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MEMORANDUM FOR: File – 1514-22d.10

FROM: for Miles M. Croom F/SE – Roy E. Crabtree, Ph.D.

SUBJECT: Endangered Species Act Section 7 Consultation on the Continued Authorization of Caribbean Spiny Lobster Fishery

The attached document constitutes the National Marine Fisheries Service's (NMFS) biological opinion based on our review of the continued authorization of Caribbean spiny lobster fishery. This opinion is based on information provided in the 2011 Caribbean ACL Amendment, censuses of commercial fishermen, communications with experts on Caribbean spiny lobster fishing, and peer-reviewed journal articles.

NMFS has analyzed the effects of the continued authorization of Caribbean spiny lobster fishery on listed species and designated critical habitat under our purview in accordance with section 7 of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. 1531 *et seq.*). It is NMFS' biological opinion that the action, as proposed, is likely to adversely affect sea turtles, staghorn corals, and the Puerto Rico Unit of *Acropora* critical habitat; however, the action is not likely to jeopardize the continued existence of these species or destroy or adversely modify critical habitat. The incidental take statement of this opinion anticipates and authorizes certain take levels by species. Authorization of take of listed species is contingent on compliance with the terms and conditions of the incidental take statement.

This concludes formal consultation on the action outlined above. 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 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.

Attachment

Ref: F/SER/2008/09173

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 Spiny Lobster Fishing Managed under the Spiny Lobster Fishery Management Plan of Puerto Rico and the U.S. Virgin Islands (SLFMP) (Consultation Number F/SER/2008/09173)
Consulting Agency:	NOAA, NMFS, SERO, Protected Resources Division (F/SER3)
Date Issued:	DEC 12 2011
Approved By:	Miles M. Croom for Roy E. Crabtree, Ph.D. Regional Administrator

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Select Abbreviations and Definition Used in this Document

ABC - Acceptable Biological Catch – The range of acceptable catch for a species or species group.

ACL - Annual Catch Limit – The annual level to which catch is limited to prevent overfishing from occurring.

EEZ – Exclusive Economic Zone

FMSY - Fishing mortality rate yielding MSY

MSY - Maximum Sustainable Yield – The greatest amount or yield that can be sustainably harvested under prevailing environmental conditions.

OFL - Overfishing limit

OY - Optimum Yield – The amount or yield that provides the greatest overall benefit to the Nation, taking into account food production, recreational opportunities and the protection of marine ecosystems.

ORCS - Only Reliable Catch Stocks

MSA – Magnuson-Stevens Fisheries Conservation and Management Act

MSRA – Magnuson-Stevens Fishery Conservation and Management Reauthorization Act

MSST - Minimum Stock Size Threshold or Overfished Threshold – The biomass level below which a stock would not be capable of producing MSY.

MFMT - Maximum Fishing Mortality Threshold or Overfishing Threshold – The maximum rate of fishing a stock can withstand or maximum yield a stock can produce, annually, while still providing MSY on a continuing basis.

NS1 – National Standard #1 – Refers to the requirement that conservation and management measures must be taken to prevent overfishing while achieving, on a continuing basis, the optimum yield (OY) from each fishery for the U.S. fishing industry.

USVI – United States Virgin Islands – Collectively this includes St. Thomas, St. John, and St. Croix.

USVI DPNR - United States Virgin Islands Department of Planning and Natural Resources

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 spiny lobster 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 spiny lobster fishing managed under the Spiny Lobster Fishery Management Plan (SLFMP) including all amendments implemented to date, as well as the actions proposed in Amendments 5. 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 to the SLFMP (hereafter Amendment 5), including a Final 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 Spiny Lobster Fishery Management Plan (SLFMP) in July 1980. 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 continued authorization of the fishery.

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 SLFMP, 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 SLFMP. NMFS concluded its proposed actions would have no anticipated impact on threatened or endangered species.

A formal consultation was conducted on Amendment 2 to the SLFMP in 2005. Amendment 2 was part of the Comprehensive Amendment to all the FMPs of the U.S. Caribbean to address required provisions of the Magnuson-Stevens Fishery Conservation and Management Act. The effects of the spiny lobster fishery were 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 was not likely to jeopardize the continued existence of green, hawksbill, leatherback, or loggerhead sea turtles. The spiny lobster fishery was determined to be likely to adversely affect only green, hawksbill, and leatherback 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 separately 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.

Amendment 3 to SLFMP considers measures to implement escape vents in traps used by the trap fishery sector. However, the implementation of the amendment has been postponed until a pilot study can be conducted on the effective size of escape vents. Amendment 3 is not being considered as part of the proposed action in this biological opinion.

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 October 8, 2008, F/SER2 requested reinitiation of Section 7 consultation for Amendment 4 to the SLFMP specifically to address adverse effects to elkhorn (*Acropora palmata*) and staghorn (*Acropora cervicornis*) corals, which were listed subsequent to completion of the 2005 Caribbean opinion. Additionally, on November 26, 2008, NMFS designed critical habitat for these species. Both of these listed species and their critical habitats overlap in some areas where fishing managed by the SLFMP is authorized; thus, they may be adversely affected by this fishery.

At the time of the reinitiation request, the first formal consultation evaluating the impacts of trap fishing on the newly listed corals species was being conducted for the Gulf of Mexico/South Atlantic spiny lobster fishery. Because of the novelty associated with analyzing the effects to corals from fisheries, F/SER3 believed it was prudent to wait for the completion of that biological opinion so the effects analysis framework used could be evaluated and used in the Caribbean spiny lobster opinion. Unfortunately, due to the paucity of data on *Acropora* abundance and fisheries information for the U.S. Caribbean region, much of the framework and effects analysis used in the Gulf of Mexico/South Atlantic spiny lobster biological opinion could not be immediately utilized.

On January 27, 2011, NMFS published a Notice of Intent that it was developing Amendment 5 to the Caribbean SLFMP. The amendment provided updated information on the fishery, making application of the effects analysis from the Gulf of Mexico/South Atlantic spiny lobster biological opinion more feasible. On July 21, 2011, the CFMC selected the preferred alternatives for Amendment 5. Formal consultation was initiated on November 12, 2011, once all necessary information on the preferred alternatives was available and all information needed to conduct an effects analysis was collected.

2.0 Description of the Proposed Action

In 2006, Congress amended the MSA with passage of the Magnuson-Stevens Fishery Conservation and Management Reauthorization Act (MSRA), and it was signed into law on January 12, 2007. 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" (MSA Section 303(a)(15)). The MSRA requires that ACLs and AMs be established in 2010 for those species classified as undergoing overfishing and in 2011 for most other managed species not classified as undergoing overfishing.

F/SER2 is proposing to continue authorizing the federal Caribbean spiny lobster fishery as managed under the SLFMP, including proposed Amendment 5 (CFMC and NMFS 2011). Amendment 5 was prepared by the CFMC and SERO and is designed to bring the Caribbean spiny lobster fishery into compliance with the 2007 revisions to the MSA.

Amendment 5 to the Spiny Lobster FMP

The purpose of Amendment 5 is to define the management reference points for spiny lobster based on an established year sequence for determining average annual landings; establish a recreational bag limit for spiny lobster harvest, and establish framework measures for the spiny lobster FMP.

Establishing a Year Sequence for Determining Average Annual Landings

This action transitions the management of the spiny lobster in the U.S. Caribbean from that established by the Caribbean Sustainable Fisheries Act (SFA) Amendment to that mandated by the MSRA. The former was dependent upon data sources of variable accuracy and precision. Under the proposed action the Council would select the longest time series of landings data that is available for each island group. The year 1988 is selected as the start year for commercial harvest in Puerto Rico because that was the first year for which a clearly defined method for calculating expansion factors to account for under-reporting, mis-reporting, and non-reporting became available for application to commercial harvest data. For St. Croix, species-group level commercial harvest data first became available for a full calendar year in 1998. Not until 2000 did species-group level commercial harvest data become available for the St. Thomas/St. John island group, so this is the first year for which species-group level commercial harvest data are available for all three island groups. Table 2.1 summarizes the proposed data periods for each island group. It is the data available from these periods that will be used to establish proxies for maximum sustainable yield (MSY) and, from those MSY proxies, OFL, ABC, OY and ACL. Commercial data would be derived from trip ticket reports collected by the state governments. Spiny lobster recreational data are not collected for Puerto Rico or USVI. Hence, MSY proxies will be determined using commercial harvest data (CFMC and NMFS 2011).

Table 2.1 Year Sequences Proposed for Determining Average Annual Landings

Island Group	Year Sequence
Puerto Rico	1988-2009
St. Croix	1999-2008
St. Thomas/St. John	2000-2008

Establishing Management Reference Points for Spiny Lobster

The MSA requires that FMPs specify a number of reference points for managed fish stocks, including: MSY, OFL, MSST, ABC, ACL, and OY. Together, these parameters are intended to provide the means to measure the status and performance of fisheries relative to established goals. Available data in the U.S. Caribbean are not sufficient to support direct estimation of MSY and other key parameters. In such cases, the NS1 guidelines direct regional fishery management councils to adopt other measures of productive capacity, including long-term average catch, which can serve as reasonable proxies. None of the parameter estimates considered by the Council represent empirical estimates derived from a comprehensive stock assessment; rather, all are calculated based on landings data averaged over alternative time series. All the reference points considered here are closely interrelated. For example, OY must be less than or equal to MSY, ACLs must be less than or equal to the ABC level recommended by a Council's SSC or other established peer-review process, and the ABC recommendation must be less than or equal to the overfishing threshold.

The proposed action would establish MSY proxies for Puerto Rico and the USVI that would equate to the median and mean of annual landings, respectively. Those estimates would be calculated using commercial landings data for the year sequence as noted previously (see Table 2.1). The proposed action also sets an OFL for Puerto Rico based on the ORCS method (see Berkson et al. 2011), and an ABC based on the OFL. Ultimately, the Council chose to set the ABC=OFL. The Council se the ACL=OY and the OY=0.9 x ABC. The proposed action would set the OFL in the USVI as the average of the available commercial landings data for the longest time series (see Table 2.1). In other words, the OFL will be equal to the MSY proxy with overfishing occurring when annual landings exceed the OFL. The proposed action also sets the ABC in the USVI equal to the OFL. The proposed management reference points for each island group are summarized in Table 2.2 (CFMC and NMFS 2011).

Table 2.2 Proposed Management Reference Points for Spiny Lobster by Island

Maximum Sustainable Yield	
Puerto Rico	MSY proxy = Median annual landings based on the year sequence selected by the Council (see Table 2.1)
STT/STJ and STX	MSY proxy = Mean annual landings based on the year sequence selected by the Council (see Table 2.1)
Overfishing Threshold	
Puerto Rico	OFL = MSY proxy adjusted using the ORCS scalar; overfishing occurs when annual landings exceed the OFL, unless NOAA Fisheries Southeast Fisheries Science Center determines the overage occurred because data collection/monitoring improved, rather than because landings actually increased.
STT/STJ and STX	OFL = MSY proxy; overfishing occurs when annual landings exceed the OFL, unless NOAA Fisheries Southeast Fisheries Science Center determines the overage occurred because data collection/monitoring improved, rather than because landings actually increased.
Acceptable Biological Catch/ABC Control Rule	
Puerto Rico, STT/STJ and STX	ABC= OFL
Optimum Yield/Annual Catch Limit	
Puerto Rico, STT/STJ and STX	OY = ACL = [ABC x (0.90)]

Establish a Recreational Bag Limit for Spiny Lobster in the U.S. Caribbean EEZ

The proposed action would also implement a recreational bag limit of 3 spiny lobster per fisher and 10 per vessel per day in the U.S. Caribbean EEZ. The goal of implementing bag limits is to ensure that the recreational ACL for spiny lobster is not reached until as near as possible to the end of the calendar year. Currently there are no recreational harvest data for spiny lobster in Puerto Rico. In the future, recreational harvest data could be gathered as part of the collection of information by Marine Recreational Fishing Recreational Survey (MRFSS) in both Puerto Rico and USVI. A bag limit quota would count against the overall ACL set for the entire spiny lobster fishery in both Puerto Rico and USVI (CFMC and NMFS 2011).

Establish Framework Provisions Specifically for Spiny Lobster

The proposed action would also establish a framework for adjusting management measures under the SLFMP.[1] Under the proposed action, management measures that could be adjusted through framework amendments include quotas, closures, limits, gear rules, and reference point modifications, among other things. 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 2.3 summarizes the proposed management measures that could be adjusted by the framework (CFMC and NMFS 2011).

[1] Framework actions involve management measures within the scope and criteria established by the FMP and implementing regulations. Framework measures are intended to describe future management actions, which would be implemented within a range as defined and analyzed in the FMP and associated analyses. The purpose of a framework action is to allow fisheries to be managed more responsively under conditions requiring "real time" management.

Table 2.3 Proposed Management Measures that Could be Adjusted by the Framework

a) Quota Requirements
b) Seasonal Closures
c) Area Closures
d) Fishing Year
e) Trip/Bag Limit
f) Size Limits
g) Gear Restrictions or Prohibitions
h) Fishery Management Unit (FMU)
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

2.1 Overview of Management and Regulations

2.1.1 The Federal Fishery Management Process

The U.S. Caribbean spiny lobster fishery is managed under the SLFMP and implementing regulations at 50 CFR Part 622, which were promulgated 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 SLFMP extend to 200 nautical miles offshore from the 9-mile nautical miles seaward boundary of the Commonwealth of Puerto Rico and the 3-mile nautical miles 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 approving, disapproving, or partially approving plans, amendments, and regulations to implement proposed plans and amendments after ensuring that management measures are consistent with the MSA, and with other applicable laws and executive orders.[2] The Secretary has delegated this authority to NMFS.

[2] 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

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 represented 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 SLFMP, and Implementing Regulations

The Council's original SLFMP (CFMC 1981) was implemented in January 1985 and was supported by an EIS. The FMP defined the Caribbean spiny lobster fishery management unit to include *Panulirus argus* (Caribbean spiny lobster), described objectives for the spiny lobster fishery, and established management measures to achieve those objectives. Primary management measures included:

- The definition of MSY as 830,000 lbs per year;
- the definition of OY as "all the non-[egg-bearing] spiny lobsters in the management area having a carapace length of 3.5 inches or greater that can be harvested on an annual basis," which was estimated to range from 582,000 to 830,000 lbs per year;
- a prohibition on the retention of egg-bearing (berried) lobsters (berried female lobsters may be kept in pots or traps until the eggs are shed), and on all lobsters with a carapace length of less than 3.5 inches;
- a requirement to land lobster whole;
- a requirement to include a self-destruct panel and/or self-destruct door fastenings on traps and pots;
- a requirement to identify and mark traps, pots, buoys, and boats; and
- a prohibition on the use of poisons, drugs, or other chemicals, and on the use of spears, hooks, explosives, or similar devices to take spiny lobsters.

The FMP acknowledged that "conclusive data regarding genetics between various geographic areas...are not available...and establishment of an international coalition will eventually be necessary to effectively manage this migratory species throughout its range" (CFMC 1981). The plan addressed only the species *P. argus* where it is limited to the geological platforms of Puerto Rico and the U.S. Virgin Islands essentially inside the 100-fathom isobath. It continued, "these shelf areas include not only the Commonwealth of Puerto Rico and the territory of the U.S. Virgin Islands, but also the entire chain of the British Virgin Islands. The lobster population recognizes none of these political entities nor the limits of territorial seas" (CFMC 1981).

Protection Act, Migratory Bird Treaty Act, National Environmental Policy Act, National Marine Sanctuaries Act, Paperwork Reduction Act, Regulatory Flexibility Act

The FMP discussed the stock unit issues as follows:

"The question of whether or not biologically distinct stocks of *P. argus* may be identified is not resolved. For purposes of this plan three biological assessment areas (distinguished by their user groups and geography) were assumed; (1) Puerto Rico, (2) St. Thomas and St. John, and (3) St. Croix. A single optimum yield is established. There is nominally one species and the source(s) of recruitment are not verified" (Section 4.2).

The original FMP also analyzed several different potential minimum sizes, ranging from 2.75 to greater than 3.5 inches carapace length. As in the Gulf of Mexico and South Atlantic FMP, the smaller minimum sizes were eliminated because they would not protect the spawning stock. The larger sizes were deemed to cost the fishery too much economically and socially, therefore, the 3.5 inch carapace length was chosen.

Amendment 1 to the SLFMP (CFMC 1990), implemented in May 1991, added to the FMP definitions of overfished and overfishing, and outlined framework actions that could be taken should overfishing occur. The amendment defined "overfished" as a biomass level below 20% of the spawning potential ratio (SPR). It defined "overfishing" as a harvest rate that is not consistent with a program implemented to rebuild the stock to the 20% SPR. That amendment was supported by an Environmental Assessment (EA) and a finding of no significant impact (FONSI).

Amendment 2 to the SLFMP (CFMC 2005), implemented in 2005 was part of the Comprehensive Amendment to the FMPs of the U.S. Caribbean to address required provisions of the Magnuson-Stevens Fishery Conservation and Management Act. This comprehensive amendment included a final supplemental environmental impact statement (FSEIS) that examined the impacts of amending the Council's FMPs to comply with several provisions of the MSA related to establishing biological reference points and stock status determination criteria, preventing overfishing and rebuilding overfished fisheries, and assessing and minimizing bycatch to the extent practicable.

A notice of intent to prepare a DEIS for Amendment 3 to the SLFMP was published in the Federal Register on October 9, 2007 (72 FR 57307). The proposed alternatives would have considered measures to implement trap escape vents in the trap fishery sector. However, Amendment 3 was postponed until a pilot study could be conducted on the effective size of escape vents.

Amendment 4 to the Caribbean SLFMP (CFMC, SAFMC, and GMFMC 2008), implemented in November 2008, required two actions to restrict imports of spiny lobster into the United States, based on minimum conservation standards to achieve an increase in the spawning biomass of the spiny lobster stock and increase long term yields from the fishery. The first action prohibited the importation of lobsters into the United States with a tail weight of less than 5 ounces, unless the importer could prove that the tail was taken from an animal with a carapace length greater than 3.0 inches, or if only the tail was present it had to be a minimum of 5.5 inches tail length. This action also required that any lobster tail imported into Puerto Rico and the USVI be no less than 6.0 ounces, unless the importer could prove that the tail was taken from an animal with a carapace length greater than 3.5 inches, or if only the tail was present it must have been a

minimum of 6.2 inches tail length (CFMC, SAFMC, and GMFMC 2008).

Amendment 4 also prohibited the importation of spiny lobster tail meat which was not in whole tail form with the exoskeleton attached; and the importation of spiny lobster with eggs attached or importation of spiny lobster where the eggs, swimmerets, or pleopods had been removed or stripped (CFMC, SAFMC, and GMFMC 2008). Table 2.1.2.1 summarizes the current regulations applicable to the federal spiny lobster fishery.

