Endangered Species Act - Section 7 Consultation
Biological Opinion

Action Agency:	U.S. Army Corps of Engineers, Jacksonville District
Activity:	Construction and Operation of the Amalago Bay Resort and Residential Community, St. Croix, U.S. Virgin Islands (Project number: SAJ-2007-06364)
Consultation Number:	SERO-2011-00004/SER-2013-12822
Consulting Agency:	National Oceanic and Atmospheric Administration (NOAA), National Marine Fisheries Service (NMFS), Southeast Regional Office, Protected Resources Division, St. Petersburg, Florida Consultation Number: SERO-2011-00004/SER-2013-12822
Approved By:	CRABTREE.ROY.E.DR.1365849559 Digitally signed by CRABTREE.ROY.E.DR.1365849559 Date: 2020.07.08 12:22:33 -04'00' Roy E. Crabtree, Ph.D., Regional Administrator NMFS, Southeast Regional Office St. Petersburg, Florida
Date:	07/08/2020

Table of Contents:

LIST OF FIGURES

LIST OF TABLES

List of Acronyms

ATONS – Aids to Navigation
BA – Biological Assessment
BMP – Best Management Practice
BUIRNM - Buck Island Reef National Monument
CCL – Curved carapace length
CFMC – Caribbean Fishery Management Council
CISS – Cast-in-Steel Shell
CO_2 – Carbon dioxide
CRCP – Coral Reef Conservation Program
DAM – Destruction or Adverse Modification
DDT – Dichlorodiphenyltrichloroethane
DEP – Division of Environmental Protection
DPNR – Department of Planning and Natural Resources
DPS – Distinct Population Segment
DWH – Deepwater Horizon
EAR – Environmental Assessment Report
EEZ – Exclusive Economic Zone
EPA – Environmental Protection Agency
ERA – Economics Research Associates
ESA – Endangered Species Act
FMP – Fishery Management Plan
FP – Fibropapillomatosis
FWCC – Florida Fish and Wildlife Conservation Commission
GHG – Greenhouse gas
HEA – Habitat Equivalency Analysis
HMS – Highly Migratory Species
HWG – Horsley Whitten Group
IMA – Institute of Marine Affairs
ITS – Incidental Take Statement
IUCN – International Union for the Conservation of Nature

JPA – Joint Permit Application
LED – Light-emitting Diode
MMPA – Marine Mammal Protection Act
MPA – Marine Protected Area
NA DPS – North American DPS
NCRMP – NOAA National Coral Reef Monitoring Program
NEFSC – Northeast Fishery Science Center
NMFS – National Marine Fisheries Service
NOAA – National Oceanic and Atmospheric Administration
NOS – National Ocean Service
NRC – National Research Council
NTU - Nephelometric Turbidity Units
ORV – Off-Road Vehicle
PCB – Polychlorinated Biphenol
PFC – Perfluorinated chemicals
PIT – Passive integrated transponder (PIT)
PRDNER – Puerto Rico Department of Natural and Environmental Resources
RC – Restoration Center
RMS – Root Mean Square
RPA – Reasonable and Prudent Alternative
RPM – Reasonable and Prudent Measure
SA DPS – South American
SE – Standard Error
SEFSC – Southeast Fishery Science Center
SEL – Sound Exposure Level
SPL – Single-strike Peak Level
STAR – Sea Turtle Assistance and Rescue
SWPPP – Storm Water Pollution Prevention Plan
TCRMP – Territorial Coral Reef Monitoring Program
TED – Turtle Exclusion Device
TEWG – Turtle Expert Working Group
TNC – The Nature Conservancy
TSS - Total Suspended Solids
UPRM – University of Puerto Rico, Mayagüez Campus
USACE – U.S. Army Corps of Engineers
USCG – U.S. Coast Guard
USFWS – U.S. Fish and Wildlife Service
USNPS – U.S. National Park Service
USVI – U.S. Virgin Islands
UVI – University of the Virgin Islands
VI – Virgin Islands
WIMARCS – West Indies Marine Animal Rescue and Conservation Service

Units of Measurement

°F	degrees Fahrenheit
°C	degrees Celsius
Ac	acre(s)
cm	centimeter
cm^2	square centimeters
ft	foot/feet
ft^2	square feet
in	inch
in^2	square inch
lb	pound
m	meter
mm	millimeter
m^2	square meter
mi^2	square mile
kg	kilogram
km^2	square kilometer
yd^3	cubic yard(s)
%	percent

INTRODUCTION

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

Consultation is required when a federal action agency determines that a proposed action "may affect" listed species or designated critical habitat. Consultation is concluded after NMFS determines that the action is not likely to adversely affect listed species or critical habitat or issues a Biological Opinion ("Opinion") that identifies whether a proposed action is likely to jeopardize the continued existence of a listed species, or destroy or adversely modify critical habitat. The Opinion states the amount or extent of incidental take of the listed species that may occur, develops measures (i.e., reasonable and prudent measures - RPMs) to reduce the effect of take, and recommends conservation measures to further the recovery of the species. Notably, no incidental destruction or adverse modification (DAM) of designated critical habitat can be authorized, and thus there are no RPMs—only reasonable and prudent alternatives (RPAs) that must avoid destruction or adverse modification. RPAs are also developed if the Opinion finds that the action is likely to jeopardize the continued existence of a listed species, or destroy or adversely modify designated critical habitat.

This document represents NMFS's Opinion based on our review of the impacts associated with the construction and operation of the Amalago Bay Resort and Residential Community located on the west end of St. Croix, U.S. Virgin Islands by William and Punch LLC with a federal permit from the U.S. Army Corps of Engineers (USACE). This Opinion analyzes the project's effects on threatened and endangered species and designated critical habitat in accordance with Section 7 of the ESA. We base our Opinion on project information provided by the USACE, William and Punch LLC and its consultants, and other sources of information, including the published literature cited herein.

1 CONSULTATION HISTORY

The consultation history for this project is as follows:

- A public notice for the William and Punch project was issued by the U.S. Army Corps of Engineers, Jacksonville District (USACE) on March 4, 2008. The project described in the original public notice consisted of a casino resort, an inland marina with 64 slips for boats ranging from 40-100 feet (ft) in length, an 18-hole golf course, residential development, beach restoration and enhancement, and other related amenities. In order to construct the marina, dredging of 29,400 cubic yards (yd^3) of materials from a 199,000 square foot (ft^2) area of marine bottom and excavation of 245,000 yd^3 of material from a 392,000 ft^2 area of uplands was proposed. The jetty construction would require depositing 33,000 yd^3 of fill over 133,000 ft^2 of marine bottom. In order to renourish and expand the existing beach, deposit of 20,000 yd^3 of sand over a 280,370 ft^2 area of nearshore hard bottom was proposed. The majority of the streams on the site would be filled as part of upland construction of the golf course, residential complexes, and associated amenities.

- NMFS sent an email response on March 17, 2008, stating our concerns regarding impacts to wetlands, streams, colonized hard bottom, coral reefs, and seagrass, as well as ESA-listed sea turtles and corals, and their habitat.
- An interagency site inspection was conducted April 29-30, 2008, with NMFS, USACE, and USFWS and representatives of the development and investment companies for the project, as well as project consultants.
- NMFS sent the USACE a letter dated July 28, 2008, as a follow-up to the site inspection regarding the need to consider potential impacts of the project to coral critical habitat, whales and sea turtles (a green and a hawksbill sea turtle were observed in the water) in the waters in the project area, seagrass beds, and listed corals (an elkhorn recruit was observed near Sprat Hole north of the proposed marina).
- By letter dated December 21, 2009, the applicant submitted a Joint Permit Application (JPA) to the USACE with copy to NMFS that included an Environmental Assessment, benthic surveys, a design report, and proposed mitigation.
- On June 2, 2010, the applicants presented the project at a USACE interagency meeting and the agencies discussed their concerns with the project.
- Due to changes in the project design, including modifications to the jetties for the marina channels and some modifications to the upland development, the USACE issued a new public notice for the Amalago Bay project on February 16, 2011. The revisions to the project described in the new public notice included an increase to a 70-slip inland marina; a reduction in the amount of material to be dredged to 19,950 yd^3 combined with a reduction in the amount of material to be excavated from uplands to 222,800 yd^3 to create the inland marina basin and navigation channel; a decrease in the amount of sand to be placed in nearshore waters to 13,100 yd^3 to renourish and expand the beach, although the footprint would still extend over a 280,370-ft^2 area; a change in the orientation of the northern jetties to reduce impacts to hard bottom; a reduction in the seaward limits and rock volumes of the jetties; and an expansion in buffer zones to protect streams and existing wetlands within the upland development footprint.
- By letter dated February 24, 2011, the applicant sent NMFS a copy of the revised JPA.
- NMFS sent a response to the new public notice on March 28, 2011.
- Moffatt & Nichols, one of the project consultants, sent a letter dated June 16, 2011, in response to NMFS's March 28, 2011, letter. The Moffatt & Nichols letter included a draft of a Biological Assessment (BA).
- On July 12, 2011, NMFS participated in a conference call with the applicant and consultants to discuss information that still needed to be included in the BA based on our review of the draft document.
- NMFS sent a letter to the USACE dated September 1, 2011, regarding information that still needed to be included in the BA.

