and research pot gear. A review of leatherback mortality documented by the STSSN in Massachusetts suggests that vessel strikes and entanglement in fixed gear (primarily lobster pots and whelk pots) are the principal sources of this mortality (Dwyer *et al.* 2002). Fixed gear fisheries in the Mid-Atlantic have also contributed to leatherback entanglements. For example, in North Carolina, two (2) leatherback sea turtles were reported entangled in a crab pot buoy inside Hatteras Inlet (NMFS SEFSC 2001). A third leatherback was reported entangled in a crab pot buoy in Pamlico Sound off of Ocracoke. This turtle was disentangled and released alive; however, lacerations on the front flippers from the lines were evident (NMFS SEFSC 2001). In the Southeast U.S., leatherbacks are vulnerable to entanglement in Florida's lobster pot and stone crab fisheries as documented on stranding forms. In the U.S. Virgin Islands, where one (1) of five (5) leatherback strandings from 1982 to 1997 were due to entanglement (Boulon 2000), leatherbacks have been observed with their flippers wrapped in the line of West Indian fish traps (R. Boulon, pers. comm. to Joanne Braun-McNeill, NMFS SEFSC 2001).

Leatherback interactions with the Southeast Atlantic shrimp fishery, which operates predominately from North Carolina through southeast Florida (NMFS 2002), have also been a common occurrence. Leatherbacks, which migrate north annually, are likely to

encounter shrimp trawls working in the coastal waters off the Atlantic coast from Cape Canaveral, Florida, to the Virginia/North Carolina border. Leatherbacks also interact with the Gulf of Mexico shrimp fishery. For many years, TEDs required for use in these fisheries were less effective at excluding leatherbacks than the smaller, hard-shelled turtle species. To address this problem, on February 21, 2003, the NMFS issued a final rule to amend the TED regulations, which required modifications to the size and design of TEDs to exclude leatherbacks and large and sexually mature loggerhead and green turtles. Mortality of leatherbacks in the shrimp fishery is now estimated at 54 turtles per year.

Other trawl fisheries are also known to interact with leatherback sea turtles. In October 2001, a Northeast Fisheries Science Center (NEFSC) observer documented the capture of a leatherback in a bottom otter trawl fishing for *Loligo* squid off Delaware; TEDs are not required in this fishery. The winter trawl flounder fishery, which did not come under the revised TED regulations, may also interact with leatherback sea turtles.

Gillnet fisheries operating in the nearshore waters of the mid-Atlantic states are also suspected of capturing, injuring, and/or killing leatherbacks when these fisheries and leatherbacks co-occur. Data collected by the NEFSC Fisheries Observer Program from 1994 through 1998 (excluding 1997) indicate that a total of 37 leatherbacks were incidentally captured (16 lethally) in drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 54-92%.

Poaching is not known to be a problem for nesting populations in the continental United States. However, in 2001 the NMFS Southeast Fisheries Science Center (SEFSC) noted that poaching of juveniles and adults was still occurring in the U.S. Virgin Islands and the Guianas. In all, four of the five strandings in St. Croix were the result of poaching (Boulon 2000). A few cases of fishermen poaching leatherbacks have been reported from Puerto Rico, but most of the poaching is on eggs.

Pollution may also represent a significant problem for leatherback sea turtles. Leatherback sea turtles may be more susceptible to marine debris ingestion than other species due to their pelagic existence and the tendency of floating debris to concentrate in convergence zones that adults and juveniles use for feeding areas and migratory routes (Lutcavage et al. 1997, Shoop and Kenney 1992). Investigations of the stomach contents of leatherback sea turtles revealed that a substantial percentage (44% of the 16 cases examined) contained plastic (Mrosovsky 1981). Along the coast of Peru, intestinal contents of 19 of 140 (13%) leatherback carcasses were found to contain plastic bags and film (Fritts 1982). The presence of plastic debris in the digestive tract suggests that leatherbacks might not be able to distinguish between prey items and plastic debris (Mrosovsky 1981). Balazs (1985) speculated that the object might resemble a food item by its shape, color, size, or even movement as it drifts about, and induce a feeding response in leatherbacks.

It is important to note that, like marine debris, fishing gear interactions and poaching are problems for leatherbacks throughout their range. Entanglements are common in Canadian waters where Goff and Lien (1988) reported that 14 of 20 leatherbacks

encountered off the coast of Newfoundland/Labrador were entangled in fishing gear including salmon net, herring net, gillnet, trawl line and crab pot line. Leatherbacks are reported captured by many other nations that participate in Atlantic pelagic longline fisheries, including Taipei, Brazil, Trinidad, Morocco, Cyprus, Venezuela, Korea, Mexico, Cuba, U.K., Bermuda, People's Republic of China, Grenada, Canada, Belize, France, and Ireland (see NMFS 2001 for a description of take records). Leatherbacks are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo et al. 1994, Graff 1995). Gillnets are one of the suspected causes of the decline in the leatherback sea turtle population in French Guiana (Chevalier et al. 1999), and gillnets targeting green and hawksbill turtles in the waters of coastal Nicaragua also incidentally catch leatherback turtles (Lageux et al. 1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six leatherbacks from 13,600 trawls (Marcano and Alio-M. 2000). A study by the Trinidad and Tobago's Institute for Marine Affairs (IMA) in 2002 confirmed that bycatch of leatherbacks is high in Trinidad. IMA estimated that more than 3,000 leatherbacks were captured incidental to gillnet fishing in the coastal waters of Trinidad in 2000. As much as one-half or more of the gravid turtles in Trinidad and Tobago waters may be killed (Lee Lum 2003), though many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NMFS 2001).

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).

Impacts on sea turtles currently cannot, for the most part, be predicted with any degree of certainty, however significant impacts to the hatchling sex ratios of leatherback turtles may result (NMFS and USFWS 2007b). In marine turtles, sex is determined by temperature in 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). However, unlike other sea turtles species, leatherbacks tend to select nest locations in the cooler tidal zone of beaches (Kamel and Mrosovsky 2004). This preference may help mitigate the effects from increased beach temperature (Kamel and Mrosovsky 2004).

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 (Daniels et al. 1993, Fish et al. 2005, Baker et al. 2006). 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 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).

Global climate change is likely to influence the distribution and abundance of jellyfish, the primary prey item of leatherbacks (NMFS and USFWS 2007b). Several studies have shown leatherback distribution is influenced by jellyfish abundance (e.g., Houghton et al. 2006, Witt et al. 2006, Witt et al. 2007). How these changes in jellyfish abundance and distribution will impact leatherback sea turtle foraging behavior and distribution is currently unclear (Witt et al. 2007).

3.3.4.3 Summary of Leatherback Status

In the Pacific Ocean, the abundance of leatherback turtle nesting individuals and colonies has declined dramatically over the past 10 to 20 years. Nesting colonies throughout the Eastern and Western Pacific Ocean have been reduced to a fraction of their former abundance by the combined effects of human activities that have reduced the number of nesting females. In addition, egg poaching has reduced the reproductive success of the remaining nesting females. At current rates of decline, leatherback turtles in the Pacific basin are a critically endangered species with a low probability of surviving and recovering in the wild.

In the Atlantic Ocean, our understanding of the status and trends of leatherback turtles is somewhat more confounded, although the overall trend appears to be stable to increasing. The data indicate increasing or stable nesting populations in all of the regions except West Africa (no long-term data are available) and the Western Caribbean (TEWG 2007). Some of the same factors that led to precipitous declines of leatherbacks in the Pacific also affect leatherbacks in the Atlantic (i.e., leatherbacks are captured and killed in many kinds of fishing gear and interact with fisheries in state, federal, and international waters). Poaching is also a problem that affects leatherbacks occurring in U.S. waters. Leatherbacks are also more susceptible to death or injury from ingesting marine debris than other turtle species.

3.3.5 Hawksbill Sea Turtle

The hawksbill turtle was listed as endangered under the precursor of the ESA on June 2, 1970, and is considered critically endangered by the International Union for the Conservation of Nature (IUCN). The hawksbill is a medium-sized sea turtle, with adults in the Caribbean ranging in size from approximately 62.5 to 94.0 cm straight carapace length. The species occurs in all ocean basins, although it is relatively rare in the Eastern Atlantic and Eastern Pacific, and absent from the Mediterranean Sea. Hawksbills are the most tropical sea turtle species, ranging from approximately 30°N latitude to 30°S latitude. They are closely associated with coral reefs and other hardbottom habitats, but they are also found in other habitats including inlets, bays, and coastal lagoons (NMFS and USFWS 1993). There are only five remaining regional nesting populations with more than 1,000 females nesting annually. These populations are in the Seychelles, Mexico, Indonesia, and two in Australia (Meylan and Donnelly 1999). There has been a global population decline of over 80% during the last three generations (105 years) (Meylan and Donnelly 1999).

3.3.5.1 Pacific Ocean

Anecdotal reports throughout the Pacific indicate the current Pacific hawksbill population is well below historical levels (NMFS 2004a). It is believed that this species is rapidly approaching extinction in the Pacific because of harvesting for its meat, shell, and eggs as well as destruction of nesting habitat (NMFS 2004a). Hawksbill sea turtles nest in the Hawaiian Islands as well as the islands and mainland of Southeast Asia, from China to Japan, and throughout the Philippines, Malaysia, Indonesia, Papua New Guinea, the Solomon Islands, and Australia (NMFS 2004a). However, along the eastern Pacific Rim where nesting was common in the 1930s, hawksbills are now rare or absent (Cliffton et al. 1982, NMFS 2004a).

3.3.5.2 Atlantic Ocean

In the western Atlantic, the largest hawksbill nesting population occurs on the Yucatán Peninsula of Mexico (Garduño-Andrade et al. 1999). With respect to the United States, nesting occurs in Puerto Rico, the U.S. Virgin Islands, and along the southeast coast of Florida. Nesting also occurs outside of the United States and its territories, in Antigua, Barbados, Costa Rica, Cuba, and Jamaica (Meylan 1999). Outside of the nesting areas, hawksbills have been seen off the U.S. Gulf of Mexico states and along the Eastern Seaboard as far north as Massachusetts, although sightings north of Florida are rare (NMFS and USFWS 1993).

Life History and Distribution
The best estimate of age at sexual maturity for hawksbill sea turtles is about 20-40 years (Chaloupka and Limpus 1997, Crouse 1999a). Reproductive females undertake periodic (usually non-annual) migrations to their natal beach to nest. Movements of reproductive males are less well known, but are presumed to involve migrations to their nesting beach or to courtship stations along the migratory corridor (Meylan 1999). Females nest an average of 3-5 times per season (Meylan and Donnelly 1999, Richardson et al. 1999). Clutch size is larger on average (up to 250 eggs) than that of other sea turtles (Hirth 1980). Reproductive females may exhibit a high degree of fidelity to their nest sites.

The life history of hawksbills consists of a pelagic stage that lasts from the time they leave the nesting beach as hatchlings until they are approximately 22-25 cm in straight carapace length (Meylan 1988, Meylan and Donnelly 1999), followed by residency in developmental habitats (foraging areas where juveniles reside and grow) in coastal waters. Adult foraging habitat, which may or may not overlap with developmental habitat, is typically coral reefs, although other hard-bottom communities and occasionally mangrove-fringed bays may be occupied. Hawksbills show fidelity to their foraging areas over several years (van Dam and Díez 1998).

The hawksbill's diet is highly specialized and consists primarily of sponges (Meylan 1988). Other food items, notably corallimorphs and zooanthids, have been documented to be important in some areas of the Caribbean (van Dam and Díez 1997, Mayor et al. 1998).

Population Dynamics and Status
Nesting within the southeastern United States and U.S. Caribbean is restricted to Puerto Rico (>650 nests/yr), the U.S. Virgin Islands (~400 nests/yr), and, rarely, Florida (0-4 nests/yr) (Eckert 1995, Meylan 1999, Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute's Statewide Nesting Beach Survey data 2002). At the two principal nesting beaches in the U.S. Caribbean where long-term monitoring has been carried out, populations appear to be increasing (Mona Island, Puerto Rico) or stable (Buck Island Reef National Monument, St. Croix, USVI) (Meylan 1999).

Threats
As with other sea turtle species, hawksbill sea turtles are affected by habitat loss, habitat degradation, marine pollution, marine debris, fishery interactions, and poaching in some parts of their range. There continues to be a black market for hawksbill shell products ("tortoiseshell"), which likely contributes to the harvest of this species.

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).

Impacts on sea turtles currently cannot, for the most part, be predicted with any degree of certainty, however significant impacts to the hatchling sex ratios of hawksbill sea turtles may result (NMFS and USFWS 2007d). In marine turtles, sex is determined by temperature in 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 a higher numbers of females (NMFS and USFWS 2007d).

The effects from increased temperatures may be exacerbated on developed nesting beaches where shoreline armoring and construction has denuded vegetation. 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 (Daniels et al. 1993, Fish et al. 2005, Baker et al. 2006). 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 increased 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., salinity, oceanic currents, dissolved oxygen levels, nutrient distribution, etc.) could influence the distribution and abundance of phytoplankton, zooplankton, submerged aquatic vegetation, coral reefs, forage fish, etc. Since hawksbills are typically associated with

coral reef ecosystems, increases in global temperatures leading to coral death (Sheppard 2006) could adversely affect the foraging habitats of this species.

3.3.5.3 Summary of Status for Hawksbill Sea Turtles

Worldwide, hawksbill sea turtle populations are declining. They face many of the same threats affecting other sea turtle species. In addition, there continues to be a commercial market for hawksbill shell products, despite protections afforded to the species under U.S. law and international conventions.

4.0 Environmental Baseline

By regulation, environmental baselines for opinions include the past and present impacts of all state, federal 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 that are contemporaneous with the consultation in process (50 CFR 402.02).

This section contains a description of the effects of past and ongoing human factors leading to the current status of the species, their habitat, and ecosystem, within the action area. The environmental baseline is a snapshot of the factors affecting the species and includes state, tribal, local, and private actions already affecting the species, or that will occur contemporaneously with the consultation in progress. Unrelated future federal actions affecting the same species that have completed consultation are also part of the environmental baseline, as are implemented and ongoing federal and other actions within the action area that may benefit listed species. The purpose of describing the environmental baseline in this manner is to provide context for the effects of the proposed action on the listed species.

4.1 Status of Elkhorn and Staghorn Coral and Designated Critical Habitat Within the Action Area

The action area comprises most of the U.S. portion of the range of staghorn and elkhorn corals. Three of the four geographic areas located off the United States where listed *Acropora* occur and where critical habitat is designated are within the action area. Within the action area itself, the majority of the area occurs within commonwealth and territorial waters.

In the USVI, coral reefs have changed dramatically over the past three decades based on data from long-term monitoring sites ranging in depth from sea level to 40 m (Rogers et al. 2008). Coral cover has declined on most, if not all, reefs in the USVI for which quantitative data are available. In the 1970s and 1980s, coral cover on some reefs was over 30% and often higher in shallow elkhorn coral zones (see Table 4.1) with macroalgae virtually absent (Rogers et al. 2008). Hurricanes David in 1979 and Hugo in 1989 caused declines in coral cover between 44 to 65% on Flat Cay Reef, St. Thomas (Hurricane David) and between 30 to 40% along transects and within quadrats in Great

Lameshur Bay, St. John (Hurricane Hugo). By the 1990s, many coral reef monitoring sites in the USVI had no more than 25% total coral cover and higher macroalgal cover than in the past. Shallow zones less than 6 m in depth have had their physical structure completely altered. Elkhorn is no longer the dominant species and only standing dead skeletons or broken dead fragments remain on many shallow reefs. Data from Thatch Cay, however, indicate that elkhorn coral colonies range in age from recent recruits to colonies up to 100 years old, assuming linear growth. The percent cover of elkhorn corals around Thatch Cay varies between 10 to 60%. Density of elkhorn corals is up to one colony per 2 m^2.

In addition to losses of elkhorn corals, monitoring data from around the USVI indicates that staghorn corals have virtually disappeared from the north side of Buck Island, St. Croix, and only a few localized areas off the southern reef contain staghorn corals, representing 2-3% of the coral cover in these areas (Rogers et al. 2002). Data from other monitoring studies around St. Croix indicate that staghorn corals are now rare around St. Croix and only isolated colonies, though numerous, exist around St. John (Rogers et al. 2002, Rogers et al. 2008). A survey in 2003 found that mixed stands of elkhorn and staghorn corals and their hybrid occur around Hans Lollick Island and Flat Cay, and Coculus Point, St. Thomas (percent cover of living *Acropora* between 11 to 13%); and Inner Brass Island, Botany Bay, and Caret Bay, St. Thomas (percent cover of living *Acropora* between 6 to 8%) (Rogers et al. 2008). However, surveys of fragments of staghorn from nearshore areas of St. Thomas and outlaying cays indicate that colonies of these corals were once much more abundant than the numbers recorded in the 2003 survey. Staghorn corals in the action area are typically only found in small, scattered colonies, except for one location off the coast of St. John (Saba Island) and the thickets around Thatch Cay. The percent cover of staghorn corals around Thatch Cay varies between 5 to 20%. Density of staghorn corals around Thatch Cay is up to one colony per 10 m^2.

Following the 2005 bleaching event, monitoring data indicate that total coral cover is now less than 12% on many reefs (Rogers et al. 2008), including a loss of 53% cover of elkhorn corals at long-term monitoring sites in the Buck Island Reef National Monument (Rothenberger et al. 2008). Coral mortality due to the 2005 bleaching event was more severe than at any time in the last 40 years of monitoring in USVI (Woody et al. 2008). Both elkhorn and staghorn corals suffered widespread mortality associated with the 2005 bleaching event and current monitoring data does not indicate significant recovery (Woody et al. 2008, Rothenberger et al. 2008). Overall, colonies of Atlantic *Acropora* have declined by up to 98% and live colonies were no longer present at many study sites in the USVI following the 2005-2006 bleaching event.

Table 4.1.1 Changes in Percent Cover of Elkhorn Coral Over Time In St. John and Buck Island, St. Croix (Aadapted from Bacle 2002 and Rogers et al. 2008).

Cover and Reef Area	Time Period	Hawksnest Bay, St. John	Windswept, St. John	Haulover Bay, St. John	Newfoundland Bay, St. John	Buck Island, St. Croix
% of Total Reef Area with 60%-80% Elkhorn Cover	1970s	11.5	9.5	26	16.1	50
	1980s	10.4	2.9*	7.1	9	N/A
	1999	5.6	6.4	0	0	0
%of Total Reef Area with <10% Elkhorn Cover	1970s	14.3	29	74	79.2	N/A
	1980s	2.3	24.4*	24.5	90.9	31
	1999	7.7	24.8	12.6	0.7	9.2
% Total Reef Area with Elkhorn	1970s	25.8	38.5	100	95.3	50
	1980s	12.7	27.3*	31.6	99.9	31
	1999	13.3	31.2	12.6	0.7	9.2

*Problem with shadow in 1983 photos leading to poor image quality that may have resulted in an overestimate of the decrease in cover in study by Bacle (2002).

NOAA's Center for Coastal Monitoring and Assessment (NCCOS) has collected biogeography data from all island areas (critical habitat units) since 2001. The NCCOS biogeography data are collected to spatially characterize and monitor the benthic habitat community through a random stratified survey (See Appendix 1, Figure 1 for site locations). Sites are randomly selected within each habitat stratum to ensure coverage of a wider study region and not just a particular reef or seagrass area. Sites are not revisited each year; rather, new sites are randomly selected each year within each stratum. The power in this type of monitoring program is the ability to incorporate spatial variability and characterize variable habitat stratum (i.e., a view of the big picture and overall trends). The NCCOS data does not include a category directly comparable to the critical habitat essential feature, like the University of the Virgin Islands (UVI) and the USVI DPNR data set, discussed further below.

In Appendix 1, we provide descriptive statistics and non-parametric Jonckheere-Terpstra (JT) tests of the NCCOS data by island and sampling period. For sites sampled in Puerto Rico, 'algae' cover was significantly increasing over the entire time series and had a percent cover of 56.6% in 2002, went to its lowest observed level (35.1%) in the summer of 2007, and its highest observed level (64.5 %) in the summer of 2009. Sites sampled in St. John indicate 'algae' cover was significantly increasing over the entire time series and had a percent cover of 47.9% in 2001, went to its lowest observed level (34.1%) in the summer of 2005, at its highest observed level (66.9%) in the summer of 2008, and was at 56.2% in the summer of 2010. Sites sampled in St. Croix indicate 'algae' cover had no significant trend over the entire time series and had a percent cover of 59.3% in 2003, was at its highest observed level (64.8%) in the spring of 2006, at its lowest observed level (35%) in the fall of 2009, and near its highest observed level (64.6%) again in the fall of 2010.

The NCCOS data do not provide an indication of the potential cause of the annual variability in the percent cover of macroalgae. These differences may be a result of natural seasonal variations in macroalgae growth (i.e., slower growth during the winter). Since the same sites are not sampled each year, the noted variability may simply reflect

localized differences in percent macroalgae cover between sites. However, the power of such sampling is its ability to provide some information on apparent larger scale (i.e., across islands) phenonmena. While these macroalgae percent cover show notable interannual changes, they also show a statistically significant increase in macroalgae over a 10-year period, indicating that while the percent cover in macroalgae can be highly variable from year to year and from site to site, all island areas appear to show a statistically significant increasing trend in macroalgae over time.

For sampled sites in Puerto Rico, 'coral' cover was significantly decreasing over the entire time series and had a percent cover of 6.1% in 2002, was at its highest observed level (9.9%) in the summer of 2002, at its lowest observed level (2.1%) in the winter of 2008, and at 4.6% in the summer of 2009. Sampled sites in St. John indicated 'coral' cover was significantly decreasing and had a percent cover of 7.9% in 2001 (the highest in the data set), was at its lowest observed level (2.4%) in the summer of 2009, and was at 3.0% in the summer of 2010. Sites sampled in St. Croix indicate 'coral' cover was significantly decreasing over the entire time series and had a percent cover of 2.8% in 2003, was at its highest observed level (3.5%) in the spring of 2004, at its lowest observed level (1.0%) in 2005, and at 2.3% in fall of 2010.

The USVI DPNR and the UVI have been monitoring the status of reefs in the USVI since 2001. As part of this monitoring, the benthic habitat community is monitored annually at thirty fixed sites. There are 17 sites around St. Thomas and St. John and 13 sites around St. Croix, and at each site six transects are sampled. The monitoring program is designed to follow trends at fixed locations on a fine spatial scale. Data collected are percent cover for all benthic habitat types. The UVI collects information on benthic coverage using highly-trained SCUBA divers who conduct visual and video transect surveys along both permanent and randomly-selected transects at each fixed site. Permanent transect locations were initially randomly selected. Long-term trends in benthic cover variables ('Coral,' 'Critical Habitat,' and 'Macroalgae') were examined using a modified version of a generalized mixed model regression (proc GLIMMIX) developed by the Florida Fish and Wildlife Research Institute in SAS v9.2 (Ruzicka et al. 2009). Four analyses were performed: 9-yr trends, 10-yr trends, post 2005 bleaching trends, and an analysis of just BIRNM. Appendix 1 contains detailed discussion of the methods and results of this analysis. In summary, output from the model showed:

- The majority of transects within sites showed no significant change in 'Coral' coverage across both 10-yr (26 of 35 transects) and 9-yr (37 of 52 transects) time series; however, for those sites where significant changes were detected at the 10-yr (9 of 35 transects) and 9-yr (15 of 52 transects) time series, each showed a declining trend. Over the both time series (i.e., 10 years and 9 years), no transects showed significant increases in coral coverage; however, in the post-2005 bleaching event subset, 2 of 52 transects had increasing coverage.

- The majority of sites showed significant declines in 'Critical Habitat' coverage across the time series for the 10-yr (5 of 6 transects) and 9-yr analysis (6 of 9 sites).

- The majority of transects within sites showed significant change in 'Macroalgae' coverage for the 10-yr time series (18 of 35 transects were increasing), while the majority of transects within sites showed no significant change for the 9-yr time series (33 of 52 transects); however, for transects where significant changes were detected for the 9-yr time series (19 of 52 transects), all were increasing. In the post-2005 bleaching event subset, the majority of transects (41 of 52 transects) showed no change in 'Macroalgae' coverage. Of those with significant changes, 1 transect was declining and 10 were increasing.

- For the post-bleaching subset, most sites had no significant trend for coral and only two sites detected some recovery; most sites had no significant trend in critical habitat, but those that did all had declining trends; the majority of sites had no significant trend in macroalgae, however, all of those that did except one, had increases in macroalgae.

- For Buck Island, St. Croix, 'Coral' coverage was significantly decreasing through time; 'Critical Habitat' was significantly decreasing at one of six transects; and macroalgae was significantly increasing at two of six transects.

In Puerto Rico, well-developed and dense thickets of staghorn coral were present through the late 1970s at many reefs surrounding the main island, and also the offshore islands of Mona, Vieques and Culebra (Almy and Carrión-Torres 1963, McKenzie and Benton 1972, Goenaga and Cintrón 1979, Boulon 1980). Later, in 1978-79 during an island-wide survey, staghorn coral was found on only 20% of those reefs (Bruckner 2002).

Unfortunately quantitative trend data sufficient for a case study to depict trends in staghorn abundance or distribution are not available from Puerto Rico. More recent description of the status of staghorn coral in Puerto Rico can be found in Bruckner (2002); a few other studies are summarized below:

- Prior to Hurricane David in 1979, 20 random 0.6 m^2 photoquadrats were selected from each of 10, 40-m long transects parallel to the depth contours across the reef (16.7 to 19.2 m depth). Based on analysis of point count data, staghorn coral had a mean total cover of 31.1% (range of 9.9 to 56.9%); after the storm, total cover of staghorn coral dropped to a mean of 0.90% (range of 0.02 to 2.7%) (Boulon unpubl. data).

- With the exception of a few reefs in the southwest and isolated offshore locations, the dense, high profile, monospecific thickets of both staghorn and elkhorn corals have disappeared from Puerto Rico coral reefs (Weil et al. unpublished data).

- In the summer of 2004, there was an epidemic outbreak of white pox disease at Los Corchos coral reef in Culebra, Puerto Rico. Prior to the outbreak, coral cover on the reef reached values of 80%. However, three weeks after Tropical Storm Jeanne, 80 to 90% of the staghorn coral colonies at permanent monitoring sites at

Los Corchos were already dead or dying; likely as a result of impacts from both disease and storm damage (Rogers, unpublished data).

As noted in Section 3.3.1, in 2005 a major bleaching event was recorded in the Caribbean that led to a coral mortality throughout the Caribbean. During the event, elkhorn corals bleached at a frequency of approximately 20% and staghorn corals at a frequency of approximately 75%, at 12 monitored locations in Puerto Rico (Garcia-Sais et al. 2008). Further, near Culebra Island, almost 100% of staghorn colonies suffered partial to complete mortality due to bleaching (Garcia-Sais et al. 2008). Similar to the situation in USVI, the bleaching event was followed by a white plague-like massive outbreak that caused mass mortality and resulted in a net 20-60% decline in living coral cover at surveyed reefs of the east coast within a period of approximately six months.

Coral cover in general has been experiencing a decline in Puerto Rico. Between 2001 and 2007 the percent total coral cover has declined and benthic algae have either increased or remained approximately stable at an average of greater than 40% with some reefs exceeding 70% (Garcia-Sais et al. 2008). The 2005 coral mortality event led to an increase in available substrate. Under optimal grazing regimes and with other anthropogenic factors under control, this newly available substrate would retain the essential features of *Acropora* critical habitat (hard, consolidated substrate, including attached, dead coral skeleton, devoid of turf or fleshy macroalgae for their larvae to settle). However, the NCCOS monitoring data indicate algal cover on Puerto Rican reefs declined in 2006, 2007, and some of 2008, before increasing to approximately 64% cover in 2008-2009.

Monaco et al. (2009) state the Virgin Islands Coral Reef Monument (VICRNM) boundaries were delineated based on legal parameters, not ecological criteria, and many of the area's most important biological resources remained unprotected. Consequently, baseline assessments revealed the reef and fish community were in better condition (i.e., higher coral cover, structural complexity, fish biomass, density and richness) in areas outside and adjacent to the VICRNM (Monaco et al. 2007, Boulon et al. 2008). As a result, ecosystem improvements within the reserve may take longer to detect since the area is more degraded than the areas adjacent to or outside VICRNM.

On-going monitoring at areas mid-shelf reef (MSR) habitats study sites inside and adjacent to VICRNM appears to support Monaco et al. (2009) conclusion. From 2003-2008, Monaco et al. (2009) reported that live scleractinian coral and rugosity at MSR sites were significantly greater outside the VICRNM, while gorgonian cover was greater inside. Throughout the study period, mean coral cover at MSR study sites showed a substantial decrease over time, particularly outside VICRNM where coral cover declined by 85% from 2003 to 2007, followed by a slight increase in 2008. A decrease of 78% was observed inside VICRNM during the same years, though the initial percent coral cover inside VICRNM was less than one third of that outside. Percent macroalgae cover showed an opposite trend over the same period, increasing 185% outside VICRNM through 2006 followed by a slight decrease in 2007. Inside the VICRNM, macroalgae decreased between 2003 and 2004 and then increased by 161% through 2008 (Monaco et

al. 2009). During the same period, reef fish species richness and density were significantly greater outside VICRNM; biomass was also greater outside, but the difference was not significant (Monaco et al. 2009).

From 2003-2008, Monaco et al. (2009) also evaluated results inside and outside VICRNM in Coral Bay (VICRNM-CB), a nearshore area which included patch reef and shallow back reef areas. Live scleractinian coral was almost twice as high outside VICRNM-CB, but values inside and outside VICRNM-CB were still relatively low (i.e. < 10%). Macroalgae cover was greater inside VICRNM-CB. Mean coral cover inside and outside VICR-CB appeared to decrease over time with local maxima in 2005 inside VICRNM-CB (8%) and 2004 outside (15%) (Monaco et al. 2009). In both datasets, coral cover decreased by over 60% in years subsequent to the maxima and remained low through the end of the study (Monaco et al. 2009). From 2003-2008, macroalgae cover increased inside the VICRNM-CB study sites in a pattern similar to mid-shelf reef habitats study sites, but the pattern was more variable outside VICRNM-CB. Highest macroalgal cover was observed in 2008 both inside and outside VICRNM-CB. Average species richness, fish density and biomass estimates were not different among samples inside and outside of VICRNM-CB.

Pittman et al. (2008) assessed ecosystem changes in the expanded Buck Island Reef National Monument (BIRNM) with a particular interest in evaluating differences inside and outside or changes in fish assemblages from 2003-2006. They found biomass over colonized hardbottom habitat was significantly higher inside BIRNM for all herbivorous fish. Biomass of parrotfishes and blue tang were also significantly higher inside BIRNM. Parrotfish biomass increased steadily inside the expanded BIRNM from 2003-2006, while parrotfish biomass decreased in areas outside the BIRNM. *Diadema* appear to be almost absent inside BIRNM and were found at low densities (approx. 8/100 m^2) on colonized hardbottom outside BIRNM. Pittman et al. (2008) also compared fish densities from sites within 500 m of Buck Island (that is, entirely within the old boundaries) against data from 1979. Mean densities of four out of the five parrotfish species reported increased between the study periods, ranging from +85% to +600%. The statistical significance of the over-time trends was not reported. Pittman et al. also reported differences in coral and macroalgae cover on colonized hardbottom inside and outside BIRNM. Mean coral cover inside BIRNM was 4.9% vs. 2.4% outside, and mean macroalgae cover was 14.5% inside vs. 15.7% outside. These differences were significant at $p<0.05$.

4.2 Factors Affecting Elkhorn and Staghorn Coral and Designated Critical Habitat Within the Action Area

Numerous activities funded, authorized, or carried out by federal agencies have been identified as threats and may affect elkhorn and staghorn corals in the action area. Although many regulations exist to protect corals, including elkhorn and staghorn corals, many of the activities identified as threats still adversely affect the species. Poor boating and anchoring practices, poor snorkeling and diving techniques, and destructive fishing practices cause abrasion and breakage to elkhorn and staghorn corals. Nutrients,

contaminants, and sediment from point and non-point sources cause direct mortality and the breakdown of normal physiological processes. Fishing alters ecosystem processes and feedback mechanisms, decreasing the resilience of *Acropora* colonies and communities. Additionally, these stressors create an unfavorable environment for reproduction and growth.

- NMFS develops fishery management plans and fishery regulations that govern fishing activities that may physically interact with the species and its habitat or that may alter ecosystem functions and the resilience of these systems through the removal of keystone species (e.g., herbivorous fish).
- The National Park Service (NPS) regulates activities that are conducted in shallow-water coral reef areas including collection of coral, alteration of the seabed, discharges, boating, anchoring, fishing, recreational SCUBA diving, snorkeling, and scientific research within the boundaries of their designated parks and monuments. The Department of the Interior, including NPS, along with NOAA and the U.S. Environmental Protection Agency (EPA), also conduct research activities using federal research vessels as part of coral reef monitoring activities within the Coral Reef Monument and National Park around St. John, as well as other areas in USVI.
- The USVI Department of Planning and Natural Resources (DPNR) issues permits for the collection of corals and other marine species for scientific and educational purposes. Through the ESA section 4(d) rule promulgated by NMFS to protect elkhorn and staghorn corals, NMFS recognized that the DPNR permit process is consistent with ESA section 10 permit requirements, and an additional permit from NMFS is not required for scientific research and enhancement activities involving either species of listed corals.
- The U.S. Coast Guard, through its Marine Event Program, permits events such as sailing tournaments, speed-boat races, fishing tournaments, fireworks displays, and swimming competitions around St. Thomas and St. John that could result in accidental groundings or accidental spills of petroleum products in areas containing listed corals and their designated critical habitat.
- The COE and the EPA permit discharges to surface waters through shoreline and riparian disturbances. These disturbances (whether in the riverine, estuarine, marine, or floodplain environment) result in discharges to surface waters that may retard or prevent the reproduction, settlement, reattachment, and development of listed corals (e.g., land development and run-off, and dredging and disposal activities, result in direct deposition of sediment on corals, shading, and lost substrate for fragment reattachment or larval settlement).
    - The U.S. Army Corps of Engineers (COE) authorizes and carries out construction and dredge-and-fill activities that may result in direct mortality or injury of elkhorn or staghorn coral through direct deposition of sediment on corals or shading, or eliminate or impede access to habitat for coral larvae or fragments.
    - EPA, through the DPNR Division of Environmental Protection (DEP), regulates the discharge of pollutants, such as oil, toxic chemicals, radioactivity, carcinogens, mutagens, teratogens, or organic nutrient-laden

water, including sewage water, from point sources into the waters of the United States. Elevated discharge levels may cause direct mortality, reduced fitness, or habitat destruction/modification.

- The EPA, through the DPNR DEP, authorizes the discharge of stormwater to surface waters as part of construction projects. This discharge may result in the release of pollutants carried in runoff that can lead to direct mortality, reduced fitness, or habitat destruction/modification.

4.2.1 Fisheries

Several types of fishing gears that have been used within the action area for decades have the potential to adversely affect elkhorn and staghorn corals. Longline gear has been documented as interacting with corals, though no data specific to listed corals in the action area are available and this gear type is generally used in waters greater than 30 meters. Available information suggests hooks and lines from other types of hook-and-line gear can become entangled in reefs, resulting in breakage and abrasion of corals but impacts are expected to be minor. Traps have been found to be the most damaging. A study of the trap fishery in the USVI found that, while most fishers deployed traps in seagrass or algae, sand, or coral rubble, a few fishers targeted corals (Sheridan et al. 2006), resulting in habitat impacts. However, less than 20% of the traps set in depths less than 30 m were in contact with hard or soft corals or sponges and damage was mainly at a scale less than the total trap footprint (Sheridan et al. 2005). Lost traps and illegal traps were found to result in greater impact to coral habitat because they cause continuous habitat damage until they degrade.

The only fisheries in the action area that may adversely affect elkhorn and staghorn corals and their critical habitat target reef fish and spiny lobster. Fisheries targeting these species in the Caribbean EEZ are managed under CFMC FMPs. HMS fisheries managed by the HMS Management Division targeting pelagic species also occur in the action area. With the exception of fisheries for HMS which occur in the EEZ and beyond the action area on the high seas, much of the fishing effort occurs in commonwealth/territorial waters.

Reef Fish Fisheries

Section 2.1.2 provides an overview of the history of the federal Caribbean reef fish fishery and its management by NMFS under the CRFFMP. The current federal reef fish fishery and its proposed continued authorization is the subject of this consultation and so is not part of the environmental baseline. However, the past and current effects of reef fish fishing in territorial and commonwealth waters are part of the environmental baseline. For decades, participants in the U.S. Caribbean reef fish fishery (both in the EEZ and USVI and Puerto Rico waters) have targeted species of all trophic levels. Amendments implemented in the past have altered gear construction and usage, closed seasons and areas, changed fishery management units, implemented size limits, placed prohibitions on the use of some fishing practices, and the harvest of some species (e.g., Nassau and goliath grouper). However, the FMP has never set catch quotas. There were no numerical estimates of the appropriate level of harvest of species or species groups

managed by the CRFFMP until the implementation of the Sustainable Fisheries Act (SFA) Amendment in 2005.

Spiny Lobster Fishery
The spiny lobster fishery in waters around Puerto Rico and the USVI occurs with pots and traps, and hand-harvest. Due to the predominance of fishable habitat in state waters, it is assumed 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). In order of gear landing predominance, SCUBA divers, fish traps, and lobster traps catch the majority of commercial lobster in Puerto Rico from 1998-2001 (NMFS 2005a). Wooden traps are used primarily for lobster, but wire mesh traps are used for both lobster and reef fish. Biodegradable panels are required for all traps, including those in commonwealth/territorial waters, but fishermen have not always followed this regulation. In the USVI, divers catch the majority (i.e., approximately 85%) of spiny lobster (Tobias 2001).

4.2.2 Federal Vessel Operations

Potential sources of adverse effects from federal vessel operations in the action area include operations of the USCG, the 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. At the present time, however, they present the potential for some level of interaction.

4.2.3 ESA Permits

Regulations developed under the ESA allow for the issuance of permits authorizing take of certain ESA-listed species for the purposes of scientific research under section 10(a)(1)(A) of the ESA. In addition, section 6 of the ESA allows NMFS to enter into cooperative agreements with states to assist in recovery actions of listed species. Prior to issuance of these permits, the proposal must be reviewed for compliance with section 7 of the ESA. The section 4(d) rule promulgated by NMFS to establish "take" prohibitions for listed elkhorn and staghorn corals enables permits issued by the Commonwealth/Territory to be used in lieu of section 10 permits issued by NMFS for activities meant to promote scientific research on Atlantic *Acropora* and enhancement of the species.

4.2.4 Vessel Traffic

Commercial and recreational vessel traffic can adversely affect listed corals through propeller scarring, propeller wash, and accidental groundings. In 1988, anchor damage from the 440-foot cruise ship *Wind Spirit* destroyed a 300-yd^2 area of coral reef in Francis Bay, St. John, in one of the worst documented cases of anchor impacts within the Virgin Islands National Park (Drayton et al. 2004, Allen 1992). Monitoring of the site over time showed that the reef did not recover fully from the damage, despite some work to repair damage to corals in the anchor scar (Allen 1992). Based on information from

the NOAA Restoration Center and NOAA's ResponseLink, reports of accidental groundings are becoming more common in the USVI and Puerto Rico. These groundings are often on reefs colonized by listed corals due to the shallow depth requirements of these species, in particular elkhorn corals.

Private vessels in the action area participating in marine events, in particular events involving motorized vessels, are an additional threat to listed corals. NMFS and the USCG have completed a section 7 consultation for the Caribbean Marine Event Program for all annually occurring marine events in the USVI and Puerto Rico. As a result of this consultation, the USCG now includes permit conditions the event participants must follow to avoid and minimize potential impacts of marine events to listed corals and their habitat. However, there are numerous other commercial and recreational vessels that transit, anchor, and moor in the action area. In addition, 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. Through the section 7 process for dock, port, and marine construction activities under the jurisdiction of the COE, NMFS will attempt to establish conservation measures to ensure that the construction and operation of these facilities avoids or minimizes adverse effects to listed species.

4.2.5 Coastal Development and Dredging

Anthropogenic sources of marine pollution, while difficult to attribute to a specific federal, state, local, or private action, may indirectly affect corals in the action area. Sources of pollutants in the action area include atmospheric loading of pollutants such as PCBs, storm water runoff from coastal towns, and runoff into rivers that empty into bays and groundwater. The pathological effects of oil spills have been documented in laboratory and field studies of corals, although effects depend on the species' tolerance and level of exposure (Hoff 2001). Following a crude oil spill in Las Minas Bay, Panama, short-term mortality to corals was documented, and long-term sublethal impacts to reproduction and growth were documented to last five years or more (Guzman et al. 1994).

Nutrient loading from land-based sources, such as coastal communities and agricultural operations, is known to stimulate plankton blooms in closed or semi-closed estuarine systems. An example is the large area of the Louisiana continental shelf with seasonally depleted oxygen levels (< 2 mg/l), caused by eutrophication from both point and non-point sources. Most aquatic species cannot survive at such low oxygen levels and these areas are known as "dead zones." Water quality monitoring studies by DEP in waters around the USVI indicate that surface waters are affected by increasing point and non-point source pollution from failing septic systems, discharges from vessels, failure of best management practices 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 DEP reports that water quality in most areas continues to decline based on monitoring data from around the USVI. This is indicated

by the designation of 69 areas as impaired in 2006 versus 50 in 2005 (Rothenberger et al. 2008).

From 2001 to 2005, eighteen coral reef monitoring locations representing a range of reef types were established around St. Thomas and St. John along an onshore to offshore gradient, and in areas of previously unstudied reef systems. The results showed that sedimentation rates were dramatically higher on nearshore coral reefs with sedimentation rates for the clay and silt fraction over 5-fold greater than for midshelf reefs and over 45-fold greater than for shelf edge reefs (Smith et al. 2008). The clay and silt fraction is an indicator of terrigenous material content of the sediments. The total combined prevalence of mortality and disease was significantly greater, by approximately 50%, in nearshore coral reefs than in the offshore coral reef complexes (Smith et al. 2008). A 4-year monitoring study of the reef complex in Caret Bay before, during, and after construction showed a significant difference among transects and depths with sedimentation rates closely tracking rainfall during the early months of construction (Nemeth and Sladek Nowlis 2001). Reef sites exposed to average sedimentation rates between 10 to 14 mg per cm^2 per day showed a 38% increase in the number of coral colonies experiencing bleaching compared to reef sites exposed to sedimentation rates between 4 to 8 mg per cm^2 per day (Nemeth and Sladek Nowlis 2001), which corresponds to findings of other studies in the USVI regarding coral tolerance thresholds for sedimentation which result in declines in coral health (Rogers et al. 1984, Rogers et al. 2008). The tolerance threshold suggested by this and other studies of 10 mg/cm^2 per day was exceeded during 6 of the 13 sample periods, indicating chronic sediment stress approximately 50% of the time (Nemeth and Sladek Nowlis 2001). Bleaching of corals was strongly correlated to sedimentation rate, indicating that bleaching can be a response to sediment stress.

Estimates were made of the peak rate of discharge and the average runoff volume for storms of various magnitudes for Hawksnest, Fish, and Reef Bays, St. John, and terrigenous sediment content of nearshore reefs was analyzed to determine the effects of runoff transporting sediment to reefs. Hubbard et al. (1987) found that, as storm intensity increases, peak discharge and average rates of runoff volume also increase dramatically. In particular, the rainfall increase between the 2- and 10-year frequency storm was 60%, while it was only 39% between the 10- and 50-year frequency storm (Hubbard et al. 1987). This is important because, while severe storms can have a substantial impact on individual reefs, the general reef distribution around St. John appears more related to events with a low periodicity (Hubbard et al. 1987). Estimates of runoff found that areas of highest runoff intensity are shoreline segments draining areas that funnel a high percentage of the runoff from a watershed, and that adjacent nearshore areas do not demonstrate reef development. Shoreline segments with less than 20 cubic feet per second of runoff intensity were more likely to contain better-developed nearshore reefs (Hubbard et al. 1987). More intense development and construction result in higher runoff intensities and corresponding inputs of high levels of sediment to nearshore areas, affecting reef development and condition. Construction in the Hawksnest watershed from 1980 to 1981 resulted in higher levels of runoff and increases in sediment and

corresponding declines in coral growth rates up to several years following development (Hubbard et al. 1987).

Measurement of erosion rates on St. John (between 1998 and 2001) indicated that unpaved roads contribute up to four orders of magnitude more sediment than undisturbed hillsides basins (Rogers et al. 2008; Ramos-Sharron and MacDonald 2007b). Runoff coefficients for St. John are approximately an order of magnitude greater than those for undisturbed tropical hillslopes in eastern Puerto Rico. This difference is due to lower canopy and litter interception rates, and the higher potential for overland flow due to the lower vegetative cover and higher proportion of rocks on the soil surface. Roads increase the frequency and magnitude of surface runoff by creating a compacted low-permeability surface and affect runoff by intercepting subsurface flows and disrupting natural drainage patterns (Ramos-Sharron and MacDonald 2007b). Using data on erosion rates, runoff, and sediment production rates, Ramos-Sharron and MacDonald (2007a; 2007b) created basin-scale erosion models. Ramos-Sharron and MacDonald (2007a) calculated that, under undisturbed conditions, the amount of sediment delivered to the marine environment (found to be from streambank erosion) ranges from 0.02-0.07 mg per hectare per year, which is similar to the measured values of 0.01 to 0.08 mg per hectare per year for undisturbed zero and first-order basins on St. John. In basins with unpaved roads, predicted sediment yields represented a 300 to 900% increase in sediment yields relative to undisturbed basins (Ramos-Sharron and MacDonald 2007a). Hillslope gullies that form through the concentration of road drainage result in another source of sediment and conduit for delivering sediment and runoff (Ramos-Sharron and MacDonald 2007b). Storm events larger than 1 cm rainfall accounted for just less than half of the total precipitation, but produced about 90% of the total runoff and sediment yield for study areas in St. John (Ramos-Sharron and MacDonald 2007b).

Sediment core data from nearshore wetland and coastal embayments around St. Thomas and St. John show that, over the past 15 to 25 years, sedimentation rates have increased from 1 to 2 orders of magnitude (Rogers et al. 2008). Nearshore waters adjacent to highly developed watersheds typically average over 10 mg per cm^2 per day, in contrast to nearshore waters adjacent to less developed watersheds, which average less than 4 mg per cm^2 per day, and offshore reefs that are not associated with a land mass that average less than 0.5 mg per cm^2 per day (Rogers et al. 2008; Smith et al. 2008). During a severe rain event, sediment load can increase to >30 mg per cm^2 per day (Rogers et al. 2008). Over the rainy season, sediment flux rates from developed watersheds were up to 360 mg per cm^2 per day (Gray et al. 2008). Developed watersheds around St. John were also found to increase the input of terrestrially derived sediments by fifteen times, in comparison to undeveloped watersheds, and mean organic matter flux rates by up to 10 times. This means that carbonate was not as common in the sediments around nearshore reefs (Gray et al. 2008), which could have significant effects on coral growth rates, as terrigenous sediments do not contain the minerals corals need to build their calcium carbonate skeletons.

The construction and maintenance of federal navigation channels may also adversely affect elkhorn and staghorn coral. The COE also permits dredge-and-fill activities that

can directly affect elkhorn and staghorn corals via fragmentation/breakage or abrasion. They can also affect the species by physically altering or removing benthic habitat suitable for elkhorn and staghorn coral colonization. Dredge-and-fill activities may also cause increases in sedimentation that may cause shading, deposition of sediment on elkhorn and staghorn coral, and/or loss of substrate for fragment reattachment or larval settlement. However, as of September 2011, no formal or informal consultations have been conducted on proposed dredging projects in the action area that may affect elkhorn and staghorn corals.