Table 2.1.2.1 Existing Federal Regulations Affecting Spiny Lobster in the U.S. Caribbean

Size Limits
Carapace length must be no smaller than 3.5 inches
Lobsters must remain whole for landing
Gear Prohibitions and Other Restrictions
The retention of egg-bearing females (i.e., "berried") is prohibited
Spears, hooks, and any other piercing device are prohibited
The use of poisons or explosives is prohibited
The use of gill and trammel nets is prohibited
Buoy, boat and trap identification and marking according to regulations are required.
Self-destruct panel and door fastenings on traps are required.
Pulling of another person's legally marked traps or pots without owner's permission is illegal, except by authorized officers.
Traps must have one escape panel, which could be the door.
At least one buoy that floats on the surface is required for all traps/pots fished individually for all fishing vessels that fish for or possess Caribbean spiny lobster in or from the EEZ.
At least one buoy is required at each end of trap lines linking traps/pots for all fishing vessels that fish for of possess Caribbean spiny lobster in or from the EEZ.

2.1.3 Fishery Data History, Monitoring and Reporting

The commercial and recreational sector data available for the U.S. Caribbean are limited and these limitations have been thoroughly described in various documents including: Caribbean SFA Amendment (2005) available at http://www.caribbeanfmc.com, SEDAR 2009 Data Workshop, and SEDAR 08A (2005) for spiny lobster. Among the primary concerns regarding the data are the scarce information on fishing effort, the lack of landings data, the lack of spatial/geographic information, missing information on life history parameters, and spatially and temporally limited fishery-independent data (SEDAR 2009) (CFMC and NMFS 2011).

Commercial sector landings data have been collected since 1974 from St. Thomas/St. John, since 1975 from St. Croix, and since 1967 (but in electronic format since 1983) from Puerto Rico. The U.S. Virgin Islands (USVI) landings data were not recorded to species group with adequate reliability until 1998 (St. Croix) and 2000 (St. Thomas/St. John). Complete and verified landings data were available through 2008 for USVI and 2009 Puerto Rico. Thus, the range of years available for calculating average landings estimates, for the purpose of setting ACLs for the pertinent commercial sector, include 2000-2008 for St. Thomas/St. John, 1998-2008 for St. Croix, and 1983-2009 for Puerto Rico (CFMC and NMFS 2011). However, for the reasons noted previously (see Section 2.0), these are not the time sequences the Council chose to use when estimating ACLs.

There are no federal licenses or permits issued for the commercial harvest of spiny lobster 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 Commonwealth of Puerto Rico requires commercial fishing licenses for fishing in commonwealth waters, with an additional permit requirement to harvest lobster. 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 NOAA's 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, and an additional permit is required to harvest spiny lobster recreationally in Commonwealth waters. In the USVI, no licenses or permits are required for fishing recreationally in territorial waters. The USVI is currently developing regulations for recreational fishing activity.

Management of Exempted Fishing 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 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) 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 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. The types and rates of interactions with listed species from the EFP and EEAA activities would be expected to be similar to those analyzed subsequently in the present 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 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 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 and EEAAs by SERO to be within the scope of this opinion. The included EFPs and EEAAs would be those involving fishing consistent with the description of the fishery in Section 2.3 and are 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 spiny lobster fishing under the Caribbean spiny lobster 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 spiny lobster), socio-economic factors (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 SLFMP as those waters 100 fathoms or shallower. The majority of fish habitat occurs in that area, as does the majority of fishing activity for spiny lobster. Beyond 100 fathoms, the sea bed drops off dramatically and is difficult to fish, as it requires larger vessels and more gear (i.e., more line for fish traps.), both of which are not typical of non-highly migratory species (HMS) U.S. Caribbean fisheries. Diving for spiny lobster becomes impractical at such depth.

The total area of fishable habitat in the U.S. Caribbean is about 2,467 nm^2 (see Figure 2.1). 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. We consider the fishable habitat in the EEZ the action area for this consulation.

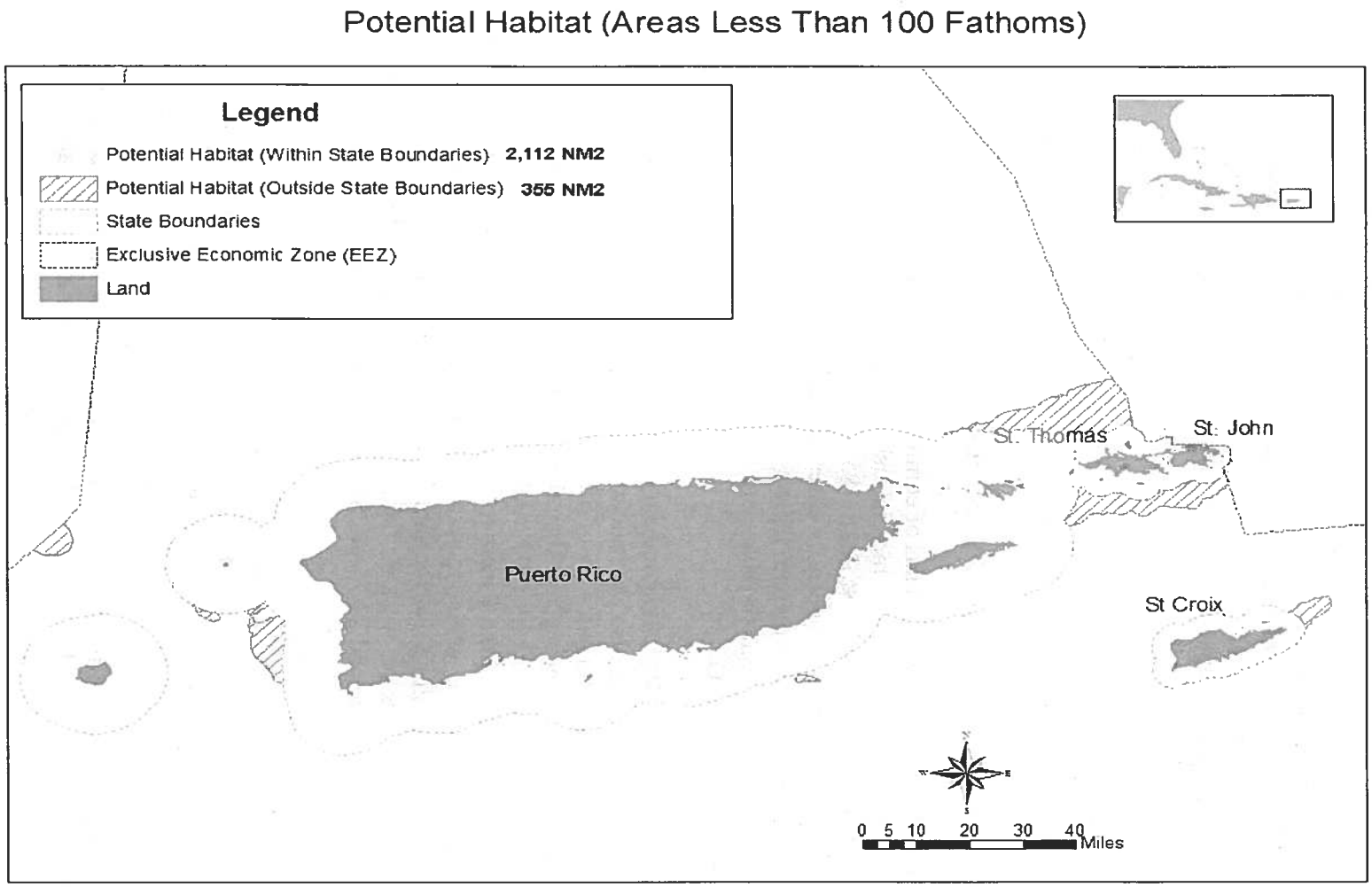

Figure 2.1. Potential Fishable Habitat Areas Less Than 100 Fathoms

2.3 Description of the Fishery

Specific fisheries information for the U.S. Caribbean for the federal fishery is generally lacking. There is a paucity of information on landings by species, commercial trip information, recreational effort, etc. for the federal fisheries. Generally speaking, landings data cannot be differentiated between the federal fisheries and those occurring in the commonwealth and territorial waters. The best source of information we have on all U.S. Caribbean fisheries comes from censuses conducted of commercial fishermen. Below is a brief description of the censuses conducted for each island area. The data in these censuses will be used throughout the effects analysis portion of the biological opinion.

2.3.1 USVI

Commercial Sector

Census of Marine Commercial Fishers of the U.S. Virgin Islands – 2011 (Kojis and Quinn 2011)

Kojis and Quinn (2011) conducted a census of commercial fishers in the USVI from July 2010 to March 2011. A total of 259 commercial fishers were interviewed, 102 in St. Thomas/St. John District (St. Thomas/St. John) (85% of licensed fishers) and 157 in St. Croix District (St. Croix) (89%). The census describes the current socioeconomic and demographic characteristics of commercial fishers in the territory and provides information on their fishing equipment (boats and fishing gear) and fishing related activities (Kojis and Quinn 2011).

The census reports that the USVI fishing industry is artisanal with most commercial fishers not only catching fish but also constructing and repairing their gear and boats. Only 30% of St. Thomas/St. John and 41% of St. Croix fishers carried out fishing related activities >36 hrs per week. This was less than the number of fishers who reported that they were full time in 2003 (i.e., Kojis 2004) (defined as >36 hrs per week). However, about 15% more St. Thomas/St. John fishers reported in 2010 than in 2003 that 100% of their income comes from fishing while the percentage of St. Croix fishers who reported 100% of their income from fishing was the same (41%). About a quarter of fishers in each district, spend 15 - 36 hrs per week on fishing related activities and slightly more (30.2% on St. Thomas/St. John and 27.7% on St. Croix) spend <15 hrs (Kojis and Quinn 2011).

Kojis and Quinn (2011) reported most commercial fishers owned a boat with a single outboard motor that they fueled with gasoline. The average boat length was 23.6ft (St. Thomas/St. John) and 22.5ft (St. Croix). Boats were constructed primarily of fiberglass (Kojis and Quinn 2011).

The reef fish fishery is the most important fishery in both St. Thomas/St. John and St. Croix. Coastal pelagic fish were the second and lobster the third most important fishery targets for St. Thomas/St. John fishers. On St. Croix spiny lobster was the second most

important fishery target and deepwater pelagic (dolphinfish, wahoo, etc.) the third (Kojis and Quinn 2011). About 50 St. Thomas/St. John licensed commercial fishers used about 5,500 fish traps, modified fish traps, and plastic lobster traps to target fish and lobster. In St. Croix, less than 1,000 traps were used by all fishers combined. Fishers, particularly those on St. Croix, have diversified into other gears such as multi-hook vertical setlines and scuba. These gears were used by fishers in St. Thomas/St. John as well, but not as commonly. Line fishing using hand lines or less commonly rods and reels was done by most fishers (Kojis and Quinn 2011).

Fishing was generally a day operation with fishers on St. Thomas/St. John fishing more hours during each trip but making fewer trips per week than St. Croix fishers. The mean fishing trip duration was 7.4 hrs for St. Thomas/St. John fishers and 6.5 hrs for St. Croix fishers. St. Thomas/St. John fishers fished 2.6 times a week and St. Croix fishers fished 3.2 times a week on average with one helper and occasionally a second helper (Kojis and Quinn 2011).

Census of Marine Commercial Fishers of the U.S. Virgin Islands – 2004 (Kojis 2004)
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 District and 217 in St. Croix District. Over 70% of licensed commercial fishers in St. Thomas/St. John District and all the licensed commercial fishers in St. Croix District were interviewed. The census described the socioeconomic and demographic characteristics of commercial fishers in the USVI at the time and provides information on their fishing equipment (boats and fishing gear) and fishing related activities (Kojis 2004).

The census reported two-thirds of fishers in the USVI identified themselves as full-time fishers and one third identified themselves as part-time or opportunistic fishers. Most commercial fishers owned a boat, most boats ranged from 16 to 25 ft, averaging 21 ft. Boats were constructed primarily of fiberglass and wood. USVI commercial fishers together and individually targeted a variety of fish and shellfish (including spiny lobster). The most commonly targeted categories of fish were reef fish and coastal pelagic fish. They used a wide range gears to target spiny lobster including traps, nets, and scuba. There were distinct differences in the gears used in each district. About 50 St. Thomas/St. John District commercial fishers used more than 7,500 fish traps, modified lobster traps, and plastic lobster traps to target fish and lobster. In St. Croix District, 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 District as well, but not as commonly (Kojis 2004).

Fishing in the USVI was generally a day operation with approximately 80% of fishing trips under nine hours. The average USVI fisher fished 3.1 times a week for 7.2 hours with one helper and occasionally a second helper.

U.S. Caribbean Fish Trap Fishery Cost and Earning Study (Agar et al. 2005)
Agar et al. (2005) conducted a socio-economic characterization of the U.S. Caribbean trap fishery, which that included information on fishing practice and gear information the trap fisheries in both the USVI and Puerto Rico. The census used in-person interviews randomly administered to 100 selected trap fishermen. These fishermen constituted nearly 25% of the estimated population at the time. The sample was stratified by geographic area and trap tier. The number of traps owned or fished to qualify for a given tier varied by island[3]. Information relevant to the USVI reported in that that study appears here. Information on that study relevant to the Puerto Rican trap fishery is discussed in the appropriate location below.

As a group, the vessels based in St. Thomas/St. John were larger vessels averaging 28 ft; vessels in St. Croix had an average length of 21 feet. Fiberglass hulled vessels were prevalent across the islands. All of the vessels sampled in St. Thomas/St. John had fiberglass hulls; in St. Croix 95% of the vessels had fiberglass hulls. The few wooden hulled vessels corresponded to the lower trap tiers of St. Croix (Agar et al. 2005).

Respondents in St. Thomas/St. John reported owning/using a total of 46 lobster traps on average, while no fishers reported using lobster traps in St. Croix. The maximum number of spiny lobster traps reported used by a fisher in St. Thomas/St. John was 460. On average, the greatest number of fish and lobster traps that a vessel would normally carry at any one time was 11 for the St. Thomas/St. John fleet, and 7 traps for the St. Croix fleet (Agar et al. 2005).

As a group, fishers in St. Thomas/St. John took 1.4 trips per week while fishers on St. Croix took 2.5 trips per week. Fishers from St. Thomas/St. John hauled 68 fish traps per trip, respectively. Fishers from St. Thomas/St. John fished an average of nine hours per trip, soaked trap for seven days. Fishers in St. Croix fished for six hours per trip, soaked traps for four days, and hauled an average of 26 traps per trip. In St. Croix, 84% of the respondents had a single trap per line, while only 10% of St. Thomas/St. John respondents indicated fishing a single trap per line (Agar et al. 2005).

Recreational Sector

The MRFSS program began in 1979 and was conducted in 1979 and 1981 in the USVI; however, it was discontinued in 1982 because of lack of funding. The MRFSS program was re-initiated in the USVI in 2000, but was subsequently discontinued due to data and statistical issues (CFMC and NMFS 2011). Currently, MRFSS/MRIP does not collect any recreational fishing data in the USVI. The National Angler Registry, which began in 2010 as part of the MRIP program, does not require anglers recreationally targeting spiny

[3] STT/STJ - Tier I = Fishers with 1 to 50 traps; Tier II = Fishers with 51-150 traps; Tier III = Fisher with 151 or more traps (Agar et al. 2005).
STX - Tier I = Fishers with 1 to 20 traps; Tier II = Fishers with 21 or more traps; Tier III = Fisher with 151 or more traps (Agar et al. 2005).
PR: Tier I = Fishers with 1 to 40 traps; Tier II = Fishers with 41-100 traps; Tier III = Fisher with 100 or more traps (Agar et al. 2005).

lobster to register. No data on the recreational harvest of spiny lobster in the USVI is available.

2.3.2 Puerto Rico

Commercial Sector

Census of Active Commercial Fishermen in Puerto Rico: 2008 (Matos-Caraballo and Agar 2011)

Matos-Caraballo and Agar (2011) provided the results of a census conducted in 2008. In all, 868 in-person, voluntary interviews were conducted with commercially active fishermen. The study separated Puerto Rico into four coastal regions: north, east, south and west and provided selected demographic characteristics, fishing dependence, fishing and marketing practices, capital investment on vessels, gear and equipment, and opinions about the biological and socio-economic health of the fishery. This selection summarizes the information presented in the study that is most germane to our effects analysis.

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. Most respondents stated that they targeted reef fish (77%), spiny lobster (49%), queen conch (33%), and/or baitfish (31%). The active commercial fleet consisted of 670 vessels, about 97% of which are vessels between 10 and 30 ft in length. Most hulls were built of fiberglass (65%) and, to a lesser extent, of fiberglass and wood (22%). Diving operations tended to have a captain and two helpers because the crew need to tend the boat and dive.

Hook-and-line gear was the most productive gear, followed by SCUBA and skin diving. Together SCUBA and skin diving they were responsible for approximately 29% of the total landings in 2008. Divers made up about 44% of the population of active fishermen, and primarily targeted queen conch and spiny lobster and, to a lesser extent, hogfish, parrotfish, boxfish (*Ostraciidae spp.*), and queen triggerfish (*B. vetula*). Skin diving mainly caught queen conch and spiny lobster. SCUBA and skin divers used 322 snares, 375 spears, 775 tanks, 578 gaffs, and 275 baskets (Matos-Caraballo and Agar 2011).

Traps or pots were the third most productive gears and accounted for almost 13% of the total 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 targeted 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 (Matos-Caraballo and Agar 2011).

U.S. Caribbean Fish Trap Fishery Cost and Earning Study (Agar et al. 2005)

Trap fishery vessels in Puerto Rico had an average length of 21 feet, 87% of which had fiberglass hulls. The few vessels with wooden hulls in Puerto Rico used very few traps. On average, respondents fished/owned 39 fish traps and an average of 11 lobster traps, and a vessel would normally carry 8 traps. Lobster traps tend to be smaller (24 x 24 x 48

inches) and have pre-cut pine or spruce wooden slats. Puerto Rican fishers took an average of 2.1 trips per week and fished six hours per trip on average. On average they hauled 27 fish traps per trip and traps soaked an average of seven days (Agar et al. 2005). Schärer et al. (2004) noted that the mean soak time for Puerto Rican fish traps was five days.

Recreational Sector

Unlike the USVI, the MRFSS does collect recreational sector data in Puerto Rico and has done so since 2000. Data are collected on recreational catch and effort targeting reef fish and on coastal and highly migratory pelagic species, but not on invertebrates such as queen conch and spiny lobster (two of the most commercially and recreationally important harvested species) (CFMC and NMFS 2011). Thus, no information on the recreational harvest of spiny lobster is available from MRFSS. The National Angler Registry, which began in 2010 as part of the MRIP program, does not require anglers recreationally targeting spiny lobster to register.

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[4]
Green sea turtle (*Chelonia mydas*)	Endangered/Threatened[5]
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).

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[4] The Northwest Atlantic Distinct Population Segment (NWA DPS).

[5] 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 2011 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 (75 FR 68468; November 8, 2010).

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 from Puerto Rico's west coast. Of the total 159 strandings, 16.3% (n=26) were attributed to human-related; 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. Applying the same percentages above 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 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. 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 easternmost records of loggerhead sea turtles in the United States were from 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 capture in fishing gear and are 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 USFWS 1991b). However, given their rarity and mainly transitory nature in the action area, we believe adverse effects on loggerhead sea turtle interactions via spiny lobster 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.