- By letter dated September 21, 2011, the USACE requested consultation with NMFS for the proposed construction and operation of the Amalago Bay project. The USACE determined that the proposed project may affect, but is not likely to adversely affect, the endangered hawksbill and leatherback sea turtles, and the threatened green sea turtle; and may affect, but is not likely to adversely affect, threatened elkhorn and staghorn corals and their designated critical habitat. However, NMFS was unable to concur with the USACE's determination that the project may affect, but is not likely to adversely affect, ESA-listed hawksbill, leatherback, or green sea turtles, elkhorn and staghorn coals or ESA-designated coral critical habitat.
- NMFS initiated formal consultation on December 13, 2011, after a site inspection with project consultants to obtain information regarding the extent of coral critical habitat in the area to be impacted by the project.
- By email dated January 11, 2012, NMFS informed the USACE that consultation would be formal and NMFS sent a letter dated February 10, 2012, to the USACE as a follow-up to the email message.
- Due to NMFS's receiving 2009 sea turtle nesting data from the Virgin Islands Department of Planning and Natural Resources (DPNR) Division of Fish and Wildlife (DFW) in June 2012, USFWS (via letter dated July 6, 2012) requested that Section 7 consultation between USACE and USFWS be reinitiated. The USACE decided to request that the applicant revise the BA for the project and informed NMFS of this decision via email dated August 12, 2012. At the USACE's request, NMFS provided the USACE with a request for additional information to be included in the supplemental BA via email dated August 16, 2012.
- USACE sent a letter dated August 31, 2012, to the applicant requesting the BA be revised and detailing the information to be included in the revised BA. Via email dated September 26, 2012, the USACE informed the agencies that the applicant received the request on September 11, 2012.
- The USACE received a response to their August 31, 2012, request from the applicant on October 25, 2012, and provided hard and electronic copies to NMFS and USFWS.
- An interagency meeting was held at the USACE offices in San Juan on January 16, 2013, to discuss issues related to the ESA consultations with the Services and the required modifications to the BA necessary to address these issues.
- The USACE sent an information request to the applicant on February 7, 2013, detailing the required modifications to the project BA.
- The USACE sent NMFS a new consultation request with the modified BA for the project by letter dated August 19, 2013.
- NMFS notified the USACE by letter dated December 20, 2013, that we would begin drafting our Opinion, but needed some additional information in order to complete it.
- NMFS received the additional information by letter dated May 14, 2014.

- On September 10, 2014, NMFS published a final rule, listing 5 additional Caribbean coral species as threatened. Consideration of effects to these species was added to the consultation.
- NMFS issued a draft Opinion with a finding of DAM on June 16, 2015, and requested that the USACE and applicant work with NMFS to develop an RPA for the project.
- NMFS participated in a July 14, 2015, conference call organized by the USACE to discuss the development of an RPA and request possible assistance from the EPA who also participated in the call.
- The USACE sent a letter with comments from the applicant regarding the draft Opinion via email dated January 20, 2016. NMFS sent a response letter dated March 1, 2016, to the USACE reminding them that we requested assistance from the USACE and applicant in developing an RPA and had not received anything toward that end to date.
- NMFS requested a copy of the applicant's July 28, 2015, response to the USACE regarding the draft Opnion referred to in the USACE's January 20, 2016, letter via email dated March 18 and received a response with the information from the USACE on March 22, 2016.
- NMFS participated in an in-person meeting with the applicant and its consultants, the USACE, and representatives of the Virgin Islands government on March 29, 2016.
- NMFS provided comments to the USACE regarding the notes from the March 29, 2016, meeting via emails on April 28 and 29, 2016. The USACE sent an email summarizing the meeting and information to be provided by the applicant in order for NMFS to draft an RPA on May 5, 2016.
- The USACE sent NMFS an email notice on May 31, 2016, indicating that the applicant had sent a response to our request for information from the March meeting and NMFS downloaded the files on June 6, 2016.
- NMFS contracted the Horsley Witten Group (HWG) to review the information from the applicant related to erosion and sediment control measures in September 2016 and received comments from HWG on October 8, 2016.
- NMFS notified the USACE via email November 4, 2016, that NMFS was completing the analysis of the applicant's May 2016 response and did not anticipate changes to our DAM determination. NMFS sent the results of the HWG's analysis of the erosion and sediment control plans to the USACE and the applicant via email November 21, 2016, and indicated that we would be developing an RPA.
- NMFS issued a draft Opinion on August 17, 2018, pursuant to Section 7(a)(2) of the ESA for the proposed action by USACE to permit the activity. NMFS concluded that the proposed action was likely to result in DAM of designated critical habitat for elkhorn corals. NMFS's draft Opinion included a RPA designed to avoid DAM.
- On March 12, 2019, the applicant submitted a response to USACE, which indicated that they had modified and updated their proposed project, and submitted the following updated documents: 1) CDR Maguire Response (dated October 2017); 2) Hydrologic Impact Assessment (dated August 2018); 3) Stormwater Pollution Prevention Plan

(SWPPP, dated August 2018); 4) Permit Plans (dated December 2018); 5) Stormwater Pre-Development Map (dated June 2018); 6) Post-Development Map (dated June 2018); and 7) Sections from the 2013 BA. USACE and NMFS reviewed the documents and agreed that the applicant did not address NMFS' concerns by submitting new information, or modifying the project in accordance with our draft Opinion and RPA, or suggest modifications that would change our finding. NMFS email dated May 6, 2019, presented examples of how the applicant's submittal did not significantly modify the project and failed to address the RPA.

- As a result, USACE requested the applicant to specifically state whether they would adopt the the 3 major elements in NMFS's RPA. On May31, 2019, the applicant submitted a response to USACE, which indicated the degree to which they would comply with the RPA. The applicant did not agree to implement the mandatory and additional elements of RPA Element 1. The applicant agreed to implement all remaining proposed material elements and additional elements set forth in RPA 1 "to the extent feasible", but the letter did not identify the material elements they would agree to. Regarding RPA Element 2, the applicant committed to implement all mandatory BMPs and elements for consideration "to the extent reasonably feasible". Lastly the applicant remained fully committed to achieving the objective of the performance criteria in RPA Element 3 "to the extent reasonably feasible".
- On June 12, 2019, USACE requested NMFS to finalize the Draft Opinion.

2 DESCRIPTION OF THE PROPOSED ACTION

The proposed development on the 594-acre (ac) site with approximately 2,800 ft of shoreline frontage (Figure 1) includes the construction and/or creation of:

- an inland marina, which requires the relocation of Route 63 and the construction of rock jetties at the marina channel entrance/exit and the mouth of the flushing channel
- an island with 56 hotel villas, pools, an events facility, and renourished and expanded beach
- a 378-room hotel with reception area and conference space, 140 waterfront condominium units, 5 hotel restaurants, a casino, and public beach parking and restrooms
- 66 fractional units (time shares) with a swimming pool and fitness club
- an 18-hole golf course, a clubhouse and restaurant associated with the golf course, 102 residential golf villas, a golf course grounds building, and a golf villa swimming pool and restrooms
- 144 subdivision lots
- a health club and spa, 17 residential spa villas, and retail stores

- 2 gatehouses, 2 tennis courts and a restroom, a service yard and maintenance building, 4 potable water storage tanks, three 1.5 megawatt emergency generators and fuel storage tank, and 1,638 parking spaces

Based on the March 4, 2008, public notice for the William and Punch project, the revised public notice for the renamed Amalago Bay project dated February 16, 2011, and the August 13, 2013, BA, the final proposed casino resort residential project includes a 70-slip inland marina for boats ranging from 40-100 ft in length; the dredging of 19,950 yd^3 of material from an 89,100 ft^2 area of marine bottom; the excavation of 222,800 yd^3 to create the inland marina basin and navigation channel (from a 384,500 ft^2 area); the construction of an expanded beach (extending over a 139,624 ft^2 area, of which 89,090 ft^2 is below mean high water) through the placement of 13,100 yd^3 of sand in nearshore waters; the construction of rock jetties at the marina entrance and flushing channel requiring 57,245 ft^2 of area seaward of mean high water and rock volumes of 17,900 yd^3 seaward of mean high water; an 18-hole golf course; a residential development, and other related amenities. The majority of the streams on the site would be filled as part of upland construction of the golf course, residential complexes, and associated amenities. Mangrove areas at the mouths of 2 of the 3 natural drainages to the sea will be eliminated. Buffer zones of at least 15 ft will be established around portions of those streams that would not be eliminated as part of the construction, and a 50-ft buffer will be established around an existing 7.61-ac mangrove wetland.