4.2.6 Natural Disturbance

Hurricanes and large coastal storms can also significantly harm elkhorn and staghorn corals. Due to their branching morphologies, they are especially susceptible to breakage from extreme wave action and storm surges. Historically, large storms potentially resulted in asexual reproductive events, if the fragments encountered suitable substrate, attached, and grew into new colonies. However, recently, the amount of suitable substrate has been significantly reduced; therefore, many fragments created by storms die. Hurricanes are also sometimes beneficial, if they do not result in heavy storm surge, during years with high sea surface temperatures, as they lower 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).

4.2.7 Conservation and Recovery Actions Benefiting Listed Corals

NMFS has implemented a section 4(d) rule to establish "take" prohibitions for listed corals. The CFMC has established regulations prohibiting the use of bottom-tending fishing gear in some seasonally and permanently closed fishing areas containing coral reefs in federal waters of the Exclusive Economic Zone (EEZ). The USVI and Puerto Rico are moving toward similar regulations for both commercial and recreational fishers, 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 on going through the Southeast Regional Office. NOAA Restoration Center has also established a contract position in Puerto Rico to participate in grounding response and carry out restoration activities. The summaries below discuss these measures in more detail.

A draft recovery plan for elkhorn and staghorn corals is in preparation. A recovery team consisting of fishers, scientists, managers, and agency personnel from Florida, Puerto

Rico, and USVI, and federal representatives has been convened and is working towards creating a draft recovery plan for public review based upon the latest and best available information.

4.2.8 Regulations Reducing Threats to Listed Corals

On October 29, 2008, NMFS published a final section 4(d) rule extending the section 9 "take" prohibitions to 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. The critical habitat designation requires that all actions with a federal nexus ensure that the adverse modification of critical habitat will not occur as part of a section 7 consultation with NMFS for the action.

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

NMFS also conducts essential fish habitat (EFH) consultations. Through EFH consultations, NMFS works with federal agencies to conserve and enhance EFH, which includes corals. Consultation is required when a federal agency authorizes, funds, or undertakes an action that may adversely affect EFH. The federal agency must provide NMFS with an assessment of the action's impacts to EFH, and NMFS provides the federal agency with EFH Conservation Recommendations to avoid, minimize, mitigate, or otherwise offset those adverse effects. Federal agencies must provide a detailed written explanation to NMFS describing which recommendations, if any, it has not adopted.

The Territory regulates activities that occur in terrestrial and marine habitats of the USVI. The V.I. Code prohibits the taking, possession, injury, harassment, sale, offering for sale, etc. of any indigenous species, including live rock (V.I. Code Title 12 and the Indigenous and Endangered Species Act of 1990). Permits can be issued by the Commissioner of DPNR for the collection and transport of indigenous or endangered species for commercial, private, educational, or scientific use. Special permits may also be issued to collectors from recognized museums, research organizations, scientific organizations, and for recovery and propagation activities. 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.

The Coral and Reef Associated Plants and Invertebrates FMP of the CFMC prohibits the extraction, possession, and transportation of any coral, alive or dead, from federal waters unless a permit is obtained from the Government of the USVI or NMFS. Similarly, the CFMC (50 CFR Part 622) prohibits the use of chemicals, plants, or plant-derived toxins and explosives to harvest coral. The CFMC also prohibits the use of pots/traps, gill/trammel nets, and bottom longlines on coral or hard bottom year-round in existing seasonally closed areas in the EEZ and Grammanik Bank in the EEZ (50 CFR Part 622). Amendment 1 to the FMP for Corals and Reef Associated Plants and Invertebrates established a marine conservation district (MCD) in federal waters southwest of St. Thomas where fishing for any species and anchoring by fishing vessels is prohibited year-round.

The National Park Service (NPS) is responsible for the management of the Virgin Islands National Park (VINP), the Virgin Islands Coral Reef National Monument (VICRNM), and BIRNM. Each of these special areas varies in the extent of protection provided.

The VINP covers slightly more than half of the island of St. John and almost nine square miles of the waters surrounding St. John (VINP 2004). In 1956, legislation was passed by Congress to authorize the establishment of the Virgin Islands National Park. This act limited the potential acreage of the Park to 9,485 acres on St. John (an island 12,500 acres) and 15 acres on St. Thomas. In 1962, the boundary of the Virgin Islands National Park was expanded to include 5,650 acres of offshore areas (waters and submerged lands). Friedlander and Beets (2008) note "Although commercial fishing is prohibited, VINP's enabling legislation allows for the "customary uses of or access" to park waters for fishing, including the use of traps of "conventional Virgin Islands design". When the park was first established, fishers usually set only a few, smaller traps but with the advent of outboard motors, line hauls, and larger fiberglass boats, fishermen now fish further offshore with a larger number of traps (Beets 1997, Garrison et al. 1998)."

The VICRNM covers 12,708 acres of federally owned submerged lands and was established in 2001 to expand protection of marine resources located near the VINP in St. John. VICRNM was created by Presidential Proclamation, calling for the area to be administered as a no-take marine reserve to protect reefs from further degradation. The new VICRNM was established largely to restore fish populations and protect reef ecosystems (NPS 2004). The area is entirely no-take except for fishing for bait fish at Hurricane Hole, St. John, and rod-and-line fishing for blue runner via permit at VICRNM. Anchoring is not permitted. Regulations to implement the new Monument took effect in April 2003.

The BIRNM "is located on the northeastern shelf of St. Croix, in the U.S. Virgin Islands and encompasses an uninhabited island of approximately 712,000 m^2 and the surrounding mosaic of coral reefs, seagrasses and sand patches. The BIRNM was originally designated by the U.S. Department of Interior in 1961 according to Presidential Proclamation 3443, in order to preserve the island and the surrounding submerged lands which at that time included "one of the finest marine gardens in the Caribbean Sea". The

original monument encompassed 880 acres (approximately 3.56 km^2) and marine areas were zoned to form a protected "Marine Garden" (259 acres or approximately 1.04 km^2), which included extensive stands of elkhorn coral and an area with restricted fishing (445 acres or approximately 1.8 km^2). The "Marine Garden" was one of the first "no-take" marine reserves in U.S. waters and in the Caribbean region. The boundaries were slightly modified in 1975 (Presidential Proclamation 4346), but it was not until 2001 that the monument was greatly expanded to 19,015 acres (approximately 77 km^2) under Presidential Proclamation 7392. At that time, new regulations were enacted making the entire monument a no-take and "restricted anchoring" zone. The BIRNM expansion was the first substantial no-take area established for the island of St. Croix and it now protects about 7.4% of the St. Croix shelf area. The expansion resulted in a 10-fold increase in protection of shallow water (<30 m) hardbottom and sand habitat types and a seven-fold increase for seagrasses when compared with the 1961 Monument (Kendall et al. 2004a). In January 2003, BIRNM became contiguous with the East End Marine Park (EEMP) through the adjoining of the southern boundary of BIRNM and northern boundary of EEMP. However, over 80% of EEMP is open to fishing including an area that extends between the southern boundary of BIRNM and the EEMP no-take coastal lagoon zone" (Pittman et al. 2008). The enlarged BIRNM now incorporates components of the marine ecosystem, which have been impacted by fishing of finfish, conch and lobster. At the time of their study, Pittman et al. (2008) reported that the expanded area was being illegally fished using hand and rod, fish traps, gill or trammels nets, and longlines in the deeper portions of the BIRNM, but that law enforcement patrols had been active since 2003 and compliance was increasing.

4.2.9 Other Listed Coral Conservation Efforts

Damage Assessment and Restoration
The final section 4(d) rule for elkhorn and staghorn corals allows restoration activities, defined in the rule as "the methods and processes used to provide aid to injured individuals," when they are conducted by certain federal, state, territorial, or local government agency personnel or their designees acting under existing legal authority.

Outreach and Education
The NOAA Coral Reef Conservation Program, through its internal grants, external grants, and grants to the Territory, Commonwealth, and the CFMC, has providing 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 elkhorn and staghorn corals, the 4(d) rule, and the designation of critical habitat. These materials have been circulated to constituents during education and outreach activities and public meetings, and as part of other section 7 consultations, and are readily available on the website: http://sero.nmfs.noaa.gov/pr/esa/acropora.htm.

4.2.10 Summary and Synthesis of Environmental Baseline for Listed Corals

In summary, several factors are presently adversely affecting elkhorn and staghorn corals and their critical habitat in the action area. Those factors that are ongoing and are expected to occur contemporaneously with the proposed action include:

- Disease outbreaks;
- Temperature-induced bleaching events;
- Major storm events;
- Upland and coastal activities that will continue to degrade water quality and decrease water clarity necessary for coral growth;
- Dredge-and-fill activities;
- Harvest of herbivorous fishes
- Interactions with some fishing gears;
- Vessel traffic that will continue to result in abrasion and breakage due to accidental groundings and poor anchoring techniques; and
- Poor diving and snorkeling techniques that will continue to abrade and break corals.

4.3 Status of Listed Sea Turtles Within the Action Area

The three species of sea turtles that occur in the action area are all highly migratory. Individual animals will likely migrate out of the action area to other parts of the North Atlantic Ocean. Therefore, the status of these species of sea turtles in the action area, as well as the threats to these species, are best reflected in their range-wide statuses and supported by the species accounts in Section 3 (Status of Species).

Within the action area, hawksbill sea turtles nest year-round in Puerto Rico and adults and hatchlings can be found in waters around the island throughout the year. Mona Island supports one of the largest nesting populations of hawksbills in Puerto Rico. For this reason, the USFWS designated the beaches of Mona Island as critical habitat for hawksbill sea turtles under the ESA and NMFS designated the waters up to three nautical miles around Mona and Monito Islands as critical habitat. A recent survey of the marine communities of Bajo de Sico (García-Sais et al., 2007) found the area to harbor a large number of adult hawksbill turtles that utilized the reef promontories as foraging and refuge habitat.

Adults and juvenile green sea turtles can often be seen in the U.S. Virgin Islands and Puerto Rico, particularly in the area of Culebra. Green sea turtle nests are reported in Manatí, Loíza, Fajardo, Ceiba, Naguabo, Culebra, Vieques, Caja de Muertos, Mona Island, and larger cays within the La Cordillera Reefs Natural Reserve off the coast of Fajardo based on annual DNER nesting surveys.

Leatherback sea turtles occur within the action area primarily during their nesting season. The Sandy Point National Wildlife Refuge in St. Croix, USVI, supports one of the largest nesting population of nesting leatherback sea turtles in the world. The greatest

concentration of leatherback nests in Puerto Rico is in the area of San Miguel, Luquillo/Fajardo. Adults and juveniles of leatherback sea turtles are observed in the area of Bajo de Sico, in particular during their nesting peak in April-August.

4.4 Factors Affecting Listed Sea Turtles Within the Action Area

Numerous activities carried out by federal, state, and private citizens in the action area were noted as adversely affecting listed coral species. Many of the same activities are identified as threats and affecting the survival and recovery of ESA-listed sea turtle species. Past and present threats in the action area 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 area 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 area 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 here. 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.

4.4.1 Sea Turtle Harvest and Poaching

Boulon (2000) summarized historic sea turtle harvest in the action area and poaching information through 1999. During the nineteenth century, the sea turtle fishery in Puerto Rico and USVI was subsistence only. Much of the harvest occurred on the beaches adjacent to the action area. For example, leatherbacks were slaughtered on their nesting beaches for their oil and their eggs were harvested for food. A substantial green turtle fishery for food and export to Europe also existed historically.

According to The Convention on International Trade in Endanggered Species of Wild Fauna and Florida (CITES), In 1999, TRAFFIC North America provided a report of the past and current status of exploitation and trade of sea turtles in the Caribbean, focusing on northern Caribbean Islands, including Puerto Rico and USVI. CITES summarized that information, which captures the status in its web publication, titled "Satus of Trade in Hawksbill Turtles) (http://www.cites.org/eng/prog/hbt/bg/trade_status.shtml).

The following excerpt from their summary describes status of trade in Puerto Rico through 1999:

> Despite protective legislation in Puerto Rico and the USVI, there has remained an unquantifiable but persistent demand for sea turtle products, especially meat and eggs. While most of the take is likely to be

opportunistic or incidental, some people fish specifically for turtles by hand, using nets, and harpoons (C. Diez, H. Horta, M. Rivera, pers. comms, 1999, Female turtles are sometimes killed on nesting beaches for their eggs and meat, and nests are poached on several beaches around the island.

Although there are no complete data on take of sea turtles in Puerto Rico, one estimate is of 1000 to 1,500 adult, sub-adult, and juvenile sea turtles poached annually for personal consumption or sale to restaurants, markets, and trusted individuals (S. Rice, in litt., 2000.). Eggs of all species are collected for food (C. Diez, pers. comm., 1999; M. Rivera, pers. comm., 1999). Researchers in Humacao reported that all nests would be likely to be lost to poachers without consistent beach patrols (L. Montero-Acevedo, pers. comm., 1999).

In 1999, there was a steady sea turtle black market in Puerto Rico that was largely organized to fill existing orders from specific buyers (C. Carreon, C. Diez, L. Santiago and M. Rivera, pers. comms, 1999). Prices for meat and eggs reported to TRAFFIC ranged from USD0.50-5.00/egg and from USD6-15/ 0.45 kg for meat for all species (C. Diez, H. Horta and L. Santiago, pers. comms, 1999). While meat and eggs have not been seen on restaurant menus since the 1980s, they have recently been offered to specific customers in certain establishments in coastal areas, including Humacao, Fajardo, Lajas, Puerto Real, Joyuda, and Mayaguez, where the price for a sea turtle steak is approximately USD25 (C. Carreon, C. Diez, S. Rice and M. Rivera, pers. comms, 1999).

The following excerpt describes status of trade in the USVI through 1999:

Despite protective legislation in the USVI, there has been a persistent demand for sea turtle meat and eggs. More poaching occurs on St. Croix than on the other islands, partly owing to a more depressed economy and a larger Hispanic population, which retains its cultural practices of eating eggs and turtles (Eckert, 1989; M. Evans, C. Farchette and Z. Hillis-Starr, pers. comms, 2000). Many of the poachers are in search of an immediate source of cash, and have often been charged with other violations such as assault and dealing in weapons and narcotics. Eggs are sold locally for USD1 each (M. Evans, pers. comm., 2000).

According to Z. Hillis-Starr (pers. comm., 2000), the only instance of egg poaching on Buck Island Reef National Monument in the last 13 years occurred when a tanker from the Dominican Republic grounded near the monument during Hurricane Hugo in 1989 - five Hawksbill nests were excavated when patrols were temporarily discontinued.

Poaching has been a traditional threat to the sea turtle nests on the East End beaches on St. Croix (Mackay and Rebholz, 1996), with up to one-third of the nests having been dug up or probed on Jack's Bay in the early 1990s. Green and Hawksbill Turtle eggs and adults are frequently taken on Sandy Point after seasonal all-night patrols for Leatherbacks Dermochelys coriacea have ended (Boulon et al., 1996). Manchenil Bay and Ha'penny Bay beaches are also subject to moderate rates of poaching, owing to the fact that they are not protected and are easily accessed (J. Rebholz, pers. comm., 2000). Hawksbill shells have been found on beaches with the meat removed, which appears to indicate that the animals were taken for the meat only (Z. Hillis-Starr, B. Kojis and A. Mackay, pers. comms, 2000).

An apparently new trend involves influential residents on St. Croix who have begun placing orders for turtle eggs as a demonstration of their personal status and authority (M. Evans and C. Farchette, pers. comms, 2000). In the last seven years, prices for turtle eggs have risen from USD15 to USD55 per dozen eggs (M. Evans, pers. comm., 2000).

Fishers in Frenchtown, on St. Thomas, have traditionally harvested turtles and eggs and periodically poach them today in the USVI (B. Kojis, pers. comm., 2000), but more often travel to the BVI to take turtles (M. Evans, pers. comm., 2000). Fishers from the BVI have also been known to take turtles from St. John.

While poaching of eggs, juveniles, and adult sea turtles in the action area has declined dramatically, isolated cases do occur, thus it is still a threat in the action area. Recent poaching is documented via strandings in both USVI and Puerto Rico. Insufficient enforcement capabilities of protective laws in nonprotected areas greatly limit the effectiveness of legal protection.

4.4.2 Fisheries

Fisheries in the action area managed via CFMC FMPs that may affect sea turtles are the reef fish, and spiny lobster fisheries. Offshore pelagic species, managed by the NMFS, Office of Sustainable Fisheries, HMS Management Division are also occur in the EEZ portion of the action area and beyond the action area on the high seas, and may affect sea turtles.

Threatened and endangered sea turtles are adversely affected by several types of fishing gears that have been used within the action area 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 all fisheries within the action area for which there is a federal FMP, impacts have been evaluated under section 7. However, the majority of fishable waters that are within the action area occur within commonwealth and territorial waters and are not subject to FMPs and section 7 consultation.

Atlantic Highly Migratory Species (HMS) Caribbean Swordfish and Tuna Fisheries
Atlantic pelagic longline fisheries targeting swordfish and tuna are also known to incidentally capture large numbers of loggerhead and leatherback sea turtles. Over the past two decades, NMFS has conducted numerous consultations on Atlantic pelagic longline fisheries, some of which required RPAs to avoid jeopardy of loggerhead and/or leatherback sea turtles. The estimated historical total number of loggerhead and leatherback sea turtles caught between 1992-2002 (all geographic areas) is 10,034 loggerhead and 9,302 leatherback sea turtles of which 81 and 121 were estimated to be dead when brought to the vessel (NMFS 2004b). This does not account for post-release mortalities, which historically was likely substantial. NMFS most recently reinitiated consultation in 2004 on the pelagic longline component of this fishery as a result of exceeded incidental take levels for loggerheads and leatherbacks (NMFS 2004b). The resulting opinion (i.e., NMFS 2004b) stated the long-term continued operation of this sector of the fishery was likely to jeopardize the continued existence of leatherback sea turtles, but RPAs were implemented allowing for the continued authorization of the pelagic longline fishing that would not jeopardize leatherback sea turtles. On July 6, 2004, NMFS published a final rule to implement management measures to reduce bycatch and bycatch mortality of Atlantic sea turtles in the Atlantic pelagic longline fishery (69 FR 40734). The management measures include mandatory circle hook and bait requirements, and mandatory possession and use of sea turtle release equipment to reduce bycatch mortality. 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 by reducing mortality attributed to this fishery.

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). This fishery is typical of most pelagic fisheries, being truly a multispecies fishery, with swordfish as a substantial portion of the total catch. Yellowfin tuna, dolphin and, to a lesser extent, bigeye tuna, are other important components of the landed catch. In some cases, traditionally utilized fishing gears and economically necessary practices, such as targeting both pelagic and reef fish species with multiple gear types during a single trip, may diverge from fishing norms in U.S. mainland fisheries. Principal ports are St. Croix, USVI, and San Juan, Puerto Rico. Many of these high quality fresh fish are sold to local markets to support the tourist trade in the Caribbean.

The distribution of HMS permits in Puerto Rico and the USVI is shown in Table 4.2.2 Currently, there are no HMS limited access permits (LAPs) held in the U.S. Caribbean and only a limited number of HMS open access fishing permits and dealer permits. The low number of HMS fishing and dealer permits has resulted in limited catch and landings data from the U.S. Caribbean fisheries. Of the 295.8 mt of tunas landed in the U.S.

Caribbean in 2007, 260.2 mt was reported as captured with PLL gear (NMFS 2008). Since no Atlantic Tunas Longline permits are held by residents of Puerto Rico or the USVI, it can be assumed that these tuna landings were reported by vessels fishing in the Caribbean, but based out of other U.S. ports. Approximately 35.6 mt of tunas were reported as harvested with handline and rod and reel gears (NMFS 2008). The handline and rod and reel landings were likely reported by Caribbean fishermen fishing under Atlantic Tunas General or HMS CHB permits. In 2007, 27.7 mt of SWO were reported as harvested from the Caribbean (NMFS 2008). All of those landings were reported as harvested with PLL gear and likely by vessels not based in Caribbean ports. Puerto Rico reported approximately 10.1 mt of commercial shark landings for 2006 (PR DNER 2007). It is not clear what portion of these landings or what species were harvested from federal waters. Currently, little information is available regarding shark catches in the USVI.

Table 4.5.2.1 Distribution of HMS permits among Puerto Rico and the USVI

Permit Type	Puerto Rico	St. Thomas	St. Croix	St. John
Atlantic Tunas General	76	4	9	1
HMS CHB	22	6	3	4
HMS Angling	529	15	16	0

* There are no other HMS fishing permits held in the U.S. Caribbean.

Reef Fish Fisheries

Section 2.1.2 provides an overview of the history of the federal Caribbean reef fish fishery and its management by NMFS under the CRFFMP. Section 1 reviews the previous consultations on the federal fishery's effects on listed sea turtles. The current federal reef fish fishery and its proposed continued authorization is the subject of this consultation so not part of the environmental baseline. However, its past effects and the past and current effects of reef fish fishing in territorial and commonwealth waters on sea turtles are part of the environmental baseline.

Spiny Lobster Fishery

The spiny lobster fishery in waters around Puerto Rico and the USVI occurs with pots and traps, and hand-harvest. Due to the predominance of fishable habitat in state waters, it is assumed 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). In order of gear landing predominance, SCUBA divers, fish traps, and lobster traps catch the majority of commercial lobster in Puerto Rico from 1998-2001 (NMFS 2005a). Wooden traps are used primarily for lobster, but wire mesh traps are used for both lobster and reef fish. Biodegradable panels are required for all traps, including those in state waters, but fishermen have not always followed this regulation. In the USVI, divers catch the majority (i.e., approximately 85%) of spiny lobster (Tobias 2001). NMFS (2005a) concluded only one leatherback trap interaction is anticipated annually and that the spiny lobster fishery is not likely to jeopardize the continued existence of listed sea turtles.

4.4.3 Vessel Traffic

Commercial and recreational vessel can adversely affect sea turtles through propeller and vessel strikes. Many records of vessel interactions have been documented within the action area. 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 and/or partially healed propeller wounds also may be associated with secondary problems such as emaciation and increased buoyancy (Walsh 1999).

Private vessels in the action area participating in high-speed marine events (e.g., boat races) may be a particular threat to sea turtles. NMFS and the USCG have completed a section 7 consultation for the Caribbean Marine Event Program for all annually occurring marine events in the USVI and Puerto Rico. As a result of this consultation, the USCG now includes permit conditions the event participants must follow to avoid and minimize potential impacts of marine events.

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. Through the section 7 process for dock, port, and marine construction activities under the jurisdiction of the COE, NMFS will attempt to establish conservation measures to ensure that the construction and operation of these facilities avoids or minimizes adverse effects to listed species.

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

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 are likely fatal.

4.4.4 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 area. 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.

Anthropogenic sources of marine pollution, while difficult to attribute to a specific federal, state, local or private action, may indirectly affect sea turtles in the action area. 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 (e.g., Mississippi River). There are some studies on organic contaminants and trace metal accumulation in green and leatherback sea turtles from other regions which indicate bioaccumulation can occur (e.g., Aguirre et al. 1994, Caurant et al. 1999, Corsolini et al. 2000). Information on detrimental threshold concentrations is not available and little is known about the consequences of exposure of organochlorine compounds to sea turtles. Research is needed on the short- and long-term health and fecundity effects of chlorobiphenyl, organochlorine, and heavy metal accumulation in sea turtles.

Nutrient loading from land-based sources such as agricultural and coastal community stormwater and sanitary discharges is known to stimulate plankton blooms in closed or semi-closed estuarine systems. Seasonally depleted oxygen levels (< 2 mg/l), caused by eutrophication from both point and non-point sources. Most aquatic species cannot survive at such low oxygen levels, thus these areas, known as "dead zones " impact the animals found there, including sea turtles, and ecosystem-level impacts continue to be investigated.

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) located on St. Croix is among the top ten largest refineries in the world and the second largest in the United States. Established in the 1960's, the oil refinery is capable of processing up to a half million barrels of oil a day. Leaks from oil process and storage have resulted in plumes of oil floating on top of the groundwater underlying the facility and oil is shipped in and out of the territory via large ocean tankers daily. 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.

4.4.5 Additional Military Activities

Military ordnance detonation has adversely affected sea turtles in the action area. 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.

4.4.6 Dredging and Beach Renourishment

The construction and maintenance of federal navigation channels has also been identified as a potential source of turtle mortality. Hopper dredges, which are frequently used in ocean bar channels and sometimes in harbor channels and offshore borrow areas, move relatively rapidly (compared to sea turtle swimming speeds) and can entrain and kill sea turtles, presumably as the drag arm of the moving dredge overtakes the slower moving sea turtle. Individual dredging and beach renourishment projects in the action area have been consulted on, but until most recently have all been conducted informally. On May 27, 1997, NMFS completed an opinion on the continued hopper dredging of channels and borrow areas in the southeast United States. This consultation determined hopper dredging would adversely affect sea turtles but would not jeopardize their continued existence and an ITS was issued. NMFS is currently reinitiating on dredging and beach renourishment activities of the U.S. Army Corps of Engineers, South Atlantic Region. The new biological opinion, expected to be completed within the next year, is being expanded to also include activities in the U.S. Caribbean Region.

4.4.7 ESA Permits

Sea turtles are the focus of research activities in the action area authorized by section 6 and 10 permits under the ESA. Regulations developed under the ESA allow for the issuance of permits authorizing take of certain ESA-listed species for the purposes of scientific research under section 10(a)(1)(a) of the ESA. In addition, section 6 of the ESA allows NMFS to enter into cooperative agreements with states to assist in recovery actions of listed species. Prior to issuance of section 6 permits, the proposal must be reviewed for compliance with section 7 of the ESA.

As of May 1, 2011, there were only two active NMFS-issued scientific research permits for in-water work directed at sea turtles in the Caribbean; several applications are under review. Permitted research activities (i.e., capture, handling, tagging, measuring, photographing, weighing, tissue and blood sampling activities) are generally expected to result in temporary stress, but are not expected to have more than short-term effects on sea turtles. Before any research permit is issued, the proposal must be reviewed under

the permit regulations (i.e., must show a benefit to the species). In addition, since issuance of the permit is a federal activity, issuance of the permit by NMFS must also undergo a section 7 analysis to ensure the issuance of the permit does not result in jeopardy to the species. The USFWS permits sea turtle research and conservation programs on land.

4.4.8 Conservation Actions Benefitting Turtles

Sea Turtle Research, Monitoring, Outreach and Education
Several USVI research projects have been ongoing in the action area 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 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.

The Sea Turtle Program of Puerto Rico is a multi-agency collaboration between DNER together with several NGO's and other agencies (Sea Grant-UPR, Rio Piedras-UPR, Mayaguez-UPR, Chelonia, WIDECAST, FWS). The main goal is to: educate, 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.

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

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 already affords the same protection to sea turtles listed as threatened under the ESA [50 CFR 223.206(b)].

4.4.9 Synthesis of Environmental Baseline for Listed Sea Turtles

In summary, sea turtles occur throughout the action area, where numerous factors adversely affect them to varying degrees. Past and present threats in the action area primarily include directed harvest and poaching, boat strikes, incidental capture and mortality in fisheries, and ingestion and entanglement in marine debris. Other activities affecting sea turtle in the action area include marine pollution, vessel and military activities, dredging, permits allowing take under the ESA; and research, outreach and education activities. These factors are ongoing and are expected to occur contemporaneously with the proposed action. Directed harvest and poaching of sea turtles, both in the action area and on nearby beaches; and incidental catch in fisheries have likely had the greatest adverse impacts on sea turtles in the action area. Over the years, the impacts associated with fisheries have likely been reduced due to declining fishing effort, coupled with increasing fishing regulations. However, interactions with commercial and recreational fishing gear not associated with the proposed action are still ongoing and are expected to occur contemporaneously with the proposed action. Other environmental impacts including the effects associated with marine debris and pollution, military activities, dredging, and permits allowing take under the ESA have also had and continue to have adverse effects on sea turtles in the action area in the past, but to a lesser degree of magnitude. The effects of overall vessel traffic on sea turtles in the action area appear to be increasing.

5.0 Effects of the Action

In this section of the opinion, we assess the probable effects of the continued authorization and operation of the Caribbean reef fish fishery on ESA-listed *Acropora* corals, *Acropora* critical habitat, and on green, leatherback, and hawksbill sea turtles. The analysis in this section forms the foundation for our destruction or adverse modification and jeopardy (risk) analysis in Section 7.

When determining the potential impacts to critical habitat this biological opinion does not rely on the regulatory definition of "destruction or adverse modification" of critical

habitat at 50 CFR 402.02. Instead we have relied upon the statutory provisions of the ESA to complete the following analysis with respect to critical habitat. Ultimately, we seek to determine if, with the implementation of the proposed action (i.e., continued authorization of fishing under the proposed ACLs), critical habitat would remain functional (or retain the current ability for the essential features to be functionally established) to serve the intended conservation role for the species.

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

A jeopardy determination is reached if we would reasonably expect the proposed action to cause, either directly or indirectly, reductions in numbers, reproduction, or distribution that would appreciably reduce a listed species' likelihood of surviving and recovering in the wild. The ESA defines an endangered species as "...in danger of extinction throughout all or a significant portion of its range..." and a threatened species as "...likely to become an endangered species within the foreseeable future..." The status of each listed species likely to be adversely affected by the continued authorization of the Caribbean reef fish fishery is reviewed in Section 3. *Acropora* species are listed because of their statuses throughout their ranges. A jeopardy determination for these species must find the proposed action will appreciably reduce the likelihood of survival and recovery for each species throughout its entire range. Sea turtle species are listed because of their global status; a jeopardy determination must find the proposed action will appreciably reduce the likelihood of survival and recovery of each species globally.

The quantitative and qualitative analyses in this section are based upon the best available scientific data on listed *Acropora* and sea turtle species biology and the effects of the proposed action. Data pertaining to the Caribbean reef fish fishery, relative to interactions with *Acropora* and sea turtles are limited, so we are often forced to make assumptions to overcome the limits in our knowledge. Frequently, different analytical

approaches may be applied to the same data sets. In those cases, in keeping with the direction from the U.S. Congress to resolve uncertainty by providing 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 select the value yielding the most conservative outcome (i.e., would lead to conclusions of higher, rather than lower, risk to endangered or threatened species).

When analyzing the effects of the proposed action, we must consider both its direct and indirect effects. Direct effects are those that caused by the proposed action and manifest themselves immediately (i.e., physical interactions between gear and listed species). As discussed in Section 2.2, the federal reef fish fishery is only authorized in the U.S. Caribbean EEZ. Since we anticipate direct effects will only occur from interactions with fishing gear, and the federal fishery is only authorized in the EEZ, our direct effects analysis only evaluates gear fished in federal waters.

We must also analyze any potential indirect effects. Indirect effects are caused by or result from the proposed action, are later in time, and are reasonably certain to occur. Indirect effects include aspects such as habitat loss and degradation, reduction of prey/foraging base, etc. Since the impacts of indirect effects are often less apparent and more difficult to quantify than direct effects, we briefly summarize the scientific literature describing the route of indirect effect between herbivory, macroalgal growth, and the impacts of that growth on coral (see Section 5.1).

As noted in Section 2.2, the proposed action may also affect populations of fishes and invertebrates in commonwealth and territorial waters because these populations are continuous, fluid, and move across imposed jurisdictional boundaries. Since species travel back and forth between the EEZ and commonwealth/territorial waters, authorization of the harvest of these species in the EEZ via the federal fishery may reduce their numbers and/or size potentially changing the amount of algae grazed throughout the EEZ and commonwealth/territorial waters. Such an impact could affect the resilience of other reef species, including threatened *Acropora* spp. For these reasons, we believe the federal reef fish fishery may also indirectly affect listed *Acropora* spp. and their critical habitat in commonwealth and territorial waters.

Basic Approach to the Assessment

The proposed action has been determined to have three primary routes of effects on listed species: via vessel and gear impacts on sea turtles and elkhorn and staghorn corals (hereafter collectively referred to as *Acropora*), and for *Acropora* and their designated critical habitat only, via authorized harvest of herbivorous fish under the CRFFMP. Each of these routes of effects will be discussed by species/habitat as applicable and a determination made whether an adverse effect is expected from that component of the proposed action; if an adverse effect is expected, an examination of that effect on the species in the action area follows. In Sections 5.1-5.6 we analyze effects on *Acropora* and *Acropora* critical habitat. In Section 5.1 we present the literature and data available to assess the effects of herbivorous fish removal on *Acropora* and critical habitat. In Sections 5.2 and 5.3, we evaluate the effects of the continued level of fish removals under

the proposed action, first to critical habitat and then to the species itself. Because the authorized harvest of herbivorous fish indirectly affects *Acropora* and *Acropora* critical habitat, our effects analyses for *Acropora* and *Acropora* critical habitat particularly focus on the proposed ACLs for the two key herbivorous reef fish species groups, parrotfishes and surgeonfishes; effects from the proposed prohibition of harvest on midnight, rainbow, and blue parrotfishes and the proposed parrotfish recreational bag limit are also analyzed. Due to the paucity of U.S. Caribbean specific data, studies examining Caribbean-wide trends are often used, with the assumption that trends in the U.S. Caribbean are consistent with general trends throughout the entire Caribbean. In Section 5.4, we examine the impacts of gear used in the Reef Fish FMP. Finally, we examine the impacts of vessel anchoring by reef fish vessels and the effects to *Acropora* and critical habitat (Section 5. 5). Section 5.6 presents our analysis of the proposed action's effects on sea turtles.

5.1 Literature and Data Available to Assess Effects of Herbivorous Fish Harvest on *Acropora* and *Acropora* Critical Habitat

5.1.1 Synthesis of Literature on Role of Herbivorous Fish in Coral Reef Ecosystems

The purpose of this section is to provide a general synthesis of the scientific literature regarding the linkage between herbivorous fish and coral reef ecosystems. While ecological processes can be complex and nuanced, many general patterns emerge. This is meant to illustrate general ecological processes and relationships and is not meant to be specific to the action area of this project. Much of the literature available on the relationships between of herbivores on corals and coral reef ecosystems is recent – from the late 1990s through 2010 – when acroporids had already become very rare on Caribbean reefs, and thus the majority of papers do not evaluate elkhorn and staghorn corals explicitly. Therefore, we are applying the findings for corals, generally, in this biological opinion.

Description of Coral, Herbivore, and Algae Feedback Loops
In general, competition between hard corals and benthic algae, particularly macroalgae and dense turf algae, is considered fundamental to the overall status of coral reefs. Algae and corals are widely considered to be in competition for available space (and light). Algal biomass interferes with coral recruitment (e.g., Steneck 1988), suppresses coral growth (Tanner 1995, Venera-Ponton et al. 2011) and fecundity (Tanner 1995, Foster et al. 2008), and may cause direct mortality (Rasher and Hay 2010). This competition can cause "feedback loops" driving high coral/low macroalgal reefs to shift toward high macroalgal/low coral cover reefs and vice versa.

Feedback loops promoting high coral/low macroalgae can begin as the amount of coral increases. As coral increases, it reduces the available space for macroalgae to colonize. Additionally, since corals (particularly large, branching corals like *Acropora*) also provide shelter from predators to herbivorous fish, increases in coral cover lead to more habitat for herbivorous fishes increasing herbivorous fish recruitment and populations.

As herbivorous fish populations increase, the grazing of macroalgae can also increase, further reducing the amount of macroalgae. Increasing amounts of the bare substrate can lead to increased incidence of successful coral recruitment. Over time these processes can act synergistically to create a feedback loop of more corals, leading to more herbivorous fish, that cause increases in grazing rates occurring on reefs leading to decreasing levels of macroalgae cover, causing an increased incidence of successful coral larvae recruitment, leading to more corals, etc. (Hay and Rasher 2010).

Conversely, feedback loops promoting low coral/high macroalgae can be established if macroalgae growth is not controlled. As macroalgae occupies greater amounts of space, it is more likely to compete directly with established colonies. Competition may stress colonies, potentially leading to reduced fitness and fecundity (Tanner 1995, Foster et al. 2008). Macroalgae can also act as a disease vector, which can also reduce the fitness of existing colonies. As macroalgae cover increases, it reduces the amount of space available for coral larvae settlement, reducing the incidence of successful coral recruitment. Macroalgae can also kill successful settled larvae via overgrowth (Mumby 2006). Reduced recruitment also means that as established colonies are lost from population they are replaced more slowly, which reduces the number of colonies available to produce larvae and compete with macroalgae for space. While macroalgae is a food source of herbivorous, it does not provide shelter from predation. Thus, even as macroalgae levels increase, herbivorous fish populations may not increase at a rate that effectively increases grazing due to a lack of the suitable shelter habitat. Over time, these processes can act synergistically to create negative feedback loop that reduces total coral cover. Increased macroalgae can reduce successful coral recruitment reducing coral cover and lead to declines in herbivorous fish populations. Declining herbivorous fish populations lead to reduced grazing rates, which can lead to the further spread of macroalgae. Macroalgal spread can cause additional reductions in the incidence of successful coral larvae recruitment, driving coral cover still lower, etc. (Hay and Rasher 2010, Mumby and Harborne 2010).

The scientific literature generally refers to these feedback loops as "phase-shifts." Phase-shifts are generally considered to be continuums ranging from high macroalgal cover/low coral cover on one side and high coral cover/low macroalgae cover on the other. For example, the negative feedback loop noted above would likely lead to a phase-shift away from a high coral cover/low macroalgae cover state toward a high macroalgal cover/low coral cover state. It is generally believed that when a system is at either end of the continuum, it is relatively resistant to any shift back toward the other end.

Effects of Exploitation on Herbivorous Fish and Diadema Grazing

In 1983, with the mass mortality of *Diadema*, Caribbean coral reefs started to experience a phase shift in benthic community structure. Immediately following the mass mortality event, algal biomass was noted to increase on reefs where *Diadema* had been the dominant grazer, and a phase shift from a coral-dominated to a macroalgal-dominated benthos began; this shift was rapid in places (Levitan 1988 and Carpenter 1990a). For *Acropora* species, the *Diadema* mass mortality and subsequent phase shift 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).

Some debate remains over why urchins acted as the primary grazers in the Caribbean reefs prior to their die off on the 1980s. Some studies suggest that urchins were the most important grazing animals in reef environments prior the 1980s (e.g., Ogden et al. 1973, Woodley 1979, Levington 1982). Others indicate that urchins only acted as the most important grazers on reefs where the herbivorous fish populations had been reduced because of fishing (Wanders 1977, Hay et al. 1983, Hay 1984, Lewis and Wainwright 1985, Lewis 1986). Regardless of the underlying reasons, urchins were clearly acting as the dominant grazers on Caribbean reefs prior to their die off in the 1980s, and that die off appears to have played a role, possibly a significant one, in triggering a phase shift toward greater macroalgae cover.

Herbivorous Fish Abundance, Biomass, and Diversity Relative to Grazing
For the phase shift that is occurring, or has occurred, on Caribbean coral reefs to be reversed some increase in the current levels of herbivory would likely be required. Given the apparent slow recovery of *Diadema*, herbivorous fish populations appear to be the most likely source for that herbivory.

Herbivory (i.e., "grazing") is an area-specific rate (i.e. proportion of grazable substrate grazed per unit time). Therefore, sudden changes to the area of grazable substrate or the abundance/size of herbivorous fish will directly affect grazing intensity (e.g., a doubling of the grazable substrate would halve the effectiveness of a given population of grazers and vice versa) (Mumby and Steneck 2008). Many studies indicate a numerically abundant, high biomass, intact size structure and diverse herbivorous fish population would likely achieve the highest herbivory rates (Lubchenco and Gaines 1981, Duffy 2002, Mumby 2006, Burkepile and Hay 2008, Lokrantz et al. 2008, Bonaldo and Bellwood 2008). Some studies present quantitative indications of how increases in herbivorous fish biomass can ultimately affect macroalgae cover (e.g., Williams and Polunin 2001; Mumby et al. 2006, 2007). However, others caution that because larger fish have a higher grazing capacity than smaller fish of the same species, biomass estimates alone may be misleading (Lokrantz et al. 2008), and high fish numbers or biomass alone cannot be viewed as evidence of an intact ecosystem with fully operational ecosystem functions (Bonaldo and Bellwood 2008).

Regardless of the overall structure and diversity of herbivorous fish stocks, it has been suggested that there may be a threshold to the amount of macroalgae an herbivorous fish guild can graze before the macroalgae growth will outpace the populations' ability to keep growth in check (Williams et al. 2001, Carpenter 1990b). This is primarily because it is conventionally believed that most species of herbivorous fish in the Caribbean do not consume macroalgae per se, or consume it in very small quantities (Paddack, unpub data). Instead, they most frequently consume new filamentous algal growth. Dietary preference for new filamentous algal growth often mean that the spread of existing algae

can be held in check, however, the density of existing, more established algae may increase.

Corallivory and Its Impacts On Acropora Relative to Herbivorous Fish Algae Removal
Caribbean parrotfish are mainly considered to be herbivorous; however, the adults of a few species such as *Sparisoma viride, Sp. aurofrenatum, and Sc. vetula* also consume some live coral as part of their diet (Bruggemann et al. 1994, Rotjan and Lewis 2006). Two forms of parrotfish corallivory have been described: 'spot biting' where individual bite-sized lesions are distributed over the entire coral skeleton, and 'focused biting' where a larger, continuous patch of coral tissue is removed by persistent predation (Bruckner et al. 2000). Regardless of the type of corallivory, bites of live coral are rare overall with only between 1-2% of all parrotfish bites occurring to live corals (Bruggeman et al. 1994, Rotjan and Lewis 2006).

Even those coral species grazed most intensely (e.g., *Montastraea* 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 (Sanchez et al. 2004, Mumby 2009). 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. Addtionally, Mumby et al. (2007) and Mumby (2006) found a strong positive correlation between parrotfish grazing and density of juvenile corals. Mumby et al. (2007), 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.

Scientific Debate Regarding the Significance of Herbivore-Mediated Algal Impacts
The scientific literature also 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/low macroalgal cover to high macroalgal/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.

5.1.2 Available Data Sets Used to Assess Effects of Herbivorous Fish Harvest on *Acropora* and *Acropora* Critical Habitat

Commercial Fishery Landings Data

Historic landings data are available at the species/species group level starting in 1998 for St. Croix, from 2000 for St. Thomas/St. John, and from 1983 for Puerto Rico (see Figure 5.1.1 and Figure 5.1.2.) through 2008. While some uncertainty surrounds these landings, they are the only information available to develop a picture of the historic fishing that has taken place in the U.S. Caribbean. USVI landings prior to 1998 were reported by gear (e.g., line fish, net fish, pot fish), thus there is no specific information about family groups (e.g., snapper, grouper, parrotfish).

Parrotfish landings data for St. Thomas/St. John are not available before 2000, but have remained relatively constant since that time, at around 50,000 lbs. In St. Croix, reported landings increased from a little over 200,000 pounds in 1998 to over 400,000 lbs in 2006 but declined to 356,000 pounds reported in 2008. In Puerto Rico, landings have generally declined over time but with periodic increases from nearly 400,000 lbs in the early- to mid-1980s to around 60,000 lbs in 2009.

Surgeonfish landings in St. Thomas/St. John increased from approximately 30,000 lbs in 2000 to 45,000 lbs by 2004, and decreased to approximately 38,000 lbs in 2008. In St. Croix, reported landings decreased from 40,000 lbs in 1998 to 35,000 lbs in 1999, then rose to 55,000 lbs in 2002, followed by another decrease in 2003 (40,000 lbs) and subsequent increase each year until 2006 (50,000 lbs). Reported landings of surgeonfish declined in 2007 and 2008 in St. Croix to 38,000 lbs. In Puerto Rico, landings of surgeonfish have essentially been zero since the early 1980s.

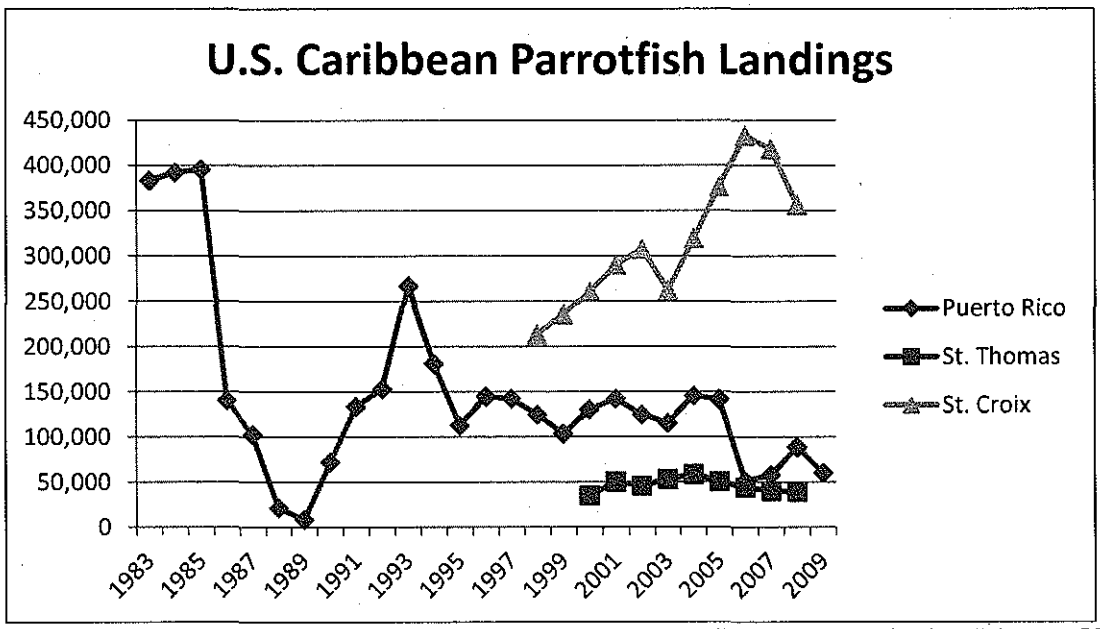


Figure 5.1.1. Commercial Landings Data for Parrotfish in the U.S. Caribbean. USVI data are reported landings only. Puerto Rico data include adjustments made by Puerto Rico DNR and SEFSC to account for reporting problems.

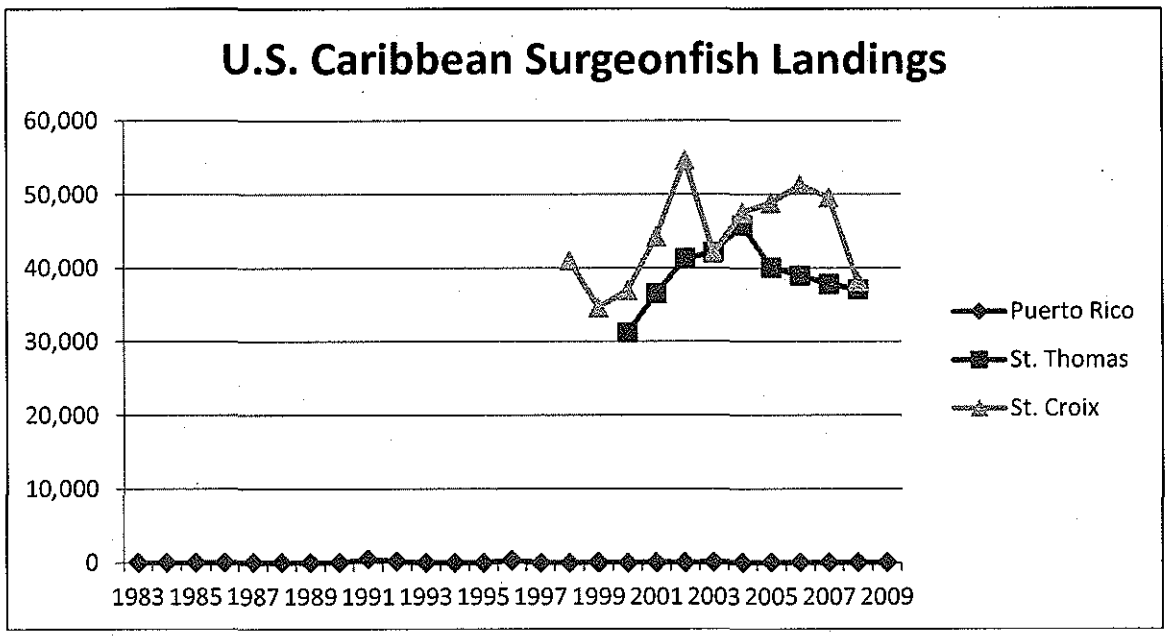


Figure 5.1.2. Commercial Landings Data for Surgeonfish in the U.S. Caribbean. USVI data are reported landings only. Puerto Rico data include adjustments made by Puerto Rico DNR and SEFSC to account for reporting problems.