Elkhorn Coral

Elkhorn colonies are flattened to nearly round, with frond-like branches that typically radiate outward from a central trunk, firmly attached to the sea floor. Historically, this species formed dense thickets at shallow (<5 m) and intermediate (10 to 15 m) depths in many reef systems. Currently, the maximum range in depth reported for elkhorn coral is <1 m to 30 m, but the optimal depth range for this coral is considered to be 1 to 5 m (Goreau and Wells 1967). The preferred habitat of elkhorn coral is the seaward face of a reef (turbulent shallow water), including the reef crest, and the shallow spur-and-groove zone (Shinn 1963, Cairns 1982, Rogers et al. 1982). Colonies are occasionally exposed during low tide. Colonies generally do not form a thicket below 5 m depth, with maximum water depths of framework construction ranging from 3 to 12 m (see Table 1 in Lighty et al. 1982).

Following the completion of our effects analysis (see Section 5.1-5.4) we concluded that the proposed action was not likely to adversely affect elkhorn coral. The analysis indicated that all the anticipated effects from fishing were likely to occur in a depth range (i.e., deeper than 15 ft) and in environmental conditions not preferred by the species (i.e., not turbulent shallow-water reef crests, or shallow spur-and-groove zones). Below is a summary of our conclusions from Section 5.1-5.4.

In Section 5.1, we determined that *Acropora* critical habitat does not occur in the EEZ off St. Thomas/St. John. Because *Acropora* corals are sessile species that only settles/re-establishes on habitat types currently designated as critical habitat, we only anticipate finding it in areas designated as critical habitat. Because designated critical habitat does not occur in the EEZ off St. Thomas/St. John, we determined that any type of commercial or recreational harvest of spiny lobster from the EEZ off these islands would not affect *Acropora* corals or their critical habitat. Therefore, we concluded there would be no effect to elkhorn or staghorn coral in the EEZ off St. Thomas/St. John.

In Section 5.2 we determined that commercial and recreational hand harvest (i.e., dip nets and snares) in both the USVI and Puerto Rico was not likely to adversely affect *Acropora* corals or critical habitat. In Puerto Rico, we determined it was extremely unlikely that recreational hand harvest was occurring the EEZ. We also determined that while commercial hand harvest may be occurring the Puerto Rico EEZ, the few number of trips and the selectivity of the gear meant it was extremely unlikely that adverse affects would occur to *Acropora* or critical habitat and any such effects are discountable.

In St. Croix, fisher censuses (i.e., Kojis 2004 and Kojis and Quinn 2011) indicate that no commercial diving for spiny lobster occurs in the EEZ. As with Puerto Rico, we also determined it was extremely unlikely that recreational hand harvest was occurring the

EEZ of St. Croix. Based on this information, we determined that *Acropora* or critical habitat in St. Croix were not likely to be adversely affected by commercial or recreational hand harvest.

Section 5.3 evaluated the effects of recreational and commercial trapping on *Acropora* and *Acropora* critical habitat. We ultimately determined it was extremely unlikely that any commercial or recreational trapping was occurring in the EEZ of St. Croix and that it was extremely unlikely that recreational trapping was occurring in the EEZ of Puerto Rico. Thus, we concluded adverse affects from these gears and sectors to elkhorn or staghorn coral were extremely unlikely to occur and discountabl. We did conclude that commercial trapping was likely to occur in the EEZ off Puerto Rico and those traps could adversely affect staghorn corals. However, because of the water depths in the EEZ (i.e., 70+ ft), we concluded it was extremely unlikely that elkhorn coral, which prefers the shallow (i.e., ~15ft or less) turbulent waters near reef crests, would overlap spatially with areas where trap effects could occur. Thus, we concluded adverse effects from spiny lobster trapping were extremely likely to occur to elkhorn coral and were discountable.

Vessel and anchoring impacts were evaluated in Section 5.4. We concluded that the lawful operation of vessels was not likely to adversely affect either species of *Acropora* or critical habitat. We determined that anchoring of hand harvest vessels in the Puerto Rico EEZ could adversely affect *Acropora*. However, our analysis determined that the only likely spot that commercial hand harvest was occurring was in an area where elkhorn coral was unlikely to exist. Therefore, we concluded that anchoring effects were also not likely to adversely affect elkhorn coral.

Based on these determinations, we ultimately concluded that proposed action was not likely to adversely affect elkhorn coral. Because we did not reach this conclusion until we finished our effects analysis, elkhorn coral are considered throughout the analyses in Section 5.1-5.6 to document the rationale and process for reaching our final determination.

Sea Turtle Critical Habitat

Critical habitat for green, hawksbill, and leatherback sea turtles occurs in the Caribbean but does not occur within 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. 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). 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. 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 have little to no overlap with the critical habitat areas and the proposed action is extremely unlikely to have any measurable effect on sea turtles' use of these areas.

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 were 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 or years from the issuance 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 spiny lobster fishery would impact sea turtle species in light of the environmental baseline effects from the DWH event.

During the response phase to the DWH oil spill (April 26 – October 20, 2010) a total of 1,146 sea turtles were recovered, either as strandings (dead or debilitated generally onshore or nearshore) or were collected offshore during sea turtle search and rescue operations (Table 3.1). Subsequent to the response phase, a few sea turtles with visible evidence of oiling have been recovered as strandings. The available data on sea turtle strandings and response collections during the time of the spill are expected to represent a fraction (currently unknown) of the actual losses to the species, as most individuals likely were not recovered. The number of strandings 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. Kemp's ridley sea turtles appear to have been the most affected sea turtle species, as they accounted for almost 71% of all recovered sea turtles (alive and dead), and 79% of all dead turtles recovered. Green turtles accounted for 17.5% of all recoveries (alive and dead), and 4.8% of the dead turtles recovered. Loggerheads comprised 7.7% of total recoveries (alive and dead) and 11% of the dead turtles recovered. The remaining turtles were hawksbills and decomposed hardshell turtles that were not identified to species. No leatherbacks were among the sea turtles recovered in the spill response area. (Note: leatherbacks were documented in the spill area, but they were not recovered alive or dead).

Table 3.1 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**	**1,146**

(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. 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 or threat to marine animal health was present in the area. 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 turtle excluder device (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. 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 undertaken on those 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 Section 3.3.3, 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

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 spiny lobster fishery based on strandings records and their capture in other southeast fisheries using similar gear. Sea turtles are also vulnerable to vessel strikes. Staghorn corals may be affected by direct injury from fishing gear/vessels. 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 staghorn corals 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-c, ; NMFS SEFSC 2001, and Marine Turtle Expert Working Group (TEWG) (2007). Information on life history and threats to staghorn 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 Staghorn coral

As stated previously, we believe the proposed action is not likely to adversely affect elkhorn corals. This section focuses on the status of staghorn corals. However, because elkhorn and staghorn corals often co-occur, have very similar life history characteristics and biology, and face the same environmental threats, in some places in the subsequent discussion both species are discussed.

Staghorn coral was listed as threatened under the ESA on May 9, 2006. The Atlantic *Acropora* Status Review presents a summary of published literature and other currently available scientific information regarding the biology and status of both elkhorn and staghorn corals. The following discussion summarizes those findings relevant to staghorn coral and our evaluation of the proposed action.

Staghorn coral is one of the major reef-building corals in the wider Caribbean. Staghorn coral is characterized by staghorn-antler-like colonies, with cylindrical, straight, or slightly curved branches. Early descriptions of Florida Keys reefs referred to reef zones, of which the staghorn zone was described for many shallow-water reefs (Figure 3.3) (Jaap 1984, Dustan 1985, Dustan and Halas 1987). The structural and ecological roles of staghorn are unique and cannot be filled by other reef-building corals (Bruckner 2002).

Life History
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 is currently observed at 5 to 15 m. In southeastern Florida, this species historically occurred on the outer reef platform (16 to 20 m) (Goldberg 1973), on spur-and-groove bank reefs and transitional reefs (Jaap 1984, Wheaton and Jaap 1988), and on octocoral-dominated hardbottom (Davis 1982). Colonies have been common in back- and patch-reef habitats (Gilmore and Hall 1976, Cairns 1982). Although staghorn coral colonies are sometimes found interspersed among colonies of elkhorn coral, they are generally in deeper water or seaward of the elkhorn zone and, hence, more protected from waves. Historically, staghorn coral was also the primary constructor of mid-depth (10 to 15 m) reef terraces in the western Caribbean, including Jamaica, the Cayman Islands, Belize, and some reefs along the eastern Yucatán peninsula (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. Thus, 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 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).

Staghorn coral, like many stony coral species, propagate sexually and asexually through fragmentation. Staghorn coral reproduce sexually by broadcast spawning, meaning that coral larvae develop externally to the parental colonies (Szmant 1986), and it is also simultaneous hermaphroditic, meaning that a given colony will contain both female and male reproductive parts during the spawning season. Despite being simultaneous hermaphrodites, staghorn corals 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 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 staghorn corals spawn within a week of the full moon in July and/or August (Lirman 2002). Annual egg production in staghorn coral populations studied in Puerto Rico was estimated to be 600 to 800 eggs per cm^2 of living coral tissue (Szmant 1986).

In staghorn corals, 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 have higher fertility rates (Soong and Lang 1992). Only colonies of staghorn coral with a branch length greater than 9 cm were fertile and over 80 percent 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).

The growth rate for staghorn coral has been reported to range from 3 to 11.5 cm/yr. This growth rate is relatively fast compared to other corals and historically enabled the species to construct significant reefs in several locations throughout the wider Caribbean (Adey 1978). Growth in staghorn coral 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 staghorn coral relative to other corals and other benthic organisms (Shinn 1976, Neigel and Advise 1983, Jaap et al. 1989).

Few data on the genetic population structure of staghorn coral exist; however, due to recent advances in technology, the genetic population structure of the current, depleted population is beginning to be characterized. Vollmer and Palumbi (2007) examined multilocus sequence data from 276 colonies of staghorn coral spread across 22 populations from 9 regions in the Caribbean, Florida, and the Bahamas. Their data were consistent with the Western-Eastern Caribbean subdivision observed in elkhorn coral populations by Baums et al. (2005).

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 (Bruckner 2002). 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).

Staghorn coral 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.

Figure 3.3.1 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 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.

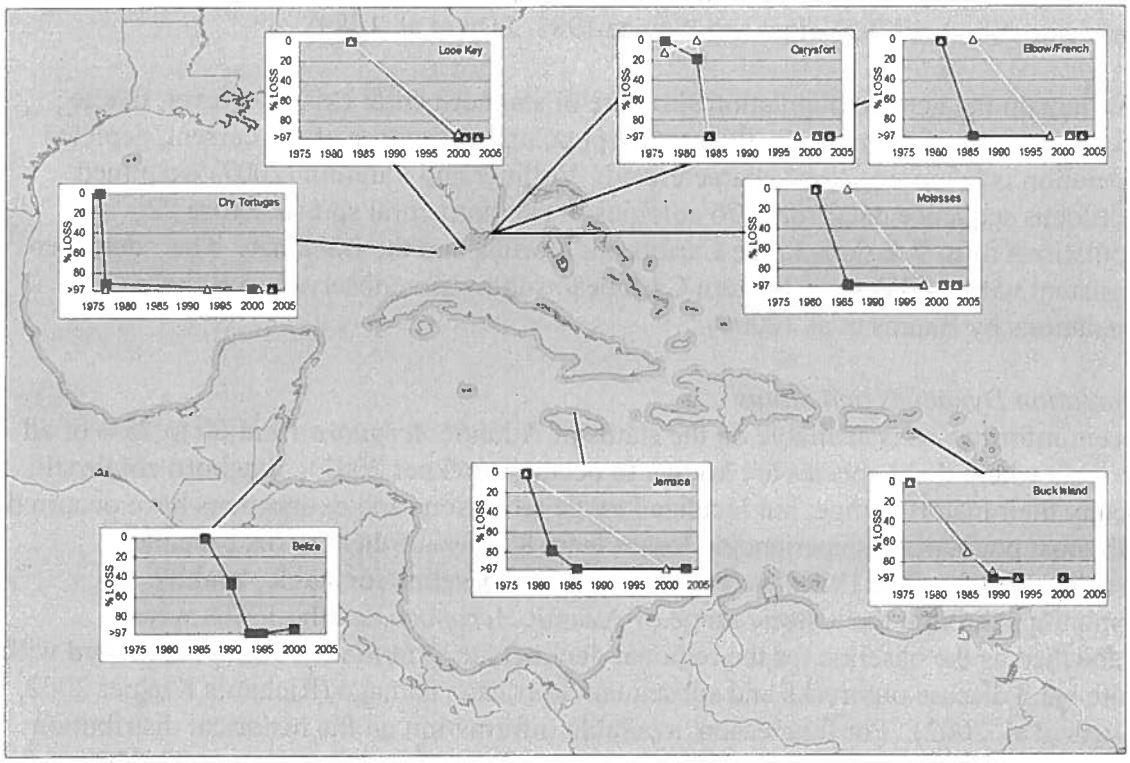

Figure 3.3.1 Percent Loss of Staghorn and Elkhorn Coral Throughout the Caribbean

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). 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). 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.

NOAA's Center for Coastal Ocean Science (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. 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.

For sites sampled in Puerto Rico by NCCOS, '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) phenomena. 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 Department of Planning and Natural Resources (DPNR) and the University of the Virgin Islands (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 30 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. Because sampling is dependent upon funding, weather, staffing, etc., not all 30 fixed stations are sampled annually and not all six transects could be conducted at each sampled station. 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. Specific benthic habitat types identified included 'Coral,' 'Critical Habitat,' and 'Macroalgae.'[6] For the habitat types 'coral' and 'macroalgae' only 35 transects could be used to evaluate trends over a 10-year period while 52 transects could be used to evaluate 9-yr trends and trends since the 2005 bleaching event. For 'critical habitat' 6 transects could be used to evaluate trends over a 10-year period, 9 transects could be used to evaluate 9-yr trends, and 52 transects could be used to evaluate trends since the 2005 bleaching event. Analyses performed on the data collected are summarized below.

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[6] "Critical Habitat' was a category created by combining the aggregated coverage categories of "dead coral with sparse turf algae" and "consolidated substrate covered with crustose coralline algae" categories reported in the USVI DPNR datasets. These data categories most closely matched the designated essential feature for *Acropora*.

- The majority of transects within sites showed no significant change in 'Coral' coverage across both 10-year (26 of 35 transects) and 9-year (37 of 52 transects) time series; however, for those sites where significant changes were detected at the 10-year (9 of 35 transects) and 9-year (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-year (5 of 6 transects) and 9-year analysis (6 of 9 sites).

- The majority of transects within sites showed significant change in 'Macroalgae' coverage for the 10-year time series (18 of 35 transects were increasing), while the majority of transects within sites showed no significant change for the 9-year time series (33 of 52 transects); however, for transects where significant changes were detected for the 9-year 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 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 (C. Rogers, unpublished data).

During the 2005 bleaching event, near Culebra Island, almost 100% of staghorn colonies suffered partial to complete mortality due to bleaching (García-Sais et al. 2008a). 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.

On-going monitoring at mid-shelf reef (MSR) habitats study sites inside and adjacent to Virgin Island Coral Reef National Monument (VICRNM) indicate that from 2003-2008, live scleractinian coral and rugosity at MSR sites were significantly greater outside the VICRNM, while gorgonian cover was greater inside (Monaco et al. 2009). 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.

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 VICRNM-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.

An additional concern about species with reduced abundance is that they are at a greater risk of extinction due to stochastic environmental and demographic factors (e.g., episodic recruitment factors). 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[7] 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). These factors are expected to further reduce successful larval recruitment to levels that will not be able to compensate for observed rates of ongoing mortality (i.e., mortality will likely outpace 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 staghorn corals 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 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).

Threats

Staghorn corals face myriad stressors that in some cases act synergistically (i.e., the total effect is greater than the sum of the individual effects). Diseases, temperature-induced bleaching, and physical damage from hurricanes are deemed to be the greatest threats to 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

[7] 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.

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 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 staghorn coral's decline by enhancing coral condition and decreasing synergistic stress effects. Table 3.3.1 summarizes the factors affecting the status of staghorn corals and the identified sources of those stressors.

Table 3.3.1 Stressors Affecting 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 causing trophic cascade natural trophic reef interactions
Nutrients Source: point-source non-point-source	**Contaminants** Source: point-source non-point-source
Competition Source: fishing causing trophic cascade	**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 *Serratia 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. While no such link has been definitely made for staghorn corals, this disease vector may be a concern for staghorn corals as well. 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 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).

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 areas are designated as anchorages 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 fish population structure 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 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. Staghorn corals are sessile, light-dependent animals that cannot move away from stressful situations (Marshall and Orr 1931, Cortes and Risk 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 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.

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, 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 staghorn coral (among others). 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 (e.g., nitrogen and phosphorus) 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 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, 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 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 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 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 staghorn coral, but the extent of such interactions is not well documented (*Acropora* BRT 2005).

3.3.2 *Acropora* 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.A and B).