The various documents describing the proposed action focus primarily on the coastal and marine components of the project, but the August 2013 BA does contain some information regarding upland construction that is contemplated as part of the Amalago Bay project. The overall concept and design of the project has not changed since it was originally proposed. It remains a casino resort with an artificially-created island, beach, and inland marina; an 18-hole golf course; and residential development along the coast and on uplands throughout the site (Figure 2).

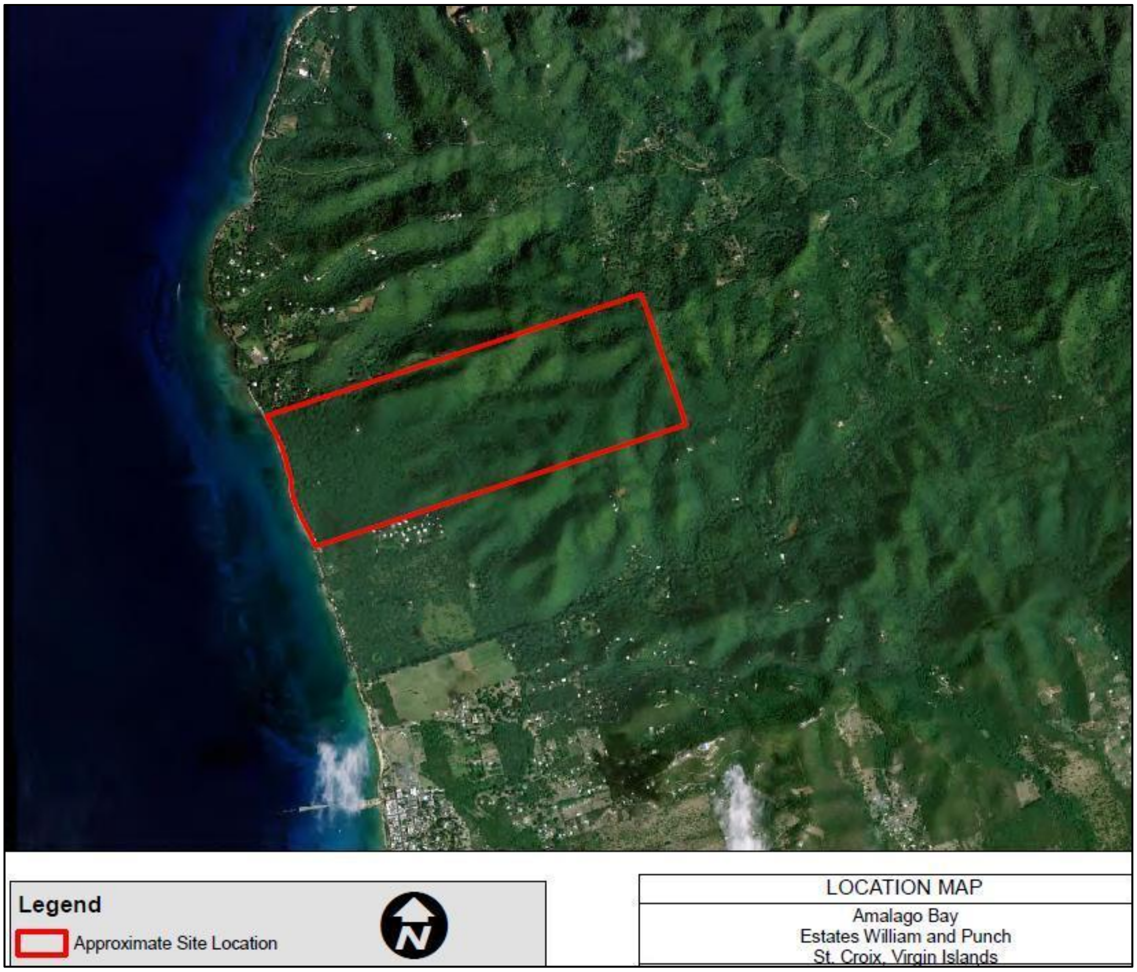

Figure 1. Approximate site location of project on the west end of St. Croix (Dial Cordy & Associates 2013)

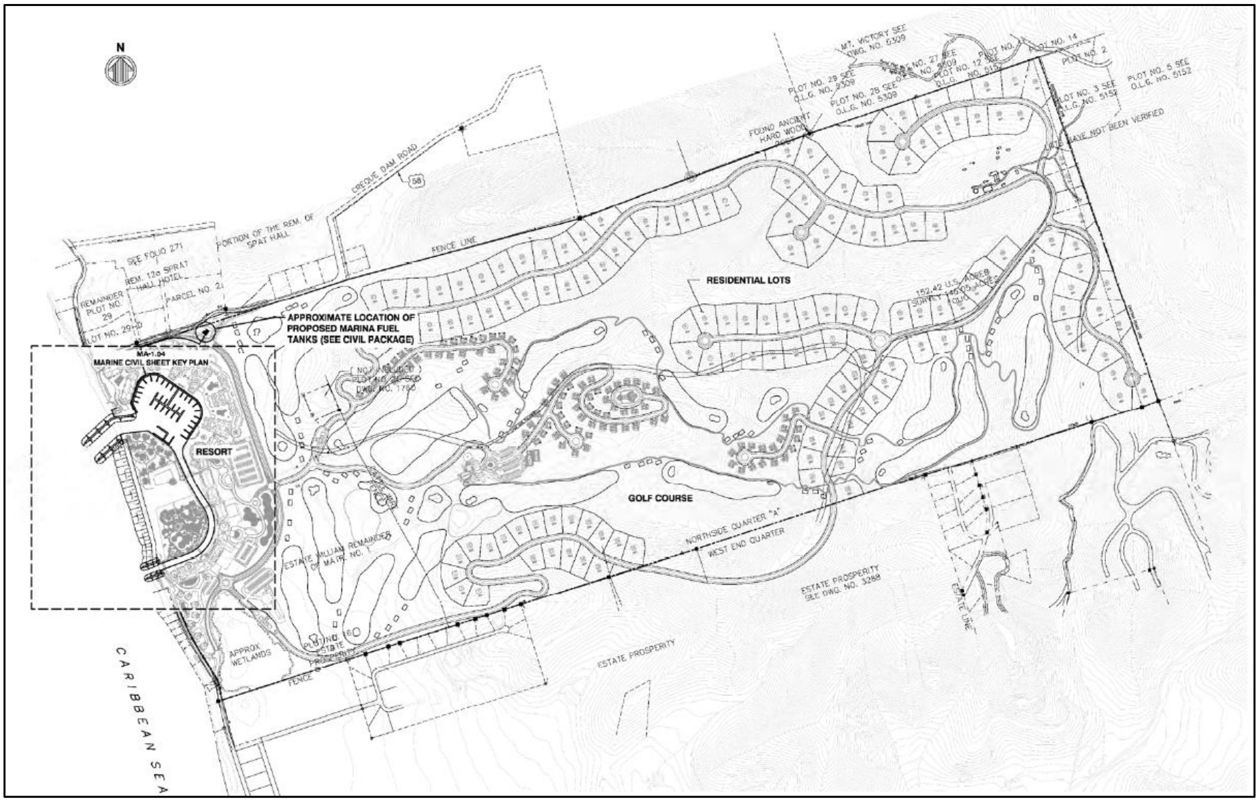

Figure 2. Site plan for entire Amalago Bay development (Dial Cordy & Associates 2013)

Marina Facilities and Created Beach

In order to construct the marina facilities, heavy equipment that is currently not available on St. Croix will be brought in via barge and/or ships. The majority of this equipment will be unloaded at Virgin Islands Port Authority facilities at Molasses Pier on the south coast of St. Croix and then transported overland to the project site. However, the construction of the jetties and dredging of the marina entrance channel may require the use of a high-capacity crane. Due to its size, this crane would have to be transported by barge to the project site. For this reason, the applicant is now proposing the construction of a temporary access trestle. This trestle will be constructed of steel pipe piles with a steel substructure and wood deck planks. The access trestle will measure 290-ft long by 40-ft wide and will be located within the footprints of the proposed marina entrance channel and jetties. The trestle will have approximately 45 steel piles spaced 20 ft apart. The piles will be vibrated in and then set-checked with an impact hammer. The applicant proposes the installation of a double row of turbidity curtains or the use of a bubble curtain around the area where pile driving will take place, as well as dry-firing of the impact hammer and reduction of the hammer energy in order to minimize noise. The trestle would be removed at the end of the marine construction activities. The trestle would allow the unloading of a crane from a deck barge. The deck barge will measure approximately 270-ft long by 70-ft wide and have an 8-ft draft. During unloading of the crane and other equipment, it may be necessary to anchor the deck barge using spuds.