Recreational Fishery Landings Data

MRFSS tracks the numbers and effort of recreational fishermen and estimates recreational landings in Puerto Rico. However, MRFSS has no data on recreational effort or harvest in the USVI. The most recent report on recreational fishing activity in USVI

waters (Tobias and Dupigny 2009) reviews the information available for the area, including the surveys on the recreational fishing activity in general, including reef fish. Most of the information on recreational fisheries for the USVI derives from offshore billfish and other pelagic fisheries since the area is well known for gamefish. Tobias and Dupigny (2009) summarize the information on the latest recreational fishing survey targeting the pelagic fleet. None of the reports on the recreational fishing activity in the USVI provide information on reef fish.

The lack of information on recreational harvest in the USVI is not detrimental to our effects analysis. Our effects analysis is primarily qualitative and uses parameters like macroalgal/coral percent cover, changes in macroalgal/coral percent cover over time, and trends in herbivorous fish biomass over time to assess the likely effects of the proposed action. While the removal of herbivorous fish from recreational fishing may ultimately affect these things, those effects would be reflected in the parameters themselves. For example, if recreational fish harvest had reduced herbivorous fish biomass, that effect would be captured in our parameter estimating herbivorous fish biomass over time. Thus, our analysis does not require accurate estimates of harvest levels, and the lack of information on USVI recreational harvest is not an impediment to an accurate effects analysis.

Coral Reef Benthic Data

The NCCOS and USVI DPNR/UVI datasets were the data sets used to evaluate recent average coverage and trends in coral reef benthic cover in the U.S. Caribbean. Trends in *Acropora cervicornis* and *A. palmata* percent coverage through time were examined using NCCOS and USVI DPNR/UVI data. Acroporid coral species were rarely encountered by either survey. Given the limited information available, no formal statistical analyses were conducted on these data (see Appendix 1 for additional detail). Throughout this effects analysis, we will focus on the available data for coral, generally. Overall coral data are more robust for analysis. Therefore, our assessment of these two monitoring datasets looked at overall coral data. Because competition between hard corals and benthic algae is considered fundamental to the overall state of coral reefs, we also believe the overall coral data to be representative of the status of coral reef health and the net balance of coral mortality, recruitment, and growth that occurs in this competitive environment.

Herbivorous Fish Biomass

No stock assessments have been conducted for parrotfish or surgeonfish in the U.S. Caribbean. NMFS SEFSC (2011) states that "existing data are insufficient to quantify current, historical, and unfished biomass levels in the US Caribbean or to accurately describe how populations would respond to changes in removals" (NMFS SEFSC 2011). While the paucity of data on herbivorous fish populations in the U.S. Caribbean confounds any attempt to quantify the impact of the proposed action on these species and ultimately, ESA-listed species, some information on herbivorous populations is available (e.g., Friedlander and Beets 2008, García-Saís et al. 2008, Pittman et al. 2008, Rothenberger et al. 2008). We recognize these data have often been collected over relatively short time periods, and from relatively small and specific locations that may or

may not be representative of all reefs. However, these data are currently the best available information from which to try and determine the effects of the action. Appendix 2 provides a more detailed discussion on these data sets.

5.1.3 Proposed Harvest Levels (ACLs)

Prohibitions of Harvest Under the Proposed Action
The 2010 ACL amendment (CRFFMP Amendment 5) would specify an ACL of zero for the three largest-bodied parrotfish species, midnight, blue, and rainbow, prohibiting their harvest for commercial or recreational purposes. These three species combined represent only 0.17% of all trip intercept program (TIP) samples for parrotfish (n=32,922) in the U.S. Caribbean for the time period 2000-2010. However, any increases resulting from this fishing prohibition could have large benefits for corals, since NMFS SEFSC (2011) and Lokrantz et al. (2008) indicate that larger individuals are often more effective grazers than smaller individuals of the same species. NMFS SEFSC (2011) indicated that these large-bodied species are potentially the most efficient grazers among Caribbean parrotfishes.

Commercial and Recreational Harvest Levels Under the Proposed Action
The proposed action sets a commercial ACL of 350,500 lbs of parrotfish in the U.S. Caribbean (240,002 lbs for St. Croix; 42,500 lbs for St. Thomas/St. John; and 52,724 lbs for Puerto Rico) and a recreational ACL of 15,259 lbs of parrotfish for Puerto Rico. The proposed surgeonfish ACL is 73,620 lbs in the U.S. Caribbean (10,768 lbs for the commercial and recreational sector of Puerto Rico; 29,249 lbs for St. Thomas/St. John; and 33,603 lbs for St. Croix). Since recreational landings information is not available in the USVI, recreational ACLs for parrotfish and surgeonfish were only specified for Puerto Rico. Since there are no recreational landings data available for parrotfish and surgeonfish in the USVI, the proposed ACL for these species is based solely on reported commercial landings and tracking of the ACL will be based only on commercial harvest. If a USVI commercial sector is determined to have exceeded its ACL, then the associated AM would apply to both the commercial and recreational sectors. For example, if the AM is to shorten the fishing season the following year, then that regulation would apply to both the commercial and recreational sectors.

The proposed action also includes a 2 fish per person/6 fish per vessel bag limit of parrotfish for the recreational sector in the U.S. Caribbean EEZ. It is believed that this bag limit, coupled with the proposed ACLs, will reduce the recreational harvest of parrotfish for two reasons. First, previous management had no restrictions on recreational harvest (i.e., an unlimited "bag limit"), thus limiting recreational harvest should reduce at least the potential for high harvest amounts. Secondly, in the USVI, when the commercial ACL is met, the recreational sector would also be closed because of the use of the commercial ACL as a proxy for recreational landings and Puerto Rico has a separate ACL for the recreational sector, which when met would shut down the fishery, again limiting harvest relative to the previous management regime.

5.2 Critical Habitat Analysis - Effects of Herbivorous Fish Harvest/Prohibition

In the Status of the Species Section (Section 3.3.1) and the Environmental Baseline Section (Section 4.1 and 4.2) we described the ecological principle that herbivory plays an important role in controlling the growth of algae on coral reefs. The NCCOS and UVI/USVI DPNR data sets indicate that the process of episodic progression towards high algae and low coral on reefs is occurring, or has occurred, throughout the U.S. Caribbean. This process has been ongoing concurrent with differing levels of parrotfish and surgeonfish harvest throughout the U.S. Caribbean. For our analysis of effects in this section, we assume the ecological principle that herbivory impacts the growth and spread of macroalgae is also valid for the U.S. Caribbean. Based on this assumption, we believe that a fishery targeting herbivorous fish reduces the effective grazing capacity of the fish population relative to that of an unexploited population. Since the essential feature for *Acropora* critical habitat is "...consolidated hardbottom or dead coral skeletons free from fleshy macroalgae or turf algae and sediment cover", by definition, macroalgae and the essential feature are mutually exclusive. Thus, we believe the harvest of herbivorous fish will reduce the populations' capacity to graze macroalgae, causing indirect effects to *Acropora* critical habitat by reducing the capacity of herbivorous fish populations to maintain the availability of the essential feature.

There is no stock assessment for Caribbean parrotfish or surgeonfish. Thus, our assessment is limited to a comparative use of landings and reported landings data and the few available studies on biomass and density of these species. Those limited surveys from discrete locations in the U.S. Caribbean appear to indicate that, in general, parrotfish and surgeonfish abundance fluctuates between years, but has not declined significantly. These studies also appear to indicate that a normal size distribution for these species is skewed toward smaller individuals not targeted by the fishery (Nemeth et al. 2006, Friedlander and Beets 2008, Garcia-Sais 2008, Pittman et al. 2008, Rothenberger et al. 2008).

Herbivorous fish are not strongly targeted in Puerto Rico and St. Thomas/St. John, while they are a major part of the reef fish fishery in St. Croix (NMFS SEFSC 2011). Landings in Puerto Rico and reported landings in St. Thomas/St. John have been consistently and significantly lower than those in St. Croix. If the herbivorous fish productivity per unit area in St. Croix is representative of the entire U.S. Caribbean, than the harvest on the larger shelves of St. Thomas/St. John and Puerto Rico are unlikely to have substantially reduced herbivorous fish populations (NMFS SEFSC 2011). Given the differences in landings across the three platforms, we analyze the potential effects to each critical habitat unit independently, with the exception of the likely effects of the prohibition of the three large-bodied species.

Three Large-Bodied Species

There are no data or current population estimates for the three large-bodied species (i.e., midnight, blue, and rainbow parrotfish) for any platform. However, we believe herbivorous fish productivity is likely the same across all three island groups. As the populations of the three large-bodied species begin to respond to a harvest prohibition,

we would anticipate populations would increase at relatively similar rates in all island groups. One possible exception is St. Croix. While no data exists on what the harvest levels of these species were across the island platforms, since parrotfish are targeted more frequently in St. Croix it is possible the stocks of the large-bodied species may be lower in that region relative to Puerto Rico and St. Thomas/St. John. If stocks are more depressed in St. Croix, the prohibition on harvest of these species is likely to cause a more noticeable increase in the populations and grazing there, relative to the other island groups. Regardless of past harvest levels, the effect of an increasing population of these three large-bodied parrotfish species increase is likely to mean an increase in grazing, which could be relatively substantial since these three species may be the most efficient grazers of all Caribbean parrotfish (NMFS SEFSC 2011). An increase is grazing is likely to have beneficial effects for all three *Acropora* critical habitat units.

5.2.1 Effects of Herbivorous Fish Harvest on the St. Croix Critical Habitat Unit

Data Regarding Herbivorous Fish Populations in St. Croix
Rothenberger et al. (2008) provide some limited information on herbivorous fish populations in St. Croix. From 2001-2006, NOAA CCMA-BB surveys found that in St. Croix, "herbivores" comprised more of the biomass than "piscivores" for all survey periods except during August 2001.[4] While this is not particularly surprising, given the generally reduced piscivores populations in the Caribbean, these data do indicate that parrotfish/surgeonfish remain relatively common. Herbivorous fish biomass in St. Croix fluctuated annually during the study period but generally remained between 1,000-2,000g/100m^2. However, during 5 of 11 survey periods herbivore biomass was greater than 2,000g/100m^2 (Rothenberger et al. 2008). While this information does not provide specifics about parrotfish or surgeonfish, it does provide an indication that herbivore biomass fluctuates annually, but does not appear to be declining over consecutive years.

The USVI DPNR-DFW also conducted a St. Croix study from 2003-2005 meant to mirror the UVI-CMES fish surveys off of St. Thomas (see below). The aggregated data (all sites pooled) among years indicate that there were no pronounced changes in reef fish assemblage structure during the monitoring period. No significant difference in average fish abundance ($p=0.086$) or average fish species richness was detected over time ($p=0.16$). This finding reflects the high variability in fish abundance among sites within any given year (Rothenberger et al. 2008). In all years, omnivores dominated the reef fish assemblage in terms of biomass. Herbivore biomass represented approximately 30% of the entire assemblage. Piscivores contributed least to assemblage biomass (10-14%) and were least abundant numerically (2.7-3.1% of all fish observed). Among the years observed, there was no clear indication of a change in trophic composition through time (Rothenberger et al. 2008). Rothenberger et al. (2008) also state "[t]he quantity of herbivorous reef fishes harvested in the St. Croix commercial fishery has increased during the past decade (W. Tobias, pers. comm.), making scarids a commercially important species group." However, a comparison of the size frequency distribution

[1] Herbivores included damselfish, parrotfish, surgeonfish, etc.; while piscivores included red hind, other groupers, snappers, etc. (Rothenberger et al. 2008). See Randall 1967 for further discussion of species included in these groups.

among years for scarids observed during 2003-2005 did not indicate a trend towards decreasing mean size during the study period. Yet, relatively few parrotfish in the >30 cm size class, which are targets of the commercial sector, were observed. The observed low frequency with which parrotfish attain large body size may be indicative of increased fishing mortality rates (Rothenberger et al. 2008).

Pittman et al. (2008) evaluated fish assemblages and benthic habitats in and around BIRNM, on the northeast side of St. Croix. Across the entire survey area (i.e., inside and outside BIRNM), ocean surgeonfish, redband parrotfish, blue tang, striped parrotfish, and princess parrotfish ranked number 3, 4, 6, 9, and 17, respectively out of the top 20 most commonly observed species. Outside BIRNM, the total parrotfish and redband parrotfish biomass remained relatively stable across the time series; however, declines over three consecutive years were recorded for striped parrotfish biomass and density. Striped parrotfish biomass was significantly lower ($p=<0.05$) in 2005 and 2006 than in 2003 (Pittman et al. 2008).

Pittman et al. (2008) also reported that from 2003-2006 parrotfish exhibited a skewed size frequency distribution towards a higher frequency of the smallest size classes. Surgeonfish size classes were also skewed toward a higher frequency of smaller size classes outside BIRNM. More specifically, blue tangs exhibited a peak in frequency for small adults and ocean surgeonfish exhibited a peak for subadults, while redband and striped parrotfish showed a strongly skewed distribution, with high frequency of the smallest juveniles (<5 cm) and gradual decline with size with very few of the largest adults (Pittman et al. 2008).

The above studies indicate that for St. Croix generally, herbivorous fish assemblages did not change significantly between 2001-2006 and parrotfish and surgeonfish remain frequently observed. The information also indicates parrotfish and surgeonfish biomasses fluctuate annually, but have not significantly decreased over time, with the notable exception of a statistically significant decline in striped parrotfish biomass outside BIRNM. The data do appear to indicate a skewing toward smaller individuals in at least two species of parrotfishes and surgeonfish outside BIRNM.

Relationship Between Herbivorous Fish Harvest and Macroalgae in St. Croix
From 1998-2002, reported landings of parrotfish in St. Croix increased steadily from approximately 200,000 to 300,000 lbs (Figure 5.1.1). A slight decline in reported landings occurred in 2003, followed by another steady increase from approximately 250,000 lbs (in 2003) to approximately 430,000 lbs in 2006, prior to the gill and trammel net prohibition. Reported landings then declined in each of the following two years. NMFS SEFSC (2011), states: "[i]n St. Croix, the recent reported landings which have been at or above 300,000 lbs. since the late 1990's are believed to be sustainable."

From 2002-2008, NCCOS data on algae cover in St. Croix indicate it was highest in 2003, the year with the lowest reported landings of parrotfish during the survey period. Algal cover declined in 2004, but increased in 2005 and 2006 as parrotfish landings increased. Reported parrotfish landings declined slightly in 2007 along with mean algal cover. In 2008, parrotfish harvest declined again, but mean algal cover increased (Figure

5.2.1.1). During the entire NCCOS time series (2002-2010) no statistically significant trend in mean algal cover was detected for St. Croix (Appendix 1).

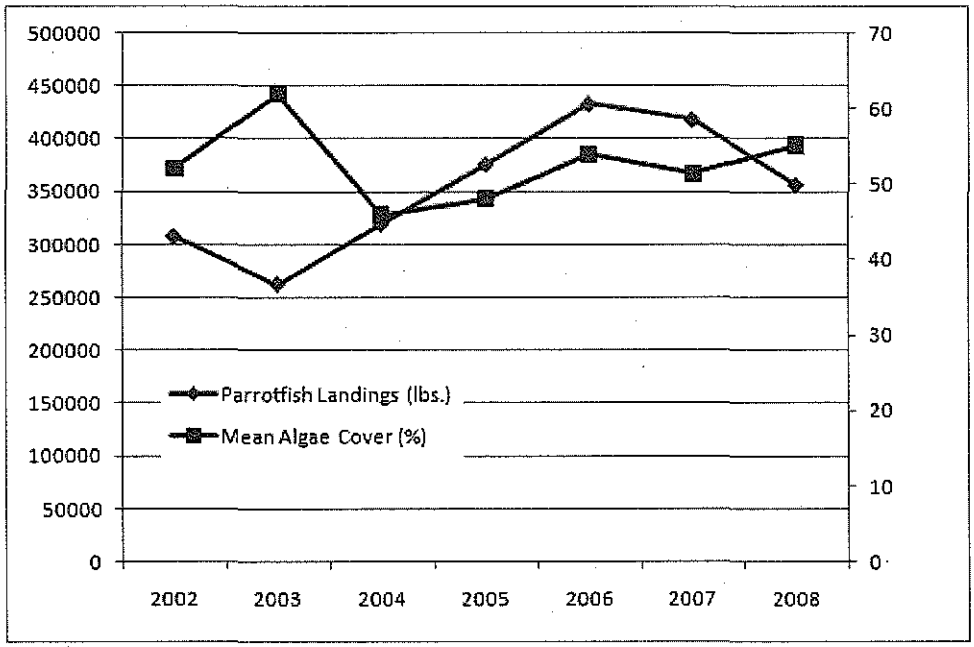


Figure 5.2.1.1 Mean Percent Algal Cover and St. Croix Parrotfish Landings 2002-2008

Reported surgeonfish landings in St. Croix declined from approximately 40,000 lbs in 1998 to approximately 35,000 lbs in 1999, but steadily increased each subsequent year until they reached a peak of approximately 55,000 lbs in 2002 (Figure 5.2.1.2). Reported landings dropped again the following year, but followed a similar pattern as before, increasing each subsequent year until reaching another peak in 2006 (approximately 50,000 lbs) prior to the gill and trammel net prohibition; lower landings were reported in 2007 and 2008 (Figure 5.2.1.2). From 2002-2008, NCCOS data on algal cover in St. Croix indicate it was highest in 2003, the year with the lowest reported landings of surgeonfish during the period. Algae cover declined in 2004, but increased in 2005 and 2006 as surgeonfish landings increased. Surgeonfish landings declined slightly in 2007 along with mean algal cover; however, mean algal cover increased again in 2008, while surgeonfish landings declined for a second straight year (Figure 5.2.1.2).

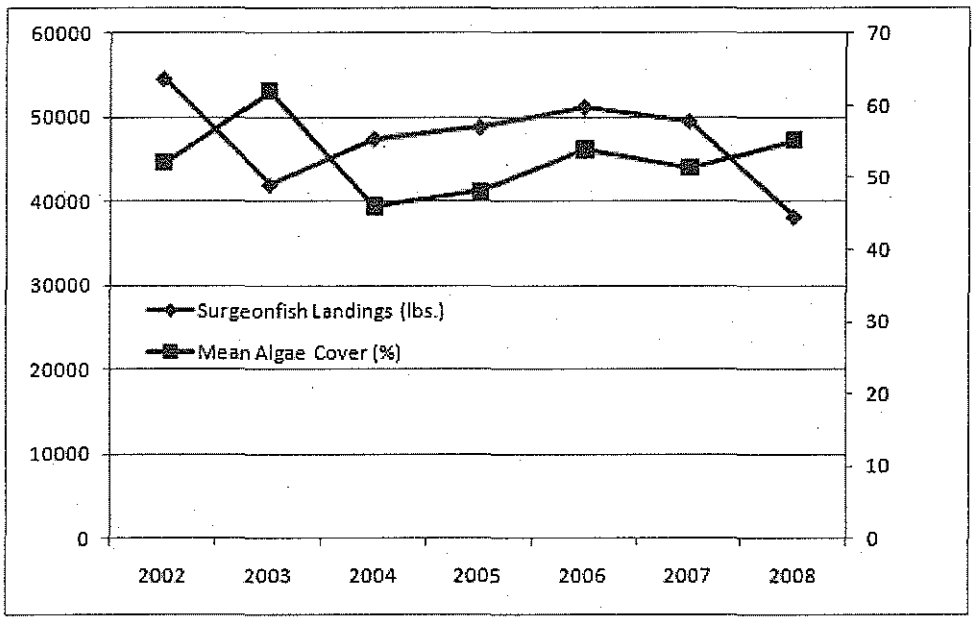


Figure 5.2.1.2 Mean Percent Algal Cover and St. Croix Surgeonfish Landings 2002-2008

Parrotfish and surgeonfish harvest in St. Croix is higher than any other island region in the U.S. Caribbean and occurs on the smallest shelf area, indicating the St. Croix fishery has the most significant impact on parrotfish and surgeonfish populations of the three island groups. As stated in the Status of the Species and Environmental Baseline sections, herbivores play an important role in mitigating algal growth on reefs and maintaining species balance. Based on the role herbivory plays in controlling macroalgal growth, and because the total reported landings of herbivorous fish are highest on St. Croix relative to the other island platforms, we anticipate that the adverse affects caused by reduced grazing are the greatest in the St. Croix *Acropora* critical habitat unit. Additionally, the data available on parrotfish and surgeonfish populations in the area indicate that the normal size distribution for these species is skewed toward smaller individuals not targeted by the fishery.

We believe this information indicates the current harvest of parrotfish and surgeonfish in St. Croix is adversely affecting *Acropora* critical habitat, by reducing grazing capacity with respect to total population numbers, and the size of the individuals making up that population. Levels of harvest authorized by the proposed ACL are expected to increase surgeonfish and parrotfish abundance and improve the size structure of those populations, reducing the adverse effects of continued harvest, when compared to current levels. However, the levels of adverse effects are highly uncertain and unquantifiable.

5.2.2 Effects of Herbivorous Fish Harvest on the Puerto Rico Critical Habitat Unit

Data Regarding Herbivorous Fish Populations in Puerto Rico
Data provided by Garcia-Sais (2008) from La Parguera, Puerto Rico, indicate that from 2001-2006 parrotfish and surgeonfish were the top two families observed, in terms of total biomass, accounting for approximately 35% of all biomass. Garcia-Sais (2008) also

reported that of all the species recorded on hardbottom habitats, parrotfishes accounted for more biomass than any other family of species. The authors defined their abundance as "moderately abundant, and generally larger bodied than most other families." Ultimately, Garcia-Sais (2008) could not determine a long-term trend in parrotfish abundance. While it is difficult to draw conclusions from these data, it does appear that parrotfish and surgeonfish remain relatively abundant in the Puerto Rico, at least in the study area.

Relationship Between Herbivorous Fish Harvest and Macroalgae in Puerto Rico
Reported landings indicate that surgeonfish harvest in Puerto Rico is essentially zero and has been that way for some time. While the proposed ACL means an increase in harvest could occur, there is no indication that landings of these species will increase. These species remain culturally and economically unimportant in Puerto Rico. Given that landings were so low when no harvest limits were in place, it is unlikely that the implementation of an ACL will lead to an increase in landings.

From 1995-2005, Puerto Rican reported landings of parrotfish remained relatively stable between 100,000 and 150,000 lbs; from 2006-2009 landings declined to between 50,000 and 100,000 lbs. NCCOS data on algae cover in Puerto Rico from 2002-2009 indicates that a statistically significant increase in algal coverage occurred from 2002-2010 despite the decrease in parrotfish landings (Appendix 1). Increases in mean algae cover during that period do not correlate closely with parrotfish landings (Figure 5.2.2.1). In 2003 and 2004, landings increased while percent cover declined slightly from the 2002 level. From 2005-2006, there was a large drop in parrotfish landings followed by two years of declines in percent algae cover. From 2007-2009 landings oscillated between approximately 60,000-80,000 lbs, far lower than landings reported in 2002-2004. Interestingly, even with relatively lower landings, the mean percent algae increased from 2007-2009 by approximately 20%.

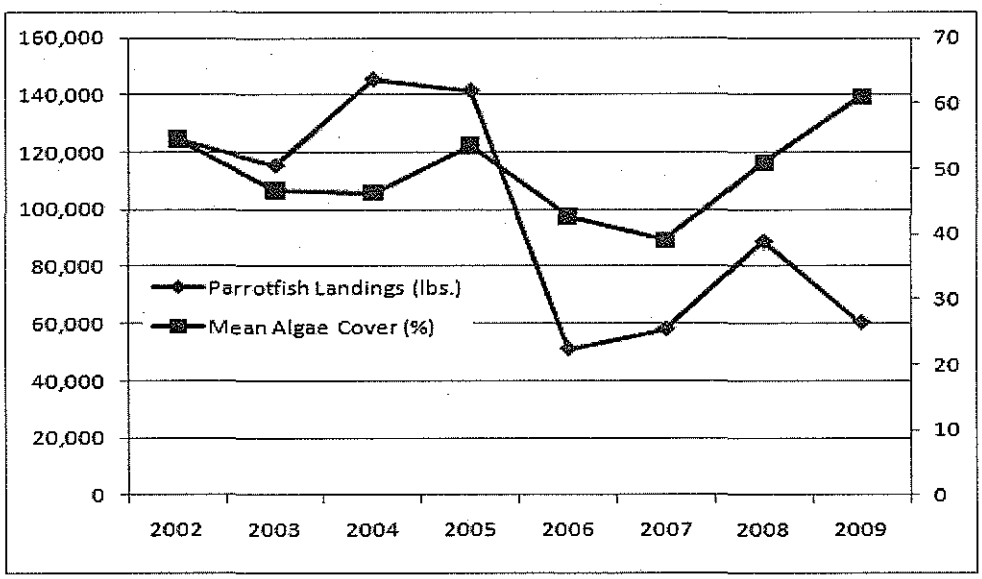


Figure 5.2.2.1 Mean Percent Algal Cover and Puerto Rico Parrotfish Landings 2002-2009

NMFS SEFSC (2011) also indicates that the stocks of parrotfish in Puerto Rico may not have been substantially reduced due to exploitation. If parrotfish productivity per unit area in St. Croix (where the harvest is much greater) is representative of the region, the removals on the larger shelves of St. Thomas/St. John and Puerto Rico are unlikely to have substantially reduced parrotfish populations due to exploitation.

The data appear to indicate a loose correlation, if any, between parrotfish landings and mean algal cover in Puerto Rico. The available data indicate that parrotfish biomass in Puerto Rico has remained relatively stable (or not significantly decreasing) (Garcia-Sais 2008), at numbers considered moderately abundant in the face of the current levels of harvest. This appears to indicate that parrotfish populations are able to replenish themselves in the face of exploitation. Likewise, mean algal cover has shown a statistically significant increase over time as parrotfish harvest has declined, indicating that harvest of parrotfish or even parrotfish population levels may not be the most significant factor contributing to changes in macroalgal cover over time. Regardless, we believe the continued harvest of parrotfish (even at reduced levels per the ACL) will reduce the overall grazing capacity of that population and thus, the proposed action is likely to adversely affect the Puerto Rico *Acropora* critical habitat unit by reducing the herbivore assemblage's ability to mitigate increasing algal cover. However, the level of adverse affects are highly uncertain and unquantifiable, but based on comparable levels of harvest are expected to be less than the effects on the St. Croix unit.

5.2.3 Effects of Herbivorous Fish Harvest on the St. Thomas/St. John Critical Habitat Unit

Data Regarding Herbivorous Fish Populations in St. Thomas/St. John
Friedlander and Beets (2008) reported the findings from a long-term monitoring study (i.e., 1988-2006) conducted inside and outside the Virgin Islands National Park (VINP) in St. John. The study found that "herbivores" were the most important trophic guild by numerical abundance among the four permanent reef sites. Parrotfishes (*Scaridae*) were the second most numerically important family among the reference reefs, only damselfish were more numerically abundant. Striped parrotfishes (*Scarus iserti*) made up nearly 62% of the numerical abundance of parrotfishes, followed by redband (*S. aurofrenatum* – 19%), stoplight (*S. viride* – 11%), and princess (*S. taeniopterus* – 3%). Surgeonfishes (*Acanthuridae*) showed consistent and level numerical abundance until 1994, followed by a notable decline from 1995-2000, followed by a return to relatively stable, albeit lower, numerical abundance from 2000-2006 (Friedlander and Beets 2008). Parrotfishes were also the family of species with the highest biomass during the period at the four reference sites, but did not show a discernible trend during that time (Friedlander and Beets 2008).

Rothenberger et al. (2008) provide some additional information on herbivorous fish populations in St. John. From 2001-2006, NOAA CCMA-BB survey found herbivores in St. John consistently comprised more of the biomass than piscivores for all years except 2003. While this not particularly surprising, given the generally reduced piscivores populations in the Caribbean, these data do indicate that parrotfish/surgeonfish remain relatively common. Herbivore biomass fluctuated between approximately 500-

3,500g/100m^2 during the study period with a spike in herbivore biomass noted in 2004 (3,500g/100m^2), but biomass declined in each of the following two years in the study, to its lowest value reported (500 g/100m^2) (Rothenberger et al. 2008).

The only information available for St. Thomas comes from fish surveys conducted by UVI-CMES between 2003 and 2006 as reported by Rothenberger et al. (2008). The pooled data from the study indicated no pronounced changes in fish assemblage structure on reef sites from 2003-2006. During that period, total fish abundance was not significantly different over time ($p=0.080$) nor was average species richness ($p=0.538$). However, a comparison of repeated sites shows fairly high variability in fish abundance between and within sites. Fish abundance by family also varied across the time series, apparently due to natural variation, seasonality and variable recruitment (Nemeth et al. 2006). In particular, *Acanthurid* and *Scarid* numerical abundance varied over time on midshelf reefs. *Scarids*, represented primarily by the princess, striped, and redband parrotfish (*Scarus iserti*, *S. taeniopterus* and *Sparisoma aurofrenatum*) were also much more abundant nearshore than offshore, with most individuals under 20 cm (Nemeth et al. 2006).

Relationship Between Herbivorous Fish Harvest and Macroalgae in St. Thomas/St. John
Parrotfish and surgeonfish landings for St. Thomas/St. John are bundled together, but mean percent algal cover data is only available for St. John. However, since the available data indicate herbivorous fish abundance and density is very similar between the two island groups, and the level of economic and coastal development is relatively the same between the two islands, we believe the environmental conditions in these areas are likely similar. Because of the similarities between the island groups we believe it is appropriate to assume the macroalgal cover estimates for St. John is also applicable to St. Thomas. The only period where data on mean algal cover and parrotfish/surgeonfish landings are both available is 2002-2007. The data indicate that as the total harvest of parrotfish and surgeonfish increased from 2002-2004, mean algal cover also increased. However, from 2005-2007 the total landings of these species landings decreased while mean algal cover spiked, and then returned to its second highest level since 2002 (Figure 5.2.3.1). NCCOS data on algal cover in St. John from 2002-2009 indicates that a statistically significant increase in algal coverage occurred from 2002-2010 (Appendix 1) despite reductions in both parrotfish and surgeonfish landings during that same time period.

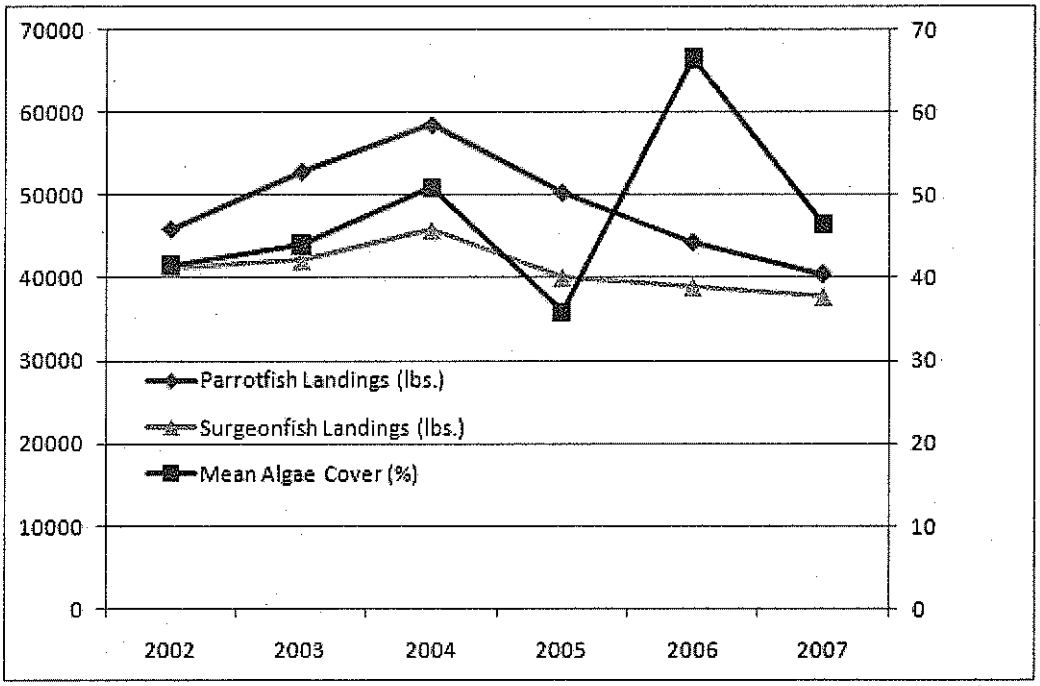


Figure 5.2.3.1 Mean Percent Algal Cover, Parrotfish, and Surgeonfish Landings in St.Thomas/St. John 2002-2007

The proposed parrotfish/surgeonfish ACLs will be lower than the levels of harvest observed for the last several years, but only marginally. Proposed future harvest levels will be very similar to the levels currently occurring. Current levels of harvest appear to have little direct impact on mean algae cover. Additionally, the limited information available on parrotfish/surgeonfish populations in the island group appears to indicate that they remain numerically abundant, albeit possibly skewed toward smaller individuals. Reduced harvest levels under the proposed action are anticipated to result in an increased number of herbivorous fish relative to the current amount. Regardless, we believe the continued harvest of parrotfish will reduce the overall grazing capacity of that population, as compared to an unfished assemblage. Thus, the proposed action is likely to adversely affect the St. Thomas/St. John *Acropora* critical habitat unit by reducing the herbivore assemblage's ability to mitigate increasing algal cover. However, the level of adverse effects are highly uncertain and unquantifiable, but based on comparable levels of harvest and the current population structure are expected to be less than effects in St. Croix but greater than the effects in Puerto Rico.

5.2.4 Synthesis of Effects to *Acropora* Critical Habitat from Continued Herbivorous Fish Harvest

As described in both the Status of the Species section and the Environmental Baseline section of this document, 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). Based on this information we believe that the continued harvest of surgeonfish and parrotfish, even at the reduced levels provided for by the ACLs is likely to adversely affect *Acropora* designated critical habitat. In this section we try to determine the magnitude of these effects.

If herbivorous fish harvest was the primary cause of the current phase-shifting (from coral dominated to algae dominated) in the three critical habitat units in the U.S. Caribbean, one would expect the St. Croix unit to show the greatest rate of phase-shift, indicated by significantly more algal cover, when compared to the Puerto Rico and St. Thomas/St. John units, based on the current harvest levels and size of the platforms being fished. The information in this section indicates that the phase-shift is just as severe in all three critical habitat units. This suggests that although harvest of herbivorous fish may be indirectly adversely affecting the essential feature, it is most likely not the driver of the phase-shift, but just one component of a larger function. As discussed in the Status of the Species section, diseases, temperature-induced bleaching, and physical damage from hurricanes are likely the greatest threats to elkhorn and staghorn corals' survival and recovery. These major threats are severe, unpredictable, likely to increase in the foreseeable future, and, at current levels of knowledge, unmanageable.

Aronson and Precht (2001) emphasize, however, that these Caribbean-wide changes in benthic assemblages were 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. In other words, the coral-dominated Caribbean reef system was resistant to reduced herbivory regimes for a period of time as long as corals maintained their occupation of space. However, when coral mortality occurred, macroalgae were able to pre-empt that space (especially following the loss of grazing by *Diadema*) and were subsequently resistant to coral re-colonization (Hughes and Connell 1999). Thus, the described shifts have been persistent on a decadal scale. Impacts to water quality (principally nutrient input) are also believed to enhance macroalgal productivity (*Acropora* BRT 2005).

Based on the above information, we believe the main reason for the phase-shift that currently affects the availability of the essential feature are effects to corals from diseases, elevated sea surface temperature, damage from hurricanes, combined with the loss of *Diadema*. The anthropogenic effects from herbivorous fish harvest and nutrient input have likely acted synergistically to exacerbate those factors. The proposed action is the continued harvest of herbivorous species at reduced levels through the ACLs. The extent to which the harvest of herbivorous fish is adversely affecting the availability of the essential feature is uncertain and currently unquantifiable; however, the information reviewed in this section indicates even unfished populations of herbivores are unlikely to completely reverse the current phase shift due to the magnitude of the other factors affecting reefs in the U.S. Caribbean. The ACLs and the prohibition of harvest of the three large-bodied parrotfish are expected to result in population increases of herbivorous fish, especially the large-bodied parrotfish (believed to be the most efficient Caribbean parrotfish grazers), throughout all three critical habitat units which means there will be

greater amounts of grazing under the proposed action than there were at the time of designation, when parrotfish and surgeonfish harvests were unrestricted. Therefore, the proposed action will continue to adversely affect *Acropora* critical habitat; however, those adverse affects are likely to be reduced by some amount that is currently unquantifiable.

5.3 Species Analysis – Effects of Herbivorous Fish Removals

The primary effect of herbivorous fish removal to *Acropora* colonies is the loss or reduction in the mitigative effect herbivorous fishes have on competitively dominant algae. Corals and macroalgae compete for space on reefs and interact through several mechanisms (McCook et al. 2001). Coral planulae cannot settle on macroalgae, and therefore the space occupied by macroalgae 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 macroalgae 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 macroalgae 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).

Of these effects from increased macroalgal cover, we believe that the most significant ones in terms of impacts to the conservation of *Acropora* are those which may affect the species' reproduction. (Although we note that all of these effects ultimately can affect reproduction: impacts on colony growth impact sexual fecundity and asexual potential, and partial and total colony mortality also obviously impact sexual and asexual reproduction potential.) As stated in the Atlantic *Acropora* Status Review Report (*Acropora* Biological Review Team, 2005):

> [T]here are several implications of the current low population sizes of Acropora spp. throughout much of the wider Caribbean. First, the number of sexual recruits to a population will be most influenced by larval availability, recruitment, and early juvenile mortality. Because corals cannot move and are dependent upon external fertilization in order to produce larvae, fertilization success declines greatly as adult density declines; this is termed an Allee effect (Levitan 1991). To compound the impact, *Acropora spp.*, although hermaphroditic, do not effectively self-fertilize; gametes must be outcrossed with a different genotype to form viable offspring. Thus, in populations where fragmentation is prevalent, the effective density (of genetically distinct adults) will be even lower than colony density. It is highly likely that this type of recruitment limitation (Allee effect) is occurring in some local elkhorn and staghorn populations, given their state of drastically reduced abundance/density. Simultaneously, when adult abundances of elkhorn and staghorn corals are reduced, the source for fragments (to

> provide for asexual recruitment) is also compromised. These conditions imply that once a threshold level of population decline has been reached (i.e., a density where fertilization success becomes negligible) the chances for recovery are low.

It was for these same reasons that facilitation of successful sexual and asexual reproduction in elkhorn and staghorn coral was identified by NMFS as the key objective to the conservation of these species. In the final *Acropora* critical habitat designation, NMFS (73 FR 72210) explained:

> Currently, sexual recruitment of elkhorn and staghorn corals is limited in some areas and absent in most locations studied. Compounding the difficulty of documenting sexual recruitment is the difficulty of visually distinguishing some sexual recruits from asexual recruits (Miller et al., 2007). Settlement of larvae or attachment of fragments is often unsuccessful, given limited amounts of appropriate habitat due to the shift in benthic community structure from coral-dominated to algae-dominated that has been documented since the 1980s (Hughes and Connell, 1999). Appropriate habitat for elkhorn and staghorn coral recruits to attach and grow consists of natural consolidated hard substrate. In addition to being limited, the availability of appropriate habitat for successful sexual and asexual reproduction is susceptible to becoming reduced further because of such factors as fleshy macroalgae overgrowing and preempting the space available for larval settlement, fragment reattachment, and recruitment. Similarly, sediment accumulating on suitable substrate impedes sexual and asexual reproductive success by preempting available substrate and smothering coral recruits. Also preempting space and exacerbating the effect of sedimentation is the presence of turf algae, which traps the sediment, leading to greater amounts of accumulations compared to bare substrate alone.

In Section 5.2, we have already assessed the likely effects of the proposed action on *Acropora* critical habitat and concluded that parrotfish and surgeonfish species continue to be relatively common components of reef assemblages under current harvest levels that are substantially higher than those for the proposed ACLs. Since there are predicted to be greater amounts of grazing under the proposed action than there were at the time of designation, when parrotfish and surgeonfish harvests were unrestricted, the proposed action would be expected to reduce the previously occurring level of adverse effects to critical habitat from herbivorous fish harvest. However, given the uncertainty in past, present, and projected parrotfish and surgeonfish populations, that conclusion had to remain qualitative. We cannot quantitatively assess the time course of the herbivorous fish population responses or the absolute or even relative increase in biomass. Thus, the magnitude and timing of any grazing-induced changes in algal cover caused by the proposed action remain uncertain.

The strength of the relationship between herbivory and the growth/spread of macroalgae in the region is currently unclear, and the population responses of herbivorous fish to lower harvest levels are also unclear. It is this uncertainty that makes it difficult to accurately estimate the magnitude and timing of grazing-induced changes in macroalgal cover in the U.S. Caribbean. However, information in the literature on the effects to fish populations following the establishment of MPAs does provide some basis for estimating the time frames required for the ecosystem benefits caused by increases in fish populations to be realized. Because the proposed action authorizes continued fishing for all parrotfish and surgeonfish, with the exception of midnight, blue, and rainbow parrotfish, we acknowledge that effects observed in these studies may well be stronger and more rapid than the changes we would anticipate in the U.S. Caribbean. However, we still believe these studies are useful because they provide some general information and reference point for evaluating population response timing.

McClanahan et al. (2007) reported scarid populations increased rapidly and reached a maximum level 15 years after implementation of an MPA. Acanthurid populations and calcifying algae cover increased steadily during 37 years of monitoring. Coincidentally, Selig and Bruno (2010) observed that after 15 years of implementation, MPAs in the Caribbean are associated with increasing coral cover. MPAs less than 15 years old and reefs open to fishing in the Caribbean demonstrated declines in coral cover. These findings suggest that, in general, with complete closures to fishing (although possibly with incomplete enforcement), improvements in coral reef health and overall coral cover brought on by the recovery of herbivorous fish populations from previous fishing pressure is likely to take over a decade. However, it should be noted that even prohibition of herbivorous fish harvest does not necessarily insure coral dominated reefs. For example, coral cover inside BIRNM is higher than outside, but macroalgal cover inside still remains high (i.e., 34-68%) and on par with macroalgal cover elsewhere in the U.S. Caribbean.

As discussed in Section 4 and Section 5.2, coral cover in the U.S. Caribbean has been declining and macroalgae cover has been increasing in conjunction with the previously unrestricted harvest of parrotfish and surgeonfish despite differing levels of parrotfish and surgeonfish harvest. Monitoring data sufficient to evaluate trends in elkhorn and staghorn corals, specifically are not available. The NCCOS and UVI/USVI DPNR data sets were not designed to monitor abundance or coverage of individual rare species. Therefore, it is not possible to derive quantitative estimates of acroporid abundance trends over the last decade. Qualitatively, we strongly believe that the same downward trends documented in overall coral cover have been also operant for these two species too. The site-specific declines in elkhorn and staghorn reviewed in Section 4 further support that conclusion. Macroalgae cover at sites surveyed had increased by 2010, with an average cover of 56% or greater. We believe the present reef conditions in the U.S. Caribbean are more likely a result of disease, elevated sea surface temperatures (a primary cause of bleaching), damage from hurricanes, and the *Diadema* die off. The anthropogenic effects from herbivorous fish harvest and nutrient input have likely exacerbated the situation, but those impacts are currently unquantifiable. Selig and Bruno's (2007) results suggest that coral recruitment, survival, and/or growth, as

evidenced by overall declining cover, are likely to continue to be negatively influenced for many years even after a complete prohibition of fishing. Also, we believe that even unfished populations of herbivores are unlikely to completely reverse the trends on the reefs due to the magnitude of the other factors affecting reefs in the U.S. Caribbean. As discussed in the Status of the Species section, the major threats (e.g., disease, elevated sea surface temperature, and hurricanes) to elkhorn and staghorn corals' persistence are severe, unpredictable, likely to increase in the foreseeable future, and, at current levels of knowledge, unmanageable (*Acropora* BRT 2005). Under the proposed action we anticipate rates of herbivory will increase, relative to those occurring currently. However, the rate at which the predicted increases in grazing will occur are likely to be slower than if harvest of parrotfish and surgeonfish were not authorized. Therefore, the proposed action is likely to adversely affect elkhorn and staghorn coral by reducing the overall grazing capacity of these populations, as compared to an unfished assemblage, by reducing the herbivore assemblage's ability to mitigate increasing algal cover which suppress both modes of their reproduction. Thus, the effects of the proposed action may do little to improve the coral recruitment, survival, and/or growth, at least in the short-term.

Current reproduction, especially sexual reproduction, is believed to be extremely low for elkhorn and staghorn corals. Due to the significant absence of quantitative information on parrotfish and surgeonfish biomass and rates of change in macroalgal cover with varying biomasses, the additional duration of these adverse effects to reproduction is not known, but, based on the discussion in this section; we believe it could be many years. However, over time, as herbivore populations continue to increase, we anticipate this effect would diminish.

5.4 Effects of Gear Deployment on Critical Habitat and *Acropora*

For the purpose of analyzing the effect of fishing gear used in the reef fish fishery, this analysis will examine the impacts to the species and critical habitat simultaneously. Without better estimates of coral colony density and location, it is difficult to accurately assess impacts to the species. Therefore, to be conservative, we assume *Acropora* colonies exist everywhere within the critical habitat area and examine the impacts of gear on both the species and critical habitat.

As noted in Section 2, while we anticipate indirect effects from the proposed action may occur in both Commonwealth/Territorial and federal waters, we anticipate direct effects will only occur to in the EEZ. Unlike our indirect effects analysis, we believe the direct effects from fishing (i.e., vessel operation, gear deployment, etc.) are just as likely to occur in all three areas of the U.S. Caribbean. Therefore, the following direct effect analysis considers the impacts of Caribbean reef fish fishing in the EEZs off Puerto Rico, St. Thomas/St. John, and St. Croix.

The reef fish fishery of the U.S. Caribbean uses hook-and-line, including bottom longline and vertical line gear; traps; and SCUBA, including speargun fishing. Hook-and-line fishing is generally not thought to have impacts to *Acropora* or critical habitat. Standard

vertical line fishing practices have the potential to impact *Acropora* or critical habitat through hooks snagging colonies or consolidated substrate/dead coral skeleton, or through the landing of weights or other pieces of gear on the benthos. However, because only 4.1% of critical habitat (the only area we anticipate *Acropora* to occur) is in the EEZ, it greatly reduces the amount of fishing effort that may occur where *Acropora* could occur. Additionally, we believe vertical line gear would not affect consolidated hard substrate in any manner that would appreciably alter the biological or physical characteristics which make them suitable for larval settlement or coral regeneration. Thus, we believe any adverse affects from vertical line fishing on *Acropora* colonies or its designated critical habitat would be discountable.

Since bottom longline fishing gear is purposely on the benthos, it has a greater chance of impacting and modifying the essential features of critical habitat or interacting with *Acropora*. However, bottom longline gear is used relatively infrequently in the U.S. Caribbean, and fishing occurs in waters deeper than where *Acropora* occur and critical habitat has been designated. Thus, bottom longline gear is very unlikely to interact with *Acropora* or its designated critical habitat, and any potential adverse effects are discountable.