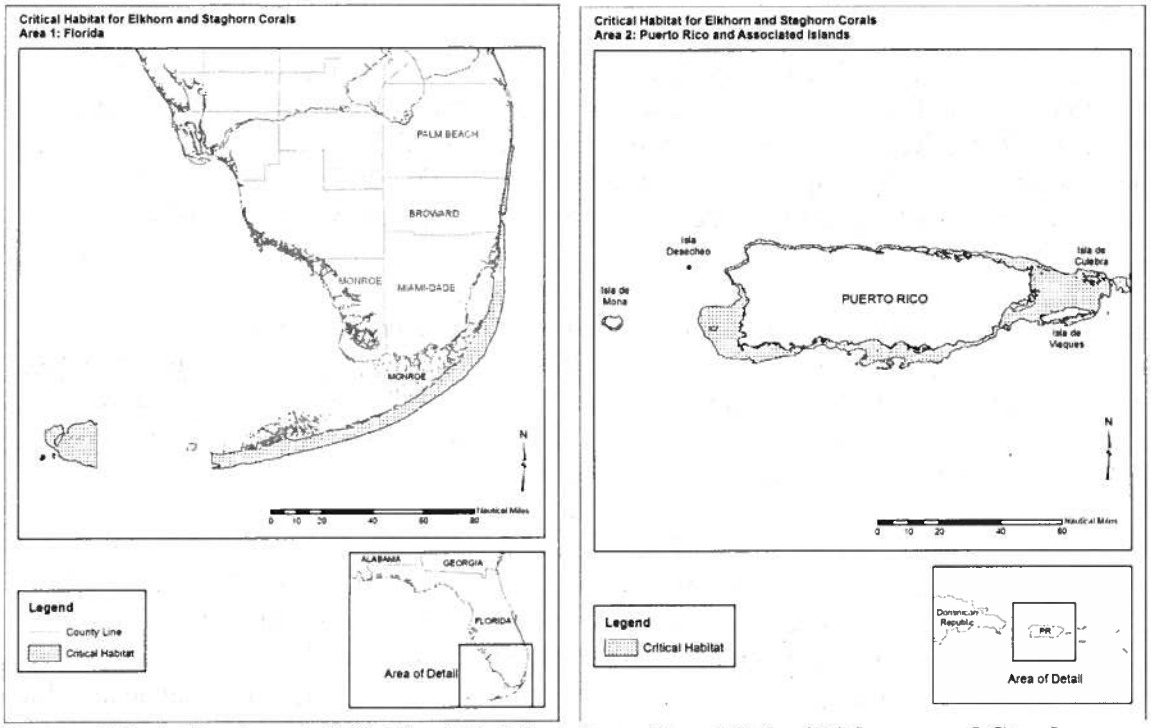

Figure 3.3.A. Designated Critical Habitat Area 1 and 2 for Elkhorn and Staghorn Corals

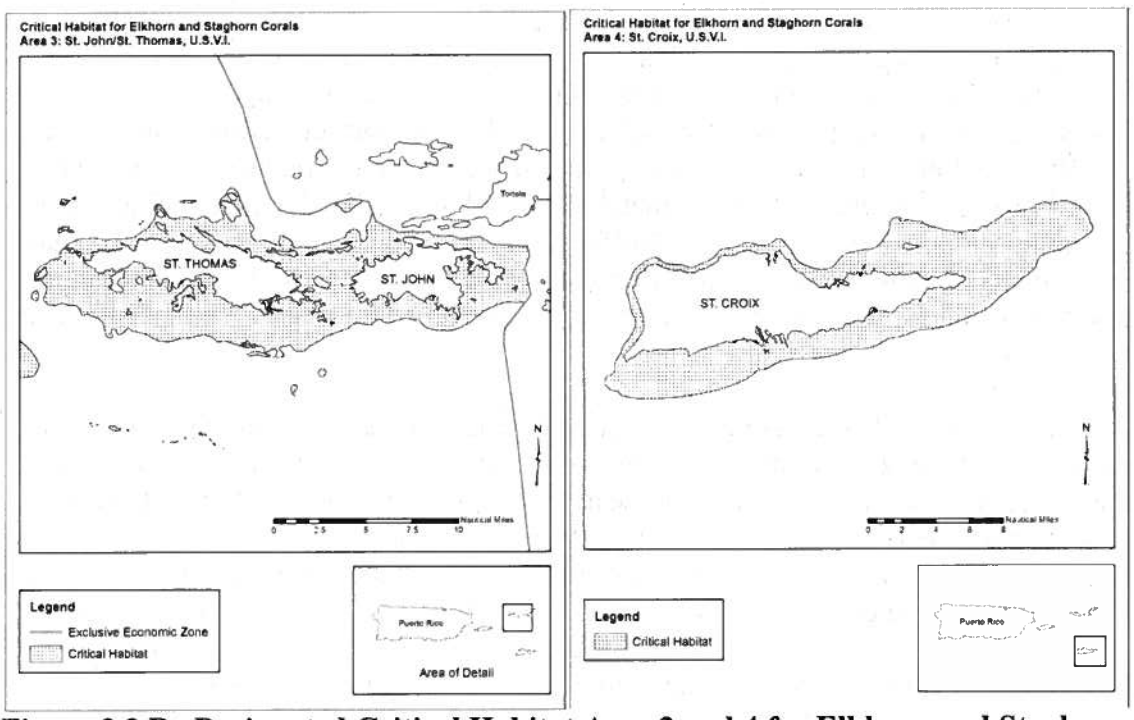

Figure 3.3.B. Designated Critical Habitat Area 3 and 4 for Elkhorn and Staghorn Corals

Acropora 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).

Shifts 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 certain species of living corals and crustose coralline algae. No information on macroalgal overgrowth of living staghorn corals is available. 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 indicates 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 1991, 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, 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: 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 in Syria annually (NMFS and USFWS 2007a). The 5-year status review noted that no nesting trends could be estimated for the Mediterranean Sea (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 Green Sea Turtles

Green sea turtles occur in the eastern, central, and western Pacific, with foraging areas 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, but the only major population (>2,000 nesting females) of green turtles in the western Pacific occurs in Australia and Malaysia. Green sea 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 sea 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).

Green sea 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 anthropogenic threats. 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. As a result, 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, biological 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 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). 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 Sea Turtle Stranding and Salvage Network (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, 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. 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).

Information obtained from satellite telemetry of post-nesting females shows variation in the distance traveled between nesting and foraging grounds (Plotkin 2003). Some turtles in the Caribbean made short range migrations (25-200 km) (Hillis-Starr et al. 2000, Horrocks et al. 2001, Lagueux et al. 2003), while other made much longer migrations (200 km or more) (Byles and Swimmer 1994, Horrocks et al. 2001, Miller et al. 1998 in Plotkin 2003, Prieto et al. 2001 in Plotkin 2003, Lagueux et al. 2003, NMFS and USFWS 2007c).

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).

Nesting occurs in at least 70 countries, although much of it now occurs only at low densities. The likely primary nesting rookeries (i.e., sites with greater than 100 nesting females per year) in the Atlantic Basin are: Antigua, Barbuda, Bahamas, Barbados, Brazil, Cuba (Doce Leguas Cays), Dominican Republic, Jamaica, Mexico (Yucatan Peninsula), Panama, Puerto Rico (Mona Island), Trinidad and Tobago, and U.S. Virgin Islands (NMFS and USFWS 2007c).

Hawksbill breeding sites have been largely affected by historical patterns of human exploitation and in general, the most significant rookeries left today are at sites that have not been permanently inhabited by humans or have not been heavily exploited until recently (Groombridge and Luxmoore 1989). More common than total extirpation, however, is for hawksbill populations to be reduced to extremely low levels (i.e., less than 10 nesting females per year). Known examples of such near extirpations include Bonaire, Costa Rica (Tortuguero National Park), Equatorial Guinea (Bioko), and Honduras (Bay Islands) (NMFS and USFWS 2007c).

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.

As noted previously, we do not believe the proposed action will affect elkhorn coral. The following sections that discuss listed corals are meant to describe the baseline conditions for staghorn corals in the action area. However, because elkhorn and staghorn corals often co-occur, have very similar life history characteristics and biology, and face the many of the same environmental threats, it is difficult to parse out general effects that occur to one species but not the other. Therefore, in the subsequent section we refer staghorn corals when we can, but in some instances end up discussing impacts to "*Acropora*" species more generally.

4.1 Status of Staghorn Coral and Critical Habitat within the Action Area

The action area comprises three of the four geographic areas located off the United States where staghorn corals occur and where critical habitat is designated. However, the majority of the area occurs within commonwealth and territorial waters and no information is currently available that is specific to colonies and designated critical habitat occurring the EEZ (i.e., the action area). The information on the Status of the Species (see Section 3.3) uses the best available information to describe the status of staghorn colonies and designated critical habitat in the areas around the U.S. Caribbean. Those data are primarily constrained to commonwealth and territorial waters. While those data describe the condition staghorn corals in commonwealth and territorial waters, it is also applicable to staghorn corals and areas of critical habitat occurring in the action area. None of the threats to staghorn corals are specific to the EEZ and the factors affecting the current status of the species and critical habitat (i.e., *Diadema* die off, reduced abundance and reproduction, etc.) are not specific to colonies or critical habitat occurring in commonwealth and territorial waters. Therefore, the status of staghorn corals and designated critical habitat described in the Section 3.3 most accurately reflects the species status within the action area.

4.2 Factors Affecting Staghorn Coral and Critical Habitat within the Action Area

Numerous activities funded, authorized, or carried out by federal agencies have been identified as threats and may affect staghorn corals in the action area. Although many regulations exist to protect corals, including 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 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 implements 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 USVI Department of Planning and Natural Resources (DPNR) has the authority to issue 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 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. Army Corps of Engineers (COE) and the Environmental Protection Agency (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 COE authorizes and carries out construction and dredge-and-fill activities that may result in direct mortality or injury of 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 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 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 Fishery

NMFS completed an ESA Section 7 consultation on the Caribbean reef fish fishery, on October 4, 2011. The reef fish fishery in waters around Puerto Rico and the USVI uses pots and traps, hook and line, longline, and spearguns. The fishery targets snapper and groupers, as well as herbivorous fish (i.e., parrotfish and surgeonfish). Herbivorous fish play a role in mediating the growth and spread of macroalgae. Because of the ecological relationship between macroalgae and *Acropora* and their designated critical habitat, the biological opinion not only evaluated the direct effects of the fishery on ESA-listed species (e.g., deployment of fishing gear, vessel operations, etc.), but also indirect adverse affects caused by the removal of herbivorous fish. The opinion determined that direct effects of the fishery (i.e., fish trap deployment and vessel anchoring) and indirect effects caused by the harvest of herbivorous fish, would adversely affect *Acropora* and their designated critical habitat. However, these adverse affects were not jeopardizing elkhorn or staghorn corals or destroying or adversely modifying their designated critical habitat.

Spiny Lobster Fishery
Section 2.1.2 provides an overview of the history of the federal Caribbean spiny lobster fishery and its management by NMFS under the SLFMP. Amendments in the past have implemented restrictions on retained egg-bearing females, size limitations altered gear construction and usage, implemented import regulations size limits, and placed prohibitions on the use of some fishing practices among other things. The current federal spiny lobster 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 spiny lobster fishing in territorial and commonwealth waters are part of the environmental baseline.

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 National Parks Service (NPS). Through the Section 7 process, where applicable, NMFS will continue to establish conservation measures for agency vessel operations to avoid or minimize adverse effects to listed species. Currently, they present the potential for some level of interaction.

4.2.3 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.

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 marine event participants must follow to avoid and minimize potential impacts 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.

4.2.4 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, 18 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 terragenous 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,b) 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 staghorn coral. The COE also permits dredge-and-fill activities that can directly affect staghorn corals via fragmentation/breakage or abrasion. They can also affect the species by physically altering or removing benthic habitat suitable for staghorn coral colonization. Dredge-and-fill activities may also cause increases in sedimentation that may cause shading, deposition of sediment on 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 staghorn corals. NMFS is currently reinitiating ESA consultation on dredging and beach renourishment activities of the U.S. Army Corps of Engineers, South Atlantic Divison, which encompasses the region from Key West, Florida, to the North Carolina-Virginia border. The new biological opinion, expected to be completed within the next year, is being expanded to also include a dredging and beach renourishment activities in the U.S. Caribbean Region (i.e., Puerto Rico and the USVI).

4.2.5 Natural Disturbance

Hurricanes and large coastal storms can also significantly harm 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).

4.2.6 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 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.

4.2.7 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 ongoing through the Southeast Regional Office. NOAA Restoration Center has also established a contract position/employee in Puerto Rico to participate in vessel grounding response and carry out restoration activities.

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

Federal Actions to Reduce Threats to 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.

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 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. NMFS also conducts essential fish habitat (EFH) consultations. Through EFH consultations, NMFS works with federal agencies to conserve and enhance EFH, which includes corals.

The Coral Reef Conservation Act of 2000 (CRCA) authorized appropriations to NOAA for coral reef protection and management through 2004 (the Act is currently up for reauthorization). The CRCA also authorized the establishment of the Coral Reef Conservation Fund. Through the Fund, NOAA works with the non-profit National Fish and Wildlife Foundation to build public-private partnerships to reduce and prevent degradation of coral reefs. The CRCA also established the Coral Reef Conservation Program (CRCP). Through the CRCP, NOAA conducts activities such as mapping, monitoring, assessment, research, and restoration that benefit coral reef ecosystems; enhancing public awareness of such ecosystems; assisting states to remove abandoned vessels and marine debris from reefs; and conducting cooperative management of coral reef ecosystems. The CRCA also authorizes CRCP to provide matching grants for coral reef conservation projects to states, territories, educational and non-governmental institutions, and fishery management councils (NOAA Coral Reef Conservation Program 2011).

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 Buck Island Reef National Monument (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 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. 2004). 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.

Commonwealth/Territorial Actions to Reduce Threats to Corals
The Commonwealth of Puerto Rico has several laws and proposed regulations that may aid in the conservation of corals. The most pertinent statute is the 2000 Law for the Protection, Conservation, and Management of Coral Reefs in Puerto Rico (Law 147). This law explicitly mandates the conservation and management of coral reefs in order to protect their functions and values, and provides for the creation of zoned areas in order to mitigate impacts from human activities. These zones will facilitate the DNER in controlling human activity, such as anchoring, that can directly impact *Acropora* spp. Law 147 also directs the DNER to identify and mitigate threats to coral reefs from degraded water quality due to pollution and additionally directs the DNER to designate priority areas as marine reserves, including a minimum of 3% of the insular platform within three years (2003). Marine reserves are defined as areas where all extractive activities are prohibited in order to help recover depleted fishery resources and protect biodiversity, and can protect *Acropora* by preventing impacts from fishery gear. There are currently an additional 13 natural reserves in Puerto Rico that have coral reefs within their boundaries, all of which are located on all coasts and offshore islands. This spatial distribution of protected areas provides an infrastructure for management measures to protect *Acropora* spp. populations.

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.

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 Staghorn Corals

In summary, several factors are presently adversely affecting 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 and are likely to be adversely affected 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 Listed Species and Critical Habitat).

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. In 1998, NMFS designated the waters up to three nautical miles around Culebra Island and its outlying cays as critical habitat for green sea turtles.

Leatherback sea turtles occur within the action area primarily during their nesting season. The Sandy Point National Wildlife Refuge in St. Croix, USVI, supports a large nesting population of leatherback sea turtles, and critical habitat for the species has been designated at Sandy Point. 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 threats that may affect 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 U.S. Caribbean 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 Endangered Species of Wild Fauna and Flora (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 "Status 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 USD 0.50-5.00/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 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 also occur in 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 4.4.1. 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 permits 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 were reported as captured with pelagic longline gear (PLL) (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 tuna 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 charter/headboat permits. In 2007, 27.7 mt of swordfish 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 4.4.1 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

NMFS completed and ESA Section 7 consultation on the Caribbean reef fish fishery, on October 4, 2011. The reef fish fishery in waters around Puerto Rico and the USVI uses pots and traps, hook and line, longline, and spearguns. The fishery targets snapper and groupers, as well as herbivorous fish (i.e., parrotfish and surgeonfish). The opinion concluded that the fishery was likely to adversely affect green, hawksbill, and leatherback

sea turtles via vessel strikes and entanglements in fishing gear, but would not jeopardize their continued existence. An ITS was issued authorizing incidental take.

Spiny Lobster Fishery
Section 2.1.2 provides an overview of the history of the federal Caribbean spiny lobster fishery and its management by NMFS under the SLFMP. Section 1 reviews the previous consultations on the federal fishery's effects on listed sea turtles. The current federal spiny lobster 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 current effects of spiny lobster fishing in territorial and commonwealth waters on sea turtles are part of the environmental baseline.

4.4.3 Vessel Traffic

Commercial and recreational vessels can adversely affect sea turtles through propeller and vessel strikes. Many records of vessel interactions have been documented within the action 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 and are susceptible to 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 polychlorinated biphenyls (PCBs) and stormwater runoff from coastal towns and cities into rivers and canals emptying into bays and the ocean. 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 1960s, 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., two spills of approximately 2,500 tons and one 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 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 and the Navy has since returned the land to the Commonwealth, 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 August 29, 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 consultation on dredging and beach renourishment activities of the U.S. Army Corps of Engineers, South Atlantic Division for East Coast activities from Florida through North Carolina. The new biological opinion, 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 Benefiting 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 NGOs and other agencies (Sea Grant-UPR, Rio Piedras-UPR, Mayagüez-UPR, Chelonia, WIDECAST, USFWS). 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 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 also 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 may adversely affect them to varying degrees. Past and present threats in the U.S. Caribbean 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 spiny lobster fishery on staghorn and elkhorn coral, *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 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. 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.

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 spiny lobster fishery is reviewed in Section 3. Staghorn and elkhorn coral 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. The sea turtle species occurring in the U.S. Caribbean 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 *Acropora* and sea turtle species biology and the effects of the proposed action. Data pertaining to the Caribbean spiny lobster fishery, relative to interactions

with staghorn coral 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 spiny lobster 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.

When analyzing any proposed action, it is important to consider not only its immediate effects to ESA-listed species, but also the effects caused by or resulting from it that are reasonably certain to occur later in time. For example, effects from the proposed action occurring later in time could include habitat degradation, reduction of prey/foraging base, etc. No such effects to sea turtles or staghorn coral are anticipated because of the operation of the Caribbean spiny lobster fishery (i.e., hand harvest via SCUBA or skin diving, vessel operations, gear deployment and retrieval). Our analysis assumes sea turtles and staghorn coral are not likely to be adversely affected by a gear type unless they interact with it. We also assume the potential effects of each gear type are proportional to the number of interactions between the gear and each species.

Effects to Elkhorn Coral

As noted previously in Section 3.1, we believe the proposed action is not likely to adversely affect elkhorn coral. However, we did not arrive at that conclusion until we completed our effects analysis (see subsequent sections). The effects analysis for elkhorn coral is included below to document how we came to our ultimate conclusion that the proposed action was not likely to adversely affect this species.

Basic Approach to the Assessment

The proposed action has been determined to have two primary routes of effects on listed species: vessel and gear impacts on *Acropora* corals and sea turtles. 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 5.1-5.4 we analyze effects on *Acropora* coral and *Acropora* critical habitat. In Section 5.1 we present our rationale for determining that *Acropora* coral and its designated critical habitat will not be affected by the federal spiny lobster fishery in the EEZ off St. Thomas/ St. John. In Sections 5.2 and 5.3 we evaluate the potential adverse affects to *Acropora* coral and its designated critical habitat from the authorized

fishing gears/techniques used in the federal spiny lobster fishery of the U.S. Caribbean. In Section 5.4, we examine the potential impacts of spiny lobster vessels and vessel anchoring on *Acropora* coral and critical habitat. Section 5.5 provides a summary of the anticipated effects of the action on *Acropora* and critical habitat in St. Croix and Puerto Rico. Section 5.6 presents our analysis of the proposed action's effects on sea turtles.

5.1 Effects on Spiny Lobster Fishing on *Acropora* Coral and Designated Critical Habitat in St. Thomas/St. John

The territorial water boundary and bathymetry around St. Thomas and St. John are such that *Acropora* critical habitat does not occur in the EEZ off these islands. Because *Acropora* coral is a sessile species and only settles/re-establishes on habitat types currently designated as critical habitat, we only anticipate finding it in areas designated as critical habitat. We do not anticipate finding *Acropora* coral everywhere within a designated critical habitat unit, but we would not anticipate finding colonies outside area designated as critical habitat. Because designated critical habitat does not occur in the EEZ off St. Thomas/St. John, we do not anticipate that commercial or recreational harvest of spiny lobster from the EEZ off these islands will affect *Acropora* coral or its critical habitat. The remainder of our effects analysis therefore focuses on the potential impacts to *Acropora* coral and their designated critical habitat occurring in the EEZs off Puerto Rico and St. Croix.