A 70-slip inland marina will be created by dredging 19,950 yd^3 from an 89,100 ft^2 area (see Figure 3). The marina entrance channel and flushing channel will be dredged using a hydraulic dredge from the shoreline seaward. All flushing channel dredging and construction will be done from land, including using a clamshell dredge from a crane. The dredged material that is found to be consistent with the material to be used to create the beach between the jetties will be used as beach fill. The rest of the excavated material will be used as land fill in upland areas of the project. A floating turbidity barrier will be placed around the dredge during operation. Movement of the dredge barge will likely occur with a self-propelled motor, but work boats may also be used to position the barge. The barge will anchor using spuds, if needed. The spuds will be located in a way that avoids impacts to existing patch reefs or coral outcroppings. A return flow pipeline will be routed from the dredge to the shoreline along an east-west alignment south of the proposed south entrance jetty

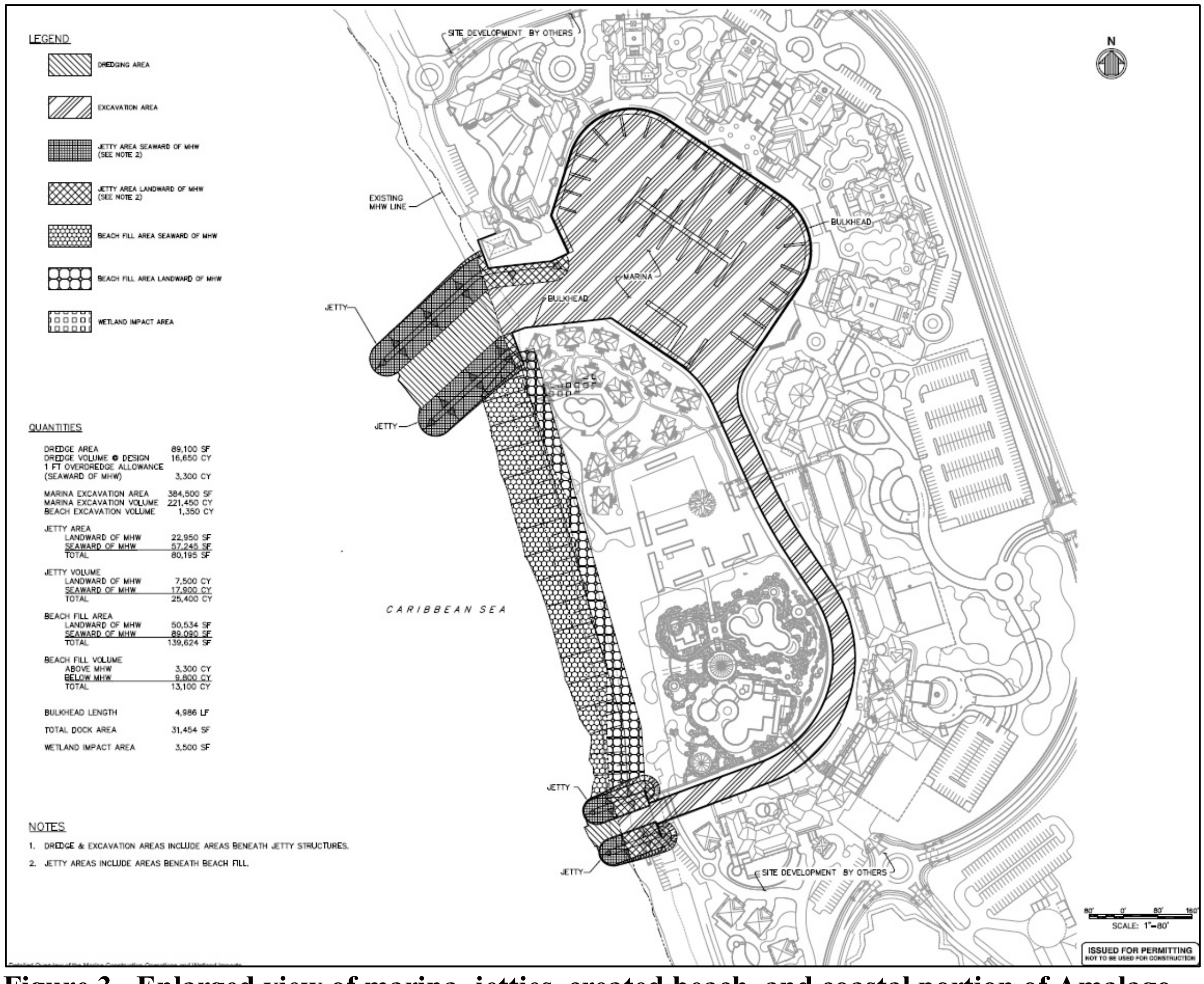

Figure 3. Enlarged view of marina, jetties, created beach, and coastal portion of Amalago Bay development (Dial Cordy & Associates 2013).

Turbidity barriers will also be installed around bulk construction materials and equipment to reduce the potential for transport of materials to nearshore waters during staging and construction. A temporary settling basin will be constructed on the beach to serve as the receptacle for the outflow of the dredge pipe such that dredged materials deposit on the beach. Material excavated from above the groundwater table on the beach as part of construction activities related to the creation of an inland marina, island, and beach and resort facilities will be placed on the new beach and graded. Material excavated below the groundwater table will be hauled to the temporary settling basin to be used as the dredge spoil pit in water-tight vehicles. Grading equipment will be used to move material to the desired beach template. Turbidity monitoring will be conducted during dredging and dredge material disposal activities. Sand will be placed on uplands and in nearshore waters to create a beach between the 2 jetties. Approximately 13,100 yd^3 of sand will be placed on the beach extending over a 139,624 ft^2 area of which 89,090 ft^2 is below mean high water.

Breakwater construction will take place once dredging is complete in order to maximize breakwater stability. Approximately 57,245 ft^2 of the total 80,195 ft^2 area of the jetties is seaward of mean high water. The construction of the jetties requires 25,400 yd^3 of rock, of which 17,900 yd^3 is seaward of mean high water (see Figure 3). The entrance and flushing channel jetties will be constructed from land using heavy equipment to construct the core, underlayer, and side slope armor of the jetties up to the offshore termini. The structure crest of the jetties was designed to be wide enough to accommodate land-based equipment. Floating turbidity curtains will be placed around the entrance jetties during construction to minimize the transport of sediments and construction materials outside the construction footprint.

Approximately 222,800 yd^3 of material will be excavated from a 384,500 ft^2 upland area to create the inland marina basin and navigation channel and an additional 19,950 yd^3 will be dredged from an 89,100 ft^2 in-water area to extend the navigation channel and control depth seaward (see Figure 3). Construction of the marina basin will be performed in the dry to the extent possible, meaning a plug of land will remain between the excavated inland basin and the sea for as long as possible. The sheet pile marina bulkhead and associated anchoring system will be constructed around the perimeter of the marina basin and flushing channel prior to excavation to support the side walls of the basin and minimize the potential for sloughing. Excavation of the inland marina basin will begin following the construction of the bulkhead. Material above the groundwater table will be deposited into a 6.8-ac marina excavation stockpile area. Material below the groundwater table will be hauled in watertight dump trucks to the 11.3-ac stockpile area. This area will have a smaller area within it (confined by earthen dikes constructed using material from the smaller stockpile area) in order to contain dredge spoil and material from below the groundwater table and allow it to de-water via natural percolation on the beach. Excavated materials transported to these 2 areas will be used as beach or upland fill, depending on the suitability of the material. Three temporary plugs will be left in place during the excavation of the inland marina basin: (1) at the entrance channel shoreline, (2) at the flushing channel shoreline, and (3) at the start of the flushing channel adjacent to the marina basin. The temporary plugs will have a minimum 25-ft crest and 1V:4H side slopes to enable the passage of equipment. Temporary slope protection will be placed on the side slopes to minimize sloughing or erosion. The middle plug will be used to separate the marina basin from the flushing channel, which will be used as a de-watering basin during construction of the marina (for pumping of water from the inland marina basin to the channel and vice versa in order to complete all excavation). Once excavation is complete for the marina basin and flushing channel, the de-watering pumps will be removed and water will be allowed to slowly enter the basin. Once the water level inside the basin has stabilized with that of the sea outside the basin, the construction of the connecting segments of the entrance and flushing channels will be completed. Turbidity barriers will be located seaward of the entrances to contain sediment from the excavation of the plugs.