Similarly, SCUBA-assisted fishing gear 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 *Acropora* corals are not a target species; thus, we believe it is likely that divers would be able to avoid directly taking these species. Additionally, because of *Acropora*'s branching morphology, SCUBA divers that do encounter these colonies would likely try to avoid them to reduce any chance of fishing gear becoming fouled. Thus, we believe adverse effects from SCUBA fishing extremely unlikely and discountable.

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 *Acropora* 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 *Acropora* critical habitat because that space has been preempted by the trap making it unavailable for the settlement and growth of corals. Thus, the use of traps may affect critical habitat and *Acropora* through breakage or other physical damage. The physical impacts of traps on *Acropora* and critical habitat in the U.S. Caribbean 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 *Acropora* critical habitat in the U.S. Caribbean. For this analysis, we assume that damage to critical habitat and *Acropora* species are equivalent.

In a survey of the distribution of trap locations, Sheridan et al. (2005), found coral to be the dominant habitat type for trap deployment (54%). Estimates of the number of traps fished in the U.S. Caribbean vary from approximately 5,500 traps (in the USVI alone) (B. Arnold, NMFS, to A. Herndon, NMFS, pers. comm. 2011) to approximately 8,500 traps

(throughout USVI and Puerto Rico) (Sheridan et al. 2003 and 2006). Regardless of the estimate of total number of traps, the available data indicate that fishers do not use all their traps simultaneously (Sheridan et al. 2006). Sheridan et al. (2005) found trap-caused damage at about 50% of all traps visited. Instances of damage (scrapes, breakage) were most prevalent among gorgonians and sponges (90 instances), followed by corals (25 instances, 13.7%). Scharer et al. (2004) report the dimensions of fish traps in Puerto Rico range between 32 and 96 inches in length by 18 to 60 inches in width (576 sq in and 5,760 sq in, respectively).

Kojis (2004) conducted a census of all commercial fishermen in the USVI. The census reported that 160 commercial fishermen were licensed in the St. Thomas/St. John and 223 were licensed in St. Croix. The census requested licensed fishermen fill out a questionnaire to gain a better understanding of the fishery. Response rates in St. Thomas/St. John were close to 70% and were approximately 100% in St. Croix. The questionnaire indicated that 69 commercial fishermen in St. Croix fished with fish traps and 29 used fish traps in St. Thomas/St. John.[5] Since only 70% of fishermen in St. Thomas/St. John responded to the questionnaire we assume the total number of trap fishermen in the area is underrepresented. Therefore, our calculation of the total traps hauled annually (see below) uses an estimate of number of trap fishermen that corrects for this underrepresentation.[6] Since 100% of all commercial fishermen in St. Croix filled out the questionnaire we will assume any fisher using fish traps in that area were identified. Agar et al. (2005) identified 324 trap fishermen in Puerto Rico.

Kojis (2004) also reported that St. Thomas/St. John fishermen fish an average of 81 traps/pots with a mean soak time of 156 hours; St. Croix fishermen used 16.2 traps/pots on average, with a soak time of 91.5 hours. Agar et al. (2005) report St. Thomas/St. John fishermen take an average of 1.4 trips/week and haul 68 traps per trip, St. Croix fishermen take 2.5 trips/week and haul 26 traps per trip; and Puerto Rico fishermen take 2.1 trips/week and haul 27 traps per trip.

Matos-Caraballo and Agar (2011) provided an updated census of the Puerto Rican fishery in 2008. The 2008 census does not provide information on number of trips per week or traps hauled per trip, or the number of traps used per fishermen. However, it does provide an indication that the total number of fish traps in the fishery has declined approximately 56%, from 10,372 in 2002 (Matos-Caraballo et al. 2002) to 4,574 fish traps in 2008 (i.e., not spiny lobster or deepwater-snapper traps) (Matos-Caraballo and Agar 2011). Since the 2008 census does not provide information on number of trips per week or traps hauled per trip, it is difficult to know what the overall impact is from a declining number of traps. If the number of fish traps has declined, but the remaining fish traps are being used more frequently (i.e., the number of trips/week or traps hauled/trip has gone up), it is possible that the potential impacts have not changed even

[5] "Fish traps" refer to traps used to target fish. These do not include spiny lobster traps, fish traps modified to catch lobster, or deepwater shrimp trap/pots.

[6] Our analysis assumes that if the 29 trap fish respondents from St. Thomas/St. John represenst only 70% of fishermen, than there were likely 41 total fish trap fishermen (29 Trap Fishermen Respondents ÷ 70% of Total Number of Trap Fishermen = 41 Total Trap Fishermen).

with fewer fish traps in the fishery. However, negative economic influences (i.e., gas prices, gear costs, vessel and dock prices, etc.) have been keeping more and more fishers from fishing (E. Pinerio, CFMC, to B. Arnold, NMFS, pers. comm. 2011). Additionally, younger fishers prefer diving because it is more exciting and economical, further reducing trapping effort (E. Pinerio, CFMC, to B. Arnold, NMFS, pers. comm. 2011). Based on this information, we believe it is likely that fewer traps likely mean fewer potential impacts. Therefore, we reduced our estimate of the total traps hauled per year in Puerto Rico by 56% to account for this updated information.

Using these data we developed several approaches for estimating potential number of fish traps used in the fishery annually. The first approach yielded the highest estimate for the number of traps used, the second approach yielded an intermediate number, and third the approach ultimately produced the lowest number of traps. All of our estimates calculated the number of traps likely used in a single week, and then multiplied that number by 52 to estimate the total number of traps used annually. While we acknowledge that it is unlikely that every fisher would fish all 52 weeks in a year, we chose this approach to avoid underestimating the potential impacts of the species.

The first estimate used the information on trips per week and the number of traps hauled per trip, to estimate of the number traps hauled per week and annually (Table 5.4.1). Agar et al. (2005) report St. Thomas/St. John fishermen take an average of 1.4 trips/week and haul 68 traps per trip, St. Croix fishermen take 2.5 trips/week and haul 26 traps per trip; and Puerto Rico fishermen take 2.1 trips/week and haul 27 traps per trip.[7] Using these data we estimated the number of traps hauled per week and annually (Table 5.4.1).

Table 5.4.1. Estimated Total Number of Traps Hauled per Week Using Number of Traps Hauled

Area	No. of fishermen	Trips/week	Hauled traps	Total traps hauled/week	Total traps hauled/yr
St. Thomas/St. John	41	1.4	68	3,903	202,956
St. Croix	69	2.5	26	4,485	233,220
Puerto Rico	324	2.1	27	18,371	420,315*
Total	435	--	121	26,854	856,491

*This number includes a 56% reduction to account for the updated 2008 census data.

The second estimate was similar to the first, but based on slightly different data. This estimate used the information available on the average number of traps fished per fisher in the USVI (i.e., Kojis 2004), in place of the number of traps hauled per trip provided by Agar et al. (2005), to recalculate the number of traps hauled for St. Thomas/St. John and St. Croix. Information for average number of fish pots fished in Puerto Rico was not provided in Kojis (2004); thus, we did not re-estimate the total traps hauled/year for this area under the second approach. Table 5.4.2 presents the new calculations for the USVI, but the information for Puerto Rico is the same as presented in Table 5.4.1.

[7] Number of Fishermen x Number Trips per Week x Number Traps Hauled x 52 weeks = Total Traps Hauled per year

Table 5.4.2. Estimated Total Number of Traps Hauled per Week Using Number of Traps Reported in the U.S.V.I

Area	No. of fishermen	Trips/week	Avg. Fish Traps Fished (Hauled Traps)	Total traps hauled/week	Total traps hauled/yr
St. Thomas/St. John	41	1.4	81	4,649	241,748
St. Croix	69	2.5	16.2	2,795	145,314
Puerto Rico	324	2.1	27	18,371	420,315*
Total	435	--	121	25,899	807,377

*This number includes a 56% reduction to account for the updated 2008 census data.

The final approach for estimating trap impacts was based on soak times as reported by Kojis (2004). As noted above, St. Thomas/St. John fishermen soaked their traps an average of 156 hours; St. Croix fishermen soaked traps 91.5 hours on average. Therefore, by dividing the average soak time by the total number of hours in a week we could estimate the total number of likely trips made. Next, we multiplied the number of fishers by the number of trips per week. Then we multiplied the average number of traps fished to estimate the number of traps hauled each week. That estimate was then multiplied by 52 to estimate the total number of traps hauled annually.[8] As with our previous estimate, soak time information was not available for Puerto Rico; thus, we did not calculate traps hauls based on soak times for that area. Adding our estimates from Puerto Rico would add an additional 420,315 traps.

Table 5.4.3. Estimated Total Number of Traps Hauled per Week Using Soak Times Reported in ihe U.S.V.I.

Area	Avg. Soak Time (Hrs)	Trips /week	No. of fishermen	Avg. Fish Traps Fished	Total traps hauled/week	Total traps hauled/yr
St. Thomas/St. John	156	1.1	41	81	3,653	189,956
St. Croix	91.5	1.8	69	16.2	2,012	104,624
Puerto Rico	--	2.1	324	27	18,371	420,315*
Total	--	2.9	434	--	24,036	714,895

*This number includes a 56% reduction to account for the updated 2008 census data.

All three methods used similar data and ultimately produced a range (i.e., 856,491 to 714,895) for the potential number of traps hauled. Since it is difficult to determine which number is the most accurate, we have chosen to select the estimate that was the highest to err on the side of the species. Selecting the higher number helps ensure that we do not underestimate the potential adverse affects from traps.

For this analysis, we assume coral habitat is analogous to designated critical habitat for *Acropora*, as coral habitat is identified as having many of the same features as those essential features identified for critical habitat. 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

[8] (Hours/week ÷ Avg. Soak Time) x Number of Fishermen x Avg. Number Traps Fished x 52 weeks = Total Traps Hauled per year

unconsolidated, nor do they cause growth of macroalgae or cause sedimentation in and of themselves, we believe it is unlikely that traps would affect this portion of the essential feature. However, we do 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) reported 54% of these traps were in coral habitat. For this analysis, we assume coral habitat is analogous to designated critical habitat for *Acropora*, as coral habitat is identified as having many of the same features as those essential features identified for critical habitat. Thus, 462,505 traps were placed and hauled from coral habitat. Sheridan et al. (2005) also indicate that of the traps deployed on coral habitat traps cause damage to 13.7% of corals. With 462,505 traps potentially affecting 13.7% of coral and critical habitat, we estimate 63,363 traps cause damage to *Acropora* and critical habitat. Traps used range in size from 576 sq in to 5,760 sq in and average 3,168 sq in. Applying this information to the 63,363 traps expected to cause damage, impacts to *Acropora* and critical habitat are expected to be no more than 0.091 sq mi (assuming all average sized = 0.050 sq mi per year, range = from 0.009 sq mi to 0.091 sq mi).[9] However, only 4.1% of designated critical habitat occurs in the EEZ in the U.S. Caribbean. Therefore, we anticipate direct effects from fishing occurring in the EEZ will occur to 0.004 mi^2 of *Acropora* and critical habitat. [10]

5.5 Effects of Vessel Anchors on Critical Habitat and *Acropora*

In this analysis we will examine the impacts to the species and critical habitat simultaneously due to the lack of data on coral colony location and the difficulty in separating out impacts to critical habitat versus impacts to species. Here we examine the number of trips taken by vessels targeting reef fish, apply factors for the percentage of those vessels deploying anchors in coral 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 U.S. Caribbean reef fish fishery.

Recreational Sector

The Marine Recreational Fishing Statistical Survey (MRFSS) data provide an attempt at accounting for the recreational harvest of reef fish in Puerto Rico, which is generally considered to be significant. A summary of all available information for Puerto Rico from the recreational sector targeting reef fish, including number of participants, number of trips taken by mode (shore, charter, and private boat), and the total catch (all species reported) from 2000 to 2008 is presented in Table 5.5.1. A relatively flat trend in number of reef fish fishing trips and pounds landed is present from 2000-2008, except for an as-yet unexplained pounds landed anomaly in 2006. The percent of shore trips (53-61%) is

[9] 462,505 traps x 13.7% of traps affecting coral = 63,363 traps expected to cause damage; 63,363 traps causing damage x 576/3,168/5,760 in^2 footprint per trap = 0.009/0.050/0.091 mi^2 of impacts.
[10] 0.091 mi^2 of critical habitat affected by traps x 4.1% of critical habitat occurring in EEZ = 0.0037 mi^2

always higher than the percent of trips in private boats (36-45%), which in turn is always higher than the number of charter trips (1-3%).

Table 5.5.1 Number of Recreational Reef Fish Trips Taken in Puerto Rico

Year	Charter	Private	Shore	Total Trips	No. of Participants
2000	16,899	522,914	792,890	1,332,703	249,868
2001	10,919	504,349	896,675	1,411,943	222,128
2002	34,277	572,844	693,938	1,301,059	237,995
2003	21,764	471,741	617,900	1,111,405	219,910
2004	22,028	389,469	638,802	1,050,299	163,833
2005	17,969	379,910	468,843	866,722	141,743
2006	16,906	386,111	493,565	896,582	213,005
2007	10,734	453,907	615,455	1,080,096	185,429
2008	12,623	362,739	423,190	798,552	149,544

The most recent report on recreational fishing activity in USVI waters (Tobias and Dupigny 2009) reviews the information available for the area, including the surveys on the recreational fishing activity in general, including reef fish. Most of the information on recreational fisheries for the USVI derives from offshore billfish and other pelagic fisheries since the area is well known for gamefish. Tobias and Dupigny (2009) summarize the information on the latest recreational fishing survey targeting the pelagic fleet. None of the reports on the recreational fishing activity in the USVI target the fleet harvesting reef fish, lobster, or conch. Telephone surveys targeting boat-based and shore fishers provide an estimate of 10% of the USVI population participating as recreational fishers (Jennings 1992, Mateo 1999). In all cases, pelagic species are the most commonly targeted (Tobias and Dupigny 2009). Based on this information, we believe recreational fishing targeting reef fish off the USVI is rare, and effects on staghorn and elkhorn coral from associated anchoring area discountable. Therefore, our analysis of the effects that recreational fishing attributed to anchoring has on staghorn and elkhorn corals is solely based on Puerto Rico fishing effort.

Using the information we have from Puerto Rico on the number of private trips and charter trips as well as information from Mac et al. (1998), we can estimate an impact from vessels fishing recreationally for reef fish. Mac et al. (1998) conducted a survey of boats in 1987 that revealed 14% of boats were anchored in coral communities and that about 40% of the anchors on these vessels caused damage. Therefore, if we sum the number of private and charter trips, average that number, and then multiply that number by the percent anchored in coral communities (i.e., 14%), we arrive at a number of anchors in coral habitat. If we then apply the percentage of anchors causing damage (i.e., 40%) to the estimate for the number of anchors in coral habitat, the result is our estimate for the total number of trips with anchor/coral interactions (Table 5.5.2). 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. Table 5.5.2 summarizes these calculations by year.

Anchoring dragging could cause additional damage to reefs. However, reef fishing vessels 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.

Table 5.5.2. Estimated Anchoring Impacts to Coral From Recreational Fishing

Year	Total trips	Trips anchoring on coral	Anchors causing damage	Area of impact (sq ft)	Area of impact (sq mi)
2000	539,813	75,574	30,230	95,525	0.00342
2001	515,268	72,138	28,855	91,182	0.00327
2002	607,121	84,997	33,999	107,436	0.00385
2003	493,505	69,091	27,636	87,331	0.00313
2004	411,497	57,610	23,044	72,819	0.00261
2005	397,879	55,703	22,281	70,409	0.00252
2006	403,017	56,422	22,569	71,318	0.00255
2007	464,641	65,050	26,020	82,223	0.0029
2008	375,362	52,551	21,020	66,424	0.00238
Total	4,208,103	589,134	235,654	744,666	0.02671
Average	467,567	65,459	26,184	82,741	0.00296

Commercial Sector

We can perform the same analysis used for recreational vessels to estimate the anchoring effects for commercial vessels. Table 5.5.3 contains Puerto Rico's commercial fleet trip information. Table 5.5.4 contains the USVI commercial fleet trip information. These estimates of reef fishing vessels do not distinguish between gear types (i.e. trapping or line) because when conducting our anchoring impacts analysis the type of reef fishing gear used by the vessel is not important.

Table 5.5.3. Reported Commercial Reef Fish Fishing Trips per Year by Coast in Puerto Rico (Matos-Caraballo 2007).

Coast	Trips Per Year			Percent of Total Trips Per Year		
	2004	2005	2006	2004	2005	2006
North	3,414	2,502	1,537	10.4	9.0	5.9
South	8,965	7,281	8,075	27.4	26.1	31.1
East	5,319	3,790	2,923	16.3	13.6	11.2
West	14,977	14,320	13,461	45.8	51.3	51.8
Total	**32,675**	**27,893**	**25,996**	**100.0**	**100.0**	**100.0**

Table 5.5.4. Estimated number of reef fish trips in the USVI.

Island	Number of fishers	Avg. trips per week	Number of trips in a year
St. Croix	223	3.3	38,267
St. Thomas/St. John	160	2.6	21,632

After we have established the number of trips, we can now estimate the impact of anchoring using the same assumptions and approach as above (Table 5.5.5).

Table 5.5.5. Estimated Anchoring Impacts to Coral From Commercial Reef Fish Vessels

Island	Total Trips	Trips anchoring on coral	Anchors causing damage	Area of impact (sq ft)	Area of impact (sq mi)
St. Croix	38,267	5,357	2,143	6,772	0.00024
St. Thomas/ St. John	21,632	3,028	1,211	3,828	0.00013
Puerto Rico	28,854	4,040	1,616	5,106	0.00018
Total	88,753	12,425	4,970	15,706	0.00056

From the analysis of recreational and commercial fishing vessels anchor impacts we see that on average 0.00352 mi^2 of coral and critical habitat are impacted each year, equivalent to 9,143 m^2. However, only 4.1% of designated critical habitat occurs in the EEZ in the U.S. Caribbean. We anticipate direct effects from fishing occurring in the EEZ will occur to 0.00014 mi^2 of to *Acropora* and critical habitat.[11]

5.6 Effect on Sea Turtles

Basic Approach to the Sea Turtle Assessment

We began our analysis of the effects of the action by first evaluating what activities and gear types/techniques are likely to adversely affect sea turtles. We determined adverse effects of the Caribbean reef fish fishery on sea turtles result from interactions (i.e., physical contact with) with fishing gear or associated moving vessels leading to the capture, injury, or death of sea turtles. In NMFS (2005a), we determined there would only be adverse effects from reef fish fishing gear on listed species; we did not expect adverse effects attributed to vessel strikes. However, with newly acquired U.S. Caribbean stranding data showing vessel strikes are the most common identifiable cause of strandings in the action area and with increasing awareness of this growing problem Atlantic-wide (e.g., see Foley et al. 2008), in this opinion we make a first attempt at estimating the Caribbean reef fish fishery's possible contribution to this problem.

No indirect effects are expected. As noted earlier, indirect effects include aspects such as habitat degradation, reduction of prey/foraging base, etc. The operation of the U.S. Caribbean fisheries (i.e., vessel operations, gear deployment and retrieval) is not expected to impact the water column or benthic habitat in any measurable manner. Unlike mobile trawls and dredges that physically disturb habitat as they are dragged along the bottom, the gears used in the U.S. Caribbean fisheries are suspended in the water column or essentially stationary on the bottom and do not affect water column or benthic habitat characteristics. Also, while sea turtles may prey on dead fish, fish are not the primary prey of sea turtles; thus, a reduction of prey/foraging base is also not likely.

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 in NMFS (2005a) we determined that

[11] 0.00352 mi^2 x 4.1% of critical habitat in the EEZ = 0.00014 mi^2 of impact in the EEZ

other fishing methods in the U.S. Caribbean (i.e., by hand and spear and power head gear) 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 these gears 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. Thus, 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 be able to avoid aiming in any direction where sea turtles are within their striking range. Our review of strandings records for this consultation did turn up some records of sea turtles being speared, but these records are believed to stem from deliberate illegal activity and not be incidental to legal fishing.

NMFS (2005a) estimated 22 (6 leatherback, 8 hawksbill, and 8 green) sea turtles would be incidentally captured as a result of the continued authorization of the Caribbean reef fish fishery managed under the CRFFMP: 4 leatherback sea turtles in trap lines, 2 green, 2 hawksbill, and 1 leatherback sea turtle in longlines; and 5 hawksbill, 5 green, and 1 leatherback sea turtle in vertical lines. In the absence of U.S. Caribbean-specific sea turtle interaction data, incidental captures were estimated via extrapolating sea turtle bottom longline and vertical line bycatch rates stemming from fishermen-reported interactions in the Gulf of Mexico reef fish fishery and Gulf of Mexico stranding data, using a proxy for the total amount of fishing effort by gear type in the U.S. Caribbean EEZ reef fish fisheries. A lot of the bycatch information and analysis methods stemmed from a 2005 opinion on the Gulf of Mexico reef fish fishery (NMFS 2005b).

In late 2008, SEFSC completed a report based on new Gulf of Mexico reef fish observer program data which indicated that sea turtle bycatch rates observed in Gulf of Mexico reef fish bottom longline gear were much higher than previously estimated using fishermen-reported interaction data. In late 2009, NMFS completed a new opinion or the Gulf of Mexico reef fish fishery which included new bottom longline and vertical line sea turtle capture estimates based on the extrapolation of the new observer program data.

The spatial overlap of fishing and individual sea turtle species in the Caribbean is likely quite different than that in the Gulf of Mexico. Given we've learned that the Gulf of Mexico hook-and-line gear interacts primary with loggerhead sea turtles, which are believed to only rarely occur in U.S. Caribbean waters, it seems inappropriate to use the updated Gulf of Mexico reef fish loggerhead bycatch data for re-estimating sea turtle bycatch in the Caribbean reef fish fishery. Therefore, in preparing this opinion, we searched extensively to uncover historic and recent sea turtles strandings and any new sea turtle bycatch information not included in the previous opinion to try and find location-specific data on which to base our effects analysis.

The Sea Turtle Stranding and Salvage Network (STSSN) was formally established in 1980 to collect information on and document strandings of marine turtles along the U.S. Gulf of Mexico and Atlantic coasts. A stranding is any dead sea turtle that is found

floating or washed ashore or any live sea turtles that are found with life-threatening problems (e.g., sick, injured, or entangled). The location of the stranding when first reported is the point location that appears in this database and may or may not be the location at the time of injury or death. Sea turtles that are known to be captured incidental to some activity (i.e., observed bycatch in commercial fisheries, research projects, power plant operations, etc.) are not included in the database.

Although STSSN technically encompasses portions of the U.S. Caribbean, queries of the online database reveal no data for the U.S. Caribbean. However, vessel and fishing-related sea turtle historic and recent data were obtained via networking, searching for related publications, and by contacting PR and USVI DNER and USFWS staff directly for any unpublished data. In Sections 5.6.1 and 5.6.2, we provide an overall summary of the data we acquired for the USVI and for Puerto Rico, respectively. Data prior to implementation of the Caribbean SFA document (pre-2005) is included here for historical perspective because these data have not been presented previously and because of the dearth of data overall. In Section 5.6.3-5, we then analyze the effects of the proposed action's traps, hook-and-line, and vessels on sea turtles, using the newly acquired Caribbean data whenever possible. The following gear and vessel analyses for sea turtles are all based on past interaction levels documented and do not anticipate any future changes associated with the proposed action. This is because the proposed changes to reef fish management are not expected to change overall trap, hook-and-line, or vessel effort in the fishery from at least the recent past levels and these data still represent the best available information on which to project future effects from the U.S. Caribbean reef fish fishery.

5.6.1. Summary of New Sea Turtle Fishery-Related Data Available for the USVI

MRAG Americas, Inc. Pilot USVI Observer Studies (MRAG 2006a and 2006b)

In 2004-2005, MRAG Americas, Inc. (MRAG) conducted feasibility studies for deploying observers in the St. Croix (MRAG 2006a) and St. Thomas fisheries (MRAG 2006b). The studies also aimed to provide preliminary assessment of the magnitude of bycatch and discards resulting from St. Croix and St. Thomas fishing. In St. Croix, at-sea observing began in late October 2004 and continued through February 2006. Of the 190 licensed fishers then registered from St. Croix, 120 were considered full-time and active (William Tobias, DFW, pers. comm.). Observer data were obtained from 20 of those licensed fishers, representing approximately 17% of full-time and 11% of total permits. During that period of observer deployment, the project observed trips, including 10 fish trap trips, 6 handline trips, 8 net trips, 5 longline trips, and 11 spear/snare trips. An additional 10 samples were taken from captain trips, in which fishers brought the total catch to shore for assessment. Of the 160 licensed fishers registered from St. Thomas, a subset of about 42% of the 50 full time fishers allowed observers on fishing vessels or agreed to bring in captain samples. Observers sampled 28 trips from nine individual captains, including 10 fish trap trips, 9 lobster traps, and 9 handline trips; data were also collected from an additional 16 fish trap trips, 11 handline trips, and 2 longline trips via captain samples (B. Trumble, pers. comm. 2010). During both studies, the observers did

not encounter fishing gear interactions with any listed species and no listed species were reported by captains. However, given the small proportion of participating fishers and voluntary nature of their participation, results may not be representative of the St. Croix and St. Thomas fleets.

Interactions Between Sea Turtles and Commercial Fishermen of the United States Virgin Islands. Per Fishermen Interviews and Analysis of 1994-2003 Stranding data (Lewis et al. 2007)

In 2004, a study was conducted to assess the interactions of sea turtles in the USVI with commercial fishing gear using fishermen interviews and an analysis of territorial stranding data collected by the USVI Department of Planning and Natural Resources Division of Fish and Wildlife (DPNR-DFW) (Lewis et al. 2007). According to the 2003-2004 commercial fishermen registry of the USVI DPNR-DFW, there were approximately 210 commercial fishermen on the island of St. Croix and about 140 commercial fishermen in the St. Thomas/St. John district. Local fishing gears include fish traps (pots), gill and trammel nets, seine nets, and hook-and-line. Fishers also free dive and SCUBA dive to collect invertebrates and to spearfish. Fishing off the coast of St. John is restricted by a number of protected and "no take" zones.

For the interview component of the study, 30% of the 210 registered commercial fishers on St. Croix (n=63) and 30% of the 140 registered commercial fishers on St. Thomas/St. John (n=42) were interviewed during the summer of 2004 to assess the interactions of sea turtles with fishing gear. Information received from interviewees included the type of gear used (i.e., trap, fishing line, gillnet, trammel net, and seine net). Areas fished were also recorded using the 13 fishing zones designated throughout the territory by USVI DPNR-DFW. Interactions were divided into five frequencies: never, rarely (once in five years), occasionally (2-5 times in five years), somewhat frequently (6-10 times in five years), and frequently (>10 times in five years). Fishers were also asked to identify the species of any entangled or hooked sea turtles and the method of release they used. For the stranding component of the study, ten years of sea turtle stranding data (1993-2003) for the USVI were analyzed to determine: (1) how frequently strandings occurred, (2) the types of injuries that caused strandings, (3) which species and age stranded most often, and (4) the distribution of strandings by island. Boating and fishery-related strandings were defined by the type of injury that caused death, which included propeller wounds, boat strikes, entanglements, hook-related injuries, poaching-related injuries, or spear-related injuries.

Approximately half of the fishers interviewed for both fishing districts in the USVI (i.e., 56% of the registered commercial fishermen interviewed on St. Croix and 47% of those interviewed on St. Thomas) reported that they had never had interactions between sea turtles and their fishing gear. Only 5 of the interviewed fishermen fished off the coast of St. John, so that data were not presented or included in analyses. Approximately 35% of the fishers interviewed for both fishing districts in the USVI reported rare and occasional interactions. Rarely occurring interactions (once in five years) were reported by 26% (n=13.38) of the fishers on St. Croix and 29% (n=12.18) of the fishers on St. Thomas.

For both islands, 7% (n=4.41 and n=2.94) of the fishers reported occasional interactions with sea turtles and their gear and none of the fishers reported somewhat frequently occurring interactions. Only 15% of the interviewees (11% [6.93] and 18% [7.56] of those fishers surveyed on St. Croix and St. Thomas, respectively) reported frequent interactions (2 or more per year). Those fishermen that reported frequent interactions were primarily net fishers, with all but one of the net fishers interviewed reporting frequent interactions with sea turtles and their gear. In fact, one fisherman recalled catching a sea turtle in his gear almost every time he set his nets.

Interactions between sea turtles and specific gear types for both islands had statistically significant differences. Twenty-one percent of the line fishers on St. Croix and 29% of those on St. Thomas reported interactions with sea turtles and their gear. Although no sea turtle interactions with trap buoy lines were reported for St. Croix, 25% of the trap fishers interviewed on St. Thomas reported interactions between sea turtles and their gear. For both islands, all but one of the net fishers had interactions with sea turtles and their gear (91% and 92% for St. Croix and St. Thomas, respectively).

On St. Croix, interactions were slightly more frequent with green sea turtles (42%) and leatherbacks (35%) than with hawksbills (23%), but there was no statistically significant difference among species that interacted with gear. On St. Thomas, interactions occurred most frequently with hawksbills (47%) and greens (37%) and less with leatherbacks (17%), and there was a statistically significant difference among species on St. Thomas.

Fishermen were also asked to describe methods of release of incidentally captured turtles. They reported removing hooks from the esophagi of sea turtles, unhooking those that were accidentally snagged, and untangling sea turtles from nets or lines. However, with gill and trammel nets, fishermen occasionally reported having to cut their nets to free incidentally captured sea turtles.

The stranding analysis conducted by Lewis et al. (2007) documented that between the years of 1994 and 2003, there were 56 boating and fishery-related sea turtle strandings reported for the USVI. More boating and fishery-related strandings (n=36) were reported for St. Croix than for the other two islands combined. There were 13 strandings reported for St. Thomas and only 6 for St. John. For all three islands, there was a statistically significant site difference among the boating and fishery related injuries that caused sea turtle strandings. Boating-related injuries (propeller wounds and boat strikes) were documented on 30 of the total strandings reported. There were 16 strandings caused by boating-related injuries on St. Croix, 9 on St. Thomas, and 5 on St. John. Of the 10 strandings caused by entanglement, 8 were on the island of St. Croix, 1 was on St. Thomas, and 1 was on St. John. Together, poaching- and spear-related injuries accounted for 13 of the stranded sea turtles over the ten year period. Remains of 8 poached and 3 speared turtles were found on St. Croix while remains of one poached and one speared turtle were found on St. Thomas.

Other Fishing and Vessel-Related Sea Turtle Stranding Data Sets/Analyses

1982-1997 (Boulon 1998)
Since 1982, the USVI Division of Fish and Wildlife (DFW) has maintained records of reported strandings of sea turtles in the Virgin Islands. The USVI DFW defines a stranding as any sea turtle which is found dead for any reason or is recovered from a compromised situation and released back into the wild. Strandings are generally reported by citizens and followed up on by DFW staff. Given the opportunistic manner in which strandings are reported, the number of stranded sea turtles reported likely does not include all of the strandings for the USVI. By relying on the reporting by individuals, some stranded sea turtles may be observed without being reported or just not observed. However, the reported strandings are probably reflective of the species composition, distribution, and relative causes of stranding for sea turtles in the Virgin Islands.

Boulon (1998) summarized 1982 through 1997 sea turtle stranding records from the USVI by species, island, and cause and then evaluated the data for trends. Strandings were sorted into five categories: boat strikes, fishing gear, poached, other, and unknown. Boat strikes included strandings with obvious crushed carapaces or deep cuts from a propeller. "Other" was assigned as the cause of the stranding when the cause was identifiable but not frequent enough to warrant its own category. Unknown was assigned when no external cause of mortality was evident and for which, if a necropsy was performed, no internal cause of mortality was determined.

At least 122 sea turtle strandings were documented during 1982-1997, including 79 green, 38 hawksbill, and five leatherback sea turtles. Of these reported strandings, 56 (46%) were from St. Croix, 46 (38%) were from St. Thomas and 20 (16%) were from St. John. Green sea turtles were the most commonly stranded species on both St. Thomas and St. Croix, while St. John had equal numbers of greens and hawksbills reported. St. Croix had the greatest number of hawksbills reported and also had all of the leatherback strandings. Annual reported strandings ranged from one to 25 sea turtles with a trend showing a gradual increase in reported strandings. Boulon (1998) in Abreu-Grobois et al. (2000) hypothesized that the increasing trend in strandings was attributed to both increases in sea turtle populations and human populations, as well as an increase in general public awareness of problems with our natural environment, resulting in more people likely to report a stranded turtle.

Boulon (1998) found boat strikes accounted for the greatest number of strandings (34.43%) followed by undetermined causes (29.51%), poaching (13.11%), "other (i.e., identifiable reasons that were not frequent enough to be in their own category) (12.3%) and fishing gear entanglement (10.66%)." Most green sea turtle strandings were due to boat strikes while hawksbill strandings were mostly from undetermined causes and leatherbacks were from poaching. The primary cause of strandings in St. Thomas and St. John was from boat strike; in St. Croix it was unknown with poaching being the second greatest cause. The numbers of reported boat strikes per year also showed an increase over time. There was no indication of any seasonality trends with the exception of leatherbacks which were all adult strandings during the nesting season.

Boulon (1998) noted that the known causes of stranding tend to follow certain logical suppositions about sea turtle habits: (1) more greens were documented stranded due to boat strikes because they are more likely to be found in shallow bays where boats are more commonly operated, (2) more boat strikes occur on St. Thomas likely because there are more boats there, (3) hawksbills are likely found poached because they are the most common nesting turtle in the USVI, (4) more sea turtles have died due to encounters with fishing gear in St. Thomas because there is more fishing activity there, and (5) leatherbacks all stranded on St. Croix where nearly all of the nesting takes place.

Table 5.6.1.1 shows more detailed information on the USVI fishery-related sea turtle strandings compiled by Mr. R. Boulon (USVI DPNR). There were 15 USVI fishery-related sea turtle strandings, including 10 green sea turtles, 4 hawksbill sea turtles, and one leatherback sea turtle. Of these, six were entangled in fishing line, five were entangled in net, two via trap use, and two were the result of poaching.

Table 5.6.1.1 USVI Fishery Related Sea Turtle Strandings: 1982-1997
(Compiled by R. Boulon, Division of Fish and Wildlife, DPNR)

Year	Island	Species	Cause
1982	St. Croix	Green	Caught in fishing line, drowned
1984	St. John	Green	Dead in fish trap tunnel
1987	St. Thomas	Hawksbill	Entangled in fishing line
1987	St. Thomas	Green	Entangled in fishing line
1988	St. Thomas	Green	Caught in an abandoned fish trap
--	St. Croix	Hawksbill	Spear through neck; poached
1991	St. Thomas	Green	Entangled in fish net
1991	St. Thomas	Green	Entangled in fish net
1991	St. Croix	Green	Entangled in fish net
1992	St. Croix	Hawksbill	Entangled in fishing line
1995	St. Croix	Hawksbill	Entangled in fishing line
1995	St. John	Green	Entangled in netting- released
1995	St. Croix	Leatherback	Entangled in netting-drowned
1995	St. Thomas	Green	Entangled in fishing line
1997	St. Croix	Green	Entangled in spear gun line, strangled, poached

2001 through 2006[12]

There were 16 strandings documented on St. Thomas, including 10 green sea turtles, 3 hawksbill, and 3 unknown. Of these, three (all green sea turtles) were attributed to vessel strikes (i.e., propeller damage) and one (an immature female green) drowned in netting. For St. Croix, 67 sea turtle strandings were documented, including 27 hawksbill, 22 green, 11 leatherback, 1 loggerhead, and 6 unknown. Of these, five strandings were attributed to vessel strikes (4 green sea turtles and 1 hawksbill sea turtle), one leatherback sea turtle was noted as possibly a boat strike, and 10 were attributed to fishing activity. Fishing related strandings included five strandings attributed to net entanglements. Of those five net entanglements, one hawksbill and one green sea turtle died from being

[12] Data received via e-mail by Jennifer Lee, Fisheries Biologist, NMFS SERO, St. Petersburg, Florida, from Ms. Claudia D. Lombard, Sea Turtle Assistance and Rescue Coordinator, U.S. Fish and Wildlife Service, Christiansted, St. Croix,USVI.

entangled in trammel net; the condition of one more hawksbill trammel net entanglement was not specified. There were also two entanglements (a green and a leatherback) involving other types of gillnet. There were also 4 stranding records described as having fishing line injuries; one resulted in amputation of a flipper. There was also one hawksbill sea turtle that was found swimming while entangled in a fish trap buoy line and unable to dive; the sea turtle was released alive.

20007-2008
No data were obtained for the years 2007 and 2008. It is believed that stranding data do exist, but our numerous attempts to obtain data were unsuccessful.

2009 and 2010[13]
In 2009, two sea turtle mortalities were documented as resulting from boat-strike injuries. One was a female hawksbill, which had propeller damage to its head and right flipper; the other sea turtle, which had neck and head injuries, was not identified to species or sex. There were no fishery-related strandings documented.

In 2010, there was at least one and possibly two dead green sea turtles (one male, one unknown) documented as caused by boat strikes. There were also three fishery-related sea turtle strandings documented. A hawksbill sea turtle stranded after ingesting a hook and was rehabilitated and released alive) net; a juvenile hawksbill drowned in a gillnet (May 2010); and a green sea turtle was found dead with rope around its right front flipper.

5.6.2. Summary of New Sea Turtle Fishery-Related Data Available for Puerto Rico

Bycatch Study of the Puerto Rico's Marine Commercial Fisheries (Matos-Caraballo 2005)

The PRDNER conducted a bycatch study of Puerto Rico's marine commercial fisheries from February 2004 through May 2005; this is the only Puerto Rico bycatch study conducted to date. The project was affected due to the poor cooperation from commercial fishers that were angry and hostile with the project personnel due to implementation of the DNER's Puerto Rico Fishing Regulation 6768. However, a total of 71 commercial fishing trips were interviewed to collect the bycatch data, including 6 beach seine trips, 13 fish trap trips, 27 trammel net trips and 25 handline trips. No sea turtle bycatch was documented during the study.

Stranding Reports

Stranding reports were the only source of new bycatch records obtained for Puerto Rico. Three separate datasets obtained contained assorted sea turtle records for Puerto Rico.

[13] Data received via e-mail by Jennifer Lee, NMFS SERO, Saint Petersburg, Florida, from Ms. Claudia D. Lombard, Sea Turtle Assistance and Rescue Coordinator, U.S. Fish and Wildlife Service, Christiansted, St Croix, USVI.

1989-1992

A list of stranding and mortality records from 1989-1992 included 66 sea turtle records (excluding hatchlings) from Puerto Rico. Each record included the date, species, number, sex, length, occurrence, and municipality where found. Each record was also identified as either a collision, stranding, capture, incidental catch, or unknown event. A summary of these records is provided in Table 5.6.2.1. The majority of the records were noted as strandings (48%), followed by captures (27%) and incidental catch (18%); only one record was a result of a boat collision.

Table 5.6.2.1 1989-1992 Puerto Rico Strandings

	Boat collisions	Strandings	Capture	Incidental catch	Unknown	Total (by species)
Green	0	18	5	9	0	32
Hawksbill	1	11	12	3	3	30
Leatherback	0	3	1	0	0	4
Total	1	32	18	12	3	66
% of Total	1.52	48.48	27.27	18.18	4.55	100.00

1993-2008[14]

Approximately 152 sea turtle stranding records from Puerto Rico were documented between 1993 through 2008, including 69 green, 76 hawksbill, 3 leatherback, and 2 loggerhead sea turtles, plus 2 additional sea turtles not identified to species. The number of strandings varied annually from none reported to as many as 21 reported, with an average of 9.5 sea turtles per year. Strandings were highest overall between 2002 and 2004, but given the potentially inconsistent manner in which reports were documented (i.e., opportunistic versus routine monitoring), this could potentially just reflect monitoring levels those years.

The suspected cause for many of the sea turtle strandings was either not documented or was unknown. For those sea turtle stranding records with a suspected cause noted, there were approximately 37 (16 green, 20 hawksbill, and 1 leatherback sea turtles) for which the identified cause was either a boat strike or bycatch-related (Table 5.6.2.2). Of those, 9 (5 green and 4 hawksbill) were boat strikes, 7 (3 green, 4 hawksbill) were hooked, 5 were entangled in net, mainly gillnet, along with 2 others we suspect were gillnet; 5 had entangling fishing line; 5 were entangled in rope, mainly around the neck; 4 were from beach seines; and one for which incidental capture in fishing was noted, but not specified further. There were 15 stranding reports for which the cause was noted as hunted; 8 for which the cause was noted as "illegal fishing, and 4 for which the cause was noted as speared; all of these are believed to be from directed fishing activities and not incidental.

[14] Data received by Lisa Marie Carrubba, NMFS SERO, San Juan, Puerto Rico, from Ms. Milagros Justiniano and Mr. Hector Horta, Puerto Rico Department of Natural and Environmental Resources.

Table 5.6.2.2 Puerto Rico Vessel and/or Bycatch Related Sea Turtle Strandings Documented Between 1999-2008.

Year	Species	Sex or Age Class	Suspected Cause	Release Condition
1999	Green	Female	Hooked	Rehabilitated
1999	Hawksbill	Female	Boat strike	Dead
1999	Hawksbill	Female	Boat strike	Dead
1999	Hawksbill	Juvenile	Hooked	Rehabilitated
2000	Green	Female	Beach Seine	Released alive
2000	Green	Female	Gillnet entanglement	Released alive
2000	Hawksbill	Female	Boat strike	Dead
2000	Hawksbill	Juvenile	Entangled in net	Dead
2001	Hawksbill	Female	Fishing line, strangled in	Dead
2001	Green	Female	Boat strike	Dead
2002	Green	--	Rope on front flipper	Dead
2002	Green	--	Boat strike	Dead
2002	Green	--	Hook	Alive
2002	Green	--	Boat strike	Dead
2003	Hawksbill		Non-identified incidental capture	Dead
2003	Green	Female	Fishing line on neck	Dead
2003	Hawksbill	Juvenile	Hooked	Rehabilitated
2003	Hawksbill	--	Hooked in esophagus	Alive
2003	Hawksbill	Female, juvenile	Rope around neck	Dead
2004	Green	--	Rope, tangled in	Dead
2004	Green	--	Rope, tangled in	Dead
2004	Green	Subadult	Rope on neck	Dead
2004	Green	--	Boat strike	Alive
2004	Hawksbill	--	Fishing line on neck	Dead
2004	Hawksbill	Female	Net, caught in	Rehabilitated
2004	Hawksbill	Female	Beach seine	Rehabilitated
2006	Hawksbill	Juvenile	Boat strike	Dead
2006	Hawksbill	Male	Beach seine	Dead
2006	Hawksbill	Juvenile	Hooked	Rehabilitated
2006	Hawksbill	Neonate	Lacerated and bitten	Rehabilitated
2007	Hawksbill	Female	Boat strike	Dead
2007	Leatherback	Female	Beach seine	Dead
2007	Green	Juvenile	Boat strike	Dead (euthanized)
2008	Hawksbill	Juvenile	Fishing line, entangled in neck and flipper	Dead
2008	Hawksbill	Female	Boat strike	Released alive
2008	Green	Male	Hooked in flipper	Released alive
2008	Green	Male	Fishing line, entangled in, neck and flipper	Dead

5.6.3 Effects of Trap Gear

Sea turtles are generally not expected to be caught inside a fish trap because the regulated opening is far smaller than any of sea turtles expected to encounter the trap. However, sea turtles encountering trap gear can become entangled in associated lines (e.g., buoy lines or floating line connecting traps set in a "string"). Records of entanglements reveal that the line can wrap around the neck, flipper, or body of a sea turtle. Constriction of the neck and flippers can result in injury, including amputation; it can also result in death by infection. If the sea turtle is cut loose with line attached, the flipper may eventually

become occluded, infected, and necrotic. If entangled below the surface of the water, the sea turtle can drown. If left entangled or with severe injuries, the entangled gear may result in reduced ability to feed, dive, surface to breath, or perform other behavior essential to survival (Balazs 1985). Entangled leatherbacks are also more vulnerable to collision with boats, particularly if the entanglement occurs at or near the surface (Lutcavage et al. 1997).

Atlantic-wide, loggerhead and leatherback sea turtles are the two sea turtle species most frequently entangled in trap lines. Thus, within the action area, leatherback sea turtles may be must vulnerable. Leatherback sea turtle susceptibility to entanglement may be the result of their body size (large size, long pectoral flippers, and lack of a hard shell), and their attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface. Records of green and hawksbill sea turtle being entangled in trap lines are less common, but do occur. Thus, all three species of sea turtles typically found in the action area (i.e., green, hawksbill, and leatherback sea turtles) are susceptible to entanglement.

No dedicated observer programs exist to provide estimates of entanglements and mortality from trap/pot fisheries anywhere in United States or U.S. Caribbean. Analyses of the effects pot/trap gear have on sea turtles in the United States have all stemmed from reported entanglements via the Sea Turtle Disentanglement Network (STDN) and the NMFS Northeast Region or the STSSN in the Southeast Region. The number of entanglements reported annually varies by species, area, and time, but entanglements are relatively rare considering the number of traps and pots fished, even in areas where both sea turtles and trap fishing effort are concentrated.

Sampson (2011) summarizes reports of sea turtles entangled in the vertical line of fixed gear fisheries throughout the Northeast Region. Since its inception in 2002, the STDN has received 126 confirmed reports of sea turtles entangled in the vertical line of fixed gear fisheries throughout the Northeast Region. Averaging 14 entanglements per year, they were reported in the region from May through December, with peak months in July (37 cases, 29.4%) and August (50 cases, 39.7%). In 74 cases, gear was identified to fishery through gear analysis and/or fisherman interviews; in these cases, 42 (56.8%) were identified as lobster, 17 (23.0%) as whelk, 10 (13.5%) as sea bass, and 4 (5.4%) as crab pot gear. Lobster and whelk gear entanglements were widely distributed; lobster gear entanglements occurred throughout New England and whelk gear entanglements occurred in states ranging from Massachusetts to Virginia. Crab and sea bass gear entanglements were more localized, with the former occurring only in Virginia and the latter only in Massachusetts. The vast majority of vertical line entanglements involved leatherback sea turtles (*Dermochelys coriacea*; 115 cases, 91.3%), but loggerhead (*Caretta caretta;* 10 cases, 7.9%) and green (*Chelonia mydas;* 1 case, 0.8%) sea turtles were also documented. All but one loggerhead entanglement occurred south of New Jersey, likely due to a higher abundance of hard shell turtles in the southern states of the region. Leatherback entanglements occurred throughout the region, but the highest incidence was in Massachusetts. The number of wraps and exact entanglement configuration varied widely between animals; however, the location of entanglement was

relatively consistent. The front flippers were involved in almost all (106 cases, 84%) and the head/neck in the majority (73 cases, 58%) of entanglements. Configurations involving the rear flippers or carapace were much less common (5 cases, 4% and 6 cases, 5%, respectively) (Sampson 2011)

Sea turtle incidental captures and strandings attributed to entanglement in trap lines are also occasionally reported to the STSSN.[15] From 1996-2007, 193 reports of sea turtles entangled in the vertical line of fixed gear fisheries in the Southeast have been documented via the STSSN. Of these, the vast majority were off Florida, the Gulf coast in particular (i.e., 142 off the west coast of Florida versus 36 of the east coast of Florida), where many crab (blue crab and stone crab) and lobster traps/pots are fished, mainly in state waters. The number of entanglements per year ranged from a low of 9 to a high of 19, with an overall average of 12. On the Gulf coast, the entanglements by species included 72 loggerhead, 31 leatherback, 14 green, 7 Kemp's ridley, 2 hawksbill, and 16 unidentified sea turtles. On the east coast of Florida, there were 21 green, 9 loggerhead, 3 leatherback, and 3 unidentified sea turtles (STSSN database).