5.2 Effects of Hand Harvest on *Acropora* Coral and *Acropora* Critical Habitat

Commercial and recreational divers (either free diving or SCUBA-assisted) fishing for spiny lobsters in both the USVI and Puerto Rico primarily use snares, their hands, or small dip nets to harvest lobster. Snares commonly consist of a long, thin pole that has a loop of coated wire on the end. The loop is placed around a lobster that may be residing in a tight overhang or other inaccessible location, and then tightened by a pull toggle at the base of the pole to capture and extract the lobster (Figure 5.2.1) (Barnette 2001). There is little difference in the techniques and gears used by recreational and commercial divers targeting spiny lobsters. Divers may affect *Acropora* coral by causing fragmentation or abrasion. However, the information available indicates that commercial and recreational dive fisheries for spiny lobster are unlikely to occur in the EEZ. Therefore, they are not likely adversely affect *Acropora* coral or critical habitat. The following sections present our rationale for those determinations

Figure 5.2.1 Example of a Spiny Lobster Snare
(Barnette 2001)

5.2.1 Puerto Rico –Hand Harvest

Commercial Sector

Data specific to the federal commercial spiny lobster dive fishery is not available. Information on the spiny lobster landings from the dive sector does exist, but it cannot be separated into federal and commonwealth effort. Because of this lack of specific information on the federal fishery we used the best available information on the percentage of commercial fishermen that target spiny lobster, the number of trips taken annually, and percentage of all fishing trips that are dive trips (Matos-Caraballo 2007, Matos-Caraballo and Agar 2011) to estimate the potential impacts from the commercial spiny lobster dive sector. Little information is available on the locations or densities of *Acropora* coral in the EEZ off Puerto Rico. However, the vast majority of the designated *Acropora* critical habitat in Puerto Rico occurs in the commonwealth waters. Since the critical habitat that occurs in the EEZ is off the west coast of Puerto Rico, our effects analysis focuses on this region.

The best source of fishing trip information in Puerto Rico comes from Matos-Caraballo (2008), which reports that from 2004-2006, between 13,461 and 14,977 total fishing trips were taken off the west coast of Puerto Rico, but those trips were not differentiated by target species. The 2008 census of Puerto Rican fishermen, conducted by Matos-Caraballo and Agar (2011), did not include fishing trip information, but it did report that 47% of respondents on the west coast of Puerto Rico indicated they were targeting spiny lobster. Together, this information indicates that up to 7,039 fishing trips off the west coast of Puerto Rico may have targeted spiny lobster. There is also no specific information on the number of spiny lobster trips that were dive trips. However, landings data indicate that from 1999-2009 diving accounted for 48.3% of all spiny lobster landings in Puerto Rico (CFMC and NMFS 2011). Given the data available, we believe using that the proportion of landings by gear as a proxy for the amount of total effort conducted with each gear type is appropriate. Assuming that proportion of landings by gear can be used as a proxy for the amount of total effort conducted with each gear type, we estimate 48.3% of the total spiny lobster fishing effort is derived from diving. Thus, of the estimated total spiny lobster trips off the west coast of Puerto Rico, we estimate 3,400 of those trips were likely dive trips.[8] Unfortunately, no information exists on the number of trips conducted in the EEZ, or the percentage of all trips that occur in the EEZ. Approximately 325 mi^2 of fishable habitat exists off west coast of Puerto Rico (NMFS unpublished data); approximately 40 mi^2 of that fishable habitat (12.3%) is *Acropora* critical habitat that occurs in the EEZ. Since there is a greater amount of fishable habitat in commonwealth waters than the EEZ, we believe it is likely that most, if not all, of the commercial spiny lobster dive trips occurring off the west coast of Puerto Rico take place in commonwealth waters. Assuming that the area of fishable habitat can be used as a proxy to estimate where fishing trips occur, we anticipate 12.3% of all spiny lobster dive trips off the west coast of Puerto Rico occur in areas designated as *Acropora* critical

[8] Maximum number of trips off the west coast of Puerto Rico (14,977) x 47% of fishermen target spiny lobster off the west coast of Puerto Rico = 7,039 west coast spiny lobster trips. Number of spiny lobster trips off the west coast of Puerto Rico (7,039) x Percentage of all spiny lobster landings coming from dive gear (48.3%) = 3,399.8 spiny lobster dive trips off the west coast of Puerto Rico

habitat. Thus, we estimate 418 spiny lobster dive trips occur annually (approximately 1 per day) in *Acropora* designated critical habitat.

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).

Since SCUBA-assisted fishing is highly selective, and *Acropora* coral is not a target species; we believe it is likely that divers would be able to avoid directly taking these species. Additionally, we believe that only one commercial spiny lobster dive trips occur in the EEZ off Puerto Rico daily on average. This estimate is far less than the number documented to cause coral injuries. Taken together, we believe that few spiny lobster dive trips are occurring in the Puerto Rican EEZ and the fishermen conducting those trips are likely to avoid *Acropora* coral. Thus, we believe adverse affects to *Acropora* coral and their designated critical habitat from spiny lobster divers in the Puerto Rican EEZ are extremely unlikely to occur and discountable.

Recreational Sector

Data on recreational spiny lobster fishing is not collected in Puerto Rico (CFMC and NMFS 2011). *Acropora* coral colonies occur only rarely and in discrete locations within the U.S. Caribbean. Little information is available on the locations or densities of *Acropora* coral in the EEZ off Puerto Rico. However, we do know that only a very small portion of the *Acropora* critical habitat (the only area we would anticipate *Acropora* coral colonies to be located) designated in Puerto Rico actually occurs in the EEZ.[9]

Divers are known to adversely affect *Acropora* coral (*Acropora* BRT 2005). However, we believe any impacts from recreational divers on *Acropora* coral and designated critical habitat in the EEZ off Puerto Rico are discountable. Divers are unlikely to cause consolidated hardbottom to become unconsolidated and they do not cause growth of macroalgae or cause sedimentation. Interactions between divers and dead coral skeletons could potentially cause breakage, adversely affecting the essential feature. However, for the reasons outlined below we believe the likelihood of interactions are so small that they are discountable.

Garcia-Moliner et al (2001) conducted a census of SCUBA schools, dive centers and shops, and dive operations in the U.S. Caribbean, which does provide some information on SCUBA diving activities in Puerto Rico including some information on recreational diving trips for lobster. Based on the information provided by Garcia-Moliner et al. (2001) and other factors, we believe the disincentives for recreational spiny lobster fishers to travel to federal waters to target these species are so great that the potential adverse affects from the fishery are extremely unlikely to occur and discountable.

[9] Approximately 40 mi^2 of 1,383 mi^2 designated in Puerto Rico occurs in the EEZ.

Only a relatively small area of fishable habitat occurs in the EEZ of Puerto Rico and it is located off the southwestern corner of the island, nine nautical miles offshore from the closest on land. The depth of the that fishable habitat ranges from approximately 80-600 ft. Garcia-Moliner et al. (2001) report that in Puerto Rico recreational diving activities generally take place between 32-108 ft, with an average depth of approximately 65 ft. These data indicate that the little fishable habitat that does occur in the EEZ is deeper than the average recreational dive depth. Additionally, Garcia-Moliner et al (2001) report that almost two-thirds of dive operators prohibit the harvest of spiny lobster fishing during their recreational dive trips.

Garcia-Moliner et al. (2001) acknowledge that the survey does not include local divers, with their own boats and equipment. However, we believe the likely size of recreational vessels provides further disincentive for traveling far offshore to reach fishable habitat in the EEZ. The size of recreational vessels is not available but data on the commercial fishery indicates that approximately 96% of vessels are 30 ft in length or smaller (Matos-Caraballo and Agar 2011).[10]

These are relatively small vessels and traveling at least nine miles offshore to recreationally target spiny lobster seems unlikely. Additionally, because the sale of recreational caught fish (including spiny lobster) is prohibited (Puerto Rico Fishing Regulation 7949; November 24, 2010), recreational fishers cannot help offset the costs of their recreational trips by selling their catch. We believe the depth of the fishable habitat in the EEZ, its distance from shore, and the relatively few number of dive operations that allow fishing, and the prohibition on harvest, all act as disincentives for recreational fishers to dive for spiny lobster in the EEZ off Puerto Rico. More specifically, we believe it is unlikely that recreational divers bypass hundreds of square miles of fishable habitat, located closer to shore, in shallower water, where the costs of a trip are lower, and weather and overall safety conditions are more favorable, to travel far offshore to target spiny lobster. Therefore, we believe it is extremely unlikely that significant amounts of recreational diving (i.e., hand harvest) for spiny lobster is occurring in the EEZ and therefore is not likely to adversely affect *Acropora* coral or designated critical habitat.

5.2.2 St. Croix – Commercial/Recreational Harvest

In St. Croix, collection of spiny lobster by hand is authorized in the EEZ of St. Croix where *Acropora* critical habitat is designated. Divers are known to adversely affect *Acropora* coral (*Acropora* BRT 2005). However, we believe any impacts from commercial and recreational divers on *Acropora* coral and critical habitat in the EEZ off St. Croix are discountable.

Divers are unlikely to cause consolidated hardbottom to become unconsolidated, nor do they cause growth of macroalgae or cause sedimentation. Dead coral skeletons free of fleshy macroalgae are another component of the essential feature for *Acropora*

[10] 0.5% of vessels were less than 10 ft long; 47.6% of vessels were 10-19.9 ft long, and 49.7% of vessels were 20-29.9 ft long (Matos-Caraballo and Agar 2011).

designated critical habitat. Interactions between divers and dead coral skeletons could potentially cause breakage, adversely affecting the essential feature. However, for the reasons outlined below we believe the likelihood of interactions are so small that they are discountable.

Commercial Sector
Kojis and Quinn (2011) report that skin and SCUBA diving is an important gear type/fishing technique in St. Croix, but no fishers reported commercially diving in the EEZ. Fishermen off St. Croix appear to prefer to fish in territorial waters because of difficult sea conditions on Lang Bank[11] and because of the price of gas, many fishermen prefer to remain closer to shore (B. Kojis to A. Herndon, NMFS, pers. comm. 2011). Since no commercial diving for spiny lobster appears to be occurring in the EEZ, we do not anticipate it will cause any adverse affects to *Acropora* coral or designated critical habitat.

Recreational Sector
There is currently no information available on the recreational dive harvest of spiny lobsters in the St. Croix EEZ. Most of the information on the recreational sector for all of the USVI fisheries derives from offshore billfish and other pelagic fisheries. 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). None of the reports on the recreational sector in all of the USVI (including St. Croix) target the fleet harvesting reef fish, lobster, or conch (CFMC and NMFS 2011). There is currently no information available that indicates a recreational dive fishery for spiny lobster in St. Croix even exists. The only rationale supporting the fishery's potential existence is simply that recreational harvest of spiny lobster by hand is authorized in the EEZ.

Since 2010, all recreational anglers fishing in the U.S. Caribbean EEZ are required to register with the National Angler Registry. As of March 2011, 37 recreational anglers had registered with National Angler Registry throughout the entire USVI (F. Darby, NMFS, pers. comm. in CFMC and NMFS 2011). In contrast, 97 anglers in the USVI held HMS permits (each permit costs approximately $15), which allowed them to harvest billfish and other large pelagics. This information supports what has been reported in previous recreational fishing surveys, which indicate that fishing for pelagic species in the USVI is by far the most popular recreational fishing pursuit.

Recreational angler registration supports the conclusion that billfish and pelagics are the preferred target species of recreational anglers in the USVI (including St. Croix). We believe there are several reasons that recreational diving for spiny lobster in the EEZ off St. Croix is unlikely to happen. There is a very narrow shelf in St. Croix that greatly reduces the amount of fishable habitat occurring in the EEZ, restricting it primarily to an area northeast of the island called Lang Bank. In general, fishers prefer to not fish Lang Bank because of difficult sea conditions (B. Kojis to A. Herndon, NMFS, pers. comm. 2011). Fishing closer to shore requires less time and fuel to access the fishing grounds,

[11] Lang Bank is where all fishable waters in the EEZ off St. Croix occur.

as well as fewer supplies. Since there is fishable habitat closer to shore than the EEZ, we believe it is extremely unlikely a recreational diver would travel much further offshore to target spiny lobster. Additionally, because the sale of recreational caught fish (including spiny lobster) is prohibited, recreational fishers cannot help offset the costs of their recreational trips by selling their catch. Taken together, the economic disincentives (i.e., costs of travel to the EEZ, not being able to sell the catch, etc.), coupled with relatively difficult sea conditions in the fishable areas of the St. Croix EEZ, and an apparent preference by recreational fishers to target billfish and pelagics, leads us to conclude that a recreational dive fishery for spiny lobster is unlikely to exist off St. Croix. Thus, we anticipate recreational hand harvest is not likely to adversely affect *Acropora* coral or designated critical habitat.

5.3 Effects of Trapping on *Acropora* Coral and *Acropora* Critical Habitat

Traps and/or trap lines can adversely affect *Acropora* coral via fragmentation or abrasion. The deployment of spiny lobster traps may adversely affect *Acropora* coral as traps drop toward the sea floor or when traps are retrieved and pulled to the surface. Abrasion may occur when traps or trap lines contact *Acropora* coral during fishing activities. Therefore, we believe trap fishing may adversely affect *Acropora* coral and their designated critical habitat. The following discussion summarizes the best available information on how *Acropora* may be impacted by these interactions with lobster trap fishing gear.

Fragmentation
Severe fragmentation can adversely affect sexual reproduction by reducing colonial biomass and/or causing a reallocation of energy away from reproduction toward stabilization, lesion repair, and growth (Van Veghel and Bak 1994, Van Veghel and Hoetjes 1995, Hall and Hughes 1996, Lirman 2000). Colony size in cnidarians[12] is directly correlated to survivorship, growth, and reproduction (i.e., the larger the colony, the greater the survivorship, growth, and reproductive potential) (Connell 1973, Loya 1976, Highsmith 1982, Jackson 1985, Karlson 1986, 1988; Hughes and Connell 1987, Lasker 1990, Babcock 1991, Hughes et al. 1992). Thus, fragmentation caused by spiny lobster trap gear could result in smaller colonies, potentially reducing their overall survivorship, and growth and reproduction potential. Mortality of coral fragments may also occur, eliminating entirely the possibility of asexual regeneration or future sexual reproduction by those fragments.

Fragmented coral colonies also frequently stop producing gametes for a period of time, due to the reallocation of energy mentioned above. Gamete production is likely to resume only once a certain level of growth and/or tissue repair/regeneration has occurred (Lirman 2000). Lirman (2000) found that *A. palmata* coral colonies that suffered fragmentation during Hurricane Andrew did not produce gametes fully three years after the event. Similar shifts in energy allocation from reproduction toward regeneration have been noted in *Montastraea annularis* (Van Veghel and Bak 1994) and other hard coral species (Kojis and Quinn 1985, Szmant 1986, Hughes et al. 1992). Thus, even surviving

[12] *Acropora* are members of the phylum cnidaria.

Acropora fragments may be removed from the spawning population for at least some period of time.

Lirman (2000) observed that the survivorship of *A. palmata* fragments was influenced by the type of substrate upon which the fragment settled. Fragments landing atop other *A. palmata* colonies showed no signs of mortality, while fragments landing on sand, rubble, and hardbottom showed 71%, 50%, and 40% losses in tissue after four months. The relative scarcity of *Acropora* colonies in the U.S. Caribbean reduces the likelihood of an *Acropora* fragment landing on another *Acropora* colony. As a result, fragments in isolated colonies may have a lower likelihood of survival. Other studies suggest a similar correlation between substrate type and survivorship in other coral species (e.g., Yap and Gomez 1984, 1985; Heyward and Collins 1985, Wallace 1985, Bruno 1998).

Abrasion

Abrasion by spiny lobster traps and trap lines can result in the loss of tissue, or tissue and skeleton. The loss of tissue can be partial or complete and the loss of tissue and skeleton can by superficial or extensive (Woodley et al. 1981, Glynn 1990, Craik et al. 1990, Hall 1997). The extent and severity of abrasion injuries is dependent upon the duration and frequency of the abrasion events.

The adverse affects to *Acropora* resulting from abrasion injuries are similar to those mentioned above for fragmentation. One of the primary impacts is the reallocation of energy away from reproduction and growth, towards regeneration or repair of the injured tissue and skeleton (Kobayashi 1984, Rinkevich and Loya 1989, Meester et al. 1994, Van Veghel and Bak 1994, Van Veghel and Hoetjes 1995, Hall and Hughes 1996, Hall 1997).

Areas injured by abrasion also provide sites for pathogens to enter and create habitable space for settlement of other organisms (e.g., algae, sponges, or other corals) (Bak 1977, Hall 1997). In many coral species, polyps defend the colony by secreting mucus, discharging nematocysts, or through the production of allelochemicals (Hall 1997). The removal of polyps reduces a colony's ability to protect itself, potentially affecting its survivorship. Abrasion injuries also reduce the surface area available to photosynthesize, feed, and reproduce (Jackson and Palumbi 1979, Wahle 1983, Hughes and Jackson 1985, Babcock 1991, Hall and Hughes 1996, Hall 1997).

The type and severity of an abrasion injury (i.e., tissue or skeleton) affects the amount of time required for healing and the amount of energy that must be allocated for regeneration. Hall (1997) states that the time needed to fully recover from tissue injuries was much faster than the time required to completely regenerate fragmented skeleton. This suggests that the loss of tissue from a branch has less impact to the colony as a whole, than the loss of a branch. Hall (1997) hypothesizes that the replacement/regeneration of soft tissue requires the commitment of fewer resources than the regeneration of skeletal material, thus soft tissue can be replaced more quickly. However, Hall (1997) also observed that the area exposed when a branch is fragmented from the colony often healed more quickly than other soft tissue injuries. This suggests that while the regeneration of a fragmented branch may take considerably longer than

healing a soft tissue injury, the colony may be exposed to disease and competitors for less time after branch fragmentation than when the colony is repairing a tissue injury.

5.3.1 Effects of Recreational Trapping

Puerto Rico
As we noted previously in Section 5.2.2, we do not believe a significant recreational hand harvest fishery exists in the EEZ off Puerto Rico. In that section we provided our rationale for that determination. There is currently no information available on the recreational trap harvest of spiny lobsters in the Puerto Rican EEZ. As with the recreational hand harvest sector, the only indication that a recreational trap sector might exist is simply that traps are an authorized gear for recreationally harvesting spiny lobster in the EEZ. However, we believe much the same rationale that support the lack of a recreational hand harvest fishery in the EEZ also applies to the recreational trap sector.

The size of recreational vessels is not available but data on the commercial fishery indicates that approximately 96% of vessels are 30 ft in length or smaller (Matos-Caraballo and Agar 2011).[13] These are relatively small vessels and traveling at least nine miles offshore simply to recreationally target spiny lobster seems unlikely. We also believe the economic disincentives (i.e., cost of fuel and supplies), the prohibition on the sale of recreational harvest (Puerto Rico Fishing Regulation 7949; November 24, 2010) sea conditions, etc. that likely act to dissuade potential recreational divers from harvest spiny lobster also act on potential recreational trappers. Recreational trappers also have additional disincentives to fish.

Traps/pots require time to build, repair, and maintain, and land to store them. These requirements and prohibition on the sale of recreational caught spiny lobster likely act to keep the number of recreational traps fished by any one fisherman relatively low. The prohibition on the sale of recreationally caught spiny lobster also means that the costs of trips to the EEZ cannot be offset by their harvest. We believe this reduced incentive to have multiple traps, also reduces the likelihood that a recreational fisher would travel offshore to the EEZ to fish a relatively small number of traps.