A portion of the marina channels and jetties and the created beach area are located over ESA-designated critical habitat for elkhorn and staghorn corals, as well as seagrass areas dominated by *Halodule beaudettei*. According to information in the June 2013 BA, based on benthic maps created from benthic surveys of the project area, 1.69 ac of hard bottom in waters greater than 10 ft deep and 1.06 ac of nearshore hard grounds will be impacted by the construction of the marina channels and jetties and the beach (for a total of 2.75 ac). Of this area, the BA states that 1.04 ac of hard bottom and 0.46 acre of nearshore habitat contain the essential feature of coral critical habitat (for a total of 1.5 ac). The essential feature of critical habitat for elkhorn and staghorn corals 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 hard bottom or dead coral skeletons free from fleshy macroalgae and sediment cover. Based on observations by NMFS's biologists during site inspections, we believe that 2.69 acres of the 2.75-acre hard bottom area to be removed by shoreline and in-water construction contain the essential feature of acroporid coral critical habitat. Only 0.06 ac of nearshore hard bottom habitat located where each of the 3 natural drainages on the Amalago Bay property currently enter the sea does not contain the essential feature of acroporid coral critical habitat due to the high sediment load from the terrestrial discharge to these areas and the dominance of macroalgal growth at these locations.

The applicant is also proposing the construction of artificial limestone reefs within the nearshore areas seaward of the beach using native limestone. Ten patch reefs measuring approximately 100 ft by 60 ft with a total height of 3 ft are proposed. These structures will have a pyramidal shape and would occupy 1.45 ac of marine bottom. Boulders with a median size of approximately 2.3 ft in diameter and a weight of approximately 1,762 pounds are proposed as the construction material for the structures. The applicant proposes the relocation of corals measuring 7 centimeters (cm) or more in diameter from the in-water construction area to these offshore breakwaters. Based on information in the benthic studies prepared for the project, a total of 382 coral colonies, including colonies of corals listed as threatened under the ESA, would be directly impacted by the marine construction. The BA does not specify how many of these colonies would be relocated outside the in-water construction footprint. The construction of these structures is meant to serve as compensation under Section 404 of the Clean Water Act for the loss of 2.75 ac of hard bottom as a result of the construction of the Amalago Bay project.

Based on information in the June 2013 BA and supplemental information provided by letter dated May 14, 2014, conservation measures that have been incorporated in the design of the marine facilities and the creation of the beach between the jetties intended to minimize potential impacts to ESA-listed species and their habitat include:

Marina and Jetty Construction

1. The marina basin will be excavated in the dry, meaning a plug of land will be left at the mouth of each of the channels (the southern flushing channel and the northern access channel) until the excavation and shoreline stabilization activities are complete.
2. Prior to anchoring the dredge barge, underwater surveys will be conducted to ensure that spuds are not placed on existing patch reefs or coral outcroppings. If patch reefs or coral outcroppings are located in the area, the barge will be repositioned to avoid impacts to reefs and hard bottom.
3. NMFS's *Sea Turtle and Smalltooth Sawfish Construction Conditions* (dated March 23, 2006) and pile driving guidelines based on recommendations from NMFS will be implemented.
4. An education program will be implemented incorporating information regarding the proper operation of vessels in areas containing ESA-listed sea turtles and corals during construction and operation of the project.
5. Sea turtle observers will be on-site daily to monitor the occurrence of sea turtles before, during, and after marine and shoreline construction activities.
6. A biological monitoring program will be implemented to monitor the effects of project construction and operation on the adjacent aquatic ecosystem. A description of this program is in Appendix B of the BA (Dial Cordy & Associates 2013) and includes water quality monitoring for pH, turbidity, total suspended solids, dissolved oxygen, salinity, and temperature; monitoring of beach profiles north and south of the jetties; marine resource monitoring for sediment cover, benthic community, fish, and sea turtles; and beach monitoring for sea turtle nesting.
7. Construction of the jetties will be done from land.
8. The marina's entrance channel will be dredged using a hydraulic dredge plant with a double floating turbidity boom around the dredging area. The dredge plant will begin at the seaward end of the jetties and proceed landward. If sea conditions limit the functional efficiency of the turbidity curtain, dredging operations will be curtailed.
9. Prior to any construction activities, all corals larger than 7 cm in diameter will be relocated to a mitigation site or other appropriate hard bottom habitat.
10. A double row of turbidity curtains or bubble curtains will be placed around the area where piles will be driven as part of the construction of the temporary trestle pier. The impact hammer will be dry-fired (meaning hammer is raised and lowered with no compression of the pistons, which produces a lower-intensity sound than the hammer under full power) and the energy of the hammer will be reduced using bubble curtains or other methods in order to minimize noise from pile driving.
11. A pre-construction survey will be conducted from the shoreline to 500-ft-seaward to estimate the amount of seagrass that will be within the footprint of the in-water construction area. The transplant of seagrass and 3-year monitoring of transplanted seagrass is proposed prior to the start of in-water construction.

Beach Creation

1. Dredged sand from marina construction will be used as part of the beach creation to minimize the need to use off-site material.
2. Sand will be placed along the shoreline template behind constructed berms.
3. Turbidity booms, velocity attenuators, and settlement basins will be used during the creation of the beach between the 2 jetties and during hydraulic dredging (dredged material will be piped to a settlement basin).
4. If sea conditions limit the functional efficiency of the turbidity curtain, dredging operations will be curtailed.
5. The proposed created beach slope from average mean high water to the top of the berm will mimic the existing slope to reduce potential impacts to sea turtle nesting.
6. The existing shoreline vegetation will remain and additional plantings of indigenous species will be done along the created beach.
7. A lighting plan was designed to minimize impacts to nesting sea turtles. The final plan will be coordinated with USFWS.
8. Any beach nourishment activities will be done in January and February to avoid peak sea turtle nesting seasons.
9. For a period of 90 days prior to construction of the channel jetties and beach, the existing beach will be inspected daily for signs of sea turtle nesting. Daily inspections will continue throughout the construction period.
10. Trained observers will be provided and authorized to cease construction activities in the event marine protected species enter the work zone during marine construction.
11. Daily shoreline inspections will be performed to ensure vegetation associated with turtle nests is not disturbed and that no mechanical beach cleaning takes place within 10 ft of any nest.

Facility Operation

1. A beach furniture protocol following the Florida Fish and Wildlife Commission's Marine Turtle Protection Program will be implemented, including the complete hand removal of all beach furniture at night, the placement of beach furniture at least 10 ft from any marked sea turtle nest, no placement of beach furniture on the beach until the daily sea turtle nesting survey has been completed and nests marked, no burial of umbrella poles within 10 ft of any marked nest, and no placement of beach furniture on vegetation or sand dunes.
2. 15 mooring buoys will be installed between Frederiksted and the project site to minimize potential anchoring impacts from vessels associated with the Amalago Bay development.
3. Use of the mooring buoys and marina will be controlled by an on-site harbormaster who will also oversee the boat-related educational plan.

4. The jetties and associated channels will be clearly marked with aids to navigation (ATONS) in coordination with the U.S. Coast Guard (USCG) to minimize the potential for accidental groundings.
5. Vegetation associated with sea turtle nests will not be disturbed and no mechanical beach cleaning will take place within 10 ft of any nest.
6. An on-site sea turtle educational program will be implemented targeting workers, owners, and guests.
7. A post-construction sea turtle monitoring program will be implemented. Nesting data will be summarized on a monthly basis and provided to DPNR, NMFS, and USFWS. An annual report analyzing project impacts to sea turtle nesting will also be provided.
8. Lighting surveys will be constructed pre- and post-construction to identify and correct any potential lighting issues.
9. Dogs will be subject to leash requirements in order to minimize potential predator effects on sea turtle nests.
10. Predator-proof trash receptacles will be installed at all beach access points.