5.6.3.1 Potential Factors Affecting the Likelihood and Frequency of Sea Turtle Interactions with Trap Lines

A variety of factors may affect the likelihood and frequency of sea turtles interacting with reef fish trap lines. The spatial and temporal overlap between fishing effort and sea turtle abundance is the most evident factor likely influencing the likelihood and frequency of entanglements. The more abundant sea turtles are in a given area where and when fishing occurs, and the more fishing effort in that given area, the greater the probability is that a sea turtle will interact with gear. Sea turtle feeding behavior and environmental conditions may also play a large part in both where sea turtles are located in the action area and whether or not a sea turtle interacts with trap lines.

Trap interactions with sea turtles may also be affected by soak time. The longer the soak time, the greater the chances a foraging turtle may encounter the gear and the longer a sea turtle may be exposed to the entanglement or hooking threat, presumably increasing the likelihood of such an event occurring.

5.6.3.2 Estimated Trap Entanglements and Associated Mortalities

Fish trap effort is concentrated along the Caribbean shelf (i.e., inshore of the 100-fathom contour), where sea turtles are generally more common. However, only about 14% of this shelf habitat occurs in federal waters of the U.S. Caribbean. Due to the regional bathymetry and jurisdictional boundaries, fishing effort in the Puerto Rico EEZ is confined to an area off the southwest coast that is approximately 116 nm^2. Since state boundaries only extend 3 nmi from the USVI, there is more shelf area in federal waters

[15] The distinction between incidental capture and stranding is whether the gear is actively fished/fishing or not. To be characterized as an incidental capture, the turtles may be either dead or alive, but the gear must be active. Sea stranding can be dead or alive and beached or floating, but the gear they are entangled in is not actively fishing (e.g. line only, old gear - disrepair/heavily fouled, gear on beach with turtle, etc.)

off the USVI (240 nmi^2) than off Puerto Rico. However, fishing in the EEZ off St. Croix is largely restricted to a small area on the end of Lang Bank off the east coast of the island.

Previous Approach to Estimating Anticipated Effects

The 2005 Caribbean opinion (NMFS 2005a) first estimated trap fishing effort in the EEZ by using the approach in the SFA Amendment and DSEIS. This calculation applied fishing effort from the USVI and Puerto Rico (i.e., the number of traps[16]) uniformly across the U.S. Caribbean (i.e., due to the lack of spatially-explicit effort data), and estimated only 3,039 traps are fished in federal waters of the U.S. Caribbean.[17] Thus, with 355 nm^2 of fishable habitat in the EEZ, the 2005 Caribbean opinion estimated a sea turtle would encounter less than 9 trap lines for every square nautical mile of area traveled, assuming all traps are buoyed as a worst case scenario. Based on available information at that time on sea turtle trap entanglements from outside of the action area (i.e., the Gulf of Mexico), the 2005 Caribbean opinion concluded sea turtle interactions with fish traps were rare, but did occur. Due to the paucity of specific stranding data in the U.S. Caribbean, the approximately 5 leatherback sea turtle entanglements documented by the STSSN during 2002-2003 in the Gulf of Mexico were used as a proxy for Caribbean trap interactions. This was believed to be acting conservatively because the number of all traps in the Gulf of Mexico likely exceeded the number of fish and lobster traps utilized in the U.S. Caribbean. The 2005 Caribbean opinion apportioned four of those five entanglements to the reef fish fishery and the remaining entanglement to the lobster fishery because the majority of traps used are fish traps (i.e., 4:1, fish traps:lobster traps).

Current Approach to Estimating Sea Turtle Trap Gear Interactions

Following the completion of the 2005 Caribbean opinion we were able to acquire strandings data specific to the U.S. Caribbean. Since those data are specific to the action area, we chose to use them in our current approach instead of relying on data from the Gulf of Mexico as done previously. None of our new Caribbean specific stranding data suggests trap line entanglements are more frequent than previously estimated. Based on our review of available Caribbean stranding data (see preceding Sections 5.6.1 and 5.6.2), reports of sea turtles interacting with fish trap gear are rare, with no more than one or two documented during any one year, and frequently none. While strandings data is available since 1982 for some regions, we only used the available strandings data from 1999-2010. Initially, we had intended on only use strandings from 2005 on because the fishery has undergone significant changes since that time, but we were concerned that this sample size was too low. We considered using all the strandings data from 1982-2010, but did not because of our concern that doing so would not properly characterize the fishery as it currently operates. Records from this period also did not generally include any information on the ultimate fate of the animal. Additionally, because of very few records reported from 1982-1998, we were concerned that using data from the time series would actually underestimate potential adverse affects. Ultimately, we chose to use strandings

[16] Trap data was compiled from Matos-Caraballo (1997) and USVI DPNR Data

[17] Number of traps in U.S. Caribbean EEZ = 21,710 total traps * 0.14 of fishable habitat in the EEZ

from 1999-2010 because it expanded our sample size. This period also matched the time series used in the ACL Amendment.

Strandings can be a valuable source of data. Stranding data are often used to monitor sea turtle nearshore mortality rates and sometimes 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/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.

In the U.S. Caribbean, strandings represent the best available information upon which to estimate potential interactions between fishing gear and sea turtles. The reported strandings data available to us indicates that not all sea turtles that become entangled in trap lines die. However, for all of the reasons just described, we are not confident in our ability to monitor non-lethal effects occurring from trap line entanglements. Thus, our analysis here acts conservatively and assumes that any trap line entanglement will result in mortality. TEWG (1998) estimates sea turtle strandings may represent as little as 5-6% of actual at-sea nearshore-mortality events.

Estimated USVI Reef Fish Trap Entanglements

From 1999 through 2010, there were two reported trap-related sea turtle strandings (one green and one hawksbill sea turtle) in the USVI. If we assume documented strandings represent only 5% of actual mortalities, then actual nearshore mortalities may have been 20 green sea turtles and 20 hawksbill sea turtles over that period. Based on 11 years of

data, we would anticipate two green sea turtles and two hawksbill sea turtles would become entangled annually in trap gear in the USVI, on average.[18]

Since the available strandings data does not differentiate between what type of trap (i.e., fish or lobster) likely caused the entanglement, we estimated the likely percentage of all traps that are fish traps to more accurately assess the effects of the proposed action.

Kojis (2004) conducted a census of all commercial fishermen in the USVI. The census reported that pot gear is widely used in the USVI, with fish pots and lobster pots being the most frequently used pot gear. Of all fishermen surveyed 88.5% responded (339 of 383 licensed fishermen); those respondents reported using 3,886 fish pots and 4,756 lobster pots. Since only 88.5% of fishermen responded, we assume the total number of lobster and fish traps reported is an underestimate. Therefore, we estimate 4,391 total fish traps and 5,374 total lobster traps were used in the USVI in 2003;[19] or 9,765 total traps. These estimates indicate that fish traps makes up 45% (4,391 of 9,765) of all traps. We acknowledge that this trap data is old, and more recent information from Puerto Rico (i.e., Matos-Caraballo and Agar (2008)) indicates that the number of traps in use there has been declining over time. If that trend also applies to the USVI it is possible the number of fish traps we estimated are in use is an overestimate. However, this information is currently the best available to estimate the potential impacts of trap fishing in the USVI.

Applying the estimate of the percentage of traps in the USVI that are likely used to target reef fish (i.e., 45%) to our estimate from above of sea turtle entanglements caused by general trap gear (i.e., two green sea turtles and two hawksbill sea turtles, annually) we estimate that one green and one hawksbill sea turtle are likely entangled in fish trap gear in the USVI annually.[20]

Estimated Puerto Rico Reef Fish Trap Entanglements

For Puerto Rico, strandings data was only available from 1999-2008. During that period five trap-related entanglements were documented (four green and one hawksbill sea turtle). Using the same approach as above to account for unreported strandings, we anticipate that between 1999-2008, the actual nearshore mortalities in Puerto Rico may have been 80 green sea turtles and 20 hawksbill sea turtles. Based on 10 years of data, we would anticipate up to eight green sea turtle entanglements and two hawksbill entanglements may occur annually in the U.S. Caribbean because of trap gear in Puerto Rico, on average.

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[18] 2 reported strandings (1 green, 1 hawksbill) ÷ 5% of actual moralities captured = 40 total possible mortalities/strandings; 20 green/20 hawksbill; 20 green/20 hawksbill ÷ 11 years of data = 1.8 green/hawksbill entanglement annually on average.

[19] Fish Traps: 3,886 fish traps reported used ÷ 88.5% of all fishermen responding = 4,391 fish traps likely reported if 100% of all fishermen had responded; Lobster Traps: 4,756 lobster traps reported used ÷ 88.5% of all fishermen responding = 5,374 lobster traps likely reported if 100% of all fishermen had responded.

[20] 2 hawksbill and 2 green sea turtle entanglements in trap gear annually x 45% of all trap gear that are fish traps = 0.9 annual hawksbill and green sea turtles entanglements caused by fish traps.

As with the strandings information from the USVI, the data we have on strandings for Puerto Rico does not differentiate between fish traps and other traps. Therefore, to estimate the number of entanglements likely caused by fish traps we estimate their proportion of all traps used in the Puerto Rico, and applied that proportion to our trap related entanglements estimates.

The 2008 census of active commercial fishermen in Puerto Rico, conducted by Matos-Caraballo and Agar (2008), is the best available data on the relative proportions of fish/lobster trap gear used Puerto Rico. That census indicated there were 9,597 traps units in use at the time of the census; 60% of those traps were fish traps (48% fish traps (4,574 traps) and 12% deepwater snapper traps (1,181 traps)) and 40% were lobster traps (3,842 traps).

Applying these proportions (i.e., 60% of all traps are fish traps) to our estimate of sea turtle entanglements caused by general trap gear from above (i.e., eight green sea turtles and two hawksbill sea turtles, annually) we estimate that five green and one hawksbill sea turtle are likely entangled in fish trap gear in the USVI annually.[21]

As discussed in Section 2.3 (i.e., Action Area) the EEZ accounts for only 14.4% of all fishable area in U.S. Caribbean. Consistent with the approach taken in NMFS 2005(a), we anticipate entanglements are proportional to the amount of fishable area in the EEZ. Therefore, we multiplied our annual estimates of entanglements for the USVI and Puerto Rico by 0.144 to calculate the likely number caused by the federal trap fishery. To act conservatively toward the species all numbers were rounded up to the nearest whole number (see Table 5.6.3.1).[22,23]

The strandings data also indicate fishery interactions with leatherback sea turtles occur, but no interactions with trap gear were reported. However, as noted in Section 5.6.3, leatherbacks are known to become entangled in trap lines. In 2009, the USFWS documented 45 leatherbacks coming to nest at Sandy Hook in St. Croix with indications of fishing gear-related injuries, including some apparently from trap gear (Garner and Garner 2009). Sea turtle strandings also often reflect nearshore species more frequently that offshore species such as leatherbacks. As we noted above, the total number of sea turtle strandings is likely far lower than the true number of incidents, so entanglements in trap gear may have occurred and were just not reported. For these reasons, we believe it is possible that leatherback sea turtles may become entangled in hook-and-line gear.

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[21] 8 green sea turtle entanglements in trap gear annually x 60% of all trap gear that are fish traps = 4.8 annual green sea turtles entanglements caused by fish traps; 2 hawksbill sea turtle entanglements in trap gear annually x 60% of all trap gear that are fish traps = 1.2 annual hawksbill sea turtles entanglements caused by fish traps.

[22] USVI: 1 hawksbill/green sea turtle entanglement in fish trap gear annually x 14.4% of fishable habitat in EEZ = 0.144 hawksbill/green sea turtle entanglements caused by fish trap gear used in the EEZ.

[23] Puerto Rico: 5 green sea turtle entanglement in fish trap gear annually x 14.4% of fishable habitat in EEZ = 0.72 green sea turtle entanglements caused by fish trap gear used in the EEZ; 1 hawksbill sea turtle entanglement in fish trap gear annually x 14.4% of fishable habitat in EEZ = 0.144 hawksbill sea turtle entanglements caused by fish trap gear used in the EEZ.

The only data available regarding overall fishery interactions by species, that is not related to strandings data, is Lewis et al. (2007). Since we believe the strandings data may be under representing leatherback interactions, we evaluated Lewis et al. (2007). The authors indicated that leatherback interactions with fishing gear are indeed more frequent than what is recorded in the strandings information. Of the three sea turtles likely to occur in the action area, leatherbacks composed between 17 and 35% of interactions with fishing gear in the USVI (Lewis et al. 2007). This indicates that leatherback interactions with fishing gear are essentially the same as the other two species, or slightly less in the USVI. Therefore, to act conservatively toward the species, we will assume one leatherback entanglement in trap gear may have occurred in the USVI EEZ.

While Lewis et al. (2007) was based on the USVI fishers, we believe the trap gear techniques used in the USVI and Puerto Rico are similar enough that we would not anticipate a large differences in the likely interactions rates between trap gears and sea turtle species. Under that assumption, we anticipate that one leatherback may also become entangled in trap gear in the EEZ off Puerto Rico. Table 5.6.3.1 summarizes our entanglement estimates for each species, including the total number of annual interactions, and the number of interactions likely to occur in the EEZ.

Table 5.6.3.1 Anticipated Annual Trap-Related Entanglements by Area

Species	USVI	Puerto Rico	Total
Total Interactions			
Green	1	5	6
Hawksbill	1	1	2
EEZ Interactions			
Green	1	1	2
Hawksbill	1	1	2
Leatherback	1	1	2

Stranding records indicate some sea turtles die as result of trap entanglements; others are found entangled and released alive in varying condition. Without reliable information on which to estimate a trap interaction mortality rate, a conservative approach will be employed and all takes will be considered lethal.

5.6.4 Effects from Hook-and-Line Gear

Hook-and-line gear is known to adversely affect sea turtles via hooking, entanglement, trailing line, and forced submergence. Captured sea turtles can be released alive or may be found dead upon retrieval of the gear as a result of forced submergence. 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, entangling, 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 following discussion summarizes in greater detail the available information on how individual sea turtles are likely to respond to interactions with hook-and-line gear.

Entanglement
Sea turtles are particularly prone to entanglement as a result of their body configuration and behavior. Fishing gear can drift according to oceanographic conditions, including wind and waves, surface and subsurface currents, etc.; therefore, depending on sea turtle behavior, environmental conditions, and set location, sea turtles can become entangled in fishing gear. Records of stranded or entangled sea turtles reveal that hook-and-line gear wrap around the neck and foreflippers most frequently, but can also wrap around the body of a sea turtle.

If sea turtles become entangled in monofilament the line can inflict serious wounds, including cuts, constriction, or bleeding anywhere on a sea turtle's body. In addition, entangling gear can interfere with a sea turtle's ability to swim or impair its feeding, breeding, or migration and can force the sea turtle to remain submerged, causing it to drown. The fishing line can also become tighter and more constricting as the sea turtle grows, cutting off blood flow and causing deep gashes, some severe enough to sever an appendage.

Hooking
Sea turtles can be injured and killed by being hooked. Hooking can occur as a result of a variety of scenarios, some of which depend on the foraging strategies and diving and swimming behavior of the various species of sea turtles. Sea turtles are either hooked externally - generally in the flippers, head, shoulders, armpits, or beak - or internally, inside the mouth; or, when the animal has swallowed the bait and the hook is ingested, hooking may occur in the gastro-intestinal tract (E. Jacobson in Balazs et al. 1995). Pelagic longline hooking data indicate entanglement and external foul hooking (usually in the front flipper, shoulder, or armpit) are the primary forms of interaction between leatherback sea turtles and longline gear, whereas internal hooking is much more prevalent in hard shell sea turtles, especially loggerheads.

Sea turtles that have swallowed hooks are of the greatest concern. The esophagus is lined with strong conical papillae directed caudally 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 sea turtle. A sea turtle's esophagus is attached firmly to underlying tissue; therefore, 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 their 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 to the sea turtle's colon or it can pass through the sea turtle entirely (E. Jacobson in Balazs et al. 1995, Aguilar et al. 1995) with little damage (Work 2000). For example, of 38 loggerheads deeply hooked by the Spanish Mediterranean longline fleet and subsequently held in captivity, 6 loggerheads expelled hooks 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 sea turtle. Tissue necrosis that may have developed around the hook may also get passed along through the sea turtle as a foreign body (E. Jacobson in Balazs et al. 1995).

Trailing Line

Trailing line (i.e., line left on a sea turtle after it has been captured and released), particularly line trailing from an ingested hook, poses a serious risk to sea turtles. Line trailing from an ingested hook is likely to be swallowed, which may occlude the gastrointestinal tract, or it may prevent or hamper foraging, leading to eventual death. Sea turtles that swallow monofilament still attached to an embedded hook may suffer from the usually fatal "accordion effect" described by Mediterranean sea turtle researchers, whereby the intestine, perhaps by its peristaltic action in attempting to pass the unmoving monofilament line through the alimentary canal, coils and wraps upon itself (Pont, pers. comm. 2001). Trailing line may also become snagged on a floating or fixed object, further entangling a sea 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 bottom, or has the potential to snag, thus anchoring them in place (Balazs 1985; Hickerson, pers. comm. 2001). Long lengths of trailing gear are likely to entangle the sea turtle eventually, leading to impaired movement, constriction wounds, and potentially death.

Forcible Submergence

Sea turtles can be forcibly submerged by hook-and-line gear when a sea turtle becomes entangled or caught on a hook on a line below the surface and is unable to reach the surface to breathe, as is most frequently the case with bottom longline gear (i.e., the line is too short and/or too heavy to be brought up to the surface by the swimming sea turtle).

Sea turtles that are forcibly submerged undergo respiratory and metabolic stress that can lead to severe disturbance of their acid-base balance (i.e., pH level of the blood). Most voluntary dives by sea turtles appear to be an aerobic metabolic process, showing little if any increases in blood lactate and only minor changes in acid-base status. In contrast, sea turtles that are stressed as a result of being forcibly submerged due to entanglement eventually consume all their oxygen stores. This oxygen consumption triggers anaerobic glycolysis, which can significantly alter their acid-base balance, sometimes leading to death (Lutcavage and Lutz 1997).

Numerous factors affect the survival rate of forcibly submerged sea turtles. It is likely that the rapidity and extent of the physiological changes that occur during forced submergence are functions of the intensity of struggling, as well as the length of submergence (Lutcavage and Lutz 1997). Other factors influencing the severity of effects from forced submergence include the size, activity level, and condition of the sea turtle; the ambient water temperature; and if multiple forced submergences have recently occurred. Larger sea turtles are capable of longer voluntary dives than small sea turtles, so juveniles may be more vulnerable to the stress from forced submergence. Gregory et

al. (1996) found that corticosterone concentrations of captured small loggerheads were higher than those of large loggerheads captured during the same season.

During the warmer months, routine metabolic rates are higher. Increased metabolic rates lead to faster consumption of oxygen stores, which triggers anaerobic glycolysis. Subsequently, the onset of impacts from forced submergence may occur more quickly during these months (Gregory et al. 1996). Sea turtles are probably more susceptible to lethal metabolic acidosis if they experience multiple forced submergence events in a short period. With each forced submergence event, lactate levels increase and require a long time (up to 20 hours) to recover to normal levels. Therefore, recurring submergence does not allow sea turtles sufficient time to process lactic acid loads (Lutcavage and Lutz 1997). Stabenau and Vietti (2003) illustrated that sea turtles given time to stabilize their acid-base balance after being forcibly submerged have a higher survival rate. The rate of acid-base stabilization depends on the physiological condition of the turtle (e.g., overall health, age, size), time of last breath, time of submergence, environmental conditions (e.g., water temperature, wave action, etc.), and the nature of any injuries sustained at the time of submergence (NRC 1990). Disease factors and hormonal status may also influence survival during forced submergence. Because thyroid hormones appear to have a role in setting metabolic rate, they may also play a role in increasing or reducing the survival rate of an entangled sea turtle (Lutcavage and Lutz 1997).

Presumably, a sea turtle recovering from a forced submergence would most likely remain resting on the surface (given it had the energy stores to do so), which would reduce the likelihood of being recaptured by a submerged bottom longline or vertical line, though possibly increasing the risk of vessel strikes. Recapture would also depend on the condition of the sea turtle and the intensity of fishing pressure in the area. For example, sea turtles in the Atlantic Ocean have been captured more than once by pelagic longliners (on subsequent days), as observers reported clean hooks already in the jaw of captured sea turtles. Such multiple captures were thought to most likely have occurred on three or four observed trips that had the highest number of interactions (Hoey 1998).

In the worst scenario, sea turtles will drown from being forcibly submerged. Such drowning may be either "wet" or "dry." With wet drowning, water enters the lungs, causing damage to the organs and/or causing asphyxiation, leading to death. In the case of dry drowning, a reflex spasm seals the lungs from both air and water. Before death due to drowning occurs, sea turtles may become comatose or unconscious.

5.6.4.1 Potential Factors Affecting the Likelihood and Frequency of Sea Turtle Interactions with Hook-and-Line Gear

A variety of factors may affect the likelihood and frequency of listed species interacting with reef fish hook-and-line gear. The spatial and temporal overlap between fishing effort and sea turtle abundance and sea turtle behavior may be the most evident variable involved in anticipating interactions. Other fishing related-factors that may influence the likelihood and frequency of hooking, entanglement, and forced submergence effects

include gear characteristics (e.g., hook sizes, bait) and fishing techniques employed (e.g., soak times). Each of these factors and its potential influence is discussed briefly below.

Spatial/Temporal Overlap of Fishing Effort and Sea Turtles and Sea Turtle Behavior
The likelihood and rate of sea turtle hookings and/or entanglements in reef fish fishing gears is at least in part a function of the spatial and temporal overlap of sea turtle species and fishing effort. The more abundant sea turtles are in a given area where and when fishing occurs, and the more fishing effort in that given area, the greater the probability is that a sea turtle will interact with gear. Environmental conditions may play a large part in both where sea turtles are located in the action area and whether or not a sea turtle interacts with hook-and-line gear.

Hook Type
The type of hook (size and shape) used in fisheries is believed to play a role in the probability and severity of interactions with sea turtles. Experiments in Atlantic pelagic longline fisheries demonstrate the best hook type for avoiding sea turtle captures is a large circle hook. The configuration of a circle hook reduces the likelihood of foul-hooking interactions because the point of the hook is less likely to accidentally become embedded in a sea turtle's appendage or shell. In some fisheries, circle hooks are wide enough to actually prevent hooking of some sea turtles if the sea turtle cannot get its mouth around the hook (Gilman et al. 2006). Circle hook configuration also reduces the severity of interactions with sea turtles because the design has a tendency to hook in the animal's mouth instead of its pharynx, esophagus, or stomach (Prince et al. 2002, Skomal et al. 2002). Caribbean reef fish fishers primarily use only J-hooks (B. Kojis, CFMC SSC Member, to J. Lee, NMFS, pers. comm. 2011).

Bait
Bait characteristics (e.g., the type, size, and texture of the bait) may also influence the likelihood and frequency of certain sea turtle species becoming incidentally hooked. For example, in pelagic longline fisheries, there has been considerable success in reducing leatherback sea turtle captures by modifying bait usage, particularly replacing squid baits with mackerel (Watson et al. 2005). There are also laboratory studies on the effect different bait characteristics have on loggerhead sea turtles' feeding behavior and preferences (Kiyota et al. 2004, Stokes et al. 2006). Caribbean reef fish fishers use round robin (scads),sprat as bait for snappers and groupers, and larger baitfish for dolphin. Ballyhoo are used for tunas, especially by the charter fleet. Fry are used in chum and bait balls for yellowtail snapper. Fry are used to chum for mutton snapper as well (B. Kojis, CFMC SSC Member, to J. Lee, NMFS, pers. comm. 2011).

Soak Time/Number of Hooks/Mainline Length
Hook-and-line gear interactions with sea turtles may be affected by both soak time and the number of hooks fished, independent of overall fishing effort. In longlining, each hook is in the water for the period of time required to set the remaining hooks, the time to haul the previously set hooks, and additional time while the longline vessel steams back to the beginning of its set and waits. Vertical line targeting reef fish generally has much shorter soak times. The longer the soak time, the greater the chances a foraging turtle

may encounter the gear and the longer a sea turtle may be exposed to the entanglement or hooking threat, presumably increasing the likelihood of such an event occurring. Likewise, as the number of hooks in the water in a given area increases, so may the likelihood of an incidental hooking event. It is probable that the more hooks used per mainline, and the longer the mainline, the greater the soak time will be, simply due to the amount of time it takes to haul back gear (i.e., retrieval of the mainline, dehooking catch, and dehooking bycatch). Thus, the two factors may interact to increase the risk of turtle encounters.

5.6.4.2 Estimated Hook-and-Line (Commercial and Recreational) Sea Turtle Captures and Associated Mortalities

Previous Approach to Estimating Interactions in Hook-and-Line Gear
NMFS (2005a) used self-reported sea turtle bycatch data from the Gulf of Mexico and available information on hook-and-line fishing effort from the U.S. Caribbean to calculate sea turtle bycatch in the U.S. Caribbean. The absence of data regarding sea turtle interactions with hook-and-line gear in the U.S. Caribbean forced the use of data from the Gulf of Mexico. The available information on hook-and-line fishing effort in the U.S. Caribbean was used to calculate total hook-and-line effort occurring on fishable habitat in the EEZ. The Gulf of Mexico sea turtle interaction rate was then applied to the estimate of the hook-and-line effort in the EEZ to calculate the likely number of sea turtle interactions occurring over fishable habitat in the EEZ.

Current Approach to Estimating Interactions in Hook-and-Line Gear
Following the completion of the 2005 Caribbean opinion we were able to acquire strandings data specific to the U.S. Caribbean. Since those data are specific to the action area, we chose to us them in our current approach instead of relying on data from the Gulf of Mexico as done previously. Our analysis here follows a similar approach to the trap analysis above. We used the available strandings data from 1999-2010 to determine the number of reported entanglements with hook-and-line gear and then accounted for the number of underrepresented entanglements to estimate total entanglements for a time period. For the same reasons noted above, our estimates here ultimately assume any entanglement will result in mortality.

Based on the information provided above (see Sections 5.6.1 & 5.6.2), five hook-and-line gear-related sea turtle strandings (four green and one hawksbill) have occurred from 1999-2010 in the USVI. For Puerto Rico, strandings data indicate that from 1999-2008 12 hook-and-line gear-related sea turtle strandings (5 green and 7 hawksbill sea turtles) occurred.

If we assume documented strandings represent only 5% of actual mortalities, then actual nearshore mortalities may have been 80 green sea turtles and 20 hawksbill sea turtles in the USVI over that period. Based on 11 years of data, we would anticipate 8 green sea

turtles and 2 hawksbill sea turtles would become entangled annually in hook-and-line gear in the USVI, on average.[24]

Applying the same steps to Puerto Rico, then the actual nearshore mortalities may have been 100 green sea turtles and 140 hawksbill sea turtles over that period. Based on 10 years of data, we would anticipate 10 green sea turtles and 14 hawksbill sea turtles would become entangled annually in hook-and-line gear in Puerto Rico, on average.

Since the EEZ accounts for only 14.4% of all fishable area in U.S. Caribbean, we anticipate that only entanglements proportionally to the amount of fishable area would actually occur in the EEZ. Therefore, we multiplied our annual estimates of entanglements by 0.144 to calculate the likely number caused by the federal fishery. All numbers were rounded up to the nearest whole number (see Table 5.6.4.1).

The strandings data also indicate interactions with leatherback sea turtles, but no interactions with hook-and-line gear were reported. However, as noted in Section 5.6.4 leatherbacks are known to become entangled in trap lines. In 2009, the USFWS documented 45 leatherbacks coming to nest at Sandy Hook in St. Croix with indications of fishing gear-related injuries, including those related to hook-and-line gear (Garner and Garner 2009). Sea turtle strandings also often reflect nearshore species more frequently that offshore specie such as leatherbacks. As we noted above, the total number of sea turtle strandings is likely far lower than true number of incidents, so entanglements in hook-and-line gear may have occurred and were just not reported. For these reasons, we believe it is possible that leatherback sea turtles become entangled in hook-and-line gear.

The only data available regarding overall fishery interactions by species, that is not related to strandings data, is Lewis et al. (2007). Since we believe the strandings data may be under representing leatherback interactions we evaluated Lewis et al. (2007). The authors indicate that leatherback interactions with fishing gear are indeed more frequent that what is recorded in the strandings information. Of the three sea turtles likely to occur in the action area, leatherbacks composed between 17 and 35% of interactions with fishing gear in the USVI (Lewis et al. 2007). This indicates that leatherback interactions with fishing gear are essentially the same as the other two species, or slightly less in the USVI. Therefore, to act conservatively toward the species, we will assume one leatherback entanglement in hook-and-line gear may have occurred in the USVI EEZ.

While Lewis et al. (2007) was based on the USVI fishers, we believe the hook-and-line techniques used in the USVI and Puerto Rico are similar enough that we would not anticipate large differences in the likely interactions rates with sea turtle species. Under that assumption, we therefore anticipate that one leatherback may also become entangled in hook-and-line gear in the EEZ off Puerto Rico. Table 5.6.4.1 summarizes our take estimates for each species, including the total number of annual interactions and the number of interactions likely to occur in the EEZ.

[24] 5 sea turtle strandings recorded ÷ 5% of all strandings reported = 100 total sea turtle strandings (95 unrecorded & 5 reported)

Table 5.6.4.1 Anticipated Annual Hook-and-Line-Related Interactions by Area

Species	USVI	Puerto Rico	Total
Total Interactions			
Green	8	10	26
Hawksbill	2	14	18
EEZ Interactions			
Green	2	2	4
Hawksbill	1	2	3
Leatherback	1	1	2

To better understand the effect the federal reef fish fishery has on each sea turtles species, it is necessary to also estimate the mortality associated with our estimated interactions. As discussed in 5.6.4, sea turtle mortality can occur prior to release (i.e., immediate mortality) or later in time, when individuals released alive die from related injuries (i.e., post-release mortality). Both types of mortality are reviewed and estimated below for sea turtles caught on reef fish hook-and-line and then overall mortality calculated.

In NMFS (2005a), we estimated mortalities for bottom longline and vertical line components of hook-and-line gear separately. For bottom longline, we estimated 27% immediate mortality and 60% and 70% post-release mortality for loggerheads and leatherback sea turtles, respectively. The immediate mortality rate was based on the old Gulf reef fish fishermen-reported sea turtle mortalities. The post-release mortality rates were based on the assumption that most sea turtles would be hooked in the esophagus and released with trailing line and NMFS' January 2004 post-release criteria (see NMFS 2005a for more detailed information). For vertical lines, we assumed no immediate mortality and that sea turtles caught on vertical line gear and released alive would presumably be in better overall health than if released alive from bottom longline gear because of the shorter soak times and ability to reach the surface of the water to breathe. We estimated 30% and 40% post-release mortality for hardshell and leatherbacks, respectively, based on the assumptions that circle hooks were used to the extent that most hardshell sea turtles caught would be hooked in the lower jaw and that sea turtles would be released with trailing line; and the January 2004 post-release criteria.

Section 5.1 summarizes the Caribbean-specific data we obtained. Early stranding data from USVI (1982-1997) and PR (1989-1992) did not include information on the stranded sea turtle's condition (i.e., lethal or non-lethal). Some of the more recent stranding reports did contain information on release condition. Between 2001 and 2006, there were 4 USVI stranding records described as having fishing line injuries. Of these, there were two that indicated abrasion injuries and one that resulted in amputation of a flipper, but these did not specify release condition. The remaining one was found entangled in 100 feet of 10 pound test fishing line and released alive. The one USVI sea turtle that stranded from ingesting a hook in 2010 was rehabilitated and released alive. In Puerto Rico between 1999 and 2008, of the 7 (3 green, 4 hawksbill) sea turtles that were hooked, 4 (2 greens and 3 hawksbills) were rehabilitated and 3 were released alive. In addition there were 5 (3 hawksbills and 2 greens) that were found dead with fishing line entangling their neck.

There is no way to differentiate between immediate and post-release mortality with stranding data. Based on the Caribbean sea turtle stranding data we compiled, very few hook-and-line interactions are lethal. However, given all the limitations associated with strandings (See Section 5.6.3.2) and the very small sample size of records that actually have release condition information available, we believe a more conservative approach is to rely on our general knowledge of immediate and post-release mortality from other hook-and-line fisheries.

In our most recent assessment of hook-and-line interactions in the Gulf of Mexico reef fish fishery, we estimated 43% of sea turtles caught on bottom longline are dead when boated, 30% and 40% post-release mortality for hardshells and leatherbacks, and an overall mortality rate of 60%, based on reef fish bottom longline observer data. For vertical line fishing (both commercial and recreational, we estimated no immediate mortality and the same 30% and 40% post-release mortality for hardshell and leatherbacks, respectively, as we had for bottom longline. The post-release mortality rates were based on our knowledge of circle hook use and anecdotal information indicating fishers typically just cut the line when sea turtles are caught and updated post-release mortality criteria (Ryder et al. 2006).

Based on the fishery description in Section 2.5, several types of hook-and-line gear are used by the Caribbean federal reef fishery, with handlines the most dominant. Because U.S. Caribbean fishermen use J-hooks, the likelihood of hooks being hooked in the esophagus and/or ingested is greater than in other areas outside of the action area where circle hooks are more common. Lewis et al. (2007) indicate that sea turtles are at least sometimes hooked in the esophagi or snagged and that most USVI fishermen do take the time to unhook and untangle sea turtles caught before releasing them.

Taking all of the information we have provided here on mortality rates into consideration we believe applying an overall rate of 60% mortality to our estimated interactions and rounding up to the nearest whole number is a conservative approach. While we acknowledge that bottom longline gear using J-hooks could cause a mortality rate higher than 60%, we do not believe bottom longline gear is commonly used in the U.S. Caribbean. Vertical line gear is the most commonly used hook-and-line gear, and even when used with J-hooks we believe the mortality rate is likely less than 60% because of shorter soak times. Therefore, applying a 60% mortality rate to interactions that are most likely caused by vertical line gear is likely conservative. An artifact of rounding up to the nearest whole number is that our mortality estimates are even more conservative than 60%.[25] Therefore, we believe this approach is conservative and appropriate. Ultimately, we estimate that of our 4 green, 3 hawksbill and 2 leatherback sea turtle interactions, the proposed action's use of hook-and-line gear will result in 3 green, 2 hawksbill, and 2 leatherback sea turtle mortalities.

[25] For example, applying a 60% mortality rate to our estimate of 4 green sea turtles interactions equals 2.4. Rounding to the nearest whole number increases that to 3, which actually equates to a higher mortality rate of 75%.

5.6.5 Effects from Fishing Vessels

Reef fish vessels transiting to and from fishing areas and moving during fishing activity pose a threat to sea turtles. Sea turtles are susceptible to vessel collisions and propeller strikes because they regularly surface to breathe and may spend a considerable amount of time on or near the surface of the water basking, mating, and resting at the surface. Both juvenile and adult sea turtles are subject to vessel strikes; young sea turtles are very alert so much less likely to be hit by a vessel.

Sea turtle stranding data also indicates sea turtle species are more susceptible to being hit by boat propellers during movements associated with reproductive activity (Foley et al. 2008). Sick and injured sea turtles typically float so are also particularly vulnerable to being struck by vessels.

5.6.5.1 Stressors and Individual Responses to Stressors if Exposed

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 cause 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 (Walsh 1999). 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 that are 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 have been documented with healed injuries; thus, we know at least some sea turtles survive without human intervention.

5.6.5.2 Potential Factors Affecting the Likelihood and Frequency of Sea Turtle Exposure to Vessel Strikes

The threat posed by moving vessels is not constant and is influenced in part by vessel type (planing versus displacement hulls), vessel speed, and environmental conditions such as sea state and visibility. Seasonal and regional variance in vessel use and sea turtle distribution and densities also are expected to affect sea turtle vessel strike rates.

Below we review how these factors may affect the likelihood and frequency of sea turtle vessel strikes.

Sea Turtle Size/Activity/Behavior
Sea turtles are highly susceptible to vessel collisions and propeller strikes because they regularly surface to breathe and may spend a considerable amount of time on or near the surface of the water basking, mating, and resting at the surface. Both juvenile and adult sea turtles are subject to vessel strikes; young sea turtles are very alert so much less likely to be hit by a vessel.

Vessel Type and Speed
Generally, vessels typically possess either a planing hull or a (semi-) displacement hull. Planing hulls, typical of smaller (e.g., 18-27 feet in length) vessels are designed to run on top of the water (i.e., on plane) at high speeds. Conversely, displacement hulls push through the water, as they have no hydrodynamic lift, and the boat does not rise out of the water as speed increases. Because of how these two hulls function, they likely introduce differing threat risks to sea turtles. For example, because operational speeds of planing hulls are typically greater than displacement hulls, they possess greater kinetic energy to transfer to an impacted sea turtle. Additionally, because most of the hull is out of the water, the running gear (including the propeller and skeg of an outboard) of a planing hull running at speed becomes a significant cutting/slashing threat, in combination with the concussive effect of a collision. This risk would be compounded by twin or triple engines, which are fairly common in small- to medium-sized (e.g., 25-34 feet in length) recreational reef fish vessels. In comparison, displacement hulls, which include most large (e.g., > 65 feet in length) vessels comprising commercial traffic (e.g., tankers, freighters, tugs, etc.), while traveling slower extend deeper into the water column. The slower speed and greater size of these vessels suggests the risk to sea turtles is largely limited to a concussive impact from the hull. It is possible that a sea turtle may avoid significant impact altogether by being pushed away by the hydrodynamic bow wave of a large vessel, and, therefore, allowed to escape before incurring an injury.

Greater vessel speed is expected to increase the probability that a sea turtle would fail to have time to flee the approaching vessel and that the vessel operator would fail to detect and avoid the sea turtle. A study on vessel speed and collisions with green sea turtles conducted in shallow water (<5 m) along the northeastern margin of Moreton Bay, Queensland, Australia, analyzed behavioral responses of benthic green sea turtles to an approaching 20-ft (6-m) aluminum vessel at slow (2 knot), moderate (6 knot), and fast (10 knot) speeds (Hazel et al. 2007). The proportion of sea turtles that fled to avoid the vessel decreased significantly as vessel speed increased, and sea turtles that fled from moderate and fast approaches did so at significantly shorter distances from the vessel than sea turtles that fled at slow approaches. Hazel et al. (2007) reported that vessel noise is within a green sea turtle's hearing range; however, they also indicated there are several factors that may impede a green sea turtle's recognition of vessel noise as a threat (e.g., directionality of the noise in the ocean and habituation to background vessel noise). The results implied that vessel operators could not rely on sea turtles to actively avoid being struck by a vessel if it exceeds 2 knots. On this basis, the authors determined that vessel

speed was a significant factor in the likelihood of a strike and implied that mandatory vessel speed restrictions were necessary to reduce the risk of vessel strikes to sea turtles (Hazel et al. 2007).

Environmental Factors
Sea state and visibility will also influence the likelihood of an interaction between a vessel and a sea turtle. Typically, most vessel operators keep watch for potential obstructions or debris, which can seriously damage or potentially sink a boat. The calmer the sea state, the easier it is to see floating objects, including sea turtles. When the sea state increases and swells are introduced, observing floating obstructions gets increasingly difficult. However, increased sea state will also compel most vessels on the water to decrease speed, which would reduce the risk of a strike and potentially the severity of a strike. Also, generally fewer recreational vessels go on trips in rough conditions, in comparison with calm seas. Thus, there may be a seasonal component to the magnitude of vessel strike risks to sea turtles in some areas. Another factor is traveling east or west during a rising or setting sun; this can dramatically limit forward visibility and inhibit an operator from seeing and avoiding a floating sea turtle or other obstruction.

Vessel Traffic and Sea Turtle Abundance
Areas with high concentrations of vessel traffic and high concentrations of sea turtles are expected to have a higher probability and frequency of vessel strikes than areas where vessels and/or sea turtles are less abundant. Data on offshore vessel traffic is still largely absent, but several recent studies have explored the issue of vessel traffic for a few coastal counties in Florida (Sidman et al. 2005, Sidman et al. 2007). The available information indicates that there is extensive traffic in inshore and nearshore waters, particularly around inlets. Additionally, there are latitudinal changes in peak use and average number of trips, with a longer peak season and higher number of monthly trips in southern counties when compared to northern counties.

5.6.5.3 Estimating Sea Turtle Vessel Strikes Attributed to Reef Fish Vessels

It is difficult to definitively evaluate the potential risk to sea turtles stemming from specific vessel traffic from any action because of the numerous variables discussed in Section 5.6.5.2 that may impact vessel strike rates. This difficulty is compounded by a general lack of information on vessel use trends, particularly in regard to offshore vessel traffic. For Puerto Rico, we were able to estimate the potential impacts for vessels based on fishing trip information. This approach was very conservative because it assumes all sea turtle vessel strikes can be attributed reef fish fishing vessels and does attempt to account for vessel strikes cause by other vessels. For the USVI, a similar-trip based approach could not be applied because of lack of trip data. Instead, we attempted to estimate vessel strike effects based on vessel registrations and reported vessel use. While we acknowledge there are potential drawbacks to this approach, the data available does not allow for any more precise estimates. Thus, the following analysis is intended to provide a gross estimate of the potential impact reef fish vessels may have on sea turtles, taking a reasoned approach to conservatively account for vessel impacts based on the best

available information. Since this approach does not allow for very precise estimates, we have acted conservatively and assumed any boat struck animal will ultimately die as a result of the interaction.

Documented Sea Turtle Boat Strikes and Estimated Sea Turtle Boat Strikes
Although the cause of death was not usually determined for stranded sea turtles, the most common, readily observable, potential mortality factor was propeller wounds. From 1999 through 2010, there are 22 sea turtle stranding records in the action area with definitive propeller injuries (green, leatherback, and hawksbill sea turtles). The USVI reported 12 boat struck sea turtles (9 green, 2 hawksbill, and 1 leatherback), an average of 1.1 annually over that 11-year period. Puerto Rico reported 10 (5 green and 5 hawksbill sea turtles), an average of 0.9 annually over 11 years. By species, the percent occurrence of boat strike wounds in USVI was 75% green, 17% hawksbill, and 8% leatherback sea turtles. In Puerto Rico, the percent occurrence of boat strike wounds by species was 50% green and 50% hawksbill.

Since we believe that only 5% of all strandings are recorded, we will follow an approach similar to that used in our gear analyses above and revise our estimates upward to account for the likely underreporting of strandings. Therefore, our revised estimates indicate that in the USVI from 1999-2010 as many as 240 sea turtles may have suffered boat strike injuries, approximately 22 annually over the 11-year period. In Puerto Rico, 200 sea turtles may have also been struck during 1999-2010, or approximately 18 annually during the 11-year period.[26]

Puerto Rico Sea Turtle/Commercial Fishing Boat Strikes
Approximately 77% of fishers in Puerto Rico target reef fish (Matos-Caraballo and Agar, 2011). The average number of hours a Puerto Rican commercial fisher devoted to different fishing tasks per week in 2008 was 45 hours: 31 hours for fishing, five hours for vessel maintenance, five hours for gear maintenance, and four hours for marketing. This estimate of hours spent fishing equates to 3.875 8-hr fishing days, which we assume is an appropriate proxy for the number of weekly trips. In 2008, there were 670 active commercial fishing vessels (Matos-Caraballo and Agar 2011). Since 77% of fishers target reef fish, we anticipate 516 of the vessels (77% of 670) used for commercial fishing are reef fishing vessels. By multiplying the estimate of commercial reef fishing vessels by the average number of weekly trips, and the number of weeks in a year, we anticipate up to 103,974 reef fish vessel trips are conducted annually.[27] The MRFSS data on recreational fishing in Puerto Rico (see Section 5.5) indicates 467,567 recreational fishing trips occurred on average from 2000-2008. In combination, these estimates indicate up to 571,541 vessel trips occur in Puerto Rico annually. Based on our revised estimate of boat-struck sea turtles in Puerto Rico, we assume that 18 sea turtles were boat struck annually. Since commercial fishing vessel trips account for 18% of all vessel trips,

[26] USVI: 12 sea turtle strandings recorded ÷ 5% of all strandings reported = 240 total sea turtle strandings (228 unrecorded & 12 reported); 240 vessel strike ÷ 11 years of data = 21.8 sea turtles annually
Puerto Rico: 10 sea turtle strandings recorded ÷ 5% of all strandings reported = 200 total sea turtle strandings (190 unrecorded & 10 reported); 200 vessel strike ÷ 11 years of data = 18.2 sea turtles annually
[27] 3.875 weekly trips x 516 commercial reef fishing vessels x 52 weeks = 103,974

we anticipate they caused 18% or 4 of the estimated sea turtle vessel strike. Based on the known percentages of boat-struck species in the Puerto Rico, we would anticipate two boat-struck sea turtle would be greens and two would be hawksbill sea turtle.

Puerto Rico Sea Turtle/Recreational Fishing Boat Strikes
Recreational and sportfishing is very popular in Puerto Rico and we believe recreational fishing vessels are also likely to cause boat strikes of sea turtles. Therefore, we also estimated the likely number of sea turtles boat-struck annually by recreational vessels targeting reef fish in Puerto Rico. As noted above, it appears that 467,567 recreational fishing trips occurred on average annually, accounting for 82% of all vessel trips annually. Applying this percentage to the 18 sea turtles struck we believe are struck annually in Puerto Rico yields an estimate of 14 vessel strikes caused by recreational vessels. Based on the known percentages of boat-struck species in the Puerto Rico, we would anticipate seven boat-struck sea turtle would be greens and seven would be hawksbill sea turtle.

USVI Sea Turtle/Commercial Fishing Boat Strikes
Since no trips information is available for the USVI recreational fishing fleet, we could not use the trip-based approach used with Puerto Rico here. Instead, we estimated effects based on vessel registrations. In the USVI, over 80% of fishers target reef fish (Kojis 2004). Two-thirds of commercial fishers in the USVI considered themselves full time commercial fishers based how much time they spent fishing and carrying out fishing related activities each week (defined as >36 hrs per week) (Kojis 2004). Kojis (2004) also report the average number of weekly trips in the USVI is 3.1. Boats are constructed primarily of fiberglass and wood (Kojis 2004). Of 323 fishers interviewed, 303 respondents stated they owned 387 vessels (1.27 vessels/fisher) (Kojis 2004). Applying that average to all the number of licensed fishers (383) reported in Kojis (2004), we estimate 487 vessels in the USVI are used for commercial fishing. Since 80% of fishers target reef fish, we anticipate 390 of the vessels (80% of 487) used for commercial fishing are reef fishing vessels.

Approximately 7,700 total vessels are registered in the USVI in 2010 (USVI DPNR Staff to J. Lee, NMFS, pers. comm. 2011). This indicates that vessels used for commercial fishing comprise 5% of the total vessels in the USVI.

Based on our revised estimate of boat struck sea turtles in the USVI, we assume that 22 sea turtles were boat struck annually. Additionally, if we assume that 5% of all vessels in the USVI are reef fishing vessels, then we would anticipate that 1 sea turtles struck annually could be attributed to reef fish fishing in the USVI.[28] Based on the known percentages of boat struck species in the USVI, we would anticipate that the 1 boat struck sea turtle would be a green, hawksbill, or leatherback sea turtle.

[28] 22 sea turtle vessel strikes annually in USVI x 5% of vessel in the USVI are commercially fishing = 1.1 sea turtle vessel strikes caused by commercial fishing vessels in the USVI annually

USVI Sea Turtle/Recreational Fishing Boat Strikes
While recreational and sportfishing is popular in USVI, no specific data is available on the fishery targeting reef fish. Since we believe some level of impact is likely occurring to sea turtles from recreational vessels targeting reef fish, we believe it is prudent to attempt to quantify those impacts. However, without specific information on the total number of recreational fishing vessels in the USVI, we used the percentage of all non-commercial fishing vessels from Puerto Rico (55%) in our calculations. Under that assumption, we estimate that of the 22 sea turtles were boat struck annually, 12 would be the result of recreational fishing vessels.[29] Based on the known percentages of boat struck species in the USVI, we would anticipate that 9 of the boat struck sea turtles would green sea turtles, 2 would be hawksbill, and 1 would be a leatherback sea turtle.