We believe the bathymetry of the continental shelf off Puerto Rico and the large area of fishable water under Commonwealth jurisdiction provides incentives for recreational spiny lobster fishers not to fish in the EEZ off Puerto Rico. For these reasons, we believe it is extremely unlikely that recreational traps for spiny lobster are being deployed in the EEZ and therefore traps are not likely to adversely affect *Acropora* coral or designated critical habitat.

St. Croix
As we noted previously in Section 5.2.3, we do not believe a significant recreational hand harvest fishery exists in the EEZ off St. Croix. In that section we provided our rationale for that determination. There is currently no information available on the recreational

[13] 0.5% of vessels were less than 10 ft long; 47.6% of vessels were 10-19.9 ft long, and 49.7% of vessels were 20-29.9 ft long (Matos-Caraballo and Agar 2011).

trap harvest of spiny lobsters in the St. Croix EEZ. As with the recreational hand harvest sector, the only indication that recreational trap sector might exist is simply that traps are an authorized gear for recreationally harvesting spiny lobster in the EEZ. However, we believe much the same rationale that supports the lack of a recreational hand harvest fishery in the EEZ also applies to the recreational trap sector. We believe the economic disincentives (i.e., cost of fuel and supplies), sea conditions, regulations (i.e., prohibition on sale of recreational harvest), etc., that likely act to dissuade potential recreational divers from harvest spiny lobster also act on potential recreational trappers. Recreational trappers also have additional disincentives to fish.

The use of traps to harvest spiny lobster recreationally is prohibited in the territorial waters of the USVI (USVI DPNR 2009). Traps/pots require time to build, repair, and maintain, and land to store them. The territorial water trap prohibition also means fishers must travel a minimum of three miles offshore to deploy their traps, which increases their fuel and supply costs. We believe these additional economic/regulatory disincentives to fish traps, in conjunction with the other issues mentioned previously in Section 5.3.3, provide a strong disincentive to recreationally fish spiny lobster traps.

We have no indication that a recreational spiny lobster trap fishery exists in St. Croix. We believe the disincentives mentioned above, and the documented fisher preference for targeting billfish and offshore pelagics are reasons that a recreational spiny lobster trap fishery is unlikely to exist. Since we do not believe spiny lobster are being recreationally targeted with traps in the EEZ, we believe this sector of the fishery will not adversely affect *Acropora* coral or designated critical habitat.

5.3.2 Effects of Commercial Trapping

Kojis and Quinn (2011) and Kojis (2004) reported that pot gear is widely used in the USVI, with fish pots and lobster pots being the most frequently used pot gear. However, all of the spiny lobster trap gear appears to be used in St. Thomas/St. John. The 2003 census reported that 82.6% of trap/pot fishers in St. Croix were contacted and none of those respondents stated they fished lobster pots.[14] The 2010 census reported that 89% of trap fishermen were contacted and there was no evidence of a directed trap fishery for spiny lobster off St. Croix in the EEZ (Kojis and Quinn 2011). Trap fishermen off St. Croix primarily deploy traps in territorial waters because of difficult sea conditions on Lang Bank and because of the price of gas, many fishermen prefer to remain closer to shore (B. Kojis to A. Herndon, NMFS, pers. comm. 2011). Landings data indicate that on average approximately 4% of all spiny lobster landings in St. Croix are caught using traps during any given year. Because landings data do not differentiate between harvest in federal or territorial waters, it is impossible to tell if those trap landings are occurring in federal waters. Information indicates that it is unlikely that these landings are the result of directed trap effort in the EEZ and are more likely lobsters caught incidentally during reef fish trap fishing. Since consecutive fishery censuses show no sign of a directed trap fishery for spiny lobster off St. Croix in the EEZ, we believe one does not

[14] In St. Croix 69 fishers reported trap landings but only 57 responded to the survey.

exist. Therefore, the following effects analysis only evaluates the potential adverse affects from trapping in the EEZ off Puerto Rico.

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* coral through breakage or abrasion, but traps can also destroy newly settled planulae during setting or hauling. For this analysis, we assume that *Acropora* coral will only occur in areas designated critical habitat. Thus, we calculate the impacts to critical habitat and then use the available data on *Acropora* coral density to estimate what area of *Acropora* coral may be affected by fishing gear.

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. 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. Since traps do not cause consolidated hardbottom to become 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.

Thus, the use of traps may affect critical habitat and *Acropora* coral through breakage or other physical damage. The physical impacts of traps on *Acropora* coral 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.

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) (Kojis and Quinn 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 spiny lobster traps in Puerto Rico are approximately 47 inches in length by 24 inches in width (1,128 sq in or 7.8 sq ft). Agar et al. (2005) identified 324 trap fishermen in Puerto Rico. Trap fishermen responding to the census used 11 spiny lobster traps on average. The report does not differentiate between fish trap trips and spiny lobster trips or soak times, but it does

report that Puerto Rican trap fishermen take an average of 2.1 trips/week and traps soak an average of 6 days. 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. The census does indicate that 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).

Using the available data on fishing effort Puerto Rico we developed two approaches for estimating potential number of fish traps used in the fishery annually. Each estimate 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. Once we calculated the total number of traps likely hauled, we then applied the proportions of all traps that are likely spiny lobster traps as reported in Matos-Caraballo and Agar (2011) (i.e., 40% of all traps are spiny lobster traps in Puerto Rico). This provided an overall estimate of the likely number of spiny lobster traps hauled.

The first estimate used the information on trips per week and the average number of spiny lobster traps fished per fisher in Puerto Rico per week and annually (Table 5.3.2.1). Puerto Rico fishermen take 2.1 trips/week and use 11 spiny lobster traps (Agar et al. 2005).

Table 5.3.2.1 Estimated Total Number of Spiny Lobster Traps Hauled per Year Based on Average Number of Spiny Lobster Traps Fished

Area	No. of fishermen	Trips/week	Avg. Lobster Traps Fished	Total lobster traps hauled/week	Total traps hauled/yr
Puerto Rico	324	2.1	11	7,484	389,168

The second approach for estimating trap impacts was based on soak times as reported by Agar et al. (2005). As noted above, Puerto Rican fishermen soaked traps 144 hours (6 days) on average. Therefore, by dividing the average soak time by the total number of hours in a week we could estimated 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 (see Table 5.3.2.2)[15]

Table 5.3.2.2. Estimated Total Number of Spiny Lobster Trap Hauled per Year Based on Reported Soak Time for all Traps

Area	Avg. Soak Time (Hrs)	Trips/ week	No. of fishermen	Avg. Lobster Traps Fished	Total traps hauled/week	Total lobster traps hauled/yr
Puerto Rico	144	1.2	324	11	4,277	222,404

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

Both methods used similar data and ultimately produced a range (i.e., 389,168 to 222,404) for the potential number of spiny lobster traps hauled in Puerto Rico. While we believe each approach is appropriate based on the data available, we chose the estimate provided by the first approach because it used the slightly more specific information on the number of spiny lobster traps fished. Therefore, we anticipate up to 389,168 lobster trap pulls may have occurred in Puerto Rico.

Sheridan et al. (2005) reported 54% of lobster and fish traps were in reported in coral habitat, information specific to spiny lobster traps was not provided. Based on the preceding trap estimate, we anticipate up to 210,151 traps were placed and hauled from coral habitat.[16] Sheridan et al. (2005) also indicate that of the traps deployed on coral habitat, 13.7% cause damage to corals. With 210,151 traps potentially affecting 13.7% of coral and critical habitat, we estimate 28,791 spiny lobster traps per year cause damage to *Acropora* coral and critical habitat.[17] The footprint of spiny lobster traps used in Puerto Rico is 7.8 sq ft (Scharer et al. 2004). Applying this information to the 28,791 traps expected to cause damage, we estimate spiny lobster traps in Puerto Rico could affect 224,570 sq ft.[18] of *Acropora* and critical habitat. However, only 2.9% of the designated critical habitat in the Puerto Rico unit occurs in the EEZ. Therefore, we anticipate direct effects from fishing occurring in the EEZ will occur to 6,513 sq. ft. of *Acropora* critical habitat in the Puerto Rico unit per year.[19]

Garcia-Sais et al. (2008b and unpublished data 2010) conducted benthic surveys at seven locations off Puerto Rico, four off western Puerto Rico (Figure 5.3.2.1). Only one of the seven survey locations (Rincon) found evidence of elkhorn coral and it was located immediately offshore in Commonwealth waters. The other three sites off western Puerto Rico found no evidence of elkhorn coral. This is not particularly surprising since elkhorn coral prefer shallow, turbulent water (i.e., 0-15 ft) and the waters depths of the fishable habitat in the EEZ off of western Puerto Rico are generally deeper (i.e., 70+ ft) than the preferred depth range. Based on this information, we do not anticipate elkhorn coral will occur in the area of fishable habitat where trap effects are anticipated; thus, traps are not likely to adversely affect this species.

Staghorn coral is known to occur deeper than elkhorn coral. Garcia-Sais et al. (2008b) reported finding some staghorn corals during transects at all four locations off western Puerto Rico. However, only one of those locations, Tourmaline Bank at Mayaguez, occurred in the area where trap effects were anticipated and we used staghorn abundance information from that site in our estimates of effects. In general, the staghorn corals were present but relatively scarce. While seen at most locations, colonies only rarely fell within the actual 10 m transects conducted by Garcia-Sais et al. (2008b). However, in

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[16] Estimated maximum number of spiny lobster trap hauls in Puerto Rico x 54% of all traps placed in coral habitat = Number of spiny lobster traps estimated hauled from coral habitat.

[17] Number of Trap Hauled From Coral Habitat x 13.7% of Traps Affecting Coral = Number of Traps Expected to Cause Damage.

[18] Number of Traps Causing Damage x Puerto Rican Trap Footprint = Area of *Acropora* and Critical Habitat Impacted by Traps

[19] Area of *Acropora* and Critical Habitat Impacted by Traps x Percentage Occurring in EEZ = Total Area of *Acropora*/Critical Habitat Affected by Traps in the EEZ.

one location staghorn corals did fall within an established transect. Garcia-Sais et al. (unpublished data 2010) report the percent cover of staghorn at that location was a mean of 0.4%. Multiplying this figure by our estimate of critical habitat that may be affected by trap fishing, we estimate up to 26 sq ft of staghorn coral may be adversely affected by fragmentation and/or abrasion in Puerto Rico each year.[20]

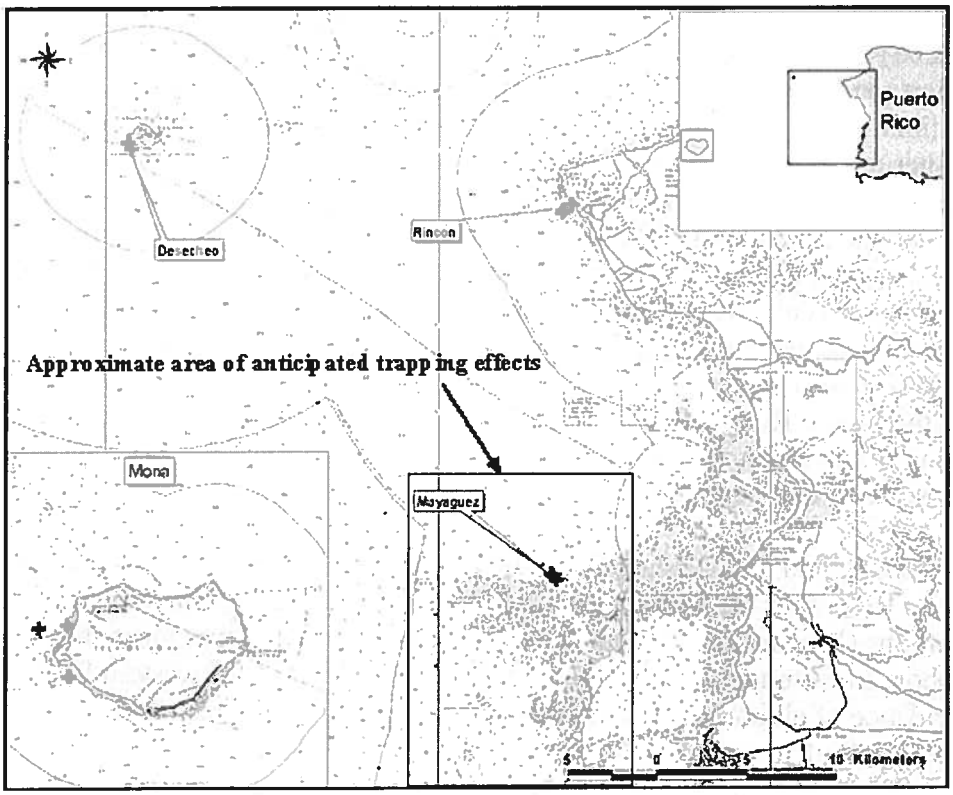

Figure 5.3.2.1 Areas Surveyed by Garcia-Sais et al. and the Area of Anticipated Trap Effects (Adapted from Garcia-Sais et al. 2008b)

5.4 Effects of Vessels and Anchors on *Acropora* Coral and *Acropora* Critical Habitat

Effects on *Acropora* coral and *Acropora* critical habitat from moving spiny lobster vessels (i.e., vessels transiting to and from fishing areas and moving during fishing activity) are discountable. *Acropora* coral and *Acropora* critical habitat are located on the benthos and would only very rarely be at risk at risk from moving vessels. Vessels need sufficient water to navigate without encountering the bottom, and when transiting shoal areas with marginal clearance vessels typically transit cautiously (i.e., slowly). Spiny lobster vessels embarking and returning from offshore fishing trips would likely travel via maintained channel waters where interactions would be even more unlikely. Thus, we believe the likely impacts from vessel strikes on *Acropora* coral and *Acropora* critical habitat are extremely unlikely to occur and are discountable.

[20] 0.4% cover of staghorn coral x 6,513 sq ft of *Acropora* critical habitat in the EEZ likely affected/year = 26.052 sq ft of staghorn coral affected/yr.

Vessel anchoring can adversely affect, via breakage/fragmentation, the dead coral skeleton element of critical habitat and live *Acropora* colonies. However, since trap vessels generally do not anchor while fishing to increase efficiency, we believe it is unlikely anchors from these vessels will damage dead coral skeletons or live *Acropora* colonies and any adverse affects are discountable.

Anchoring by vessels using hand harvest techniques could pose a threat to *Acropora* coral and *Acropora* critical habitat. In Section 5.2.3, we concluded that commercial or recreational hand harvest of spiny lobster will not adversely affect *Acropora* or critical habitat in the St. Croix EEZ. Likewise, we do not believe the recreational harvest of spiny lobster is likely to adversely affect *Acropora* or critical habitat in the EEZ off Puerto Rico. Therefore, the following analysis only focuses on commercial diving for spiny lobster in the EEZ off Puerto Rico. Our commercial dive analysis for Puerto Rico estimated that up to one trip daily maybe occurring in the EEZ where *Acropora* coral and critical habitat could occur. The deployment of an anchor is often not precise, particularly during rough weather conditions or during periods of poor water visibility, and could potentially affect *Acropora* coral or their critical habitat. Commercial dive trips in the EEZ likely have a crew of two or more. While we believe multi-man crews can fish without the use of an anchor, our analysis will assume they do use an anchor. The vast majority of vessels in Puerto Rico are less than 30 ft in length.[21] We assume that most vessels of this size use of a 4-lb. aluminum anchor with measurements of 24 in x 19 in (3.17 ft^2). Multiplying the area of an anchor by the total number spiny lobster dive trips to the Puerto Rican EEZ, we estimate that up to 3.2 sq. ft. of *Acropora* coral and/or designated critical habitat may be affected each day, or 1,168 sq ft. annually. Applying our staghorn density estimate by the area of critical habitat that may be affected annually, we anticipate up to 5 sq ft of staghorn coral may be adversely affected by fragmentation and/abrasion.[22]

5.5 Anticipated Annual Impacts to Staghorn Coral and Designated Critical Habitat

Our effects analysis determined that traps and anchoring contacting staghorn coral and *Acropora* critical habitat may cause adverse affects via fragmentation and/or abrasion. The anticipated interactions resulting from traps and vessel anchoring are summarized in Table 5.5.1

Staghorn coral often reproduce asexually via fragmentation but the substrate upon which it lands plays a significant role in whether the fragment survives (see previous discussion in 5.3). Benthic habitat data from Tourmaline Bank indicates that 4.8% of the benthos there is comprised of habitats likely to be suitable for reattachment.[23] We consider hardbottom habitat suitable for reattachment. Lirman (2000) reports that even fragments landing hardbottom habitat still suffered tissue loses of approximately 40%. The remaining 95.2% of benthic habitat is unlikely to support reattachment; fragmented

[21] 97% of vessels in Puerto Rico are 30ft or less (Matos-Caraballo and Agar 2011).
[22] 1,168 sq ft x 0.4% coverage of staghorn coral = 4.67 sq ft of staghorn coral affected/yr
[23] 4.2% reef over hang, 0.1% coralline algae, 0.1% *Hamileda tuna* and 0.4% *Acropora cervicornis* (Garcia-Sais et al. unpublished data 2010).

colonies landing on unsuitable habitat are unlikely to survive. Because the likelihood of landing on habitat suitable for reattachment is so low (i.e., less than 5%) we will assume that all fragmented colonies result in mortality. Table 5.5.1 summarizes the amount of staghorn coral we anticipate will suffer mortality each year in the Puerto Rico EEZ.

Table 5.5.1 Annual Area of Impacts to Staghorn Coral and Designated Critical Habitat in Federal Waters

Corals	Trapping Impact (Sq. Ft)	Anchoring Impact (Sq. Ft)	Total Area Impacted (Sq. Ft)
Acropora Critical Habitat	6,513	1,168	7,681
Staghorn Coral *(Acropora cervicornis)*	26	5	31

5.6 Effects of Caribbean Spiny Lobster Fishing 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 spiny lobster 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 spiny lobster 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 spiny lobster 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 Caribbean spiny lobster fishery (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 Caribbean spiny lobster fishery are suspended in the water column or essentially stationary on the bottom and do not affect water column or benthic habitat characteristics. Also, spiny lobster 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 on sea turtles. This is because in NMFS (2005a) we determined that 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. NMFS (2005a) estimated one leatherback sea turtle would be incidentally captured as a result of the continued authorization of the Caribbean spiny lobster fishery managed under the SLFMP. In the absence of U.S. Caribbean-specific sea turtle interaction data, incidental captures were estimated via extrapolation by using data on sea turtle entanglement data from the Gulf of Mexico and a proxy for the total amount of fishing effort by gear type in

the U.S. Caribbean EEZ trap fisheries. Much of the bycatch information and analysis methods stemmed from a 2005 opinion on the Gulf of Mexico reef fish fishery (NMFS 2005b).

In the previous biological opinion on the Caribbean spiny lobster fishery, the Caribbean specific data available for our effects analysis was so limited that the available information on interactions between sea turtles and fishing gear in the Gulf of Mexico was the best available information. Since the completion of that opinion, additional historic and recent sea turtle strandings data specific to the U.S. Caribbean has become available.