Construction of Roadways, Utilities, Residential and Hotel Units, and Other Amenities
The project will be constructed in 3 phases over a 48-month period. Phase I will consist of the demolition of the majority of the historic buildings and constructed features of the former William Plantation; the relocation of Route 63; and the construction of the marina, the resort hotel and casino, 56 beach villas, 140 waterfront condominium units, retail facilities, subdivision roads A, B, I, J, and K, 18-hole golf course, golf course clubhouse and restaurant, gatehouses, sewer and water lines to Frederiksted, sewerage pumping station, health club and spa, resort swimming pools and restaurant, 3 condo swimming pools, 2 tennis courts, public beach parking and restroom, service yard and maintenance building, 3 potable water storage tanks, 2 1.5-megawatt emergency generators and fuel storage tank, and golf course grounds building all over the course of 24 months. The construction of the waterfront condominiums may extend beyond this 24-month period as it will be dependent on sales.

Phase II will consist of the construction of Roads C, D, E, F, and G, a local restaurant, 66 fractional units and swimming pool, 18 spa villas, 102 golf villas and associated amenities, and 1 potable water tank, all over the course of 18 months.

Phase III will consist of the construction of Road H, subdivision residence lots, and associated amenities, all over the course of 18 months. It is expected that the phases will overlap, which is why a 48-month total construction time is anticipated. The marina basin and flushing channel will be the main stormwater discharge point for the project. The network of ridgelines in the central to eastern sections of the property is intended for the subdivision road system featuring single-family homes overlooking the golf course.

The western portion of the golf course area will be adjacent to the relocated Route 63 on slopes of 5-10% sloping from east to west toward the ocean but without well-defined drainage patterns (Figure 2). The eastern portion of the golf course will be located on a series of hills and ridges and cross well-defined drainage courses between ridges. The defined drainage courses run generally from east to west and then fan out onto the gently sloping areas in the western portion of the site. The driving range and front nine of the course will be on the western slopes to minimize the need for grading. The back nine will wind through hills and valleys on the eastern portion of the site with the golf playing predominantly on ridge tops or from ridge to ridge with golf features such as tees and greens benched into the sides of hills. The holes adjacent to the relocated Route 63 will serve as sedimentation basins during construction and then as stormwater management basins post-construction. The initial clearing of the golf course will be done in a narrow strip along the centerline of the hole to identify whether significant vegetation or terrain features are present within the golf hole safety corridors so that the architect can make adjustments to the hole where possible to minimize impacts to these features. The next phase of clearing will use sight lines from elevated tees across low areas to determine where taller vegetation needs to be trimmed or selectively removed to allow a view from the tees to the fairways or greens. The clearing limits will be marked on the site at this time. Preliminary grading will then take place and the architect will review the grading to make modifications where possible to minimize cuts, fills, and rock excavation, and blend the grading into the surrounding terrain. After rough grading is complete, the detailed golf construction work will take place, including drainage, irrigation, construction of features such as tees, greens, and bunkers, and cart path installation. All disturbed areas outside the areas to be grassed will receive final grading and landscaping. The golf course will also be planted with turf grass developed for golf course applications.

The hotel will be located in the area bordered by the relocated Route 63 and the marina facilities (Figure 3). The main hotel building follows the path of the marina flushing channel. The lower level of the hotel building will house restaurants, bars, and retail shops. The support areas will be buried beneath a grass terrace. The beach villas and the new beach are located on the island that will be created as part of the marina construction. The villa elevations will not exceed that of 2-story structures. A health club and spa will also be located on the created island and hotel spa suites will be on the upper floors of the same building. A pool complex will be located across the flushing channel from the hotel and consists of an adult and a family pool with a lazy river, a water slide, a bar and grill, and a beach bar.

Marina condos will be located adjacent to the marina, casino, fractional units, and re-routed Route 63 (Figure 3). The units will be in 4-story structures with 3 stories of residential units above a story containing retail shops. The units gradually step down to 1 story toward the marina. There are also fractional units proposed for this area in the western portion of the property that will be accessed from the relocated Route 63 (Figure 3). The buildings will be 4 stories with parking on the first story. The buildings will be attached in V- or U-shaped layouts.

There will be 36 ¾-acre lots and 108 1-acre lots. These lots will be located on ridges throughout the project (Figure 2). The owner is proposing development guidelines to minimize disturbance due to home construction, but will not be responsible for construction of any of the single-family homes. There will be 3 groups of golf villas throughout the uplands of the project. The individual units will be terraced on grades stepping down from high points. The villas will be 2-story and step down the hill in clusters. There will also be a golf maintenance facility adjacent to the residential entry along relocated Route 63. The main building consists of storage bays for mechanical equipment and support offices. The building will be single-story with a mezzanine, making it a maximum of 2 stories in height. The golf clubhouse will be located at the western portion of the center cluster of golf villas close to the resort core (Figure 2). The building will be 2-stories and will be surrounded by terraces.

Based on information in the June 2013 BA, conservation measures that have been incorporated in the design of the roadways, utilities, residential and hotel units, and other amenities (e.g., the golf course) intended to minimize potential impacts to ESA-listed species and their designated critical habitat from stormwater runoff and associated transport of sediment include:

1. A 50-ft natural vegetative buffer and exclusion fencing, as applicable, will be established around the 7.61-ac mangrove wetland except in the area of Route 63, where the buffer will be 25 ft.
2. The stormwater management system for the entire project will be designed and constructed to manage the discharge rate and retention time of stormwater into the wetland and to address sediment discharge into the 7.61-ac mangrove wetland.
3. Mangrove shelves will be constructed, planted, and maintained along the island (seaward side) the length of the flushing channel.
4. Bottomless culverts and small-span bridge crossings will be used where the construction of these structures does not affect the functionality of the stormwater management system.
5. Initially, only the roads and service area will be excavated. Road alignment and the location of the golf course and buildings will be adjusted in the field to avoid significant trees.
6. Upon completion of all grading of all roadways and the installation of underground utilities, Route 63 and the subdivision roads to be used during the first phase of construction will be paved. The parking lot at the service area will also be paved and used for storage of construction material. All other roadways will be either paved or sealed using liquid polymer emulsion. All roads to be used for access to a construction area will be paved before starting construction of buildings in that area.
7. Structural footprints will be manually cleared to maximize the preservation of large trees and existing vegetation. All areas where there will be site disturbance will be pre-cleared of brush by hand in phases. All major trees will remain initially. All vegetation will be left a height of 1 ft to allow the existing root system to hold the soil.
8. All areas to be excavated will be laid out by a surveyor prior to excavation.
9. Prior to the start of any earth work, silt fences will be installed with rebar or steel fence posts and wire mesh where needed, following the guidelines in the *VI Environmental Protection Handbook* (University of the Virgin Islands Cooperative Extension Service 2002). Both single and double rows of silt fences will be used throughout the site due to the topography.
10. Flags marking the limits of site disturbance for structures and major earth work will be attached to trees or stakes at a height that is clearly visible to construction equipment operators.

11. All cleared brush will be used as berms and installed upland of the single and double rows of silt fences. Silt fences will be installed downslope from all roadways at the downhill limit of all site work, road, and utility construction.
12. For each phase of the project, additional double metal reinforced silt fences will be placed below each group of buildings being constructed in that phase. As each building is completed, the grading will be stabilized by erosion control blankets or final landscaping will be completed.
13. A 10-ft minimum natural vegetative buffer will be established around ephemeral streams with the intent to maintain an average buffer of 25-ft based on "top of bank" adjacent to ephemeral streams. This assumes a 10 ft buffer to either side of the stream bank and a 5-ft-wide stream.
14. The clearing area around building foundation footprints will be limited to 5 ft outside of the building foundation line. The intent is to excavate the least amount of the site as possible to minimize site disturbance and allow existing vegetation to remain until final landscaping is completed.
15. Site grading (i.e., road construction) will have water or wetting agents applied to minimize potential particulate emissions and erosion.
16. Landscaped areas will be incorporated as bio-retention filters and/or supplemented by engineered stormwater systems.
17. Cut or fill material will be covered with tarpaulins or straw erosion control blankets or treated (e.g., hydroseeding) wherever it is stockpiled.
18. Best management practices (BMPs) will include silt fencing, straw waddles, or downstream turbidity curtains, stabilized construction entrances, bio-retention swales, and immediate upland stabilization by hydroseeding, geo-textiles, or sodding.
19. Natural on-site ponds and additional velocity attenuation/sediment basins will be incorporated within multiple golf course fairway designs.
20. When roads are in a side hill cut/fill situation and stabilized with retaining walls/gabions, roads will be sloped to the downhill side where possible to allow runoff into the bio-retention area between the roadway and the retaining wall. When roads are in a cut/fill situation and do not need retaining walls, the roadway will be sloped to the downhill side to allow the water to sheet flow.
21. Parking areas in the resort portion of the project will drain into landscape areas where possible so as to act as bio-retention filters. Parking areas that cannot be drained to a landscape area will drain to mechanical treatment units.
22. The existing Horse Pond will remain and act as a sediment basin. Once the Horse Pond is one-third full, the sediment will be removed and stockpiled at the maintenance area, covered with tarpaulins, and used in landscaping. A new outlet weir will be constructed at the Horse Pond to better manage the water level and control downstream flow during storm events.
23. Silt fencing and erosion control blankets, liquid polymers, and landscaping will be used on exposed slopes. Erosion control mats will be used on all cut, fill, or regraded slopes