5.6.6 Anticipated Total Number of Sea Turtle Interactions

The proposed action is expected to continue to adversely affect listed sea turtle via entanglement, hooking, and vessel strikes. Anticipated interactions resulting from traps, hook-and-line, and vessels are summarized in Table 5.6.6.1

Table 5.6.6.1 Summary of Anticipated Annual Sea Turtle Interactions (Mortalities) By Species

Sea Turtles	Traps	Hook-and-Line	Vessels	Entire Fishery
Green	2 (2)	4 (3)	19 (19)	25 (25)
Hawksbill	2 (2)	3 (2)	9 (9)	17 (16)
Leatherback	2 (2)	2 (2)	2 (2)	6 (6)

[29] 22 sea turtle vessel strikes annually in USVI x 55% of vessel in the USVI are recreational fishing = 12.1 sea turtle vessel strikes caused by recreational fishing vessels in the USVI annually

6.0 Cumulative Effects

Cumulative effects include the effects of future state, tribal, local, or private actions reasonably certain to occur within the action area considered in this opinion (i.e., commonwealth, territorial, and EEZ waters of the U.S. Caribbean). Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

6.1. Sea Turtles

Human-induced injury and mortality of sea turtles occurring in the action area are reasonably certain to occur in the future. Sources of injury and mortality include territorial- and commonwealth-regulated fishing activities, vessel collisions, marine debris, pollution, and global climate change. While the combination of these activities may prevent or slow the recovery of populations of sea turtles, the magnitude of these effects is currently unknown.

Fisheries
Commonwealth and territorial fisheries described as occurring within the action area (see Section 4) are expected to continue as described into the foreseeable future, concurrent with the reef fish fishery. NMFS is not aware of any proposed or anticipated changes in these fisheries that would substantially change the impacts each fishery has on the sea turtles covered by this opinion. At present, NMFS anticipates that Commonwealth and Territorial governments will implement fisheries management regulations consistent with those described in the proposed action. However, if they fail to do so, or landings increase (as opposed to decrease as anticipated under the proposed ACLs), this may represent new information indicating effects are occurring that are not considered in this opinion; potentially requiring reinitiation of consultation.

Vessel Interactions
Strandings data indicate that vessel interactions are responsible for a large number of sea turtles stranding within the action area each year. Such collisions are reasonably certain to continue into the future. Collisions with boats can stun or easily kill sea turtles, and many stranded turtles have obvious propeller or collision marks (Dwyer et al. 2003). However, it is not always clear whether the collision occurred pre- or post-mortem. NMFS believes that sea turtles takes by vessel interactions will continue in the future. Aside from what we just estimated in Section 5.6, a meaningful estimate of the total number of sea turtles that will likely be killed by vessels is not possible at this time with data available.

Marine Debris and Other Pollution
Human activities in the action area causing pollution are reasonably certain to continue in the future, as are impacts from them on sea turtles. However, the level of impacts cannot be projected. Marine debris (e.g., discarded fishing line or lines from boats) can entangle sea turtles in the water and drown them. Sea turtles commonly ingest plastic or mistake debris for food. Excessive turbidity due to coastal development and/or construction sites

could influence sea turtle foraging behavior. As mentioned previously, sea turtles are not very easily affected by changes in water quality or increased suspended sediments, but if these alterations make habitat less suitable for turtles and hinder their capability to forage, eventually they would tend to leave or avoid these areas (Ruben and Morreale 1999). Noise pollution has been raised primarily as a concern for marine mammals but may be a concern for other marine organisms, including sea turtles. The potential effects of noise pollution on sea turtles range from minor behavioral disturbance to injury and death. The noise level in the ocean is thought to be increasing at a substantial rate due to increases in shipping and other activities, including seismic exploration, offshore drilling, and sonar used by military and research vessels. While there is no hard evidence of a sea turtle population being adversely impacted by noise, masking[30] could possibly interfere with their ability to feed and to communicate for mating. Concerns about noise in the action area of this consultation include increasing noise due to increasing commercial shipping and recreational vessels.

Global Climate Change

Global climate change is likely adversely affecting sea turtles. 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 sea turtles are unknown at this time, but there are multiple hypothesized effects to sea turtles including changes in their range and distribution, as well as changes in prey distribution and/or abundance due to water temperature changes. Ocean acidification may also negatively affect marine life, particularly organisms with calcium carbonate shells which serve as important prey items for many species. Global climate change may also affect reproductive behavior in sea turtles including earlier onset of nesting, shorter inter-nesting intervals, and a decrease in the length of nesting season. Sea level rise may reduce the amount of nesting beach available. Changes in air temperature may also affect the sex ratio of sea turtle hatchlings. A decline in reproductive fitness as a result of global climate change could have profound effects on the abundance and distribution of sea turtles in the action area. Beyond the threats noted above, NMFS is not aware of any proposed or anticipated changes in other human-related actions (e.g., poaching, habitat degradation) or natural conditions (e.g., overabundance of land or sea predators, changes in oceanic conditions, etc.) that would substantially change the impacts that each threat has on the sea turtles covered by this opinion.

6.2. *Acropora*

Activities affecting corals are highly regulated federally; therefore, any future activities within the action area will likely require ESA section 7 consultation. However, much of the development occurring on the USVI and Puerto Rico has been shown to affect water quality, in particular through increases in sedimentation rates. In the USVI, upland development in Tier 2 of the Coastal Zone Management Program usually has no federal permit requirements and development in Tier 1 may not have a federal nexus if the project is located on uplands and is small in size. Depending on the number and location of these developments, sediment and nutrient loading to nearshore waters could become a

[30] "Masking" refers to one sound covering or interfering with another.

chronic stressor. Indeed, results of water quality monitoring from sites around USVI by DEP indicate that this is becoming the case with the number of impaired sites increasing each year (Rothenberger et al. 2008). As the rate of development continues to accelerate in the USVI and Puerto Rico, it is likely that the sedimentation rates in nearshore waters at the outlets of developed watersheds will continue to increase, leading to continued impacts to elkhorn and staghorn 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 area 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.

7.0 Destruction or Adverse Modification/Jeopardy Analysis

Section 5 outlined how the Caribbean reef fish fishery may affect designated *Acropora* critical habitat, elkhorn and staghorn corals, and sea turtles. Now we assess each species' response to these impacts. The assessment considers the effect on designated critical habitat and the entire population of the listed species noted above from these anticipated effects. We also consider whether those effects, in the context of the status of the species (Section 3), the environmental baseline (Section 4), and the cumulative effects (Section 6), will destroy or adversely modify designated critical habitat or jeopardize the continued existence of any ESA-listed species known to interact with the Caribbean reef fish fishery.

"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 conclusion for each species, we first look at whether there will be a reduction in the reproduction, numbers (areal coverage for *Acropora* species), or distribution. Then, if there is a reduction in one or more of these elements, we explore whether it will cause an appreciable reduction in the likelihood of both the survival and the recovery of the species.

The NMFS and USFWS' ESA Section 7 Handbook (USFWS and NMFS 1998) provides further definitions for *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.

Thus far throughout the document we have referred to the two listed *Acropora* species collectively as "*Acropora*." A jeopardy analysis requires that we evaluate the impacts of the proposed action to each species. Therefore, below we describe the likely response of both elkhorn and staghorn corals individually to the proposed action.

7.1 Designated Critical Habitat for *Acropora*

Our analysis seeks to determine whether or not the proposed action is likely to destroy or adversely modify designated critical habitat, based on the information provided in the Status of Species (Section 3.0), the Environmental Baseline (Section 4.0), and the Effects of the Action (Section 5.0) sections. When determining the potential impacts to critical habitat this biological opinion does not rely on the regulatory definition of "destruction or adverse modification" of critical habitat at 50 CFR 402.02. Instead we have relied upon the statutory provisions of the ESA to complete the following analysis with respect to critical habitat. Ultimately, we seek to determine if, with the implementation of the proposed action (i.e., continued authorization of fishing under the proposed ACLs), critical habitat would remain functional (or retain the current ability for the essential features to be functionally established) to serve the intended conservation role for the species.

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

NMFS determined the feature essential to the conservation of *Acropora* is substrate of suitable quality and availability, in water depths from the mean high water line to 30 m, which supports successful larval settlement, recruitment, and reattachment of fragments. Substrate of suitable quality and availability means consolidated hardbottom or dead coral skeletons free from fleshy macroalgae or turf algae and sediment cover. On November 26, 2008, (73 FR 72210) critical habitat containing those features was designated in four areas. 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 above, we anticipate direct effects will only occur in the EEZ. We believe the direct effects from fishing (i.e., vessel operation, gear deployment, etc.) are likely to occur in all three areas of the U.S. Caribbean. The results of our direct effect analysis in Section 5 indicate that 0.004 mi^2 of *Acropora* critical habitat is adversely affected via contact with traps in the U.S. Caribbean EEZ annually. An additional 0.0001 mi^2 of critical habitat is also likely to be adversely affected annually by vessel anchors in the U.S. Caribbean EEZ. Thus, on average, we believe 0.0041 mi^2 of *Acropora* critical habitat is adversely affected in the U.S. Caribbean EEZ annually. However, we also anticipate contact between trap and critical habitat that does not break destroy dead coral skeleton will be temporary in nature. A trap could temporarily cover an area with the appropriate essential feature, impeding its function. However, once that trap is retrieved the function will be restored. Since 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. 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 direct effects from the proposed action would affect less than one-one-thousand of one percent (i.e., 0.0003%) of the designated critical habitat in the U.S. Caribbean, which we believe it too small to constitute an appreciable impact.

Now our analysis turns to the indirect effects of the proposed action on *Acropora* critical habitat. As noted in Section 4.1, 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 Caribbean reef fish fishery would destroy or adversely modify critical habitat depends on what role the proposed action plays overall in the condition of critical habitat.

The *Acropora* 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, 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/threats (including the three major stressors). As noted in Section 5, it appears that a phase shift toward high macroalgae/low coral cover is occurring, or has occurred in the U.S. Caribbean. We believe it is 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 that 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/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/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 action is likely exacerbating the phase-shift that is occurring or has occurred, but is not a primary cause. This indicates that the proposed action may be causing indirect adverse affects to the essential feature to *Acropora* critical habitat, but it is unlikely to be a primary factor causing the degradation of that essential feature. Likewise, the existing data appear to show that herbivorous fish remain relatively abundant in the U.S. Caribbean, indicating that the current levels of herbivorous fish harvest are unlikely to be a primary contributor to growth and spread of macroalgae. If herbivorous fish harvest was a primary cause of the current phase-shifting in the three critical habitat units in U.S. Caribbean, one would expect the St. Croix unit to show the most severe degree of phase-shift, based on the current harvest levels and size of the platforms being fished. More specifically, we would anticipate St. Croix would have significantly higher levels of macroalgal cover when compared to the Puerto Rico and St. Thomas/St. John units. The information in Section 5.2 indicates that the phase-shift is just as severe in all three critical habitat units. Additionally, if herbivory was a primary cause in macroalgal growth we would anticipate the macroalgal cover inside BIRNM, where herbivorous fish harvest is prohibited, would be significantly lower than outside where it is not. We would anticipate that this effect would be particularly apparent in the case of BIRNM because it is located on St. Croix, the platform with highest level of herbivorous fish harvest. However, we do not see any obvious difference between macroalgal cover inside or outside the monument. From 2006-2010, macroalgal cover levels inside BIRNM were highly variable ranging from 36.0-67.8%. During the same time period, macroalgal levels on St. Croix, outside the monument, ranged from 35.0-64.8%.

The ACLs and the prohibition of harvest of the three large-bodied parrotfish are expected to result in population increases of herbivorous fish, especially the large-bodied parrotfish (believed to be the most efficient parrotfish grazers in the Caribbean), throughout all three critical habitat units, likely leading to a greater amount of grazing under the proposed action than there has been at any time in the last decade. 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 biological opinion will require certain research and monitoring actions be taken to help NMFS better understand the extent to which the harvest of herbivorous fish is affecting macroalgal growth. However, we believe the available information on the growth/spread of macroalgae in the U.S. Caribbean indicates that herbivorous fish harvest is unlikely to currently be one of the primary causes of its growth. Thus, the proposed action appears to have only a small, incremental role in what is believed to be only a moderate threat to the species.

At the time of designation in 2008, *Acropora* critical habitat was functional. Since that time, macroalgal cover in the U.S. Caribbean has fluctuated annually, but has remained within the same relative range across all three island platforms. Since the proposed action is anticipated to increase grazing relative to its current level, and since we believe the effects of the proposed action on designated critical habitat are not currently a primary cause in macroalgal growth, we believe the incremental impacts caused by the proposed action are not in and of themselves appreciably reducing capacity of critical habitat to provide an increased incidence of successful sexual/asexual reproduction (i.e., remain functional) currently or in the future. Additionally, the direct effects of the proposed action will affect less than one-one-thousandth of one percent of all critical habitat in the U.S. Caribbean annually. Therefore, we believe the direct and indirect effects from the proposed action are likely adversely affecting, but not destroying or adversely modifying *Acropora* critical habitat in the U.S. Caribbean.

7.2 Jeopardy Analysis for Elkhorn Coral and Staghorn Coral

Our jeopardy analysis now considers whether the direct effects caused by physical contact with fishing gear and the indirect effects caused by reduced herbivory caused by the proposed action will jeopardize the continued existence of elkhorn and staghorn corals. First, we evaluate whether the anticipated direct and indirect effects will result in any reduction in distribution, reproduction, or aerial coverage (numbers) that may appreciably reduce the species' likelihood of survival in the wild. Second, we consider how those effects are likely to affect these species' recovery in the wild.

Reduction in Area Coverage (Numbers) Analysis
Our analysis of direct effects in Section 5 indicates 0.0041 mi^2 (10,618 m^2) of *Acropora* critical habitat is likely to be adversely affected annually via contact with traps and anchors in the U.S. Caribbean EEZ. Those effects account for less than one one-thousandth of one percent (i.e., 0.0003%) of all *Acropora* critical habitat. Because of the very close relationship between elkhorn coral, staghorn coral, and their critical habitat, we do not believe colonies of those species will exist in the U.S. Caribbean in areas outside designated critical habitat. Thus, we anticipate only elkhorn and staghorn colonies occurring within the area of affected critical habitat would be impacted. Since elkhorn and staghorn colonies are rare in the U.S. Caribbean, we anticipate that the number of elkhorn or staghorn colonies, if any, actually affected by traps and anchors would be much smaller than 10,618 m^2. Additionally, since asexual reproduction via fragmentation is the primary mode of reproduction for elkhorn and staghorn, contact with traps/anchors does not necessarily mean a reduction in numbers would occur. Contact with a trap/anchor that causes damage significant enough to remove the potential for asexual reproduction (i.e., traps pulverizing corals) is unlikely. Traps are generally not dropped directly on corals, since most fishers target seagrass, algae, sand, or rubble habitats (Sheridan 2006). Fishers are also not likely to drop gear on reef habitat because doing so could cause damage to gear. Thus, no appreciable reduction in numbers is anticipated from the direct effects of the action.

It is currently unclear whether the indirect effects from the reduction of macroalgae mediation by herbivorous fish is directly reducing the areal coverage (numbers) of elkhorn and staghorn coral. To date, there is little convincing evidence to suggest that algae can act as a direct cause of coral mortality. 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). Recent studies that specifically investigated the correlation between macroalgae and corals 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, only one species (*Agaricia agaricites*) experienced greater tissue mortality after exposure to *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 and staghorn coral mortality, but it appears unlikely that macroalgae is causing direct morality of these species. Thus, we believe a reduction in number of elkhorn and staghorn colonies is not likely as a result of indirect effects caused by the proposed action, except through impacts to reproduction, as discussed below.

Reduction in Reproduction Analysis

As noted above, we believe direct contact from fishing gear is likely to cause an extremely small reduction in the total area cover (number) of elkhorn and staghorn coral in the U.S. Caribbean. Only a portion of this contact is likely to affect sexual reproduction by causing breakage or mortality of large, fecund colonies, which are rare. Effects to elkhorn and staghorn corals from contact with fishing gear is also likely to have a discountable effect on asexual reproduction. Since these species commonly reproduce asexually via fragmentation, contact with fishing gear that causes fragmentation does not necessarily indicate a reduction in asexual reproduction will occur. The success of those fragments is more likely dependent upon the substrate upon which it lands. Impacts surrounding the amount of suitable substrate are addressed during our discussion of indirect effects below. Contact with a trap that causes damage significant enough to remove the potential for asexual reproduction (i.e., traps pulverizing corals) is unlikely. Traps are generally not dropped directly on corals, with most fishers targeting seagrass, algae, sand, or rubble habitats (Sheridan 2006). Fishers are also not likely to drop gear on reef habitat because doing so could cause damage to gear. In summary, we believe impacts to reproduction from trap and anchor injury will occur very rarely, have minor overall impacts, and will not result in an appreciable impact on reproduction.

We do anticipate possible effects to reproduction of both species via indirect effects. Critical habitat was designated to promote sexual and asexual reproduction; therefore, it is appropriate to look at the proposed action's effects to critical habitat to aid in the

determination as to whether the proposed action will cause an appreciable reduction in reproduction for each species.

As noted in Section 5, it appears that a phase shift toward high macroalgae/low coral cover is occurring, or has occurred in the U.S. Caribbean. 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 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/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/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 action is likely exacerbating the phase-shift that is occurring or has occurred, but is not a primary cause. This indicates that the proposed action may be causing indirect adverse affects to the essential feature to *Acropora* critical habitat, but it is unlikely to be a primary factor causing the degradation of that essential feature. Likewise, the existing data appear to indicate that the current levels of herbivorous fish harvest are unlikely to be a primary contributor to growth and spread of macroalgae. If herbivorous fish harvest was a primary cause of the current phase-shifting in the three critical habitat units in U.S. Caribbean, one would expect the St. Croix unit to show the most severe degree of phase-shift, based on the current harvest levels and size of the platforms being fished. More specifically, we would anticipate St. Croix would have significantly higher levels of macroalgal cover when compared to the Puerto Rico and St. Thomas/St. John units. The information in Section 5.2 indicates that the phase-shift is just as severe in all three critical habitat units. Additionally, if herbivory was a primary cause in macroalgal growth we would anticipate the macroalgal cover inside BIRNM, where herbivorous fish harvest is prohibited, would be significantly lower than outside where it is not. We would anticipate that this effect would be particularly apparent in the case of BIRNM because it is located on St. Croix, the platform with highest level of herbivorous fish harvest. However, we do not see any obvious difference between macroalgal cover inside or outside the monument. From 2006-2010, macroalgal cover levels inside BIRNM were highly variable ranging from 36.0-67.8%. During the same time period, macroalgal levels on St. Croix, outside the monument, ranged from 35.0-64.8%.

The proposed action is the continued harvest of herbivorous species at reduced levels through the ACLs. 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 biological opinion will require certain research and monitoring actions be taken to help NMFS better understand the extent to which the harvest of herbivorous fish is affecting macroalgal growth. However, as we also noted above, the *Acropora* BRT (2005) does not even consider competition from macroalgae a "major" threat to elkhorn and staghorn coral and we believe herbivorous fish harvest is only one of many factors that have led to high macroalgal levels. Thus, the proposed action appears to have only a small, incremental role in what is believed to be only a moderate threat to the species.

Since the proposed action is anticipated to increase grazing relative to its current level, and since we believe the effects of the proposed action on designated critical habitat are not currently a primary cause in macroalgal growth, we believe the incremental impacts caused by the proposed action are not in and of causing an appreciable reduction in the species reproduction.

Macroalgae appear unlikely to cause direct mortality to elkhorn and staghorn coral. Thus, the number of colonies available for asexual/sexual reproduction appears unlikely to change. Unlike lethal takes of other species that may forever remove a sexually mature individual for the population, macroalgae appears to affect sexual/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 action is likely adversely affecting elkhorn and staghorn coral, but those effects are not in and of themselves causing an appreciable reduction in reproduction.

Reduction in Distribution Analysis

Our analysis above indicates a very small amount of elkhorn and staghorn coral is likely to be adversely affected directly by the proposed action. This potential loss of elkhorn and staghorn colonies would not have a measurable effect on the distribution of the species within the U.S. Caribbean or throughout its range. Therefore, we do not believe direct effects from the proposed action will reduce the distribution of the species.

Additionally, we believe indirect effects from the proposed action may impact all elkhorn and staghorn colonies in the U.S. Caribbean, but because the harvest of herbivorous fish is so much greater off St. Croix than the other island platforms we believe the majority of the adverse affects are likely occurring to colonies off St. Croix. As explained in the preceding reduction in numbers analysis, we do not anticipate that any colonies will actually be lost due to macroalgal growth associated with the indirect effects of the action. Since we anticipate the colonies experiencing adverse affects will not be lost, we do not anticipate any change in the overall distribution of elkhorn and staghorn corals. Since we do not anticipate a change in the overall distribution of these species within the U.S. Caribbean we also do not believe indirect effects from the proposed action will reduce the distribution of the species throughout its range.

Macroalgal growth can also affect elkhorn and staghorn coral larval settlement, which could affect distribution. The proposed action is the continued harvest of herbivorous species at reduced levels through the ACLs. We acknowledge that many factors have worked synergistically to reduce the amount of substrate suitable for successful *Acropora* settlement. We also acknowledge that this reduction in suitable substrate has degraded the species chances of surviving and recovering in the wild. However, as we also noted above, the *Acropora* BRT (2005) does not even consider competition from macroalgae one of the "major" threat to elkhorn and staghorn coral and we believe herbivorous fish harvest is only one of many factors that have led to high macroalgal levels. Since the proposed action is anticipated to increase grazing relative to its current level, and since we believe the effects of the proposed action are not currently a primary cause in macroalgal growth, we believe the incremental impacts caused by the proposed action are not in and of themselves causing an appreciable reduction in the amount of substrate available for settlement. Therefore, we do not believe the proposed action will cause an appreciable reduction in the species' distribution.

Now our analysis must evaluate whether or not the impacts we discussed above are likely to result in an appreciable reduction in the likelihood of survival of elkhorn and staghorn coral.

Above we indicated that a very small reduction in numbers resulting from direct effects, and a reduction of sexual reproduction from macroalgae may result from the direct and indirect effects of the proposed action. However, the reduction in areal coverage is very small and the ultimate result of proposed action should lead to increases in grazing over time, thereby reducing the remaining effects of the proposed action and reducing the effects of macroalgae on sexual reproduction. Therefore, based on the evaluations above, we anticipate the proposed action will continue to have adverse affects on elkhorn and staghorn coral, but we do not anticipate those adverse effects on numbers and reproduction will appreciably reduce the likelihood of elkhorn and staghorn survival in the wild.

A recovery plan for elkhorn and staghorn coral is not yet available, though a list of threats and causal listing factors exists (Table 7.2.1). Diseases, temperature-induced bleaching, and physical damage from hurricanes are deemed to be the greatest threats to elkhorn and staghorn corals' survival and recovery (*Acropora* BRT 2005). The impact of disease, though clearly severe, is poorly understood in terms of etiology and possible links to anthropogenic stressors. These major threats are persistent, severe, unpredictable, likely to increase in the foreseeable future, and, at current levels of knowledge, unmanageable. Direct and indirect effects from the proposed action (i.e., anthropogenic abrasion and breakage and competition) are currently considered moderate threats to elkhorn and staghorn coral. However, managing these less severe threats may assist in decreasing the rate of elkhorn and staghorn corals' decline by enhancing coral condition and decreasing synergistic stress effects. The *Acropora* BRT concluded that these secondary stressors should be the main focus of regulatory and recovery actions such that the species would be better able to adapt to and recover from the continuing impacts of primary stressors such as diseases and rising sea surface temperatures.

The proposed action is not anticipated to appreciably reduce the likelihood of survival in the wild for elkhorn and staghorn coral. "Recovery is the process by which species' ecosystems are restored and/or threats to the species are removed..." (USFWS and NMFS 1998). The proposed action is not anticipated to increase any of the major threats, and may reduce impacts from some of the moderate threats. Reductions in harvest of parrotfish and surgeonfish are predicted to increase grazing over time and because the proposed ACLs restrict future harvest to levels we anticipate will lead to increased grazing, we believe the continued harvest into the future will not result in any appreciable affects on macroalgal growth. Thus, we do not believe the proposed action is acting to appreciably reduce the species' capacity to recover in the wild. Therefore, NMFS believes that the proposed action is not likely to jeopardize the continued existence of elkhorn and staghorn coral.

Table 7.2.1 Rank of Elkhorn and Staghorn Stressor Severity

Stressor	Elkhorn Coral *(A. palmata)*		Staghorn Coral *(A. cervicornis)*	
	Rank without Regulations	Rank with Regulations	Rank without Regulations	Rank with Regulations
Disease	5+	5+	5+	5+
Temperature	5	5	5	5
Over-harvest	5*	1	5*	1
Natural abrasion and breakage	4	4	4	4
Anthropogenic abrasion and breakage	3	2	2	1
Competition	3	3	3	3
Predation	3	3	3	3
Sedimentation	3	2	3	2
African Dust	1	1	1	1
CO_2	1	1	1	1
Nutrients	1	1	1	1
Sea level rise	1	1	1	1
Sponge boring	1	1	1	1
Contaminants	U	U	U	U
Loss of genetic diversity	U	U	U	U

*A rank of 5 represents the highest threat, 1 the lowest, and U undetermined/unstudied.
Source: *Acropora* BRT 2005

7.3 Green Sea Turtles

The proposed action may result in up to 75 green sea turtle takes every three years, all are estimated to be lethal.

The potential lethal take of up to 75 green sea turtles over consecutive 3-year periods would reduce the number of green sea turtles, compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. Lethal takes could also result in a potential reduction in future reproduction, assuming the individuals were females, and would have survived to reproduce. For

example, an adult green sea turtle can lay 1-7 clutches (usually 2-3) of eggs every 2 to 4 years, with 110-115 eggs/nest. Thus, the loss of up to 75 sea turtles could preclude the production of thousands of eggs and hatchlings, of which a fractional percentage are expected to survive to sexual maturity. The anticipated takes are expected to occur anywhere in the action area and sea turtles generally have large ranges in which they disperse; thus, no reduction in the distribution of green sea turtles is expected from these takes.

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. The 5-year status review for green sea turtles stated that of the seven green sea turtle nesting concentrations in the Atlantic Basin for which abundance trend information is available, all were determined to be either stable or increasing (NMFS and USFWS 2007a). Additionally, the pattern of green sea turtle nesting shows biennial peaks in abundance, with a generally positive trend during the 20 years of regular monitoring since establishment of index beaches in Florida in 1989. An average of 7,560 green turtle nests were laid annually in Florida between 2003 and 2010 with a low of 2,622 in 2003 and a high of 13,225 in 2010 (FWRI 2011). That review also stated that the annual nesting female population in the Atlantic basin ranges from 29,243-50,539 individuals.

Although the anticipated mortalities would result in an instantaneous reduction in absolute population numbers, the U.S. populations of green sea turtles would not be appreciably affected. For a population to remain stable, sea turtles must replace themselves through successful reproduction at least once over the course of their reproductive lives, and at least one offspring must survive to reproduce itself. If the hatchling survival rate to maturity is greater than the mortality rate of the population, the loss of breeding individuals would be replaced through recruitment of new breeding individuals from successful reproduction of non-taken sea turtles. Since the abundance trend information for green sea turtles is either stable or increasing, we believe the loss of up to 75 green sea turtles over consecutive 3-year periods will not have any measurable effect on that trend. As described in the Environmental Baseline section, although the DWH oil release event is expected to have resulted in impacts to green sea turtles, there is no information to indicate, or basis to believe, that a significant population-level impacts have occurred that would have changed the species' status to an extent that the expected takes from the Caribbean Reef Fish Fishery would result in a detectable change in the population status of green sea turtles in the Atlantic.

Although no change in distribution was concluded for green sea turtles, lethal takes would result in a reduction in absolute population numbers that may also reduce reproduction, but these reductions are not expected to appreciably reduce the likelihood of survival of green sea turtles in the wild. The following analysis considers the effects of the anticipated take on the likelihood of recovery in the wild.

The Atlantic Recovery Plan for the population of Atlantic green sea turtles (NMFS and USFWS 1991b) lists the following relevant recovery objectives over a period of 25 continuous years:

- The level of nesting in Florida has increased to an average of 5,000 nests per year for at least 8 years;
  - Green turtle nesting in Florida over the past six years has been documented as follows: 2003 – 2,622; 2004 – 3,577 nests; 2005 – 9,644 nests; 2006 – 4,970 nests (NMFS and USFWS 2007a); 2007 – 12,752 nests; 2008 – 9,228 nests; 2009 – 4,462 nests; and 2010 – 13,225 nests (FWRI 2011). This averages 7,560 nests annually over the past eight years.
- A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.
  - Several actions are being taken to address this objective; however, there are currently no estimates available specifically addressing changes in abundance of individuals on foraging grounds.
  - Takes of juvenile green sea turtles during hopper-dredging activities at Kings Bay, Georgia, in 2009, indicate that juvenile green sea turtle abundance in nearshore/inshore waters of U.S. south Atlantic waters may be increasing (E. Hawk, NMFS, pers. comm. 2009).

The potential lethal take of up to 75 green sea turtles over consecutive 3-year periods will result in reduction in numbers when takes occur but it is unlikely to have any detectable influence on the trends noted above. Non-lethal takes of sea turtles would not affect the adult female nesting population or number of nests per nesting season. Additionally, our estimate of future take is based on our belief that the same level of take occurred in the past. It is worth noting that this level of take has already occurred in the past, yet we have still seen positive trends in the status of this species. Thus, the proposed action is not in opposition to the recovery objectives above and will not result in an appreciable reduction in the likelihood of green sea turtles' survival or recovery in the wild.

7.4 Hawksbill Sea Turtles

The proposed action may result in up to 51 hawksbill sea turtle takes every three years of which 48 are estimated to be lethal.

The potential non-lethal takes are not expected to have any measurable impact on the reproduction, numbers, or distribution of these species. The individuals are expected to fully recover such that no reductions in reproduction or numbers of hawksbill sea turtles are anticipated. Since the takes may occur anywhere in the action area and would be released within the general area where caught, no change in the distribution of hawksbill sea turtles is anticipated.

The potential lethal take of up to 48 hawksbill sea turtles over consecutive 3-year periods would reduce the number of hawksbill sea turtles, compared to the number that would

have been present in the absence of the proposed action, assuming all other variables remained the same. These lethal takes could also result in a reduction in future reproduction, assuming the individual was a female and would have survived to reproduce in the future. For example, an adult hawksbill sea turtle can lay 3-5 clutches of eggs every few years (Meylan and Donnelly 1999, Richardson et al. 1999) with up to 250 eggs/nest (Hirth 1980). Thus, the loss of up to 48 hawksbill sea turtles could preclude the production of thousands of eggs and hatchlings, of which a fractional percentage is expected to survive to sexual maturity. The anticipated takes are expected to occur anywhere in the action area and sea turtles generally have large ranges in which they disperse; thus, no reduction in the distribution of hawksbill sea turtles is expected from these takes. Likewise, as explained in the Environmental Baseline section, while a few individuals were found to have been impacted, hawksbill turtles as a species are not expected to have been significantly impacted by the DWH oil release event.

Although we believe no change in distribution is likely because of the proposed action, we concluded lethal takes would result in a reduction in absolute population numbers that may also reduce reproduction, but these reductions are not expected to appreciably reduce the likelihood of survival of any species in the wild. The following analysis considers the effects of the anticipated take on the likelihood of recovery in the wild.

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:

- The adult female population is increasing, as evidenced by a statistically significant trend in the annual number of nests at five index beaches, including Mona Island and Buck Island Reef National Monument.
    - Of the rookeries regularly monitored: Jumby Bay (Antigua/Barbuda), Barbados, Mona Island, and Buck Island Reef National Monument all show increasing trends in the annual number of nests (NMFS and USFWS 2007b).
- The numbers of adults, subadults, and juveniles are increasing, as evidenced by a statistically significant trend on at least five key foraging areas within Puerto Rico, USVI, and Florida.
    - In-water research projects at Mona Island, Puerto Rico, and the Marquesas, Florida, which involve the observation and capture of juvenile hawksbill turtles, are underway. Although there are 15 years of data for the Mona Island project, abundance indices have not yet been incorporated into a rigorous analysis or a published trend assessment. The time series for the Marquesas project is not long enough to detect a trend (NMFS and USFWS 2007b).

The potential lethal take of up to 48 hawksbill sea turtles over consecutive 3-year periods is not likely to reduce population numbers over time due to current population sizes and expected recruitment. Additionally, our estimate of future take is based on our belief that

the same level of take occurred in the past. It is worth noting that this level of take has already occurred in the past, yet we have still seen positive trends in the status of these species. Thus, we believe the proposed action is not in opposition to the recovery objectives above and will not result in an appreciable reduction in the likelihood of hawksbill sea turtles' recovery in the wild.

7.5 Leatherback Sea Turtles

The proposed action may result in up to 18 lethal leatherback sea turtle takes every three years. Potential non-lethal effects were discussed in our analyses, but in making conservative decisions in their fate, ultimately we assumed all interactions were lethal.

The potential non-lethal takes are not expected to have any measurable impact on the reproduction, numbers, or distribution of these species. The individuals are expected to fully recover such that no reductions in reproduction or numbers of leatherback sea turtles are anticipated. Since the takes may occur anywhere in the action area and would be released within the general area where caught, no change in the distribution of leatherback sea turtles is anticipated.

The potential lethal take of up to 18 leatherback sea turtles over consecutive 3-year periods would reduce the number of leatherback sea turtles, compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. Lethal takes could also result in a potential reduction in future reproduction, assuming one or more of these individuals was a female and would have survived to reproduce in the future. An adult female leatherback sea turtle can produce up to 700 eggs or more per nesting season (Schultz 1975). Although a significant portion (up to approximately 30%) of the eggs can be infertile, the loss of an adult female leatherback sea turtle could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. Thus, the death of up to 18 leatherback sea turtles would eliminate those individuals' contributions to future generations, and the action will likely result in a reduction in sea turtle reproduction. The anticipated takes are expected to occur anywhere in the action area and leatherback sea turtles generally have large ranges in which they disperse; thus, no reduction in the distribution of leatherback sea turtles is expected from proposed action.

Whether the estimated reductions in numbers and reproduction of these species would appreciably reduce their likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

The Leatherback Turtle Expert Working Group estimates there are between 34,000-95,000 total adults (20,000-56,000 adult females) in the North Atlantic. Of the five leatherback populations or groups of populations in the North Atlantic, three show an increasing or stable trend: Florida, Northern Caribbean, and Southern Caribbean. This includes the largest nesting population, located in the Southern Caribbean at Suriname and French Guiana. Of the remaining two populations, there is not enough information

available on the West African population to conduct a trend analysis, and for the Western Caribbean, the annual population growth rate is essentially stable (TEWG 2007).[31]

Although the up to 18 anticipated mortalities would result in a reduction in absolute population numbers, it is not likely this small reduction would appreciably reduce the likelihood of survival of either of these sea turtle species. If the hatchling survival rate to maturity is greater than the mortality rate of the population, the loss of breeding individuals would be replaced through recruitment of new breeding individuals from successful reproduction of non-taken sea turtles. Considering that nesting trends for the Florida and Northern Caribbean populations and the largest nesting population, the Southern Caribbean population, are all either stable or increasing, we believe the loss of up to 18 leatherback sea turtles over consecutive 3-year periods will not have any measurable effect on overall population trends. As described previously, although some impacts may be expected to leatherbacks from the DWH oil release in the northern Gulf of Mexico, there is no information to indicate that this species has experienced significant population-level impacts. Any impacts are not thought to alter the population status to a degree in which mortality from this fishery could be seen as reducing the likelihood of survival and recovery of the species.

The Atlantic recovery plan for the U.S. population of the leatherback sea turtles (NMFS and USFWS 1992) lists the following relevant recovery objective:

- The adult female population increases over the next 25 years, as evidenced by a statistically significant trend in the number of nests at Culebra, Puerto Rico; St. Croix, U.S. Virgin Islands; and along the east coast of Florida.
    - In Puerto Rico, the main nesting areas are at Fajardo on the main island of Puerto Rico and on the island of Culebra. Between 1978 and 2005, nesting increased in Puerto Rico from a minimum of 9 nests recorded in 1978 and to a minimum of 469-882 nests recorded each year between 2000 and 2005. The annual intrinsic rate of population increase[32] was estimated to be 1.1 (95% confidence interval between 1.04 and 1.12), using nest numbers between 1978 and 2005 (TEWG 2007).
    - In the U.S. Virgin Islands, researchers estimated a population growth of approximately 13% per year on Sandy Point National Wildlife Refuge from 1994 through 2001. Between 1990 and 2005, the number of nests recorded has ranged from 143 (1990) to 1,008 (2001). The average annual intrinsic rate of population increase was calculated as approximately 1.10 (95% confidence interval between 1.07 to 1.13) (TEWG 2007). In 2006, the number of nest was 373; in 2007 there were 989 nests; in 2008 195 nests; in 2009 there were 944 nests, and preliminary data from 2010 indicate there were 337 nests (Garner and Garner 2010).

[31] An annual intrinsic rate of population growth equal to 1.0 is considered a stable population; the rate of population growth for the two nesting populations in the Western Caribbean were 0.98 and 0.96 (TEWG 2007).

[32] An intrinsic rate of population growth equal to 1.0 is considered stable; less than 1.0 is considered a declining rate of population growth, and a value greater than 1.0 is considered increasing.

- In Florida, a statewide nesting beach survey program has documented an increase in leatherback nesting numbers from 98 (1989) to 800-900 (early 2000s). Based on standardized nest counts made at index nesting beach survey sites surveyed with constant effort over time, there has been a substantial increase in leatherback nesting in Florida since 1989. The estimated annual intrinsic rate of population increase was approximately 1.18 (95% confidence interval between 1.07 to 1.21) (TEWG 2007).

The potential lethal take of up to 18 leatherback sea turtles over consecutive 5-year periods will result in reduction in numbers when takes occur but it is unlikely to have any detectable influence on the trends noted above. Additionally, our estimate of future take is based on our belief that the same level of take occurred in the past. It is worth noting that this level of take has already occurred in the past, yet we have still seen stable or increasing trends in the status of the species in most Atlantic populations. Thus, we believe the proposed action is not in opposition to the recovery objectives above and will not result in an appreciable reduction in the likelihood of leatherback sea turtles' survival or recovery in the wild.

8.0 Conclusion

We have analyzed the best available data, the current status of the species, environmental baseline, effects of the proposed action, and cumulative effects to determine whether the proposed action is likely to destroy or adversely modify designated *Acropora* critical habitat. We have also used those data to determine if the proposed action is likely to jeopardize the continued existence of green, leatherback, and hawksbill sea turtles and *Acropora* corals.

Designated Critical Habitat for Acropora
Our analyses of the impacts to *Acropora* critical habitat analyzed the direct and indirect effects from Caribbean reef fish fishing. Based on these analyses it was determined the proposed action is not likely to destroy or adversely modify designated *Acropora* critical habitat in the U.S. Caribbean.

Acropora
Our *Acropora* analysis focused on the direct and indirect effects from Caribbean reef fish fishing on listed *Acropora* species. Based on these analyses, it was determined the Caribbean reef fish fishery is not likely to jeopardize the continued existence of *Acropora cervicornis* or *Acropora palmata.*

Green, Hawksbill, and Leatherback Sea Turtles
Our sea turtle analyses focused on the impacts to and population response of sea turtles in the Atlantic basin. However, the impact of the effects of the proposed action on the Atlantic populations must be directly linked to the global populations of the species, and the final jeopardy analysis is for the global populations as listed in the ESA. Because the proposed action will not reduce the likelihood of survival and recovery of any Atlantic

populations of sea turtles, it is our opinion that it is also not likely to jeopardize the continued existence of green, hawksbill, or leatherback sea turtles.

9.0 Incidental Take Statement (ITS)

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. Take that occurs while not fishing in compliance with the requirements of the CRFFMP does not constitute authorized incidental take because it is not incidental to an otherwise lawful activity. Accordingly, such take is not covered by the ITS and constitutes unlawful take. Under the terms of Section 7(b)(4) and Section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the RPMs and terms and conditions of the ITS.

Regulations at 50 CFR §402.14, state that if NMFS determines incidental take is likely as a result of a proposed action, but will not jeopardize a species, NMFS must provide a statement that: specifies the impact (i.e., the amount or extent) of such incidental taking on the species; specifies those reasonable and prudent measures that are necessary or appropriate to minimize such impact; and sets forth the terms and conditions (including, but not limited to, reporting requirements). Additionally, in order to monitor the impacts of incidental take, the action agency must report the progress of the action and its impact on species as specified in the incidental take statement. If those reports indicate the amount or extent of incidental taking is exceeded, formal consultation must be reinitiated immediately.

Section 7(b)(4)(c) of the ESA specifies that to provide an ITS for an endangered or threatened species of marine mammal, the taking must be authorized under Section 101(a)(5) of the MMPA. Since no incidental take of listed marine mammals is expected or has been authorized under Section 101(a)(5) of the MMPA, no statement on incidental take of protected marine mammals is provided and no take is authorized. Nevertheless, F/SER2 must immediately notify (within 24 hours, if communication is possible) NMFS' Office of Protected Resources should a take of a listed marine mammal occur.

9.1 Anticipated Amount or Extent of Incidental Take

The direct effects to sea turtles, elkhorn coral, and staghorn coral were estimated above and we anticipate the following incidental takes may occur in the future as a result of the continued operation of Caribbean reef fish fishery. Our analyses of direct effects in Section 5 calculated annual takes. However, annual take estimates can have high variability because of natural and anthropogenic variation. Based on our experience monitoring fisheries and the frequency of changes in fisheries management, we believe a three-year time period is more appropriate for meaningful monitoring. This approach

will allow us to reduce the likelihood of requiring reinitiation unnecessarily because of inherent variability in take levels, but still allow for an accurate assessment of how the reef fish fishery is performing versus our expectations. Thus, NMFS anticipates the following incidental takes may occur over consecutive 3-year periods as a result of the continued authorization of the reef fish fishery.

We chose this metric because traps affect an area of the seafloor, and using this parameter made quantification of adverse affects more meaningful and it expresses the impacts in a metric that is more easily indentified and monitored. The morphology of the species also makes using an areal metric necessary. Since the polyps that make up elkhorn and staghorn corals 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 *Acropora* are branching, colonial species, that use asexual 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 one 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 *Acropora* takes by area. Table 9.1 summarizes these estimates.

Table 9.1 3-Year Anticipated Take in the Caribbean Reef Fish Fishery

Marine Turtles	**Number of Takes**		
	Lethal	Non-Lethal	Total
Green	75	0	75
Hawksbill	48	3	51
Leatherback	18	0	18
Corals	**Area Directly Affected**		
Staghorn Coral *(Acropora cervicornis)*	10,619 m^2 (0.0041 mi^2)*		
Elkhorn Coral *(Acropora palmata)*			

* These estimates are for both species in combination, not each species individually

Beyond the direct effects to species noted above, we also anticipate that indirect effects from the proposed action are likely to result in take in the form of harm to elkhorn and staghorn coral via reduced sexual/asexual reproductive success. However, our jeopardy analysis indicates those indirect effects are not likely to jeopardize the continued existence of those species. While we cannot quantitatively determine how much elkhorn and staghorn coral will be indirectly affected, we are able to monitor those adverse affects via a take proxy (i.e., herbivorous fish biomass); prescribe reasonable and prudent measures that are necessary or appropriate to minimize such impact; and sets forth the terms and conditions (including, but not limited to, reporting requirements).

The indirect effects we anticipate are primarily the reduced success of coral larvae settlement and fragment reattachment because of increases in macroalgae leading to a reduction in suitable substrate. However, we believe many factors have worked synergistically to reduce the amount of substrate suitable for successful *Acropora*

reproduction, and believe herbivory has played a secondary role in that process. The incremental impact the harvest of herbivorous fish has on the reduction of suitable substrate is uncertain and currently unquantifiable. We determined the existing information indicated that those incremental impacts were not destroying or adversely modifying *Acropora* critical habitat, or jeopardizing the continued existence of either species. However, because of the uncertainty of surrounding these potential impacts, we believe it is prudent and necessary to require certain research and monitoring actions be taken (through the RPMs and their implementing terms and conditions) to help NMFS better understand the extent to which the harvest of herbivorous fish is related to macroalgal growth.

Since impacts to reproduction cannot be monitored directly (visual detection of acroporid sexual recruitment is not possible (Miller et al. 2007)), monitoring the factors that affect the amount of substrate suitable for coral larvae settlement and fragment reattachment is appropriate. Since we believe herbivorous fish harvest is not a primary factor affecting macroalgal growth, and it is possible that marcoalgal growth will continue even as herbivorous fish populations increase, we believe it is imprudent to monitor macroalgae directly. Instead, we believe monitoring herbivorous fish biomass is a better metric for judging whether the proposed action is ultimately leading to an increase in grazing, and whether excessive indirect take of *Acropora* is occurring as a result of the harvest of herbivorous fishes. We believe an increase in herbivorous fish biomass will lead to increased grazing on the reefs, potentially reducing the impacts of macroalgae on corals. Conversely, we believe that if herbivorous fish biomass decreases in the future, the overall grazing rate will be lower than what is currently occurring and we believe this reduction would contribute to even greater macroalgal dominance. Therefore, we will use herbivorous fish biomass as a proxy for monitoring the indirect incidental take of elkhorn and staghorn corals.

There is currently no quantitative information available on herbivorous fish stocks in the U.S. Caribbean. More specifically, no data exists on the likely number of herbivorous fish across the U.S. Caribbean, nor do any data exist on the biomass of herbivorous fish across the region. The qualitative data available suggest that herbivorous fish remain numerically abundant and current harvest levels may be sustainable. Because stocks are believed to be in relatively good shape, we believe the proposed lower harvest levels will lead to a readily observable increase in herbivorous fish biomass; however, we currently cannot estimate the likely extent of that increase. Because data on number and biomass do not currently exist, it is impractical to try and estimate what changes in these metrics represent a decline over time (the reinitiation trigger). Therefore, we will base our estimates of changes in herbivorous fish (and ultimately the impacts to elkhorn and staghorn coral) on biomass changes occurring over time and monitored as described below.

The Terms and Conditions of this biological opinion will require that an assessment of herbivorous fish biomass is conducted within one year from the completion of this biological opinion. Once that assessment is completed, NMFS will monitor the biomass of herbivorous fish during consecutive three-year periods to insure that it is not

decreasing. If during that period, herbivorous fish biomass does decrease, reinitiation of consultation will be required, because we believe this would indicate that the effects of herbivorous fish removal were greater than currently believed, resulting in greater than anticipated adverse effects on reproduction of *Acropora*.

This biological opinion prescribes reasonable and prudent measures that will allow NMFS to establish the baseline herbivorous fish populations and monitor their trends, to monitor changes in algal cover and coral reef health, to monitor *Acropora* populations in the U.S. Caribbean, and further minimize impacts from the proposed action's effects on herbivorous fish populations. If NMFS determines, based on biomass estimates derived from stock assessments, visual surveys, etc. that herbivorous fish biomass is decreasing, reinitiation of consultation will be required.

9.2 Reasonable and Prudent Measures (RPMs)

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

The RPMs and terms and conditions are specified as required by 50 CFR 402.14 (i)(1)(ii) and (iv) to document the incidental take by the proposed action and to minimize the impact of that take on sea turtles and *Acropora*. These measures and terms and conditions are non-discretionary, and must be implemented by the NMFS in order for the protection of section 7(o)(2) to apply. NMFS has a continuing duty to regulate the activity covered by this incidental take statement. If NMFS fails to adhere to the terms and conditions of the incidental take statement through enforceable terms, and/or fails to retain oversight to ensure compliance with these terms and conditions, the protective coverage of section 7(o)(2) may lapse.

NMFS has determined that the following RPMs are necessary and appropriate to minimize impacts of the incidental take of sea turtles and *Acropora* during fishing.