The Sea Turtle Stranding and Salvage Network (STSSN) was formally established in 1980 to collect information on and document strandings of sea 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 the 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 the STSSN technically encompasses portions of the U.S. Caribbean, queries of the online database reveal no data for the U.S. Caribbean. However, historic and recent data on vessel and fishing-related interactions with sea turtle 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. The following summaries are general and include information on fishing gears that are not necessarily used in the Caribbean spiny lobster fishery. We have included this information because these data have not been widely reported and because of the general dearth of information on the sea turtle interactions with U.S. Caribbean fisheries overall. In Section 5.6.3-5.6.5, we then analyze the effects of the proposed action's traps 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 spiny lobster management are not expected to change overall trap effort or vessel trips 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 spiny lobster 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 unobserved 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 samples. Observers sampled 28 trips from nine individual captains that allowed observers on board, 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 report 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.

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 (1994-2003) 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 five 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% of the fishers on St. Croix and 29% of the fishers on St. Thomas. For both islands, 7% 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% and 18% 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. 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 frequent 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. 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. Boating-related injuries (propeller wounds and boat strikes) were documented on 30 of the total strandings reported. There were 16 strandings exhibiting boating-related injuries on St. Croix, 9 on St. Thomas, and 5 on St. John. Of the 10 strandings exhibiting 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 2000)
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 the relative causes of stranding for sea turtles in the Virgin Islands.

Boulon (2000) 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 (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 sea turtle.

Boulon (2000) 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 apparent 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 (2000) 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 (R. Boulon, USVI DPNR, to J. Lee, NMFS, pers. comm. 2011). 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

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

1998 through 2000
No data from "other fishing and vessel-related sea turtle stranding data sets/analyses" were available for this period.

2001 through 2006[24]
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 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[25]
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

[24] Data received via e-mail to J. Lee, NMFS, from C. Lombard, USFWS, St. Croix, USVI.
[25] *Ibid*

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; 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 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" (Matos-Caraballo 2005). 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 contained assorted sea turtle records for Puerto Rico.

1989-1992
A list of stranding and mortality records from 1989-1992 included 71 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 (45%), followed by captures (30%) and incidental catch (18%); only one record was a result of a boat collision (NMFS unpublished data).

Table 5.6.2.1 1989-1992 Puerto Rico Strandings

	Boat collisions	Strandings	Capture	Incidental catch	Unknown	Total (by species)
Green	0	18	5	10	0	33
Hawksbill	1	11	15	3	4	34
Leatherback	0	3	1	0	0	4
Total	1	32	21	13	4	71
% of Total	1.41%	45.07%	29.58%	18.31%	5.63%	100.00%

1993-2009

Approximately 261 sea turtle stranding records from Puerto Rico were documented between 1993 through 2009, including 110 green, 141 hawksbill, 4 leatherback, and 4 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 34 reported, with an average of 16.3 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 just reflect monitoring levels those years (NMFS unpublished data).

The suspected cause for many of the sea turtle strandings was labeled "human impacts" (i.e., interactions with legal fishing gear, poaching, boat strike, etc.) (39.4%) followed by strandings caused by "unknown" causes (36.7%) and "natural" causes (36.7%) (NMFS unpublished data). Of the 102 strandings noted as "human impacts" approximately 56 (33 green, 22 hawksbill, and 1 leatherback sea turtles) were reported as either a boat strike or bycatch-related (Table 5.6.2.2). Of those, 15 (7 green and 8 hawksbill) were boat strikes, 17 (10 green, 7 hawksbill) were hooked or entangled in fishing line, 18 were entangled in net (11 green, 6 hawksbill, 1 leatherback), mainly gillnet; and 6 were entangled in rope/trap gear, mainly around the neck. Two additional reports could not be assigned to a specific gear type. Of the remaining strandings noted as caused by "human impacts" 32 were noted as "illegal fishing;" 8 reports (4 green, 4 hawksbill) noted the cause was "hunting", 15 reports (3 green, 12 hawksbill) noted the cause was "harpoon"; one report (hawksbill) just indicated "shooting", while the remaining 8 records simply stated the cause was "illegal fishing." All of these are believed to be from directed poaching activities and not incidental.

Table 5.6.2.2 Puerto Rico Vessel and/or Bycatch Related Sea Turtle Strandings Documented Between 1993-2009

Year	Species	Sex or Age Class	Suspected Cause	Release Condition
No reported bycatch/vessel related strandings reported from 1993-1995				
1996	Green	--	Captured in Net	Alive
1997	Hawksbill	Juvenile	Captured in Net	Alive
1997	Hawksbill	Juvenile	Captured in Net	Alive
1997	Loggerhead	M	Captured in Net	Alive
1998	Green	--	Captured in Net	Dead
1999	Green	F, Sub adult	Hooking	Rehabilitated Released Alive
1999	Hawksbill	F	Vessel Strike	Dead
1999	Hawksbill	Juvenile	Hooking	Rehabilitated Released Alive
1999	Hawksbill	F	Vessel Strike	Dead
2000	Green	F juvenile	Captured in Net	Alive
2000	Green	F, Adult	Captured in Net	Alive
2000	Hawksbill	F	Vessel Strike	Dead
2000	Green	--	Vessel Strike	Dead
2000	Hawksbill	F, Adult	Vessel Strike	Dead
2000	Hawksbill	Subadult	Captured in Net	Dead
2000	Hawksbill	--	Vessel Strike	Dead

Table 5.6.2.2 Puerto Rico Vessel and/or Bycatch Related Sea Turtle Strandings Documented Between 1993-2009 (cont'd)

Year	Species	Sex or Age Class	Suspected Cause	Release Condition
2001	Hawksbill	Adult	Captured in Net	Dead
2001	Green	--	Captured in Net	Alive
2001	Green	--	Rope entanglement	Dead
2001	Green	--	Captured in Net	Dead
2002	Green	--	Rope entanglement	Dead
2002	Green	--	Vessel Strike	Dead
2002	Green	--	Captured in Net	Dead
2002	Green	Juvenile	Hooking	Alive
2002	Green	--	Vessel Strike	Dead
2002	Green	--	Fishing line entanglement	Dead
2002	Green		Hooking	Alive
2003	Hawksbill	--	Rope entanglement	Dead (Skeletal remains)
2003	Green	F, Adult	Fishing line entanglement	Rehabilitated Released Alive
2003	Green	F, Adult	Fishing line entanglement	Dead
2003	Hawksbill	Juvenile	Hooking	Alive
2003	Hawksbill	F, Juvenile	Fishing line entanglement	Dead
2004	Green	Juvenile	Vessel Strike	Alive
2004	Hawksbill	Adult	Fishing line entanglement	Dead
2004	Hawksbill	F, Adult	Captured in Net	Rehabilitated Released Alive
2004	Hawksbill	Adult	Fishing line entanglement	Dead
2005	Green	--	Hooking	Dead
2005	Green	--	Rope entanglement	Dead
2006	Hawksbill	M, adult	Captured in Net	Dead
2006	Hawksbill	Juvenile	Hooking	Rehabilitated Released Alive
2006	Green	F, Adult	Captured in Net	Dead
2006	Green	--	Fishing line entanglement	Alive
2006	Green	F, Adult	Lobster trap entanglement	Dead
2007	Hawksbill	Adult	Vessel Strike	Dead
2007	Leatherback	F, Adult	Captured in Net	Dead
2007	Green	--	Lobster trap entanglement	Dead
2008	Hawksbill	--	Vessel Strike	Dead
2008	Green	--	Vessel Strike	Alive
2008	Green	Juvenile	Vessel Strike	Euthanized
2008	Green	--	Vessel Strike	Dead
2008	Green	Adult	Captured in Net	Dead
2008	Hawksbill	Adult	Vessel Strike	Dead
2008	Hawksbill	Juvenile	Fishing line entanglement	Dead
2008	Green	Juvenile	Hooking	Rehabilitated Released Alive
2008	Green	M, adult	Captured in Net	Dead
2009	Green	Adult	Fishing line entanglement	Dead

5.6.3 Effects of Hand Harvest on Sea Turtles

In Section 5.2.1, we determined it was extremely unlikely that recreational hand harvest was occurring the Puerto Rico EEZ. Since we do not anticipate recreational spiny lobster diving is occurring in Puerto Rico EEZ, we expect any adverse affects to sea turtles from those activities are extremely unlikely to occur and are discountable. We did determine that commercial hand harvest may be occurring the Puerto Rico EEZ. Divers only occasionally encounter sea turtles and anecdotal information from encounters indicates some sea turtles voluntarily change their route to avoid coming in close proximity to divers, whereas others appear unaware of their presence. There are no reports of incidental sea turtle takes by spiny lobster divers. Given the selectivity of the gears used and the visual nature of the hunt and capture of spiny lobsters, spiny lobster divers will easily be able to avoid sea turtles. Any behavioral effects on sea turtles from the presence of spiny lobster divers are expected to be insignificant. We therefore conclude that diving for spiny lobster is not likely to adversely affect sea turtles.

In Section 5.2.2, we concluded that no commercial diving for spiny lobster occurs in the EEZ off St. Croix based on fisher responses to censuses (Kojis and Quinn 2011). In that section we also concluded that it was extremely unlikely that recreational hand harvest was occurring the EEZ of St. Croix. As a result, we anticipate any adverse affects from recreational diving for spiny lobster to sea turtles in the St. Croix EEZ are extremely unlikely to occur and discountable.

5.6.4 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 most 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. Entanglement data from the Gulf of Mexico and South Atlantic regions indicate green and hawksbill sea turtle can also be entangled in trap. 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 the 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) in 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.[26] 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

[26] 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 turtle strandings 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.)

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).

5.6.4.1 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 spiny lobster 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 sea turtle may encounter the gear and the longer a sea turtle may be exposed to the entanglement threat, presumably increasing the likelihood of such an event occurring.

5.6.4.2 Recreational Trap Effects to Sea Turtles

USVI
In Section 5.3.1, we described our rationale behind our belief that no recreational trapping for spiny lobster is occurring in the EEZ off St. Croix. We believe that rationale also supports the determination that recreational trap fishing for spiny lobster will not adversely affect sea turtles occurring the EEZ off St. Thomas/St. John. There is currently no information available that indicates a recreational trap fishery for spiny lobster in St. Thomas/St. John even exists. Therefore, we anticipate that no adverse affects will occur to sea turtles as a result of recreational trap fishing in the EEZ off St. Croix, or St. Thomas/St. John.

Puerto Rico
In Section 5.3.1, we described our rationale behind our belief that no recreational trapping for spiny lobster is occurring in the EEZ of Puerto Rico. We believe that rationale also supports the determination that recreational trap fishing for spiny lobster will not adversely affect sea turtles. Therefore, we anticipate that recreational trap fishing is not likely to adversely affect sea turtles in the EEZ of Puerto Rico

5.6.4.3 Estimated Commercial Trap Entanglements and Associated Mortalities

Previous Approach to Estimating Anticipated Trap 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[27]) 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.[28] 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 was 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 (Sections 5.6.1 and 5.6.2), reports of sea turtles interacting with trap gear are rare, with no more than one or two documented during any one year, and frequently none.

In the USVI, some strandings data was available as far back as 1982. However, we only used the available strandings data from 2001-2010. Initially, we had intended on only using 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 the earlier time period also did not generally include any information on the ultimate fate of the animal. Additionally, because of very few records reported from 1982-2000, we were concerned that using data from the time series would actually underestimate potential adverse affects. Ultimately, we chose to use strandings from 2001-2010.

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[27] Trap data was compiled from Matos-Caraballo (1997) and USVI DPNR Data
[28] Number of traps in U.S. Caribbean EEZ = 21,710 total traps * 0.14 of fishable habitat in the EEZ

In Puerto Rico, strandings data is available from 1989, but we only used the available strandings data from 1996-2010. Initially, we had intended to 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 small. We considered using all the strandings data from 1989-2010, but did not because we were concerned that doing so would not properly characterize the fishery as it currently operates. Additionally, records from 1989-1992 did not include any information on the ultimate fate of the animal (i.e., dead or alive) and there were no records of interactions with legal fishing gear from 1993-1995. Because of the very few records reported from 1989-1995, we were concerned that using data from the time series may underestimate potential adverse affects. Ultimately, we chose to use strandings from 1996-2009 because it expanded our sample size. Additionally, since these data were not available for use in the 2005 Caribbean opinion, we felt it prudent to use them here to provide a more specific Caribbean-based analysis.

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, dead, or distressed sea turtles strand; thus, sea turtle stranding data represent only a subset of all sick, dead, or distressed sea 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 data indicate that all interactions with trap gear have resulted in mortality; however, given the small sample

size, we also believe it is possible that some sea turtles may have non-lethal interactions with trap gear. 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 Spiny Lobster Trap Entanglements
From 2001 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.[29] Since we do not believe trapping for spiny lobster occurs in the EEZ off St. Croix, this analysis only evaluates potential effects from fishing off St. Thomas/St. John.

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 spiny lobster traps to more accurately assess the effects of the proposed action.

Kojis and Quinn (2011) 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 87.2% responded (259 of 297 licensed fishermen); those respondents reported using 4,211 fish pots and 2,259 lobster pots, indicating that lobster traps make up 35% (2,590 of 7,419) of all traps.

Applying the estimate of the percentage of traps in the USVI that are likely used to target spiny lobster (i.e., 35%) 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 spiny lobster trap gear in St. Thomas/St. John annually.[30]

Estimated Puerto Rico Spiny Lobster Trap Entanglements
The data we have on strandings for Puerto Rico indicates that at least two strandings were the result of entanglement in lobster gear and an additional four had "rope" entanglements for a total of six sea turtles (5 green, 1 hawksbill). Using the same approach we used in our USVI calculations to account for unreported strandings, we

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[29] 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.
[30] 2 hawksbill and 2 green sea turtle entanglements in trap gear annually x 35% of all trap gear that are spiny lobster traps = 0.7 annual hawksbill and green sea turtles entanglements (rounded up to 1) caused by spiny lobster traps.

estimate the actual nearshore mortalities in Puerto Rico may have been 100 green and 20 hawksbill sea turtles entangled in "rope". Based on 14 years of data, we would anticipate up to seven green sea turtles and one hawksbill sea turtle would become entangled in "rope" annually in Puerto Rico, on average.[31]

The 2008 census of active commercial fishermen in Puerto Rico, conducted by Matos-Caraballo and Agar (2011), 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., 40% of all traps are spiny lobster traps) to our estimate of sea turtle entanglements (i.e., seven green sea turtles and one hawksbill sea turtles, annually) we estimate that three green and one hawksbill sea turtle are likely entangled in spiny lobster trap gear in the Puerto Rico annually.[32] General rounding rules would indicate we should round our estimate of annual hawksbill entanglements down to zero. However, because we have documented entanglements, albeit rare, we feel it is prudent to assume an additional entanglement could occur in the future. Therefore, we will assume that one hawksbill entanglement could occur.

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 St. Thomas/St. John 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 5.6.4.1).[33,34]

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

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[31] 5 green/1 hawksbill reported sea turtle strandings ÷ 5% of actual moralities reported = 100 total possible green and 20 hawksbill mortalities/strandings; 100/20 green/hawksbill ÷ 14 years of data = 7.1 green/1.4 hawksbill sea turtle entanglement annually on average.

[32] 7 green sea turtle entanglements in trap gear annually x 40% of all trap gear that are fish traps = 2.8 annual green sea turtles entanglements caused by fish traps; 1 hawksbill sea turtle entanglements in trap gear annually x 40% of all trap gear that are fish traps = 0.4 annual hawksbill sea turtles entanglements caused by fish traps.

[33] 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.

[34] Puerto Rico: 3 green sea turtle entanglement in fish trap gear annually x 14.4% of fishable habitat in EEZ = 0.43 green sea turtle entanglements caused by spiny lobster 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.

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 trap 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 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 reported 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 fisheries, we believe the trap gear techniques used in the USVI and Puerto Rico are similar enough that we would not anticipate 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 5.6.4.1 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 5.6.4.1 Estimated Annual Trap-Related Entanglements by Area

Species	USVI	Puerto Rico	Total
Total Interactions			
Green	1	3	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.

5.6.5 Effects from Fishing Vessels of Sea Turtles

Fishing 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, or resting.

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 and so are also particularly vulnerable to being struck by vessels.

5.6.5.1 Impacts of Vessel Strikes

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 were the cause of 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.

Vessel Type and Speed
Generally, vessels 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 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. 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 Florida counties when compared to northern counties.

5.6.5.3 Estimating Sea Turtle Vessel Strikes Attributed to Spiny Lobster Fishing 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 5.6.4.2 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. Unlike the 2011 Caribbean reef fish biological opinion, trip based information from which to calculate the potential impacts from spiny lobster fishing was not available. Instead, we attempted to estimate vessel strike effects based on vessel registrations and reported vessel use. While 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 vessels used to fish for spiny lobster 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. Because we do not believe significant recreational fishing for spiny lobster occurs in the EEZ of the U.S. Caribbean, the following analysis only evaluates the potential adverse effects from commercial vessels targeting spiny lobster.

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. In the USVI, a reported 12 sea turtles (9 green, 2 hawksbill, and 1 leatherback) strandings showed definitive signs of vessel strikes from 2001-2010, an average of 1.2 annually over that 10-year period. By species, the percent occurrence of boat strike wounds in USVI was 75% green, 17% hawksbill, and 8% leatherback sea turtles (See Section 5.6.1 for summary of USVI strandings information).

From 1996 through 2009, there are 15 sea turtle stranding records with definitive propeller injuries (7 green, 8 hawksbill sea turtles) in Puerto Rico, an average of 1.1annually over 14 years. In Puerto Rico, the percent occurrence of boat strike wounds

by species was 47% green and 53% hawksbill (See Section 5.6.2 for summary of Puerto Rico strandings information).

Since we believe that as few as 5% of all nearshore at-sea mortalities result in strandings, we will follow an approach similar to that used in our gear analyses above and make our estimates accordingly. Therefore, we estimate that in the USVI from 2001-2010 as many as 240 sea turtles may have suffered boat strike injuries, approximately 24 annually over the 10-year period. In Puerto Rico, 300 sea turtles may have also been struck during 1996-2009, or approximately 20 annually during the 15-year period.[35]

USVI Sea Turtle/Commercial Fishing Boat Strikes
NMFS (2011) used information on the number of commercial and recreational reef fish trips taken annually to calculate the likely proportion of vessel strikes that were attributable to the commercial sector of the fishery. However, no information on the recreational sector of the spiny lobster fishery in the USVI is available. Since no trip information is available, we could not use the trip-based approach we used in the in the 2011 Caribbean reef fish biological opinion. Instead, we estimated effects based on vessel registrations.

In the USVI, 47% of commercial fishers target spiny lobster and the average number of vessels owned per fisher was 1.22 (Kojis and Quinn 2011). Applying that average to licensed fishers (296) reported in Kojis and Quinn (2011), we estimate 361 vessels in the USVI are used for commercial fishing. Since 47% of fishers target spiny lobster, we anticipate 170 of the vessels (47% of 361) used for commercial fishing are spiny lobster 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 spiny lobster fishing comprise 2.2% of the total vessels in the USVI.

We estimated that 24 sea turtles were boat struck annually in the USVI. Additionally, if 2.2% of all vessels in the USVI are spiny lobster fishing vessels, then 1 sea turtle struck annually would be attributed to commercial spiny lobster fishing in the USVI.[36] Based on the known percentages of boat-struck species in the USVI, we would anticipate that the 1 boat struck sea turtle would be either a green, hawksbill, or leatherback sea turtle.