steeper than 3 horizontal to 1 vertical and whenever a slope extends for more than 30 vertical ft without a bench, swale, or other feature to reduce velocity and divert flow.

24. Stabilized construction entrances will be constructed at all points where construction and earth moving equipment enters or exits the site from paved roads. Prior to the start of earth work for the roads and cluster areas, the beginning of the roads into the site will have a 50-ft-long by 15-ft-wide area filled to 9 inches (in) of depth with stone or a pre-fab trackout control unit.
25. Temporary sediment basins will be constructed at multiple locations throughout the project area as construction of various components begins. Large semi-permanent ponds will be constructed on the lower portions of the golf course to act as sediment basins to treat runoff from golf course construction. Holes 4 and 5, adjacent to the relocated Route 63, will be the last holes to be constructed to allow the sediment basins to remain in service until all up-gradient golf holes are constructed and vegetation is established. The permanent irrigation pond will be constructed early in the project to also function as a sediment basin during the start of construction. Four permanent basins will be constructed near the tees on hole 10 and on hole 16, while 2 will be constructed along hole 18. These basins will be incorporated into the golf course construction. The basins will be designed to drain within 36 hours following storm events.
26. During construction, areas of exposed soil that have been graded to final pre-landscaped elevations and those that will not be worked for at least 3 weeks will be covered with straw erosion control blankets, liquid polymer emulsion, or hydro-seeded.
27. Outlets for basins and culverts will be stabilized with riprap aprons or gabion mattresses. Wherever practical, swales will be grassed to provide biotreatment and sediment control. In areas where erosive velocities could occur, riprap, gabions, or paved channels will be constructed.
28. Stone check dams will be constructed in ghuts immediately downstream from all construction activity in or immediately adjacent to ghuts. Stone check dams will also be constructed in swales and other locations where stormwater runoff concentrates.
29. Erosion control mats will be used in vegetated swales, slope convergences, or depressions where stormwater flow can concentrate.

The June 2013 BA also indicates that erosion and sediment control measures have been developed for all subdivision lots that will be developed by the individual homeowners upon approval by the William & Punch Homeowners Association, after having obtained all required federal and local permits, as applicable.

Based on the applicant's May 27, 2016, submission, the following additional measures were incorporated in the project to minimize potential impacts to ESA-listed species and designated critical habitat from sediment transport during project construction and operation:

1. An additional temporary sediment basin will be installed in the northeastern corner of the property within the Creque Dam watershed in the Creque Dam tributary that flows through this portion of the site.

2. Residential Development Phasing Protocols will be implemented and adherence to same will be ensured by the applicant. These protocols include:
    a. A 5-year moratorium will be imposed on the development of any residences on lots 28-45 (located along the easternmost border of the property including around the proposed golf course hole 15). This moratorium will not prevent the use and development of these lots for any non-residential alternative use such as a renewable energy project.
    b. No more than 15 single family detached residences will be permitted to be under active development at any one time.
    c. A 75 ft setback will be applied from the mean high water line to any hotel unit (e.g., rooms, villas, or suites) located on the created beach island.
    d. A 5-year moratorium will be imposed on the development of the 17 spa villas proposed on the eastern side of the access road for coastal residences and public parking area. This moratorium shall not prevent the development of this area for non-residential or non-hotel inventory purposes such as tennis courts.
    e. The development of any above ground structure timeshare/fractional buildings on the timeshare development site located on the northern coastal portion of the property will not commence until the marina condominium development is completed.
3. A Water Quality Testing Program will be developed and implemented during construction and operation of the project based on the draft document included in the May 2016 submission.
4. The covenants and restrictions that will apply to the residences at Estates William and Punch were updated. The covenants and restrictions ensure that William and Punch LLC remains responsible for oversight of the environmental quality of all aspects of the project, particularly those related to the on-going residential development that will occur until the proposed full site build out is achieved.

2.1 ACTION AREA

The action area is defined by regulation as "all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action" (50 CFR 402.02). The project site is a 594-ac area in Estates William and Punch, St. Croix (Figure 4).

The waters and beach where the in-water construction footprint is located are part of a connected system of coral reefs, colonized pavement and hard grounds interspersed with some areas of sand on the west side of St. Croix known as the Frederiksted Reef System, that extends from Sprat Hole south to King's Corner offshore of the Westend Saltpond. This is the only reef system on the west end of St. Croix and represents the dominant benthic habitat along most of the west coast of the island. The majority of the marine area to be affected is between the areas known as Rainbow Beach (south of the southern property line of Estates William and Punch) and Sprat Hole (to the north of the northern property line). In-water activity and marine habitat effects from the project will extend the action area beyond the immediate project site into all waters below mean high water from approximately the Frederiksted Pier (where the construction vessels will be traveling from) to Sprat Hole out to a depth of approximately 90 ft at the shelf edge (Figure 5 area C).

Area A in Figure 5 is the area expected to experience the greatest impacts from the proposed action; this area includes the 46 ac area (which extends 500 ft from the 4000 ft shoreline) surveyed for the BA. This area contains 2.75 acres of hard bottom that will be lost due to construction and dredging activities, and 30.31 acres of hardbottom that is expected to experience significant chronic impacts resulting from construction and maintainence activities and stormwater runoff. Area B extends an additional 2000 ft from Area A (for a total of 2500 ft from the shoreline) to the shelf's edge and contains 77 ac of shallow water reef/colonized bedrock and colonized pavement based on NOAA National Ocean Service (NOS) benthic habitat maps. This area is expected to receive episodic impacts due to stormwater runoff and other activities. Area C extends from the Fredriksted Pier out past the shelf edge to Sprat Hole and encompases approximately 828 acres of which approximately 588 acres are shallow water reef/colonized bedrock and colonized pavement. Impacts are expected from possible grounding of construction vessels or strikes by these vessels.

Figure 4. Map of the expanded area showing the west coast where the Frediksted Reef Complex is located from the Westend Saltpond south of the Frederiksted Pier to Sprat Hole and the approximate location of the shelf edge reef (©2015 Google, TerraMetrics)