1. <u>Minimizing Sea Turtle Take and Mortality Through Outreach and Education</u>
In Section 5.6.3 and 5.6.4, we described how fishing gear can adversely affect sea turtles via hooking, entanglement, trailing line, and/or forced submergence. In Section 5.6.5, we described how moving reef fish vessels are also likely to adversely affect sea turtles via collision impacts or propeller wounds. Most, if not all, sea turtles released after capture have experienced some degree of physiological injury from forced submergence and/or abrasions/lacerations caused by hooking or entanglement. 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 also greatly affects its chance of

recovery. Therefore, the experience, knowledge, ability, and willingness of fishers to remove gear, is crucial to the survival of sea turtles and following release. Certain behavior by fishermen may also help to reduce the likelihood of takes. For these reasons, NMFS shall conduct outreach and education to ensure that sea turtle takes and mortalities are minimized to the extent practicable.

2. Assessing Herbivorous Fish Populations Response

The proposed action is predicted to reduce indirect adverse affects on *Acropora* from the removal of herbivorous fish. However, if the qualitative guidance on the likely population responses of herbivorous fishes is incorrect, we risk having (1) overestimated the reduction in indirect effects, (2) underestimated the potential adverse effects to *Acropora*, and (3) possibly reached the incorrect conclusion in our jeopardy analysis. Based on NMFS SEFSC (2011), a detectable herbivorous fish population response to the proposed decreases in harvest levels is anticipated only for St Croix. Thus, it is imperative that we monitor and track the herbivorous fish population response in St. Croix following the implementation of the proposed action.

3. Monitoring the Frequency, Magnitude, and Impact of Incidental Take

The jeopardy analyses for sea turtles and *Acropora* are based, in part, on the assumption that the frequency, magnitude, and impact of incidental take estimated in this opinion are accurate. 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, particularly status information for listed corals and the herbivorous fish populations that lead to effects on listed corals. If our estimates regarding the frequency and magnitude of incidental take by the federal reef fish fishery prove to be an underestimate, or the status of listed species 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 occurring specific to the reef fish fishery and the status of listed corals. Therefore, NMFS must ensure that monitoring and reporting related to sea turtle and *Acropora* take and effects associated with the proposed action: (1) detect any adverse effects resulting from the Caribbean reef fish fishery; (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 of Trap Damage to Corals

The proposed action is predicted to result in physical damage to corals via trap use. Our effects analysis describes how the use of traps can temporarily preclude new settlement of some planulae where traps occupy the seabead and destroy new growth. Given these expected impacts and the importance of increasing coral recruitment in the action area, NMFS must reduce the frequency of trap damage to the extent practicable.

9.3 Terms and Conditions

In order to be exempt from liability for take prohibited by section 9 of the ESA, NMFS must comply with the following terms and conditions, 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 CFMC, must work with the USVI and Puerto Rico DNR to develop and implement an outreach program to educate commercial and recreational fishermen on the benefits of using circle hooks and use of available sea turtle release equipment and/or sea turtle handling protocols and guidelines.
2. NMFS, in cooperation with the CFMC, must work with the USVI and Puerto Rico DNR to distribute information to reef fish fishermen on sea turtle vessel strikes in the U.S. Caribbean and any vessel strike avoidance measures. NMFS must also work with its partners to promote research for a better understanding on U.S Caribbean vessel traffic, sea turtle vessel strikes, and how to minimize them.

The following terms and conditions implement RPM No. 2.

3. Within three months of the completion of this biological opinion, F/SER and NMFS SEFSC must convene a meeting that outlines the best and most appropriate way to monitor herbivorous fish stocks in the U.S. Caribbean.
4. Within one year of the conclusion of the meeting described in term and condition 3, a multi-species assessment of herbivorous fish stocks, or an assessment of the most abundant herbivorous fish stocks in St. Croix must be completed, at a minimum. This will act as baseline against which future changes in herbivorous fish stocks can be measured.
5. Following the completion of the initial assessment, NMFS SEFSC will conduct follow-on assessments to document trends no later than 3 years from the previous assessment. Terms and conditions 4-7 are intended, in part, to support this requirement and to facilitate rapid completion of a stock assessment(s).
6. NMFS SEFSC, in conjunction with NOAA's Coral Reef Conservation Program (CRCP), must establish a trap survey or visual census to establish a baseline for current herbivorous fish populations in St. Croix and if feasible, in other U.S. Caribbean areas. NMFS must also maintain those surveys/censuses to ensure data is collected that can indicate the rate of herbivorous fish population recovery in St. Croix.
7. The SEFSC must monitor the length composition of herbivorous fish (from landings) in St. Croix. The SEFSC must supply F/SER with a report annually describing the mean lengths of herbivorous fishes. Increasing mean

8. NMFS SEFSC, in conjunction with the CRCP will ensure that a project investigating the importance of herbivorous fish in the recovery of corals be fully funded to investigate herbivorous fish grazing preferences, grazing rates, and demographic processes. These data will further refine our understanding of the ecological processes resulting in indirect effects to *Acropora* spp. The project entitled: "The importance of parrotfish (fam. Scaridae) on the maintenance and recovery of coral-dominated reefs" is a potential project that could meet this requirement.

9. NMFS SEFSC, in cooperation with SERO, will work with the commonwealth and territorial governments on improving fisheries reporting. Because of the importance of herbivorous species diversity and complementary grazing preferences of herbivorous species, the improved program must include species-specific landings data to the extent that is needed to monitor if diversity and complementary feeding are maintained.

The following terms and conditions implement RPM No. 3.

10. NMFS must ensure that the "Monitoring and mapping of threatened acroporid corals in U. S. jurisdiction: Development of a multi-state conservation program," initiated in 2011 through the ESA Species Recovery Grants Program continue to be funded. This project provides cross-jurisdictional comparable demographic and synoptic monitoring data for *Acropora palmata*. It also will develop methods to collect similar data for *A. cervicornis*. These data will provide the currently lacking information to determine species status and trajectory, recruitment and mortality, and response to threats.

11. NMFS SEFSC, in coordination with the CRCP, must ensure that surveys are conducted to collect sufficient information to determine the status and trends of the benthic communities within the boundaries of designated critical habitat for *Acropora*. Specifically, emphasis should be placed on collecting information on percent cover of: (1) sceleractinian and soft corals, (2) macroalgae; (3) turf algae; and (4) bare substrate. Currently, surveys conducted by NCCOS, UVI, and University of Puerto Rico are fulfilling this requirement. NMFS SEFSC must compile the data and submit a report to F/SER annually. Those reports must provide a summary to the previous year's findings, as well as a trend analysis for the entire time series for which data is available.

12. NMFS must work with USVI and Puerto Rico on implementing a category to record sea turtle discards as part of its standardized bycatch-reporting program. To ensure the quality of the sea turtle data reported under the trip ticket system, NMFS, in cooperation with the CFMC, must distribute educational outreach materials regarding the specific information to be reported and sea turtle identification to commercial fishermen.

13. NMFS must work with the Puerto Rico and USVI sea turtle stranding coordinators to improve collection and reporting of incidental capture and

strandings data 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.

14. As the primary source of data on which to monitor effects of authorized fishing on sea turtles, NMFS must work with the Puerto Rico and USVI stranding coordinators to 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).

15. NMFS, in collaboration with PRDNR and other local partners, must develop a proposal to conduct a survey on interactions between sea turtles and Puerto Rico commercial fishermen similar to Lewis et al. (2007).

The following terms and condition implement RPM No. 4.

16. NMFS must assist fishermen and the territorial 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.

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 F/SER3 to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, F/SER3 requests notification of the implementation of any conservation recommendations.

Sea Turtles:

1. To better understand sea turtle populations and the impacts of incidental take in the reef fish fishery, NMFS should support in-water abundance estimates of sea turtles 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 population modeling or other risk analyses of the sea turtle populations affected by the reef fish fishery. This will help improve the accuracy of future assessments of the effects of different levels of take on sea turtle populations.

Acropora:

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 *Acropora* 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 *Acropora*.
5. NMFS should conduct or fund *Acropora* restoration efforts in the U.S. Caribbean.
6. NMFS should conduct or fund research designed to better understand the importance of the *Diadema* die off and whether their loss 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*.
8. NMFS, in collaboration with the CFMC, should implement escape vents in the trap fishery of the U.S. Caribbean to reduce bycatch of undersized herbivorous fishes in the trap sector of the reef fish fishery.

Both Sea Turtles and *Acropora*:

9. NMFS should encourage the USVI and Puerto Rico to develop and implement programs aimed at helping conserve sea turtles and *Acropora* species occurring in commonwealth and territorial waters.
10. 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.
11. 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 sea turtles and *Acropora* species occurring in state waters.

11.0 Reinitiation of Consultation

This concludes formal consultation. As provided in 50 CFR 402.16, reinitiation of formal consultation is required if discretionary federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) The amount or extent of the taking specified in the incidental take statement is exceeded; (2) new information reveals effects of the action that may affect listed species or critical habitat (when designated) in a manner or to an extent not previously considered; (3) the identified action is subsequently modified in a manner that causes an effect to listed species or critical habitat that was not considered in the biological opinion; or (4) a new species is listed or critical habitat designated that may be affected by the identified action. In instances where the amount or extent of incidental take is exceeded, F/SF1 must immediately request reinitiation of formal consultation.

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Appendix 1 – Data Methods and Results

METHODS

NCCOS Data

Data on habitat composition were obtained from NOAA's Center for Coastal Monitoring and Assessment (NCCOS) Biogeography program. The NCCOS data was queried from: http://www8.nos.noaa.gov/biogeo_public/query_habitat.aspx. Data was queried by 'Cover type' = 'biotic' and 'Cover group' = 'Algae' and 'Hard Corals.' NCCOS has collected biogeography data from all U.S. Caribbean island areas (critical habitat units) since 2001. The NCCOS biogeography data is collected to spatially characterize and monitor the benthic habitat community through a random stratified survey, though the regions sampled are not a complete assessment around each island (Figure A1.1). Sites are randomly selected within each habitat stratum to ensure coverage of a wider study region and not just a particular reef or seagrass area. Sites are not revisited each year; rather, new sites are randomly selected each year within each stratum. The power in this type of monitoring program is the ability to incorporate spatial variability and characterize variable habitat stratum (i.e., a view of the big picture and overall trends).

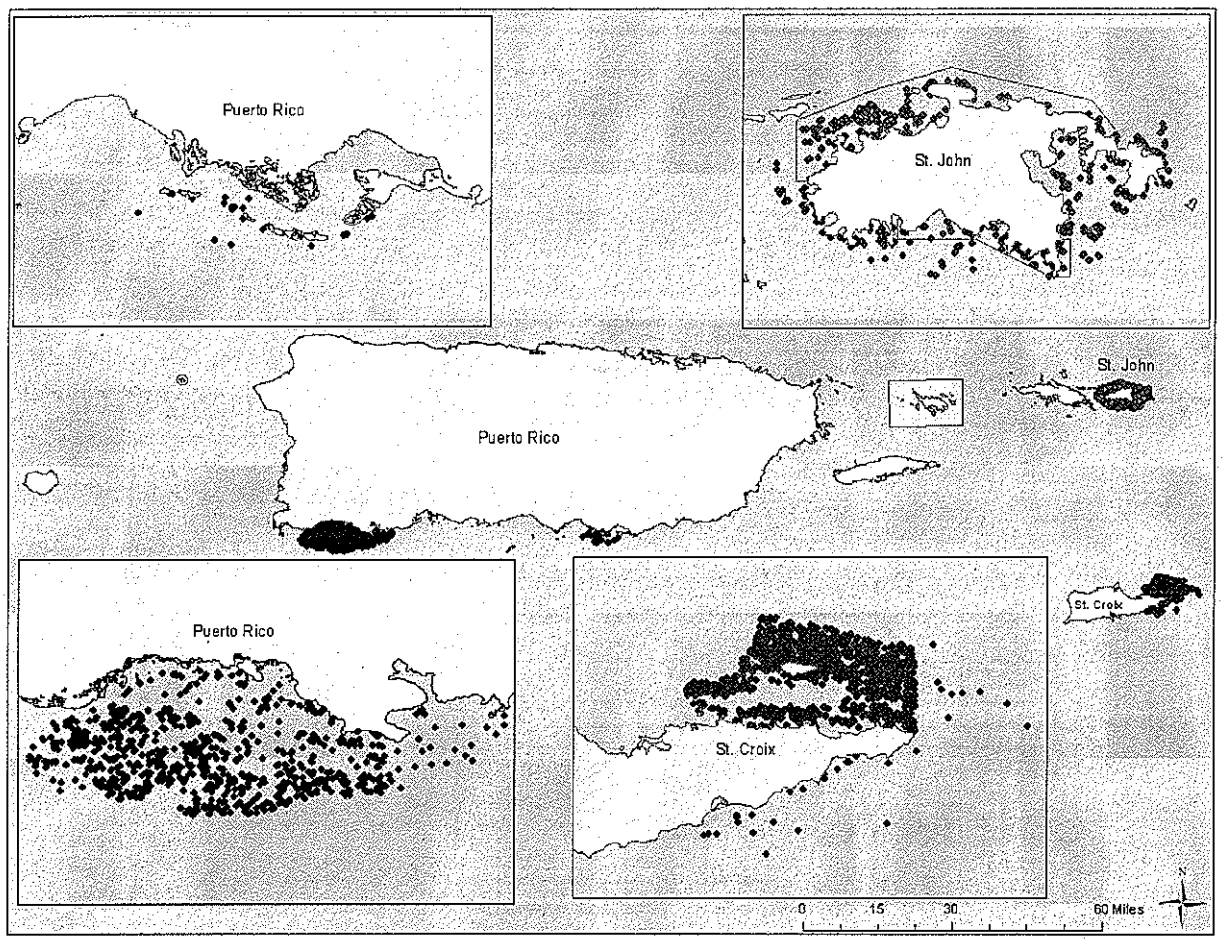

Figure A1.1. NCCOS survey locations.

Sampling sites for the NCCOS program are randomly selected from stratified habitat maps (e.g., 'hard', 'soft', or 'mangrove' habitat) for each island (Pitmann et al. 2010). NCCOS data for the La Parguera study area in southwest Puerto Rico, St. John, and St. Croix 'hard' substrate benthic coverages (2001-2010) were downloaded via Web query (www.nos.noaa.gov/biogeo_public/query_habitat.aspx, accessed May 2011). Data collected in 2001 in Puerto Rico and St. Croix were excluded from analyses due to differences in sampling methods between 2001 and subsequent years. Data for each island were summarized by survey date and station as total 'coral' cover (i.e., NCCOS 'Hard Corals' species group) and 'algae' cover (i.e., NCCOS 'Algae' species group, excluding *Rhodophyta cru. Spp.*). The 'algae' group includes algae, filamentous algae, red algae, green algae, and brown algae. As such, it may represent an overestimate of macroalgal coverage. Due to concerns about the potential bias introduced by the Caribbean-wide 2005 coral bleaching event, pre- and post-bleaching time periods were also evaluated separately. Due to differences in habitats and protection status, Buck Island National Marine Refuge (NMR) in St. Croix was evaluated separately. Sites in Buck Island NMR were identified using GIS. Due to concerns about inequality of error variances and inconsistent temporal breaks between sampling periods, non-parametric Jonckheere-Terpstra (JT) tests were used to evaluate directional trends in benthic coverages for each island (Jonckheere 1954, Pitmann et al. 2010). To reduce the risk of Type I errors over multiple comparisons (e.g., three sets of years, two of which were overlapping), a standard Bonferroni-corrected alpha value of 0.025 (k=2) and a sequential Bonferroni correction (k=3; Rice 1989) were used to evaluate significance of within island trends. As the JT test compares ranks rather than means, trends in the data are presented as box plots.

USVI-DPNR/UVI Data

The USVI DPNR and the UVI have been monitoring the status of reefs in the USVI since 2001. As part of this monitoring, the benthic habitat community is monitored annually at thirty fixed sites. There are 17 sites around St. Thomas and St. John and 13 sites around St. Croix, and at each site six transects are sampled. The monitoring program is designed to follow trends at fixed locations on a fine spatial scale. The UVI collects information on benthic coverage using highly-trained SCUBA divers who conduct visual and video transect surveys along both permanent and randomly-selected transects at each fixed site. Permanent transect locations were initially randomly selected.

Long-term trends in benthic cover variables ('Coral,' 'Critical Habitat,' and 'Macroalgae') were examined using a modified version of a generalized mixed model regression (proc GLIMMIX) developed by the Florida Fish and Wildlife Research Institute in SAS v9.2 (Ruzicka et al. 2009). Percent coverage data were square-root transformed. Transects were nested within sites, and sites nested within islands to provide long-term trend information at the site and island level. For all datasets, a regression analysis for each transect, each site (transect) grouping, and each island (site (transect)) grouping was conducted on annual percent cover values; and the slope was identified as increasing or decreasing by t-tests demonstrating that the slope was significantly different from zero. To reduce the possibility of Type I errors due to repeating the same test on multiple island (site) groupings or sites, a Bonferroni

correction was used to adjust the p-value for identifying a trend as significantly increasing or decreasing. At the island (site) level, the p-value was adjusted to $p<0.004$ and at the site level to $p<0.002$. Residual diagnostics were examined to ensure that regressions met assumptions of normality. The Akaike Information Criterion (AIC; Akaike 1974) was used to determine the best model.

For our analysis, aggregated coverage categories of 'Coral,' 'Critical Habitat,' and 'Macroalgae' were defined by summing the coverages of the species and habitat types shown in Table A1.1. We defined our 'Critical habitat' aggregated coverage category to include the habitat categories recorded in the surveys which correspond to the designated essential feature for *Acropora* (i.e., dead coral with sparse turf algae and consolidated substrate covered with crustose coralline algae that would be suitable for coral recruitment). Although *Acropora cervicornis* and *A. palmata* were included in the 'coral' aggregated category, there were only 11 observations of these species in the 2001-2010 data, and percent coverage per site was <1% for most of these observations. Only sites with annual measurements at a permanent transect were considered in the analyses. If sites had multiple measurements within a single sampling year, these measurements were averaged into one annual measurement for the site. Buck Island Reserve off of St. Croix was analyzed separately due to substantial differences in its benthic habitat composition and protection. Three subsets of the aggregated coverage data were evaluated: (1) All transects with 10 years of samples 2001-2010; (2) All transects with at least 9 years of samples 2001-2010; and (3) All transects with 5 years of samples following the 2005 bleaching event.

RESULTS

NCCOS Data

Descriptive statistics for NCCOS data by island and sampling period are presented in Table A1.4 and A1.5 and Figures A1.1, A1.2, and A1.3. In general, NCCOS-observed 'algae' coverage was highly variable through time. For sites sampled in Puerto Rico, 'algae' cover was significantly increasing over the entire time series and had a percent cover of 56.6% in 2002, went to its lowest observed level (35.1%) in the summer of 2007, and its highest observed level (64.5 %) in the summer of 2009. Sites sampled in St. John indicate 'algae' cover was significantly increasing over the entire time series and had a percent cover of 47.9% in 2001, went to its lowest observed level (34.1%) in the summer of 2005, at its highest observed level (66.9%) in the summer of 2008, and was at 56.2% in the summer of 2010. Sites sampled in St. Croix indicate 'algae' cover had no significant trend over the entire time series and had a percent cover of 59.3% in 2003, was at its highest observed level (64.8%) in the spring of 2006, at its lowest observed level (35%) in the fall of 2009, and near its highest observed level (64.6%) again in the fall of 2010.

For sampled sites in Puerto Rico, 'coral' cover was significantly decreasing over the entire time series and had a percent cover of 6.1% in 2002, was at its highest observed level (9.9%) in the summer of 2002, at its lowest observed level (2.1%) in the winter of 2008, and at 4.6% in the summer of 2009. Sampled sites in St. John indicated 'coral'

cover was significantly decreasing and had a percent cover of 7.9% in 2001 (the highest in the data set), was at its lowest observed level (2.4%) in the summer of 2009, and was at 3.0% in the summer of 2010. Sites sampled in St. Croix indicate 'coral' cover was significantly decreasing over the entire time series and had apercent cover of 2.8% in 2003, was at its highest observed level (3.5%) in the spring of 2004, at its lowest observed level (1.0%) in 2005, and at 2.3% in fall of 2010. It should be noted that mean 'coral' cover is extremely low relative to its variability, and thus measurement error may play a significant role in the observed trends.

USVI-DPNR/UVI Data

Outputs from the mixed model regression for the USVI/UVI data are presented in Table A1.3 and Figure A1.4. The AIC indicated that the best models for capturing variability in annual trends in benthic 'Coral' and 'Macroalgae' coverage for all three sampling periods were at the site (transect) level. The AIC indicated the best model for capturing variability in annual trends in benthic 'Critical Habitat' coverage was at the site level for the 10-yr and 9-yr sampling periods. The AIC indicated that a site-level fit was appropriate for Buck Island, St. Croix for 'Coral' coverage; a transect-level fit was appropriate for 'Critical Habitat' coverage; and a transect-level fit was appropriate for macroalgae coverage. In summary, output from the model (Table A1.3) showed:

- The majority of transects within sites showed no significant change in 'Coral' coverage across the time series (Table A1.3; 26/35 for 10-yr and 37/52 for 9-yr were not significant); however, all significant changes detected were declining trends (9/35 transects for 10-yr and 15/52 transects for 9-yr were declining. Over the 10-yr and 9-yr time series, no transects showed significant increases in coral coverage; however, in the post-2005 bleaching event subset, 2/52 transects had increasing coverage.

- The majority of sites showed significant declines in 'Critical Habitat' coverage across the time series (Table A1.3; 5/6 sites for 10-yr and 6/9 sites for 9-yr were declining); however, the percentage of sites exhibiting this trend following the 2005 bleaching event was substantially reduced (8/52 transects were declining with 44/52 transects not significant; no transects had increasing coverage of critical habitat).

- The majority of transects within sites showed significant change in 'Macroalgae' coverage for the 10-yr time series (18/35 were increasing); while the majority of transects within sites showed no significant change for the 9-yr time series (33/52 were not significant; Table A1.3); however, all significant changes detected for the 9-yr time series were increasing trends (19/52 transects). In the post-2005 bleaching event subset, the majority of transects (41/52 transects) showed no change in 'Macroalgae' coverage. Of those with significant changes, one transect was declining and ten transects were increasing.

For Buck Island, St. Croix, the AIC indicated that a site-level fit was appropriate for 'Coral' coverage. 'Coral' coverage at Buck Island, St. Croix was significantly decreasing

through time ($F_{1,47}$=93.0, p<0.0001). The AIC indicated that a transect-level fit was appropriate for 'Critical Habitat' and 'Macroalgae' coverage. 'Critical Habitat' was significantly decreasing at only one of six transects ($F_{6,42}$=2.11, p>0.05); macroalgae was significantly increasing at two of six transects ($F_{6,42}$=3.45, p<0.01).

Table A1.1. Species contained within the aggregated categories of 'Coral,' 'Critical Habitat,' and 'Macroalgae.'

CORAL		CRITICAL HABITAT	MACROALGAE
Acropora cervicornis	Montastraea faveolata	Coralline algae	Macro Algae
Acropora palmata	Montastraea franksi	Dead coral with turf algae	Amphiroa spp.
Acropora prolifera	Montastraea species	Boulder	Cladophora spp.
Agaricia agaricites	Mussa angulosa	Pavement	Dictyota spp.
Agaricia fragilis	Mycetophyllia aliciae		Halimeda spp.
Agaricia grahamae	Mycetophyllia danaana		Microdictyon spp.
Agaricia humilis	Mycetophyllia lamarckiana		Liagora spp.
Agaricia lamarcki	Mycetophyllia ferox		Lobophora variegata
Agaricia tenuifolia	Mycetophyllia species		Lyngbia spp.
Agaricia undata	Oculina diffusa		Sargassum spp.
Agaricia species	Porites astreoides		Schizothrix spp.
Colpophyllia natans	Porites branneri		Filamentous Cyanobacteria
Dendrogyra cylindrus	Porites divaricata		
Diploria clivosa	Porites furcata		
Diploria labyrinthiformis	Porites porites		
Diploria strigosa	Porites branching species		
Dichocoenia stokesii	Scolymia cubensis		
Eusmilia fastigiata	Scolymia lacera		
Favia fragum	Scolymia species		
Isophyllia sinuosa	Siderastrea radians		
Isopyhyllastrea rigida	Siderastrea siderea		
Leptoseris cucullata	Siderastrea species		
Manicina areolata	Solenastrea bournoni		
Madracis decactis	Solenastrea hyades		
Madracis formosa	Stephanocoenia intercepta		
Madracis mirabilis	Tubastraea coccinea		
Meandrina meandrites	Millepora alcicornis		
Montastraea annularis	Millepora complanata		
Montastraea annularis complex	Millepora squarrosa		
Montastraea cavernosa	Coral juvenile		
Hard Coral, unknown spp.			

Table A1.2. Descriptive statistics for percent cover trends by sampling period.

			CORAL		ALGAE	
Island	Sample Period	N	Mean	SE	Mean	SE
Puerto Rico	Winter-2002	17	6.1%	6.0%	56.6%	12.4%
Puerto Rico	Summer-2002	17	9.9%	7.5%	43.9%	12.4%
Puerto Rico	Fall-2002	28	5.7%	4.5%	56.8%	9.5%
Puerto Rico	Spring-2003	40	5.2%	3.6%	42.6%	7.9%
Puerto Rico	Fall-2003	40	8.9%	4.6%	47.8%	8.0%
Puerto Rico	Spring-2004	42	6.4%	3.8%	53.7%	7.8%
Puerto Rico	Summer-2004	40	6.8%	4.0%	36.7%	7.7%
Puerto Rico	Winter-2005	41	5.6%	3.6%	51.5%	7.9%
Puerto Rico	Summer-2005	51	5.0%	3.1%	51.6%	7.1%
Puerto Rico	Winter-2006	46	2.7%	2.4%	40.6%	7.3%
Puerto Rico	Summer-2006	50	4.5%	3.0%	43.4%	7.1%
Puerto Rico	Winter-2007	50	5.0%	3.1%	38.9%	7.0%
Puerto Rico	Summer-2007	49	3.4%	2.6%	35.1%	6.9%
Puerto Rico	Winter-2008	43	2.1%	2.2%	37.5%	7.5%
Puerto Rico	Summer-2008	43	3.1%	2.7%	61.1%	7.5%
Puerto Rico	Winter-2009	46	3.0%	2.5%	56.4%	7.4%
Puerto Rico	Summer-2009	48	4.6%	3.1%	64.5%	7.0%
St. John	Summer-2001	25	7.9%	5.5%	47.9%	10.2%
St. John	Summer-2002	50	6.4%	3.5%	39.0%	7.0%
St. John	Summer-2003	34	6.9%	4.4%	40.0%	8.5%
St. John	Summer-2004	63	4.2%	2.5%	45.6%	6.3%
St. John	Summer-2005	31	5.1%	4.0%	34.1%	8.7%
St. John	Summer-2006	29	4.1%	3.7%	62.5%	9.1%
St. John	Summer-2007	30	4.5%	3.9%	44.3%	9.2%
St. John	Summer-2008	32	2.5%	2.8%	66.9%	8.4%
St. John	Summer-2009	30	2.4%	2.8%	66.1%	8.8%
St. John	Summer-2010	30	3.0%	3.2%	56.2%	9.2%

			CORAL		ALGAE	
Island	Sample Period	N	Mean	SE	Mean	SE
St. Croix*	Spring-2003	42	2.8%	2.6%	59.3%	7.7%
St. Croix*	Fall-2003	37	2.9%	2.8%	63.8%	8.0%
St. Croix*	Spring-2004	16	3.5%	4.7%	62.2%	12.5%
St. Croix*	Fall-2004	37	3.1%	2.9%	43.3%	8.3%
St. Croix*	Spring-2005	36	2.9%	2.9%	41.6%	8.3%
St. Croix*	Fall-2005	46	1.0%	1.5%	47.8%	7.4%
St. Croix*	Spring-2006	41	1.8%	2.1%	64.8%	7.5%
St. Croix*	Fall-2006	42	1.3%	1.8%	48.7%	7.8%
St. Croix*	Fall-2007	40	1.5%	2.0%	57.0%	7.9%
St. Croix*	Spring-2008	39	2.0%	2.3%	44.4%	8.1%
St. Croix*	Fall-2008	38	1.4%	1.9%	59.2%	8.1%
St. Croix*	Spring-2009	26	2.6%	3.2%	52.9%	10.0%
St. Croix*	Fall-2009	40	1.5%	1.9%	35.0%	7.6%
St. Croix*	Fall-2010	39	2.3%	2.4%	64.6%	7.8%
Buck Island NMR	Winter-2002	36	3.9%	3.3%	48.8%	8.4%
Buck Island NMR	Fall-2002	32	7.5%	4.7%	45.8%	8.9%
Buck Island NMR	Spring-2003	42	7.3%	4.0%	60.0%	7.7%
Buck Island NMR	Fall-2003	37	3.9%	3.2%	57.0%	8.3%
Buck Island NMR	Spring-2004	18	6.2%	5.9%	46.1%	12.1%
Buck Island NMR	Fall-2004	38	4.4%	3.4%	33.9%	7.8%
Buck Island NMR	Spring-2005	38	4.3%	3.3%	53.8%	8.2%
Buck Island NMR	Fall-2005	48	2.4%	2.2%	41.7%	7.2%
Buck Island NMR	Spring-2006	48	4.6%	3.0%	66.3%	6.9%
Buck Island NMR	Fall-2006	51	3.3%	2.5%	36.0%	6.8%
Buck Island NMR	Fall-2007	50	3.7%	2.7%	44.3%	7.1%
Buck Island NMR	Spring-2008	42	4.5%	3.2%	65.0%	7.4%
Buck Island NMR	Fall-2008	44	2.3%	2.3%	48.6%	7.6%
Buck Island NMR	Spring-2009	43	2.8%	2.5%	40.8%	7.6%
Buck Island NMR	Fall-2009	47	1.9%	2.0%	58.0%	7.3%
Buck Island NMR	Fall-2010	42	4.3%	3.2%	67.8%	7.3%

*Excludes Buck Island

Table A1.3. Summary statistics for mixed model regression analysis of changes in annual benthic coverage macro-variables for different subsets of data under different variable nesting configurations. Best model fit per AIC in bold.

Factor	Subset	Model	Block	Inc.	Dec.	NSD	AIC	NumDF	DenDF	F-Value	Pr > F
coral	10-yr	**y*transect(site)**	**transect(island)**	**0**	**9**	**26**	**724.03**	**35**	**280**	**5.24**	**<.0001**
		y*site	transect(site)	0	2	4	749.6	6	309	18.4	<.0001
		y*island	transect(site)	0	2	0	815.53	2	313	16.95	<.0001
		y	transect	0	1	0	995.26	1	342.9	14.38	0.0002
		y	transect(island*site)	0	1	0	812.97	1	314	30.67	<.0001
coral	9-yr	**y*transect(site)**	**transect(island)**	**0**	**15**	**37**	**1045.86**	**52**	**400**	**5.63**	**<.0001**
		y*site	transect(site)	0	5	4	1091.3	9	443.2	18.3	<.0001
		y*island	transect(site)	0	2	0	1184.54	2	450.1	37.08	<.0001
		y	transect	0	1	0	1527.45	1	497.3	20.72	<.0001
		y	transect(island*site)	0	1	0	1186	1	451.2	64.95	<.0001
coral	post-bleaching	**y*transect(site)**	**transect(island)**	**2**	**0**	**50**	**451.9**	**52**	**156**	**1.2**	**0.2016**
		y*site	transect(site)	3	0	6	511.75	9	199	2.71	0.0054
		y*island	transect(site)	1	0	1	532.96	2	206	3.95	0.0206
		y	transect	1	0	1	652.61	1	252.9	3.08	0.0804
		y	transect(island*site)	1	0	1	532.11	1	207	7.34	0.0073
critical habitat	10-yr	y*transect(site)	transect(island)	0	15	20	1007.67	35	280	3.77	<.0001
		y*site	**transect(site)**	**0**	**5**	**1**	**980.45**	**6**	**309**	**21.4**	**<.0001**
		y*island	transect(site)	0	2	0	1038.71	2	313	52.06	<.0001
		y	transect	0	1	0	1316.29	1	343.5	36.28	<.0001
		y	transect(island*site)	0	1	0	1071.07	1	314	99.51	<.0001
critical habitat	9-yr	y*transect(site)	transect(island)	0	13	39	1521.39	52	400	2.73	<.0001
		y*site	**transect(site)**	**0**	**6**	**3**	**1509.34**	**9**	**442**	**15**	**<.0001**
		y*island	transect(site)	0	2	0	1585.99	2	450.6	46.58	<.0001
		y	transect	0	1	0	1846.45	1	497.4	45.42	<.0001
		y	transect(island*site)	0	1	0	1612.51	1	451.6	90.99	<.0001
critical habitat	post-bleaching	**y*transect(site)**	**transect(island)**	**0**	**8**	**44**	**587.91**	**52**	**156**	**2.28**	**<.0001**
		y*site	transect(site)	0	3	6	652.89	9	199	11.29	<.0001
		y*island	transect(site)	0	2	0	776.43	2	206	20.89	<.0001
		y	transect	0	1	0	919.35	1	253.5	12.35	0.0005
		y	transect(island*site)	0	1	0	788.85	1	207	31.84	<.0001
macroalgae	10-yr	**y*transect(site)**	**transect(island)**	**18**	**0**	**17**	**1132.14**	**35**	**280**	**8.51**	**<.0001**
		y*site	transect(site)	5	0	1	1147.26	6	309	50.29	<.0001
		y*island	transect(site)	2	0	0	1212.73	2	313	121.76	<.0001
		y	transect	1	0	0	1479.1	1	343.1	64.21	<.0001
		y	transect(island*site)	1	0	0	1298.22	1	314	140.95	<.0001
macroalgae	9-yr	**y*transect(site)**	**transect(island)**	**19**	**0**	**33**	**1665.62**	**52**	**400**	**5.83**	**<.0001**
		y*site	transect(site)	7	0	2	1694.03	9	443.1	33.53	<.0001
		y*island	transect(site)	2	0	0	1815.01	2	450.2	109.7	<.0001
		y	transect	1	0	0	2109.87	1	496.8	73.46	<.0001
		y	transect(island*site)	1	0	0	1869.99	1	451.5	154.56	<.0001
macroalgae	post-bleaching	**y*transect(site)**	**transect(island)**	**10**	**1**	**41**	**607.52**	**52**	**156**	**3.48**	**<.0001**
		y*site	transect(site)	2	2	5	712.65	9	199	16.08	<.0001
		y*island	transect(site)	1	0	1	853.22	2	206	28.56	<.0001
		y	transect	1	0	0	1008.6	1	253	6.35	0.0123
		y	transect(island*site)	1	0	0	885.85	1	207	15.57	0.0001

Note: 'Inc.' denotes significantly increasing regression slope; 'Dec.' denotes significantly decreasing regression slope; 'NSD' denotes no significant trend in regression slope.

Table A1.4. Jonckheere-Terpstra (JT) test statistics for percent algae cover trends for Puerto Rico, St. Croix, and St. John, by sampling period.

Island	*Years*	*n*	*Direction**	*J*	*Z*	$p_{(one\text{-}sided)}$
Puerto Rico	2002-2004	224	Decreasing*†	9305	-2.2744	0.0115
	2002-2010	691	Increasing*	117460	1.8353	0.0332
	2006-2010	375	Increasing*†	38177	6.1761	0.00001
St. John	2001-2004	172	Not Significant	5520	0.5153	0.3032
	2001-2010	354	Increasing*†	33559	5.1363	0.00001
	2006-2010	151	Not Significant	4743	0.6048	0.2727
St. Croix	2002-2004	236	Decreasing*†	9644	-2.4868	0.0064
	2002-2010	959	Not Significant	216705	0.5801	0.2809
	2006-2010	581	Not Significant	74852	0.4618	0.3221
Buck Island NMR, St. Croix	2002-2004	99	Not Significant	1765	-1.4937	0.0676
	2002-2010	216	Not Significant	11051	0.4062	0.3423
	2006-2010	91	Not Significant	1670	-0.859	0.1952

*denotes significant trend with sequential Bonferroni correction ($P_i \leq (1\text{-}[1\text{-}\alpha]^{1/(1+k-i)})$; k=3
†denotes significant trend with standard Bonferroni correction ($P_i \leq \alpha/k = 0.025$); k=2

Table A1.5. Jonckheere-Terpstra (JT) test statistics for percent coral cover trends for Puerto Rico, St. Croix, and St. John, by sampling period.

Island	*Years*	*n*	*Direction**	*J*	*Z*	$p_{(one\text{-}sided)}$
Puerto Rico	2002-2004	224	Not Significant	11109	0.9886	0.1614
	2002-2010	691	Decreasing*†	100150	-3.8889	0.00001
	2006-2010	375	Not Significant	32031	1.0669	0.143
St. John	2001-2004	172	Decreasing*†	4606	-2.0161	0.0219
	2001-2010	354	Decreasing*†	24006	-3.5117	0.0002
	2006-2010	151	Not Significant	4070	-1.609	0.0538
St. Croix	2002-2004	236	Not Significant	11542	0.718	0.2364
	2002-2010	959	Decreasing*†	198263	-3.1528	0.0008
	2006-2010	581	Not Significant	76143	1.0188	0.1542
Buck Island NMR, St. Croix	2002-2004	99	Not Significant	1851	-0.961	0.1683
	2002-2010	216	Decreasing*†	8770	-3.9026	0.00001
	2006-2010	91	Not Significant	1813	0.1282	0.449

*denotes significant trend with sequential Bonferroni correction ($P_i \leq (1\text{-}[1\text{-}\alpha]^{1/(1+k-i)})$; k=3
†denotes significant trend with standard Bonferroni correction ($P_i \leq \alpha/k = 0.025$); k=2

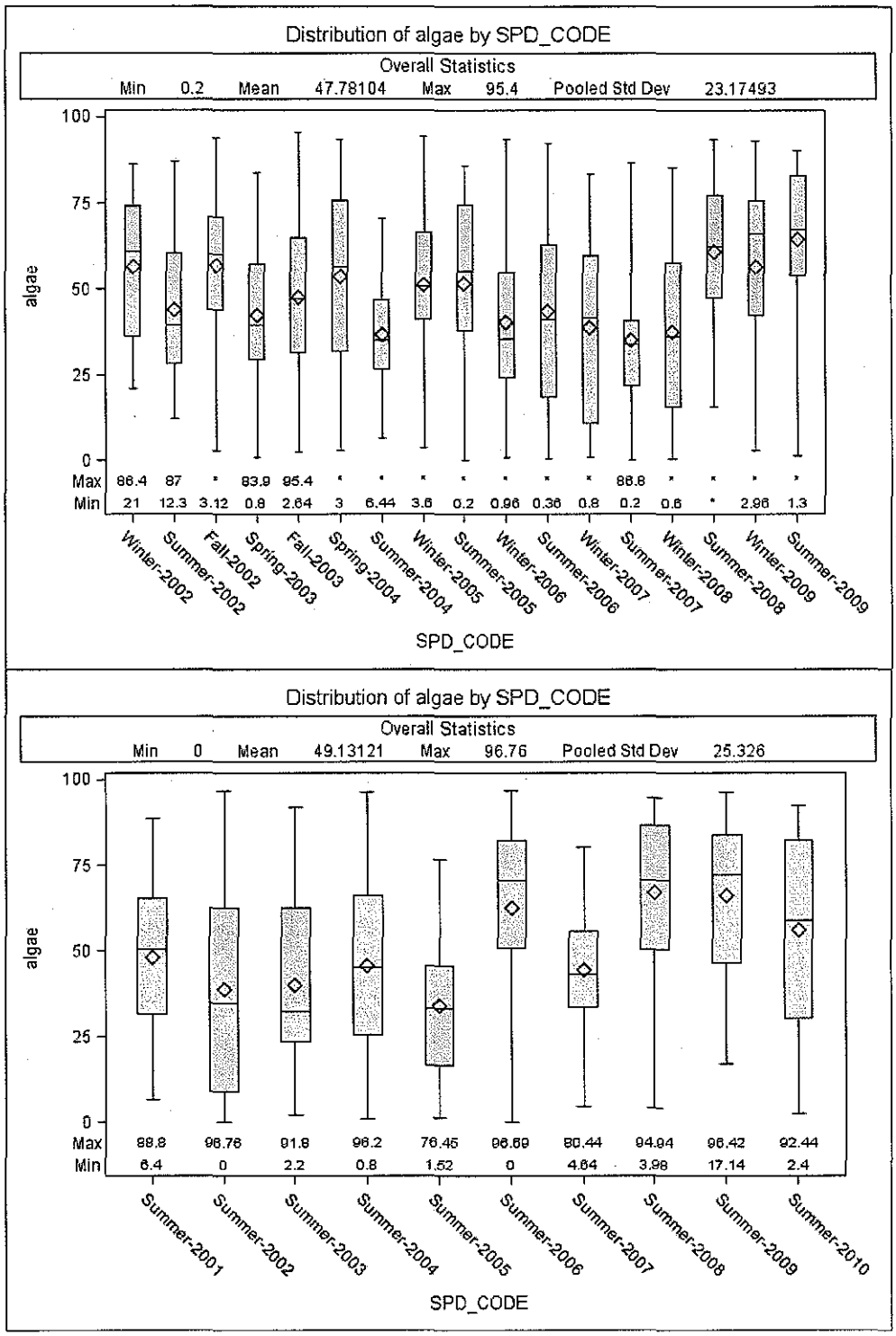

Figure A1.1. Trends in percent cover of 'algae' relative to sampling period for Puerto Rico (top) and St. John (bottom).

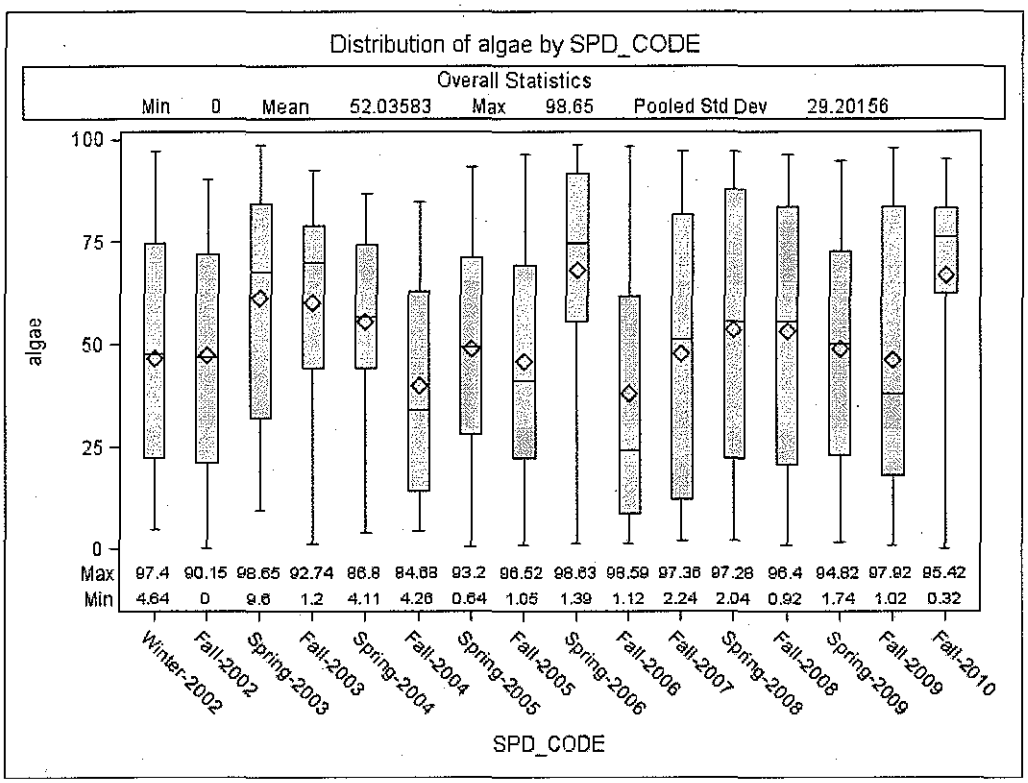

Figure A1.1 con't. Trends in percent cover of 'algae' relative to sampling period for St. Croix.

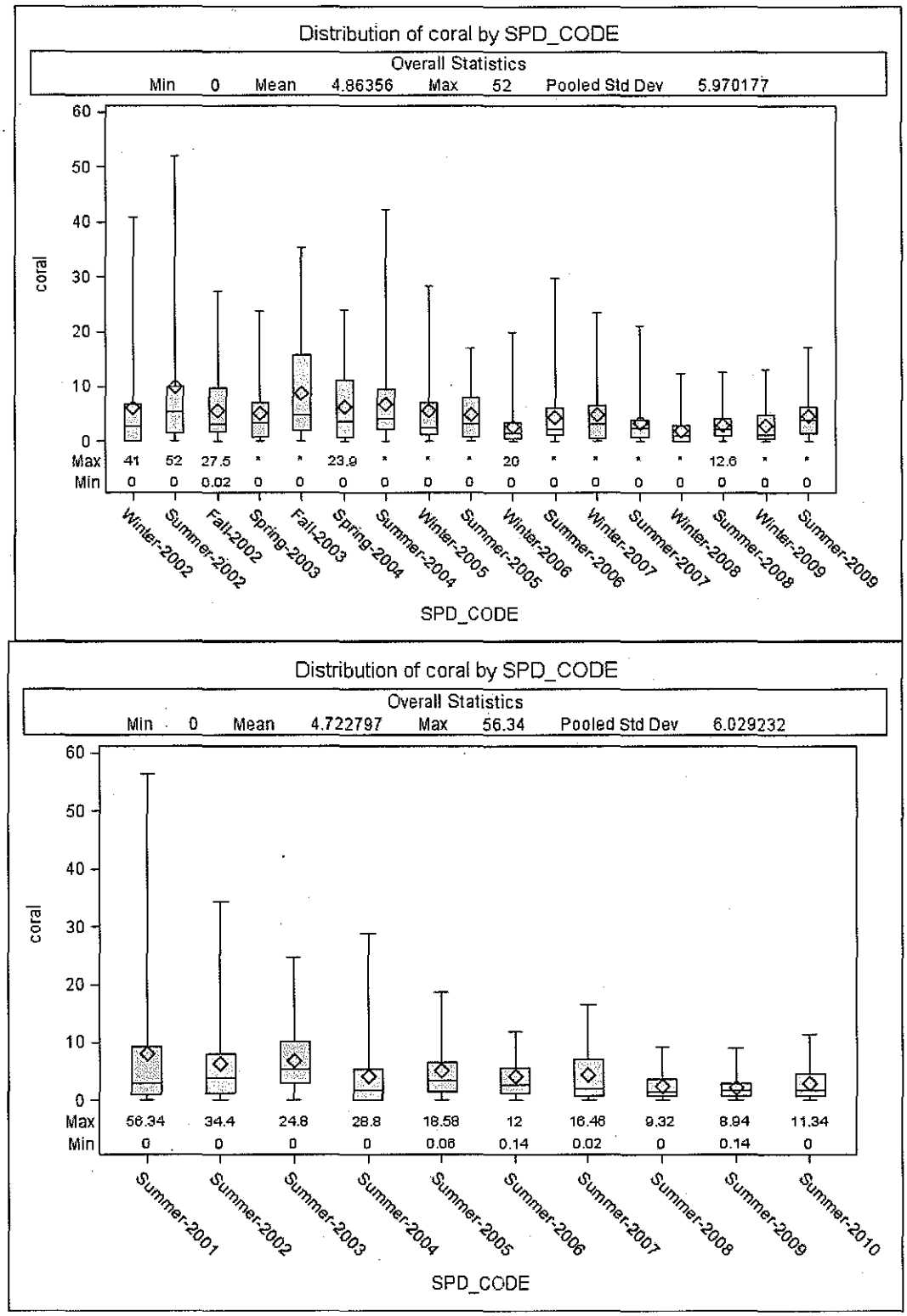

Figure A1.2. Trends in percent cover of 'coral' relative to sampling period for Puerto Rico (top) and St. John (bottom).