Puerto Rico Sea Turtle/Commercial Fishing Boat Strikes
For Puerto Rico we used an approach similar to what was done with the USVI, using vessel registrations to estimate the potential effects from the commercial fishery. Instead, we estimated effects based on vessel registrations.

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[35] USVI: 12 sea turtle strandings recorded ÷ 5% of nearshore at-sea mortalities stranding = 240 total sea turtle strandings; 240 vessel strike ÷ 10 years of data = 24 sea turtles annually
Puerto Rico: 15 sea turtle strandings recorded ÷ 5% of nearshore at-sea mortalities stranding = 300 total sea turtle strandings; 300 vessel strike ÷ 15 years of data = 20 sea turtles annually
[36] 24 sea turtle vessel strikes annually in USVI x 2.2% of vessel in the USVI are commercial spiny lobster fishing = 0.5 sea turtle vessel strikes caused by commercial fishing vessels in the USVI annually

In Puerto Rico, 49% of commercial fishers target spiny lobster (Matos-Caraballo and Agar 2011). Matos-Caraballo and Agar (2011) report the commercial fishing fleet consists of 670 vessels. Since 49% of fishers report targeting spiny lobster, we estimate 328 of the vessels (49% of 670) used for commercial fishing are spiny lobster vessels.

There are 60,640 USCG registered vessels in Puerto Rico (CFMC and NMFS 2011). This indicates that vessels used for commercial spiny lobster fishing comprise 0.5% of the total vessels in Puerto Rico.[37]

We estimated that 20 sea turtles were boat struck annually. If we assume that 0.5% of all vessels in Puerto Rico are commercial spiny lobster fishing vessels, then we would anticipate that 1 sea turtles struck annually could be attributed to commercial spiny lobster fishing in Puerto Rico.[38] Based on the known percentages of boat struck species in Puerto Rico, we would anticipate that the 1 boat struck sea turtle would be a green or a hawksbill sea turtle.

5.6.6 Anticipated Total Number of Annual Sea Turtle Interactions

The proposed action is expected to continue to adversely affect listed sea turtle via entanglement and vessel strikes. Anticipated interactions resulting from traps 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		Vessels		Maximum Anticipated Takes for the Entire Fishery
	USVI	Puerto Rico	USVI	Puerto Rico	
Green	1 (1)	1 (1)	1 (1)*	1 (1)*	4 (4)
Hawksbill	1 (1)	1 (1)			4 (4)
Leatherback	1 (1)	1 (1)		0	3 (3)

*Takes of these species are in combination ,this does NOT indicate a take for each species

5.7 Anticipated Future Take After Implementation of Amendment 5

In the preceding sections, we estimated the likely impacts to sea turtles, staghorn coral, and *Acropora* critical habitat over past years resulting from operation of the Caribbean spiny lobster fishery. We now must consider what effect, if any, implementation of Amendment 5 would have on future levels of take; i.e., whether the estimated past take and mortality levels would increase or decrease and by how much, or whether the same levels would continue in the future. We do this by looking at how the Caribbean spiny

[37] 60, 640 registered recreational vessels + 670 commercial vessels = 61,310 total vessels in Puerto Rico; 328 commercial spiny lobster vessels ÷ 61,310 total vessels = 0.5% of all vessels in Puerto Rico are commercial spiny lobster vessels.

[38] 20 sea turtle vessel strikes annually Puerto Rico x 0.5% of vessel in Puerto Rico are commercial spiny lobster fishing = 1 (0.1 rounded up) sea turtle vessel strikes caused by commercial fishing vessels in Puerto Rico annually

lobster fishery will potentially be affected by Amendment 5, and whether the anticipated effects will result in any changes to the overall operation of the spiny lobster fishery.

Amendment 5 would define the management reference points for spiny lobster based on an established year sequence for determining average annual landings; establish a recreational bag limit for spiny lobster harvest, and establish a framework for adjusting management measures in the spiny lobster FMP. As noted in Sections 5.2, there are no data available on recreational fishing for spiny lobster in the U.S. Caribbean and we believe there is no significant level of harvest from that sector of the fishery in the EEZ. NMFS is proposing a recreational bag limit to ensure that any harvest that may occur in the EEZ is done sustainably. The Commonwealth and Territorial governments are also considering implementing recreational bag limits for spiny lobster. Implementing a bag limit in the EEZ ensures that fishers cannot escape proposed regulations in the commonwealth/territorial waters by traveling to the EEZ.

None of the management measures proposed in Amendment 5 would immediately impact sea turtles, staghorn coral, and *Acropora* critical habitat because they do not specifically address these species/critical habitat, nor are they designed to specifically address fishing impacts on these species/critical habitat. Localized indirect impacts are possible from the proposed management measures because they could influence where and when fishing effort will occur. However, the proposed measures would not alter the techniques used in the Caribbean spiny lobster fishery.

Some of the catch limits set under Amendment 5 are anticipated to reduce overall harvest of spiny lobster by approximately 10% (NMFS and CFMC 2011). Assuming that those reductions result in a similar reduction in fishing effort, our level of past effects may be proportionately lower going forward. However, because it is unclear to what extent future fishing effort maybe reduce and how quickly an effort reduction may occur, we believe it is prudent to assume that the sea turtle, staghorn coral, and *Acropora* designated critical habitat interaction patterns that existed in the recent past will continue into the future.

5.8 Summary of Anticipated Incidental Take

The effects to sea turtles, staghorn coral, and critical habitat were estimated above. Under Amendment 5 we anticipate no changes from recent take levels and the following incidental takes may occur in the future as a result of the continued operation of Caribbean spiny lobster fishery. We calculated annual takes but actual annual take likely has very high variability because of natural and anthropogenic variation. Based on our experience monitoring fisheries 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 spiny lobster 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 spiny lobster fishery. We quantified the adverse affects to staghorn coral by estimating the area likely

to die as a result of contact with spiny lobster trap gear. 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 staghorn coral also makes using an areal metric necessary. Since the polyps that make 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 staghorn coral is a branching, colonial species, that uses 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 staghorn coral takes by area. Table 5.8.1 summarizes these estimates; mortal takes are denoted in parenthesis.

Table 5.8.1 Anticipated Future 3-Yr Incidental Take

Sea Turtles	Traps		Vessels		Anticipated Takes for the Entire Fishery
	USVI	Puerto Rico	USVI	Puerto Rico	
Green	(3)	(3)	(3)*	(3)*	(12)
Hawksbill	(3)	(3)			(12)
Leatherback	(3)	(3)		0	(9)
Corals and Critical Habitat	**Trapping Impact (Sq. Ft)**		**Anchoring Impact (Sq. Ft)**		**Total Area Impacted (Sq. Ft)**
Acropora Critical Habitat	0	19,539	0	3,504	23,043
Staghorn Coral	0	78	0	15	93

*Takes of these species are in combination ,this does NOT indicate a take for each species

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. 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 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.

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.

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[39] 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.

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

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 Staghorn Coral

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 chronic stressor. 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 staghorn coral colonies that may 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 spiny lobster fishery may adversely affect designated *Acropora* critical habitat, 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 spiny lobster 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 staghorn coral 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.

7.1 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.

In Section 5 we estimated the Caribbean spiny lobster fishery was likely to adversely affect up to 23,043 sq. ft (0.001 mi^2) of *Acropora* critical habitat every three years, all of which is likely to be in the Puerto Rico critical habitat unit. However, we also anticipate contact between traps, anchors, and critical habitat that does not break destroy dead coral skeleton will be temporary in nature. A trap/anchor could temporarily cover an area with the appropriate essential feature, impeding its function. However, once that trap/anchor

is retrieved the function will be restored. Since function is likely to be restored as soon as a trap/anchor is removed, we do not expect any cumulative effects from trap/anchor 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 Puerto Rico critical habitat unit is 1,383 mi^2. Thus, the potential direct effects from the proposed action would affect less than one-ten-thousandth of one percent (i.e., 0.00007%) of the designated critical habitat in Puerto Rico every three years.[40] Therefore, we believe the effects from the proposed action are likely adversely affecting, but not destroying or adversely modifying *Acropora* critical habitat in the Puerto Rico critical habitat unit. We do not believe critical habitat is being affected in any other critical habitat unit. Therefore, we do not believe the designated critical habitat in the U.S. Caribbean will be adversely modified or destroyed by the continued operation of the Caribbean spiny lobster fishery.

7.2 Jeopardy Analysis for Staghorn Coral

Our jeopardy analysis now considers the effects of physical contact with fishing gear and anchors. First, we evaluate whether the anticipated effects will result in any reduction in distribution, reproduction, or aerial coverage (numbers) may appreciably reduce the species' likelihood of survival in the wild. Second, we consider how those effects are likely to affect the species' likelihood of recovery in the wild.

We anticipate up to 93 ft^2 of staghorn coral may have contact with traps and anchors every three years in Puerto Rico and lead to morality. Benthic habitat cover reported at Tourmaline Bank indicates that 4.8% of habitat could be suitable for staghorn coral (adapted from Garcia-Sais et al. unpublished data 2010). Multiplying the percent cover of habitat potentially suitable for staghorn by the amount of critical habitat occurring in the Puerto Rico EEZ (the only areas we anticipate finding staghorn coral) yields an estimate of the amount of area where staghorn coral may occur. Multiplying that estimate by the percent cover of staghorn coral described in Section 5.3.2 (i.e., 0.4%), we estimate 0.0077 mi^2 (214,664 ft^2) of staghorn coral may exist around in the EEZ off Puerto Rico.[41] The area we estimate may suffer mortality every three years because of the spiny lobster fishery accounts for less than one percent (i.e., 0.04%) of total amount of staghorn corals that may be in the Puerto Rico EEZ. The potential mortality of up to 93 ft^2 of staghorn coral over consecutive 3-year periods would reduce the amount of staghorn coral, compared to the amount that would have been present in the absence of the proposed action, assuming all other variables remained the same.

We anticipate the proposed action will cause mortality to only a very small amount of staghorn coral and those effects are only anticipated in the Puerto Rico EEZ, which is an even smaller portion of the species range in the U.S. Caribbean. Since these potential impacts are small and likely to occur only off Puerto Rico, we believe these impacts

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[40] 0.001 mi^2 of critical habitat affected by spiny lobster fishing in Puerto Rico ÷ 1,383 mi^2 of critical habitat designated in Puerto Rico x 100 = 0.00007% critical habitat affected every 3 years

[41] Approximately 40 mi^2 of critical habitat in the Puerto Rico EEZ x 4.8% potential staghorn habitat = 1.92 mi^2 of potential staghorn habitat; 1.92 mi^2 x 0.4% staghorn percent cover = 0.0077 mi^2 of staghorn coral.

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 the proposed action will reduce the distribution of the species.

The proposed action is anticipated to cause a reduction of 93 ft^2 in the total area cover (number) of staghorn coral in the Puerto Rico EEZ every three years. Only a portion of this contact is likely to affect sexual reproduction by causing breakage or mortality of large, fecund colonies, because colonies of this size are rare. Since smaller, immature colonies are numerically more abundant, it is more likely that trap effects will occur to these smaller colonies. Since trap impacts are likely to occur to non-sexually mature colonies, no reduction in sexual reproduction is anticipated. Additionally, each colony, whether sexually mature or not, has the capacity to reproduce asexually. While trap impacts may affect the asexual reproductive capacity of an affected colony, those impacts are unlikely to affect the asexual reproductive capacity of the population as a whole.

Whether the estimated reductions in numbers and the possible reduction reproduction of the 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. Staghorn corals occur throughout the Caribbean basin and the U.S. Caribbean accounts for a very small portion of the total area of staghorn coral. Since the potential area affected only occurs in a small portion of the U.S. Caribbean, the total area of staghorn coral affected by the proposed action will likely be incredibly small relative to the total area currently in existence. We believe the potential loss of up to 98 ft^2 over consecutive 3-year periods will not have any measurable effect on overall population and is not likely to reduce the species likelihood of survival in the wild.

The following analysis considers the effects of the anticipated loss of areal coverage on the likelihood of recovery in the wild. Recovery plans delineate actions that the available information indicates are necessary for the conservation and survival of listed species. Actions deemed necessary for the conservation and survival of the species are developed after considering the threats and causal listing factors. A recovery plan for *Acropora* corals (including staghorn corals) is not yet available; though a list of threats and causal listing factors exists (Table 7.2.1). We can compare the proposed action to this list, to get a sense of how all fishing (classified as anthropogenic abrasion and breakage, below) ranks as a stressor to these species. Diseases, temperature-induced bleaching, and physical damage from hurricanes are deemed to be the greatest threats to staghorn corals' survival and recovery (*Acropora* BRT 2005). These major threats are persistent, severe, unpredictable, likely to increase in the foreseeable future, and, at current levels of knowledge, unmanageable. Anthropogenic abrasion and breakage is currently considered a moderate threat to staghorn corals, and is likely less of a threat with protective regulations in place. The continued authorization of the Caribbean spiny lobster fishery represents only a tiny fraction of all fishing operations that may be affecting *Acropora* throughout its range, and the threat from fishing in general represents only a portion of the larger anthropogenic abrasion and breakage threat category. Additionally, we concluded the continued authorization of the Caribbean spiny lobster fishery is not likely to reduce the chances of staghorn coral's survival in the wild. Therefore, we do not

believe the continued authorization of the Caribbean spiny lobster fishery will appreciably reduce the likelihood of staghorn coral's recovery in the wild.

Table 7.2.1 Rank of Staghorn Stressors and Their Severity

Stressor	Elkhorn Coral		Staghorn Coral	
	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 12 green 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 lethal take of up to 12 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 12 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). Data in that review also stated that the annual nesting female population in the Atlantic basin ranges from 29,243-50,539 individuals (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).

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 12 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 spiny lobster 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 for the endangered Florida breeding population;
    - 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 12 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. Additionally, our estimate of future take is based on our belief that the same level of take 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 12 hawksbill 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 lethal take of up to 12 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 12 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 12 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 nine 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 lethal take of up to nine 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 nine 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 the 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 size and trend. 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).[42]

Although the up to nine 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 this 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 nine 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 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.

[42] 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).

- 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 to a minimum of 469-882 nests recorded each year between 2000 and 2005. The annual intrinsic rate of population increase[43] 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 nests 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).
- 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 nine leatherback sea turtles over consecutive 3-year periods will result in a 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, 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' recovery in the wild.

[43] 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.

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 *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.

Critical Habitat for Acropora
Our analyses of the impacts to *Acropora* critical habitat analyzed the effects from Caribbean spiny lobster fishing. Based on these analyses NMFS determined the proposed action is not likely to destroy or adversely modify *Acropora* critical habitat in the U.S. Caribbean.

Staghorn Coral
Our staghorn coral analysis focused on the effects from Caribbean spiny lobster fishing on listed staghorn coral. Based on these analyses NMFS determined the Caribbean spiny lobster fishery is not likely to jeopardize the continued existence of staghorn corals.

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 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 SLFMP 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.

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 NMFS' Office of Protected Resources should a take of a listed marine mammal occur.

9.1 Anticipated Amount or Extent of Incidental Take

The effects to sea turtles and staghorn coral were estimated previously. NMFS anticipates the following incidental takes may occur in the future as a result of the continued operation of the Caribbean spiny lobster fishery. As noted in Section 5.8, incidental take for staghorn coral tissue that is likely to die following contact with spiny lobster fishing gear is issued as an area because of the species' unique morphology, and because of the accepted practice of monitoring coral species using areal parameters.

Table 9.1 3-Year Authorized Incidental Take

Marine Turtles	**Number of Takes**	
	Lethal	Total
Green	12	12
Hawksbill	12	12
Leatherback	9	9
Coral	**Area Lethally Affected**	
Staghorn Coral *(Acropora cervicornis)*	93 ft^2	

9.2 Effect of the Take

NMFS has determined the level of anticipated take specified in Section 9.1 is not likely to jeopardize the continued existence of green, hawksbill, or leatherback sea turtles, or staghorn corals.

9.3 Reasonable and Prudent Measures (RPMs)

Section 7(b)(4) of the ESA requires NMFS to issue, for 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 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 5.7.4, we described how trap gear can adversely affect sea turtles via entanglement, and/or forced submergence. In Section 5.7.5, we described how moving spiny lobster vessels are also likely to adversely affect sea turtles via collision impacts or propeller wounds. Most sea turtles released after capture have experienced some degree of physiological injury from forced submergence and/or abrasions/lacerations caused by entanglement. Experience with other fisheries has shown that the ultimate severity of interactions with fishing gear 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 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. Monitoring the Frequency, Magnitude, and Impact of Incidental Take

The jeopardy analyses for sea turtles and staghorn coral 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 with respect to the amount of spiny lobster fishing likely occurring in federal waters. If our estimates regarding the frequency and magnitude of incidental take by the federal spiny lobster fishery prove to be an underestimate, we risk having misjudged the potential adverse effects to these species. Thus, it is imperative that we monitor and track the level of take occurring specific to the spiny lobster fishery. Therefore, NMFS must ensure that monitoring and reporting related to sea turtle and staghorn coral take and effects associated with the proposed action: (1) detect any adverse effects resulting from the Caribbean spiny lobster 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.

3. Reducing the Frequency of Trap Damage to Corals and Sea Turtle Entanglements

The proposed action is predicted to result in physical damage to corals via trap use and likely entanglements with sea turtles due to trap lines. Our effects analysis describes how the use of traps can cause entanglements, damage and kill colonies, and temporarily preclude new settlement of some coral 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.

9.4 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 DPNR and Puerto Rico DNER to distribute information to spiny lobster 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.

2. Currently no information is collected on recreational spiny lobster fishing in the U.S. Caribbean. NMFS must work with CFMC, Puerto Rico, and the USVI, to ensure at least some amount of information is collected on spiny lobster recreational fishing through MRIP.

3. 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. NMFS should also remind commercial and recreational fishermen that the incidental capture of a sea turtle and reporting that capture is not illegal so long as they are complying with all applicable fishing regulations.

4. 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.

5. 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).

6. NMFS, in collaboration with PR DNER 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).

7. NMFS must ensure that sufficient information is collected to determine the potential impacts from spiny lobster fishing. The project entitled "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, could be a mechanism for collecting such data.

The following terms and condition implement RPM No. 3.

8. NMFS must assist fishermen and the territorial government with efforts already underway in the USVI to control fishing effort through a trap certificate program. NMFS must also assist fishermen and the territorial government to consider whether a similar effort is appropriate and feasible in Puerto Rico.

9. The spiny lobster fisheries in the U.S. Caribbean are most likely to occur in commonwealth/territorial waters. As such, the greatest conservation value to *Acropora* and sea turtles will come from minimizing adverse impacts from spiny lobster trap fishing occurring in commonwealth/territorial waters. Therefore, NMFS must work with the Commonwealth and Territories to try and develop changes in those fisheries that reduce impacts to ESA-listed species. Specifically, NMFS should encourage the Commonwealth and Territories to pursue an ESA section 10(a)(1)(B) Incidental Take Permit and develop a Conservation Plan for their spiny lobster fisheries.

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 spiny lobster 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 spiny lobster 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, 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 spiny lobster fishery.

Both Sea Turtles and *Acropora*:

7. 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.
8. 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.

9. 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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