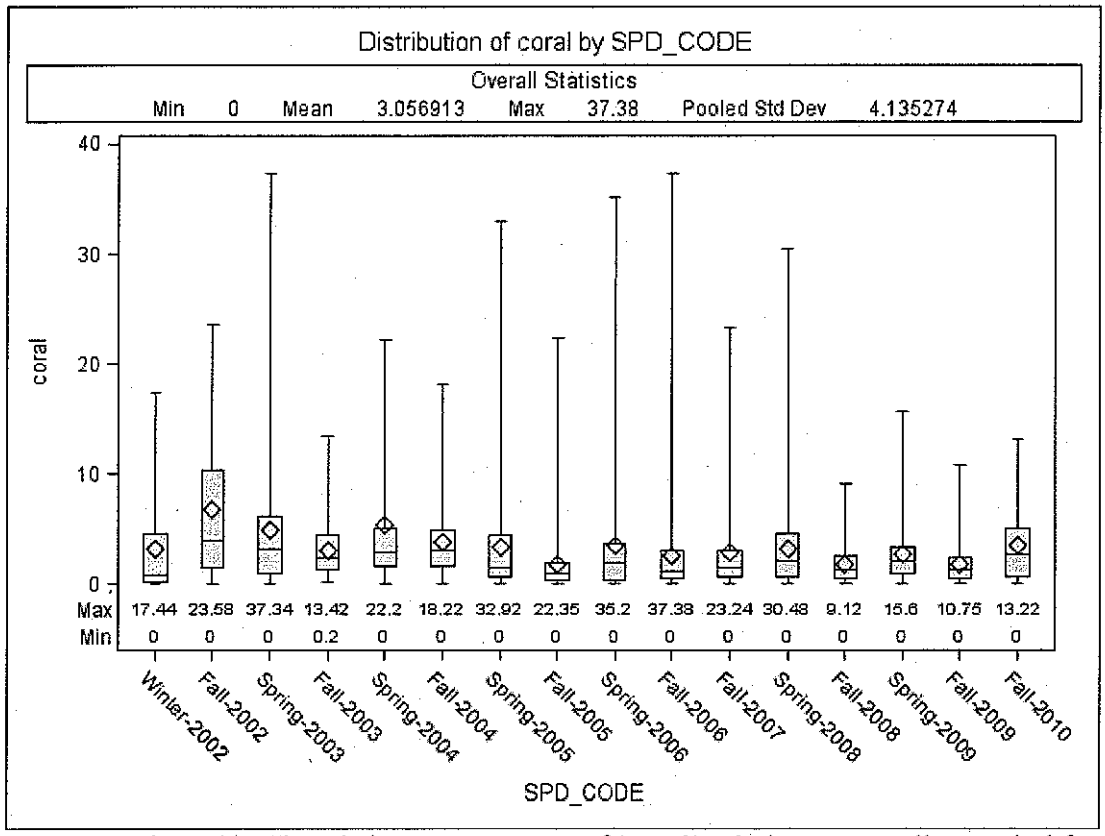

Figure A1.2 con't.. Trends in percent cover of 'coral' relative to sampling period for St. Croix.

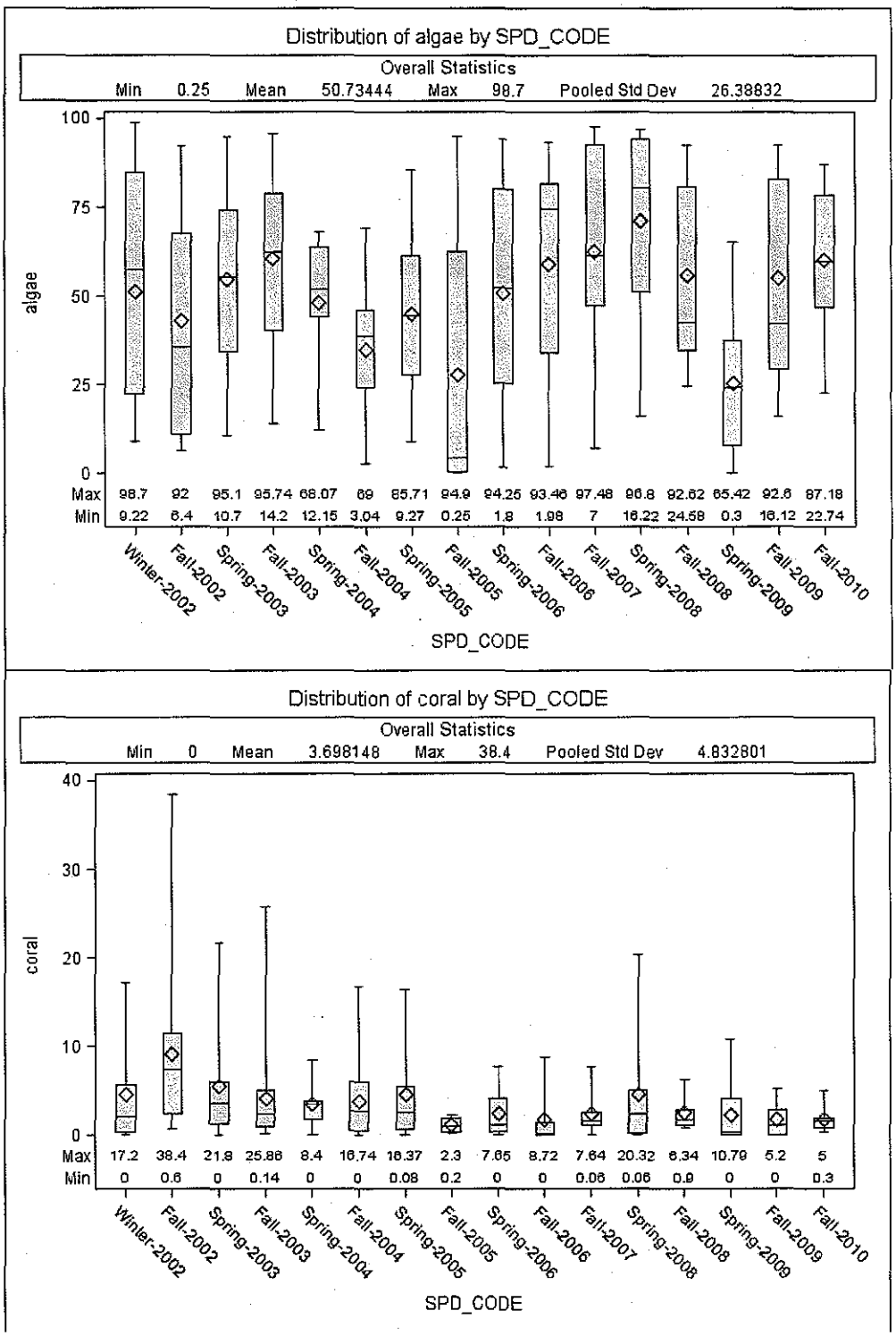

Figure A1.3. Trends in benthic coverage of 'algae' (top) and 'coral' (bottom) in Buck Island Reef National Monument, St. Croix relative to sampling period.

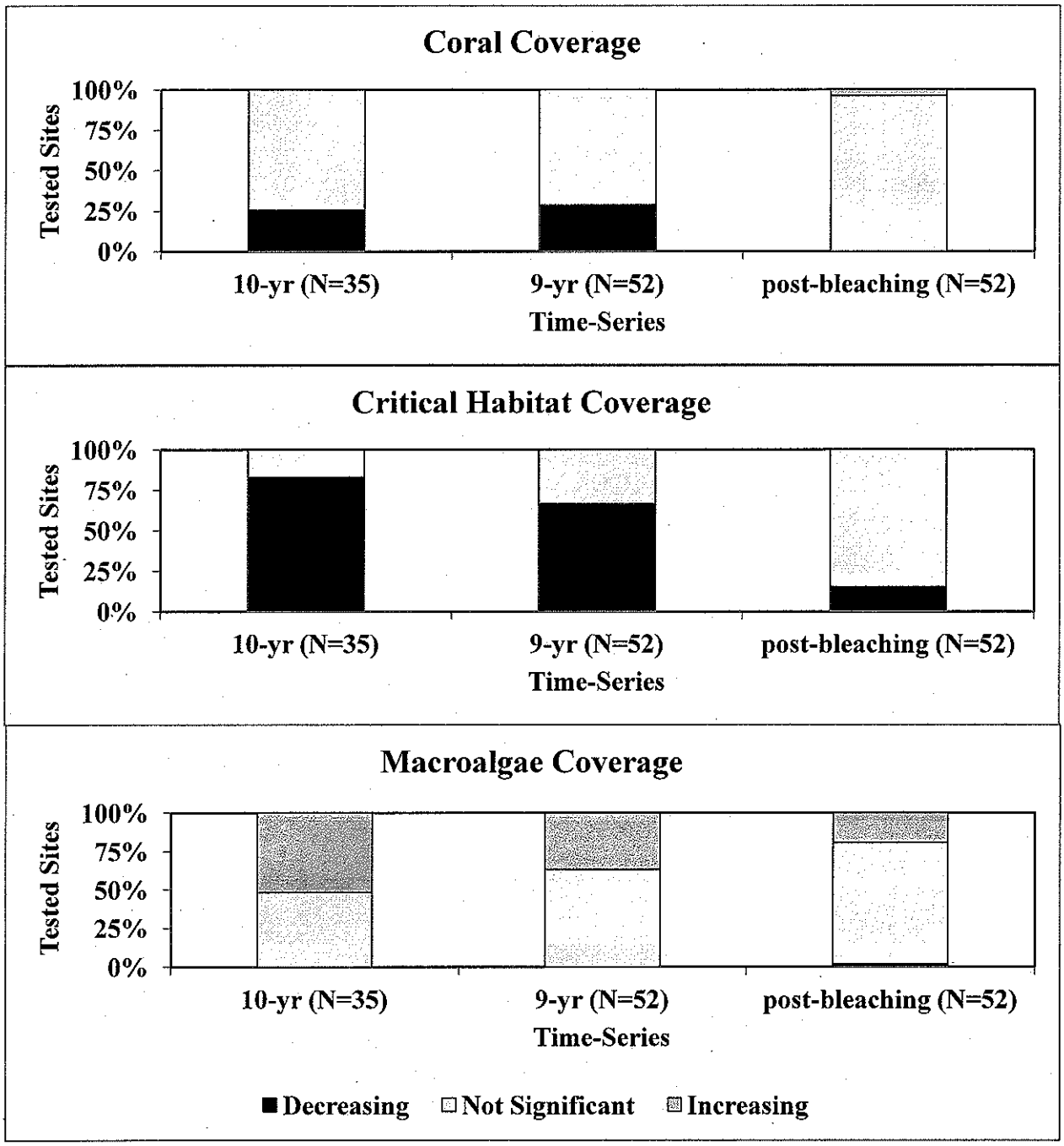

Figure A1.4. Trends in benthic habitat coverage by transect through time for A) 'Coral', B) 'Critical Habitat', and C) 'Macroalgae' under three different subsets of the data (10-yr, 9-yr, and post-bleaching (5-yr) time frames). Colors denote percentage of evaluated sites showing significantly decreasing (black), significantly increasing (dark gray), or no significant change (light gray) regression slopes across the time series.

Appendix 2 – Herbivorous Fish Biomass Information

Herbivorous Fish Biomass Information in Puerto Rico
No stock assessments have been conducted for parrotfish or surgeonfish in the U.S. Caribbean. The SEFSC states that "existing data are insufficient to quantify current, historical, and unfished biomass levels in the US Caribbean or to accurately describe how populations would respond to changes in removals." While the paucity of data on herbivorous populations in U.S. Caribbean confounds any attempt to truly understand the impact of proposed action on these species and ultimately, ESA-listed species, some information on herbivorous populations is available. We recognize these data have often been collected over relatively short time periods, and from relatively small and specific locations that may or may not be representative of all reefs. However, these data are the best available from which to try and determine the effects of the action.

García-Saís et al. (2008) reported some herbivorous fish abundance information that was collected from the La Paguera region of Puerto Rico from 2001-2006. The sampling was conducted as part of the NCREMP and Caribbean Coral Reef Monitoring Project. From 2001-2006 a total of 1,035 randomly selected locations (including coral reef/hardbottom, mangrove, and seagrass habitats) were sampled in southwestern Puerto Rico to collect information on fish assemblages across time and habitat type. Much of data reported provide information in broad terms herbivore families (i.e., *Acanthuridae* (surgeonfishes) and *Scaridae* (parrotfishes)).

García-Saís et al. (2008) reported data from 57 taxonomic families. The proportion of the top 17 families are reported below (Figure A2.1). Biomass and abundance were unevenly distributed throughout these taxonomic groupings. The majority of surveyed individuals came from one family (*Clupidae*), which made up less than 1% of the biomass. Parrotfishes (family *Scaridae*) were typically the most numerically abundant group at reefs/hardbottom, followed by wrasses (family *Labridae*), damselfishes (family *Pomacentridae*), gobies (family *Gobiidae*) and surgeonfishes (family *Acanthuridae*) (Garcia-Sais 2008).

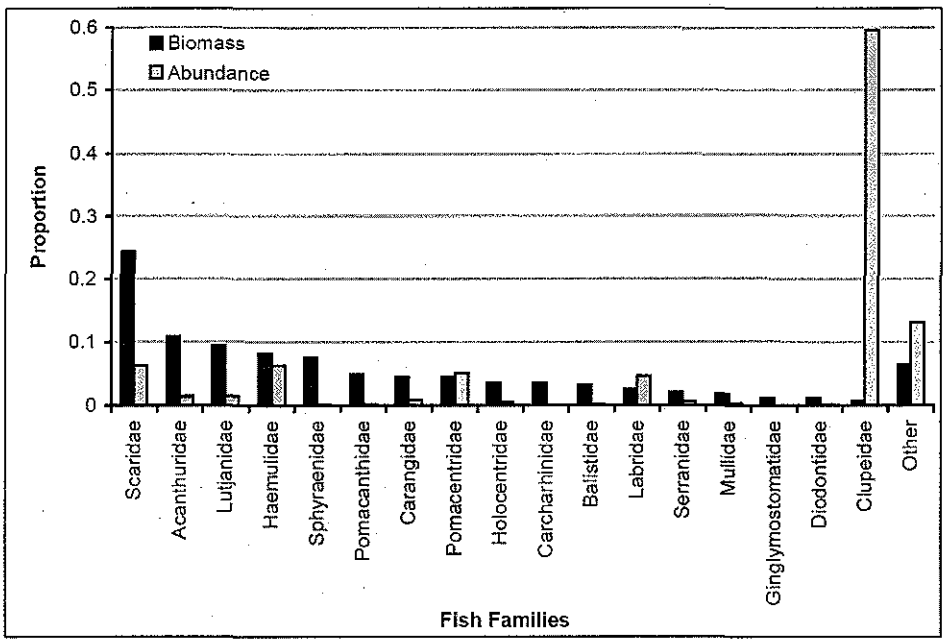

Figure A2.1 Proportional distribution of biomass and abundance of major families in La Parguera study area. (Source: CCMA-BB in Garcia-Sais (2008)).

The survey also documented parrotfishes (*Scaridae* family) as making up more of the reef fish community biomass than any other family in the study area. They were moderately abundant, but were generally larger bodied than most other families. Their density was generally greater on hardbottom sites, but during June 2001 was greatest in mangroves and in January 2005 was greatest on soft bottom habitats (Figure A2.2). Across all habitat types their densities typically ranged from 25 to 45 individuals per transect. The spike in density found during September 2003 was attributed to a detection of several large schools of princess parrotfish (*Scarus taeniopterus*), which were infrequently sighted during other surveys. No long-term trends could be identified from the available data (Garcia-Sais 2008).

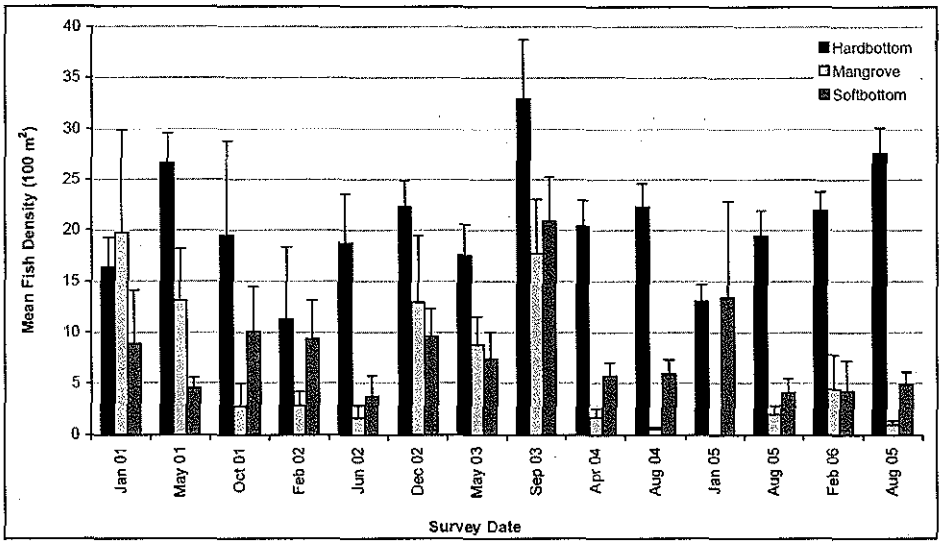

Figure A2.2 Density estimates of parrotfish (family *Scaridae*) in mangrove, soft bottom and reef/hardbottom habitats. (Source: CCMA-BB in Garcia-Sais 2008).

Herbivorous Fish Biomass Information in for the USVI

One source of information on herbivorous fish populations in the USVI comes from a long-term monitoring study conducted inside and outside the Virgin Islands National Park (VINP) in St. John. Initially established in 1989, the project monitored 18 reef sites until 1994. In 1995, a change in methodology was adopted and annual monitoring was restricted to four reference sites that were identified for their high biodiversity and conservation value (Friedlander and Beets 2008).

Commercial fishing is authorized in VINP. The parks enabling legislation allowed for the "customary uses of or access" to park waters for fishing, including the use of traps of "conventional Virgin Islands design" (Friedlander and Beets 2008). When the park was first established, fishers usually set only a few, smaller traps, but with the advent of outboard motors, line hauls, and larger fiberglass boats, fishermen now fish further offshore with a larger number of traps (Beets 1997, Garrison et al. 1998).

Comparison of fish trap catches at Yawzi Point, St. John, (inside VINP) between 1982-1983 and 1993-1994 showed substantial changes among the top 10 families, with an average decrease of 75.5% in numbers of fish caught among these families (Figure A2.3). However, parrotfishes (*Scaridae*) and surgeonfishes (*Acanthuridae*) showed a 61% and 39% increase, respectively in number of fish caught between 1982-83 and 1993-4.

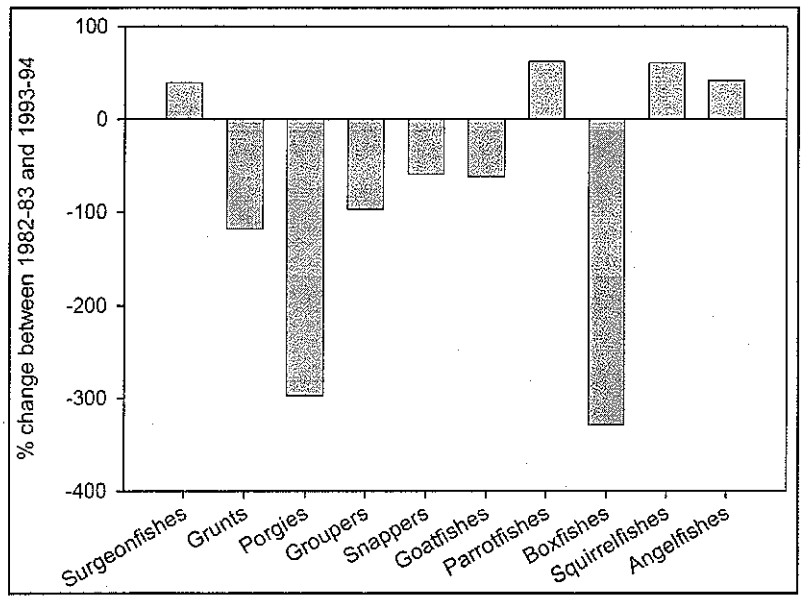

Figure A2.3 Percent change in number of individuals captured in fish traps at Yawzi Point, St. John between 1982-3 and 1993-4 for each of the top ten families
(Source: Adapted from Beets 1997 in Friedlander and Beets 2008).

The proportional catch at Yawzi Point by family also changed dramatically during this time (Figure A2.4). The largest proportional increase was for parrotfishes (+185%) and squirrelfishes (+170%), followed by surgeonfishes (+76%). Among the top 11 taxa, the average numbers of redband parrotfish and ocean surgeonfish caught both increased by approximately 50%; the average number of doctorfish caught during the period decreased by 26% (Figure A2.5)(Friedlander and Beets 2008). While the numerical proportion and number of individuals of parrotfish and surgeonfishes increased between 1982-83 and 1993-94, their mean lengths on decreased on average; blue tangs decreased approximately 22%, ocean surgeonfish decreased approximately 20%, doctorfish decreased approximately 26%, and redband parrotfish decreased approximately 21% (Friedlander and Beets 2008).

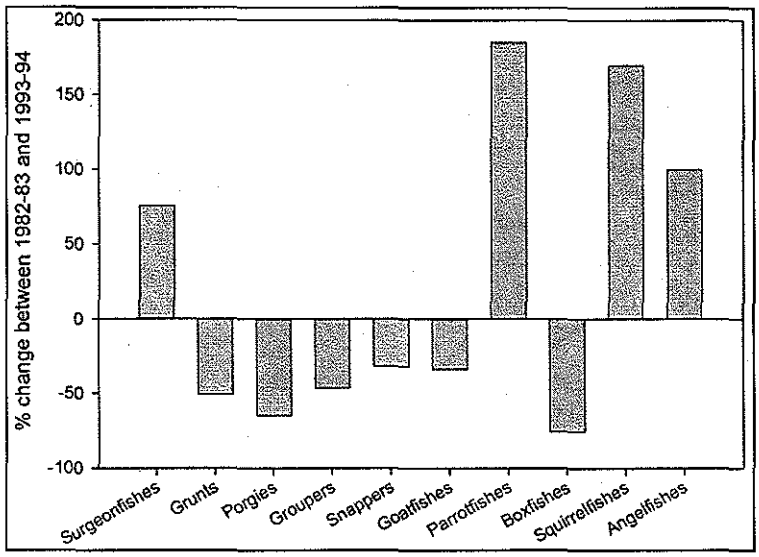

Figure A2.4 Percent change in numerical proportion of each of the top ten families captured in fish traps at Yawzi Point, St. John between 1982-3 and 1993-4
(Source: Adapted from Beets 1997 in Friedlander and Beets 2008).

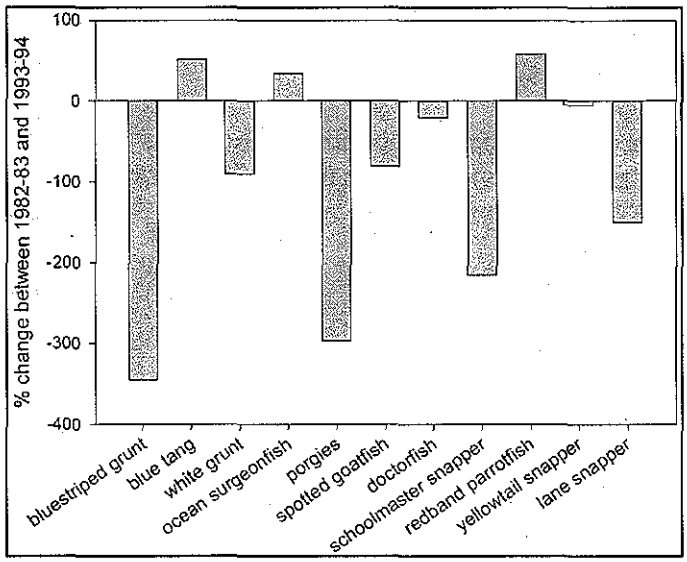

Figure A2.5 Percent change in numerical abundance of the top 11 taxa captured in fish traps at Yawzi Point, St. John between 1982-3 and 1993-4
(Source: Adapted from Beets 1997 in Friedlander and Beets 2008).

The VINP study also documented changes in temporal tropic dynamics, which the authors believed were closely tied to weather events (Friedlander and Beets 2008). The study found that herbivores (Herb.), followed by planktivores (Plank.) and secondary consumers (Sec. Consumer), were the most important trophic guilds by numerical abundance among the four permanent reefs (Figure A2.6). Following Hurricane Hugo, all trophic groups except piscivores (Pisc.) increased in numerical abundance but began to decline approximately five years after this storm. Herbivores and planktivores declined further following Hurricane Marilyn but quickly rebounded, with planktivores showing the most dramatic increase in numerical abundance. Herbivores continued to

increase in abundance, while planktivores increased and declined twice during this time period (Friedlander and Beets 2008).

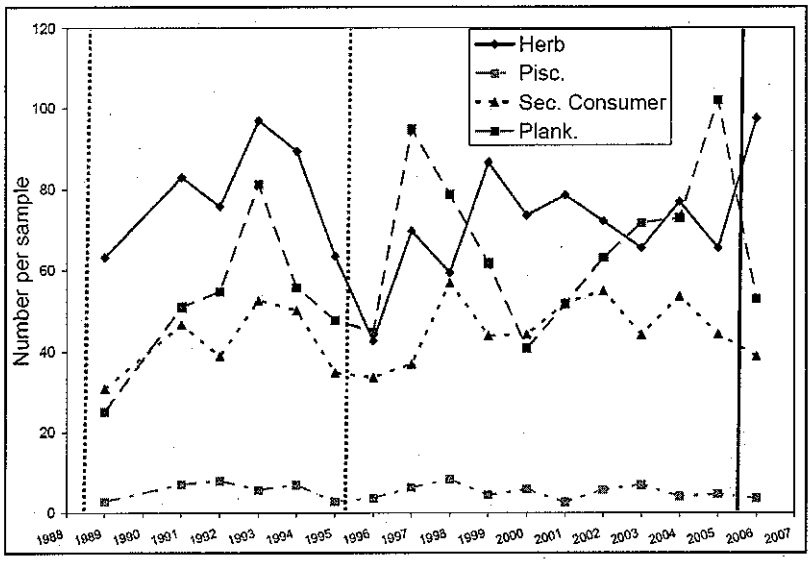

Figure A2.6 Mean number of individuals for each major trophic grouping. Vertical dotted lines denote periods of major hurricanes and solid line marks period of major bleaching event. (Source: Friedlander and Beets (2008))

Friedlander and Beets (2008), also reported that numerical abundance increases were noted for small benthic herbivores (i.e., damselfishes) and large mobile herbivores (i.e., parrotfishes and surgeonfishes) (Figure A2.7) during the course of the study.

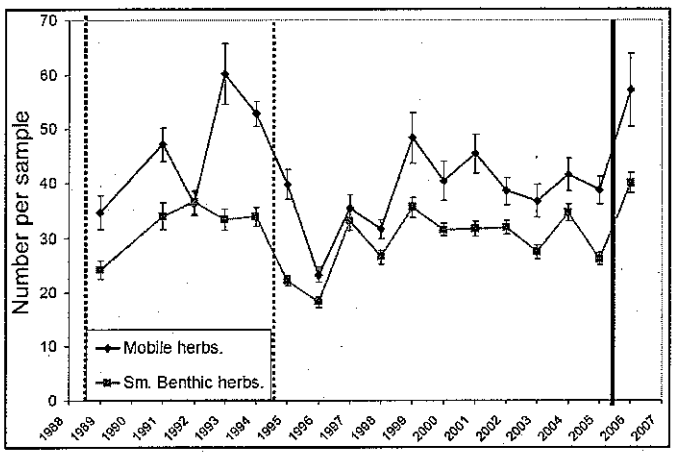

Figure A2.7 Mean number of mobile and small benthic herbivores for the four reference reefs pooled.
(Source: Friedlander and Beets (2008))

Parrotfishes (*Scaridae*) the second most numerically important family among the reference reefs, only damselfish were more numerically abundant. Striped parrotfishes (*Scarus iserti*) made up nearly 62% of the numerical abundance of parrotfishes, followed by redband (*S. aurofrenatum* – 19%), stoplight (*S. viride* – 11%), and princess (*S. taeniopterus* – 3%). Surgeonfishes (*Acanthuridae*) showed consistent and level

numerical abundance until 1995, but have shown a notable decline in numbers since 1996 (Figure A2.7) (Friedlander and Beets 2008). Parrotfishes were also the family of species with the highest biomass during the period at the four reference sites, but did not show a discernible trend during that time (Figure A2.8) (Friedlander and Beets 2008).

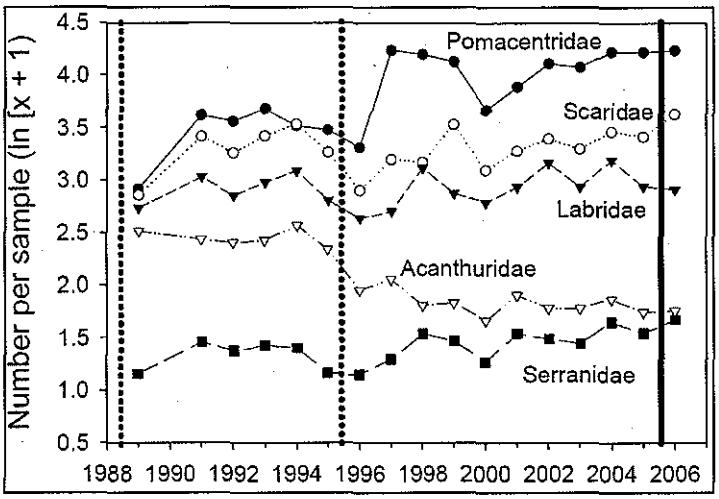

Figure A2.7 Mean number of individuals (ln[x+1]) of the five most commonly observed families from 1989 to 2006. (Source: Friedlander and Beets (2008))

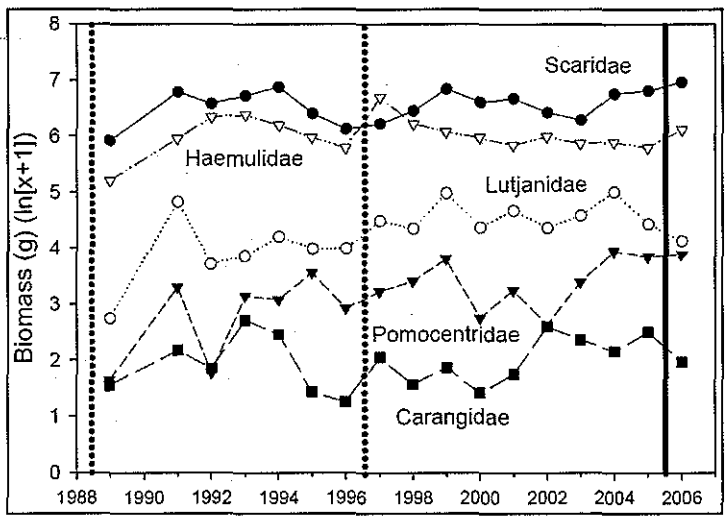

Figure A2.8 Mean biomass (g) (ln[x+1]) of the five most commonly observed families based on weight from 1989 to 2006. (Source: Friedlander and Beets (2008))

Rothenberger et al. (2008) also provide some limited information on herbivorous fish populations in the St. Thomas and St. Croix. From 2001-2006, NOAA CCMA-BB survey found that in St. Croix, "herbivores" comprised more of the biomass than "piscivores" for all survey periods except during August 2001 (Figure A2.9).[33] Herbivores in St. John also consistently comprised more of the biomass than piscivores

[1] Herbivores included damselfish, parrotfish, surgeonfish, etc.; while piscivores included red hind, other groupers, snappers, etc. (Rothenberger et al. 2008). See Randall 1967 for further discussion of species included in these groups.

for all years except 2003 (Figure A2.9). Fluctuations in relative biomass of herbivores and piscivores most likely relate to the occurrence of large schooling jacks or snappers during surveys (Rothenberger et al. 2008). Herbivorous fish biomass in St. Croix fluctuated annually but remained between 1,000-2,000g/100m^2. However, during 5 of 11 survey periods herbivore biomass greater than 2,000g/100m^2. In St. John, herbivore biomass fluctuated between approximately 500-3,500g/100m^2. A spike in herbivore biomass was noted in 2004, but biomass declined in each of the following two years in the study, to its lowest value reported in Rothenberger et al. (2008).

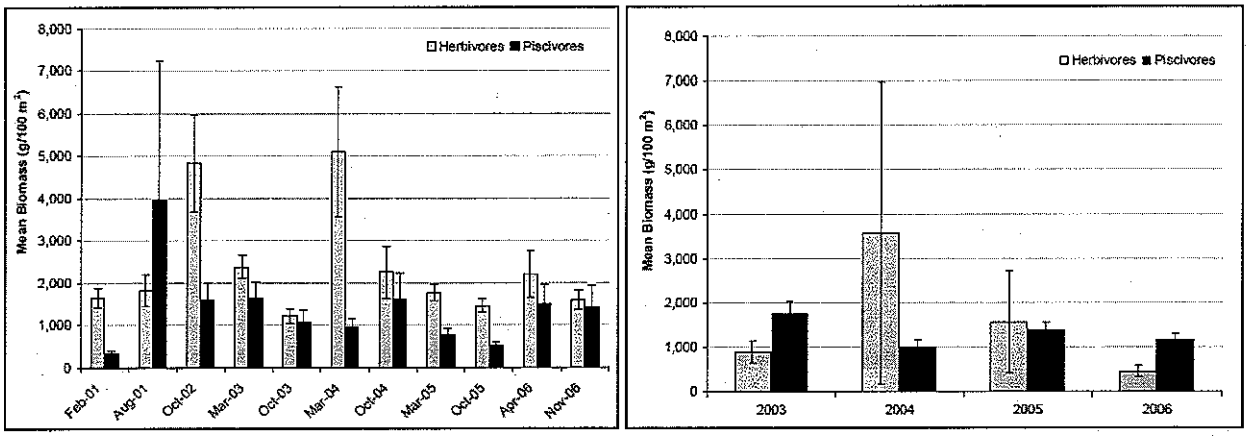

Figure A2.9 Estimates of mean (± SE) biomass of herbivores and piscivores from all surveys in St. Croix (left) and St. John (right). (Source: NOAA CCMA-BB in Randall et al. 2008).

UVI-CMES conducted fish surveys off of St. Thomas between 2003 and 2006. For St. Thomas, the 2003 surveys were conducted on six sites south of St. Thomas within three strata (nearshore, mid-shelf and shelf-edge). In 2004, nearshore sites were dropped from the survey and in 2005 and 2006 one additional mid-shelf site and one shelf-edge site were added Rothenberger et al. (2008).

The pooled data from the St. Thomas study indicated no pronounced changes in fish asssemblage structure on reef sites from 2003-2006. During that period, total fish abundance was not significantly different over time (p=0.080) nor was average species richness (p=0.538). However, a comparison of repeated sites shows fairly high variability in fish abundance between and within sites. Fish abundance by family also varied across the time series, apparently due to natural variation, seasonality and variable recruitment (Nemeth et al. 2006). In particular, *Acanthurid* and *Scarid* numerical abundance varied over time on midshelf reefs. *Scarids*, represented primarily by the princess, striped, and redband parrotfish (*Scarus iserti*, *S. taeniopterus* and *Sparisoma aurofrenatum*) were also much more abundant nearshore than offshore, with most individuals under 20 cm (Nemeth et al. 2006).

The USVI DPNR-DFW conducted the St. Croix study from 2003-2005 meant to mirror the UVI-CMES fish surveys off of St. Thomas. During the study period eight St. Croix reef sites were analyzed. The aggregated data (all sites pooled) among years indicate that there were no pronounced changes in reef fish assemblage structure during the

monitoring period. No significant difference in average fish abundance (p=0.086)] or average fish species richness was detected over time (p=0.16). This finding reflects the high variability in fish abundance among sites within any given year (Rothenberger et al. 2008). In all years, omnivores dominated the reef fish assemblage in terms of biomass. Herbivore biomass represented approximately 30% of entire assemblage. Piscivores contributed least to assemblage biomass (10-14%) and were least abundant numerically (2.7-3.1% of all fish observed). Among the years observed, there was no clear indication of a change in trophic composition through time (Rothenberger et al. 2008). Figure A2.10 illustrates the size frequency distribution for *Scarids* observed during 2003-2005. These data did not indicate a trend towards decreasing mean size during the study period. However, a relatively few number of parrotfish in the >30 cm size class, which are targets of the commercial fishery, were observed. The observed low frequency with which parrotfish attain large body size may be indicative of increased fishing mortality rates (Rothenberger et al. 2008).

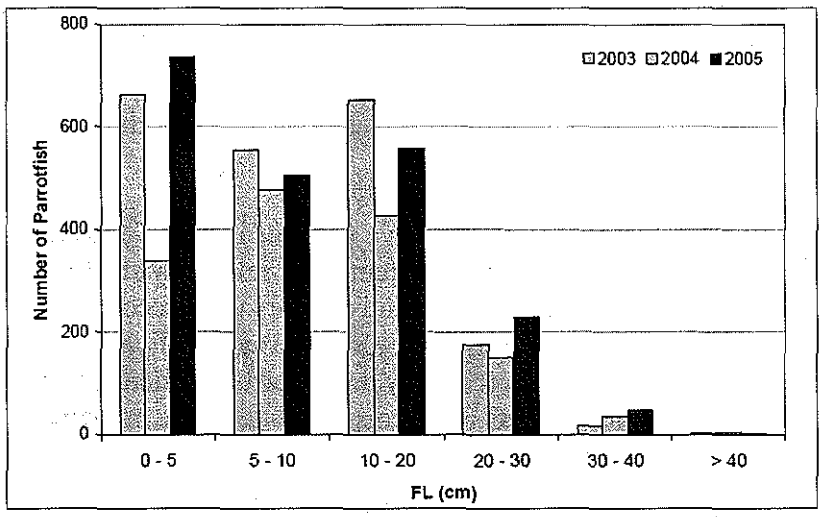

Figure A2.10 Size-frequency distribution of parrotfishes (all *Scarids* pooled) observed in St. Croix monitoring reef fish surveys, 2003-2005. (Source: W. Toller, ASI, in Rothenberger et al. 2008)

Finally, Pittman et al. (2008) evaluated fish assemblages and benthic habitats in and around BIRNM (Figure A2.11). That assessment is somewhat informative of parrotfish and surgeonfish populations in eastern St. Croix.

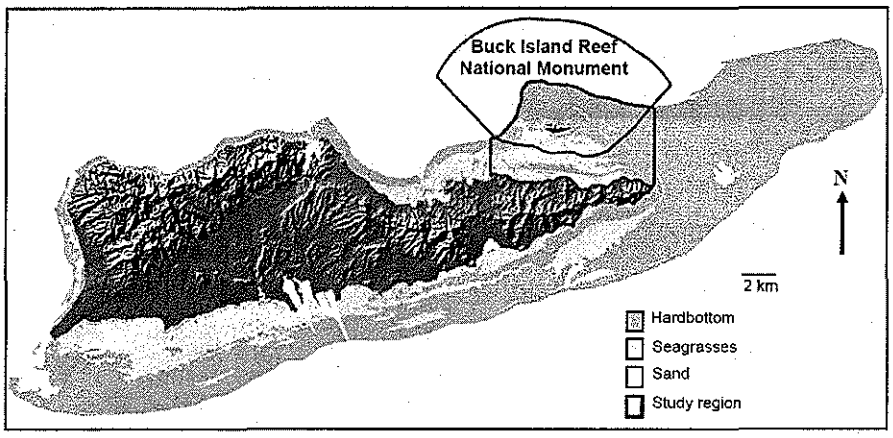

Figure A2.11 Map of the Study Area Evaluated by Pittman et al. (2008)

Pittman et al. (2008) reported that across their entire study area the density of striped parrotfish exhibited a year after year decline across three consecutive years (Table A2.1) and its biomass was significantly ($p=<0.01$) lower in 2006 than 2003. The mean parrotfish density and biomass fluctuated for year to year, though the mean biomass of redband parrotfish increased year after year across three consecutive years, with 2006 significantly higher than 2003 (Pittman et al. 2008).

Unlike inside BIRNM, no metric exhibited three consecutive years of increase outside BIRNM. Total parrotfish and redband parrotfish biomass remained relatively stable across the time series; however, declines over three consecutive years were recorded for striped parrotfish biomass and density. Striped parrotfish biomass was significantly lower ($p=<0.05$) in 2005 and 2006 than 2003. Interestingly, total parrotfish density also declined in each consecutive year. Given there was not a correlated decrease in total parrotfish biomass during the same period, this may indicate a shift toward larger individuals. Blue tang biomass and density fluctuated for year to year (Pittman et al. 2008).

Table A2.1 Reported Density (# fish/100m^2) of Redband Parrotfish, Striped Parrotfish, Blue Tang, and All Parrotfish in and Around BIRNM, 2003-2006 (Source: Pittman et al. 2008)

	Outside BIRNM				Entire Study Area				Inside BIRNM			
	2003	2004	2005	2006	2003	2004	2005	2006	2003	2004	2005	2006
Redband Parrotfish *(S. aurofrenatum)*	4.1	2.6	5.6	3.9	3.7	2.4	5.4	4.7	3.2	1.7	5.2	5.6
Striped Parrotfish *(S. iseri)*	5.4	3.9	1.8	1.0	4.4	3.4	3.1	2	3.4	2.4	4.5	3
Total Parrotfish	18.6	14.4	13.3	10.8	16.1	13.8	14.8	13.1	13.7	11.5	16.6	15.4
Blue tang *(A. coeruleus)*	2.1	3	2.6	2.9	4.1	8.3	6.8	6.9	6.3	13.7	11.2	10.9

Table A2.2 Reported Biomass (g/100m^2)of Redband Parrotfish, Striped Parrotfish, Blue Tang, and All Parrotfish in and Around BIRNM, 2003-2006 (Source: Pittman et al. 2008)

	Outside BIRNM				Entire Study Area				Inside BIRNM			
	2003	2004	2005	2006	2003	2004	2005	2006	2003	2004	2005	2006
Redband Parrotfish *(S. aurofrenatum)*	190	312	185	213	181	200	204	255	175	136	221	304
Striped Parrotfish *(S. iseri)*	49	44	20	6	60	51	31	46	72	38	43	84
Total Parrotfish	407	601	410	407	646	837	718	828	909	998	1039	1261
Blue tang *(A. coeruleus)*	97	372	113	260	444	1200	484	789	818	2091	874	1305

Pittman et al. (2008) also reported that from 2003-2006 parrotfish exhibited a skewed size frequency distribution towards a higher frequency of the smallest size classes, while surgeonfish were more normally distributed inside BIRNM, with size frequency distribution outside skewed towards a higher frequency of smaller size classes (Figure A2.12).

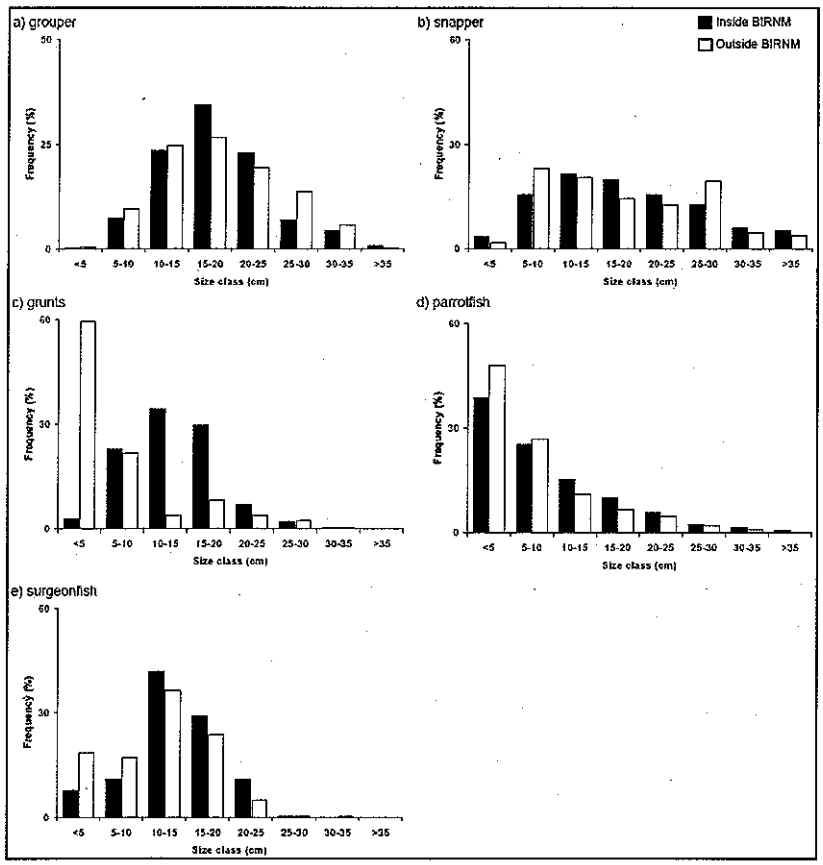

Figure A2.12 Length frequency histograms fish families over hardbottom sites inside and outside BIRNM: (a) grouper, (b) snapper, (c) grunts, (d) parrotfish and (e) surgeonfish

(Source: Pittman et al. 2008).

More specifically, blue tang exhibit a peak in frequency for small adults and ocean surgeonfish exhibit a peak for subadults, while redband and striped parrotfish showed a strongly skewed distribution, with high frequency of the smallest juveniles (<5 cm) and gradual decline with size with very few of the largest adults (Figure A2.13) (Pittman et al. 2008).

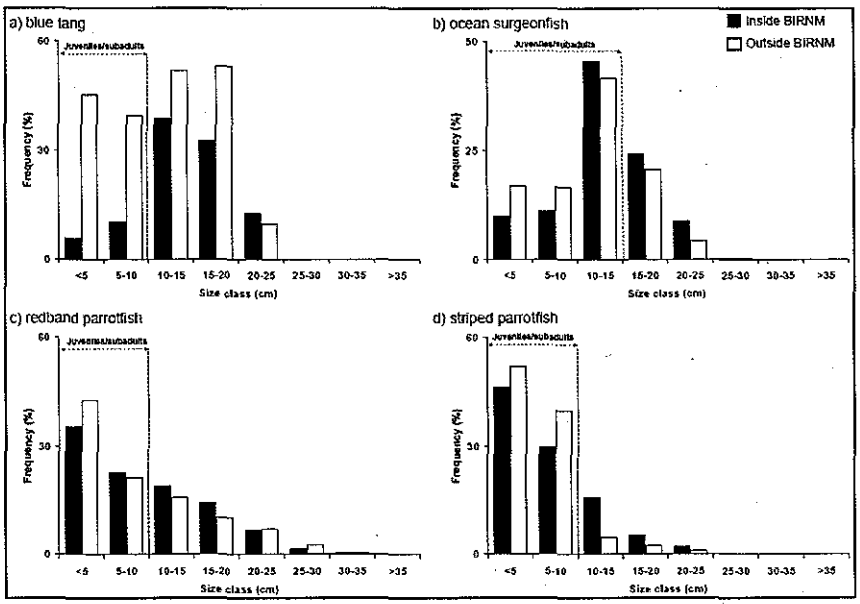

Figure A2.13 Size class frequency histogram for selected fish species over hardbottom sites inside and outside BIRNM. (a) Blue tang (*A.coeruleus*), (b) ocean surgeonfish (*A. bahianus*), (c) redband parrotfish (*S. aurofrenatum*), and (d) striped parrotfish (*S. iseri*).
(Source: Pittman et al. 2008)