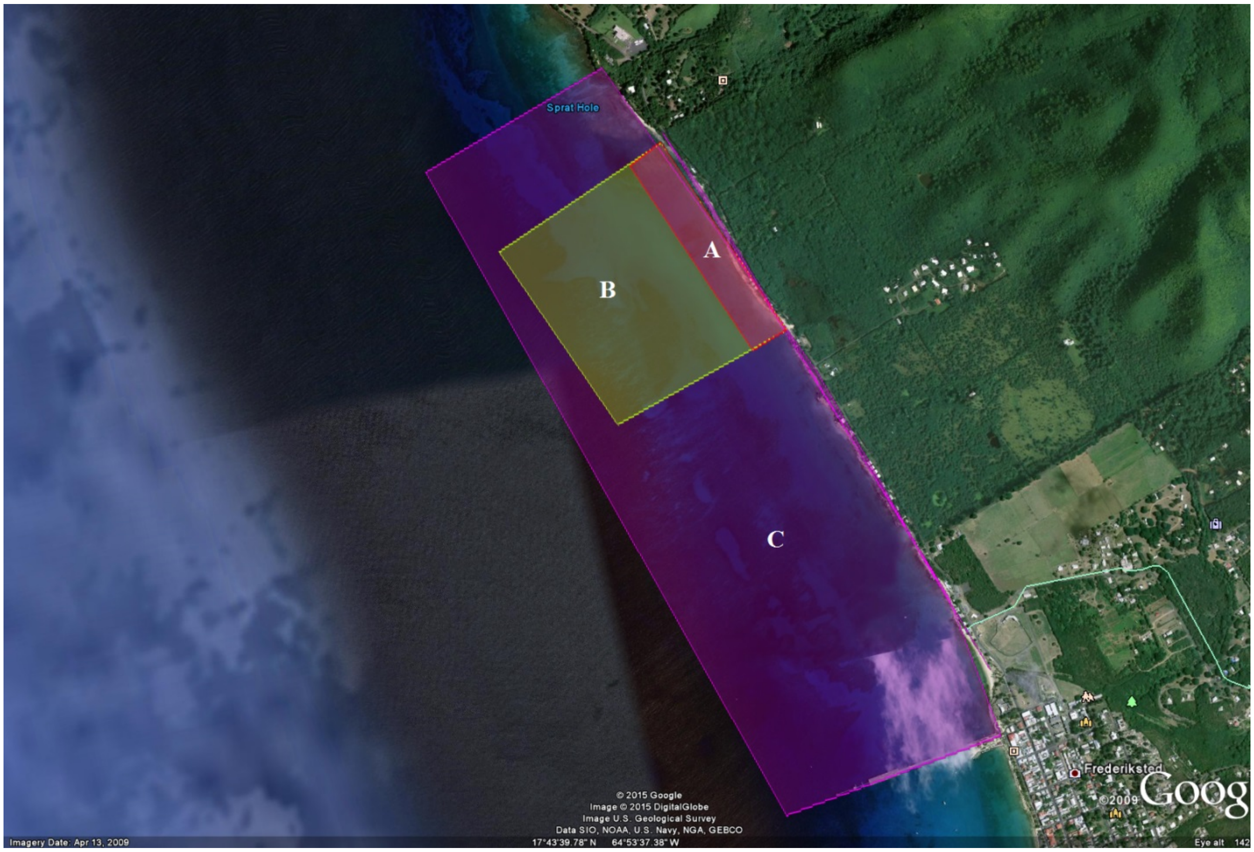

Figure 5. Map of the action area including approximate location of surveys and zones of potential impacts. Area A = Benthic habitat surveyed in the Biological Assessement (beach to 500 ft from shore); Area B = Area identified by NOS's benthic habitat maps (beach to 2,500 ft from shore); Area C = Area anticipated to be directly or indirectly affected by the project (i.e., action area). (©2015 Google, Digitial Globe)

3 STATUS OF LISTED SPECIES AND CRITICAL HABITAT

Table 1 lists the endangered (E) and threatened (T) sea turtle and coral species under the jurisdiction of NMFS that occur in or near the action area. Table 2 lists the designated critical habitat that occurs in or near the action area.

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

Species	ESA Listing Status	Action Agency Effect Determination	NMFS Effect Determination
Marine Mammals			
blue whale, *Balaenoptera musculus*	E[1]	NE	NLAA
fin whale, *Balaenoptera physalus*	E	NE	NLAA

[1] E = endangered, T = threatened, NLAA = may affect, not likely to adversely affect, LAA = may affect, likely to adversely affect, ND = no determination, NE = no effect

Species	ESA Listing Status	Action Agency Effect Determination	NMFS Effect Determination
sei whale, *Balaenoptera borealis*	E	NE	NLAA
sperm whale, *Physeter macrocephalus*	E	NE	NLAA
Sea Turtles			
green sea turtle North Atlantic DPS, *Chelonia mydas*	T	LAA	LAA
green sea turtle South Atlantic DPS[2]	T	LAA	LAA
loggerhead sea turtle Northwest Atlantic DPS, *Caretta caretta*	T	NLAA	NLAA
hawksbill sea turtle, *Eretmochelys imbricata*	E	LAA	LAA
leatherback sea turtle, *Dermochelys coriacea*	E	LAA	LAA
Fish			
Nassau grouper[3], *Epinephelus striatus*	T	ND	NLAA
Scalloped hammerhead shark (Central Atlantic and Southwest Atlantic DPS)[4], *Sphyrna lewini*	T	ND	NE
Oceanic whitetip shark[5], *Carcharinus lonigmanus*	T	ND	NE
Giant manta ray[6], *Manta birostris*	T	ND	NE
Invertebrates			
elkhorn coral, *Acropora palmata*	T	NLAA	LAA
staghorn coral, *Acropora cervicornis*	T	NLAA	LAA
pillar coral, *Dendrogyra cylindrus*	T	ND	NLAA
lobed star coral, *Orbicella annularis*	T	ND	LAA
mountainous star coral, *Orbicella faveolata*	T	ND	LAA
boulder star coral, *Orbicella franksi*	T	ND	NLAA
rough cactus coral, *Mycetophyllia ferox*	T	ND	NLAA

NMFS published a final rule on September 8, 2016 (81 FR 62260) identifying 14 DPS's for humpback whales. The West Indies DPS, which includes Puerto Rico, was found not to merit listing under the ESA. Therefore, no effects determination is needed for humpback whales.

[2] Green sea turtles nesting in Puerto Rico are now within the North Atlantic DPS and green sea turtles nesting in the Virgin Islands are now within the South Atlantic DPS based on the final listing rule designating 11 DPSs published on April 6, 2016. However, because of the mobility of sea turtles, we consider both DPSs in this Opinion as it is not possible to separate animals observed in the action area into one or the other of the DPSs given the small geographic separation between Puerto Rico and the Virgin Islands.
[3] Nassau grouper were listed as threatened on June 29, 2016 (81 FR 42268).
[4] The Central and Southwest Atlantic DPS and the Indo-West Pacific DPS of scalloped hammerhead shark were listed as threatened and the Eastern Atlantic DPS and Eastern Pacific DPS were listed as endangered on July 3, 2014 (79 FR 38214).
[5] Oceanic whitetip sharks were listed as threatened on January 30, 2018 (83 FR 4153).
[6] Giant manta ray were listed as threatened on January 22, 2018 (83 FR 2916).

NMFS published a final rule on January 30, 2018 (83 FR 4153) to list the oceanic whitetip shark as threatened under the ESA. This species is pelagic and generally found offshore in the open ocean or on the outer continental shelf of tropical and subtropical waters around the world in water depths greater than 600 ft, which are conditions that do not exist in the action area. Because of this, we believe the proposed action will have no effect on oceanic whitetip shark.

NMFS published a final rule on January 22, 2018 (83 FR 2916) to list the giant manta ray as threatened under the ESA. Giant manta ray are typically found offshore in the open ocean waters though they sometimes may be found around nearshore reefs and estuarine waters. Based on information from scientists who frequently work in waters around USVI, giant manta rays have not been sighted in the action area and are extremely rare around St. Thomas, St. John and St. Croix (A. Dempsey, BioImpact, personal communications to L. Carrubba, NMFS, January 26, 2018 and February 26, 2018; R. Nemeth, University of the Virgin islands, personal communication to L. Carrubba, NMFS, January 26, 2018). No giant manta rays were reported as part of benthic surveys conducted for the proposed action and the species was not observed by NMFS' biologists during site inspections in the action area.[7] Therefore, given the rarity of the species in USVI waters, the lack of sightings in the action area, and the species' preference for open ocean waters, we believe the proposed action will have no effect on giant manta rays.

Table 2. Designated Critical Habitat in the Action Area

Species	Critical Habitat Unit	Action Agency Effect Determination	NMFS Effect Determination
elkhorn and staghorn coral	St. Croix unit	LAA[8]	LAA

3.1 Analysis of Species Not Likely to be Adversely Affected

3.1.1 Whales

There are 4 species of ESA-listed whales (blue, fin, sei, and sperm) that can possibly be found in or near the action area. These species could be affected by the construction and operation of the Amalago Bay project by vessels transiting to and from the project either during construction, as part of dredging operations, or operations as part of the use of the marina. Sighting and stranding data for USVI are limited. However, information from previous consultations, such as the Marine Events Program consultation with the USCG, which included annually occurring events throughout USVI, indicated that whales have not been sighted during events. Some of the events include sailing, swims, and other activities around St. Croix.

Up to 70 vessels will be able to use the marina and these vessels range in size from 40-100 ft, on average. Therefore, there is a potential for vessel strikes or interference with whales by

[7] Even in the unlikely event that giant manta rays were to enther the actiona area, the major stressors associated with this action – pollution and sedimentation – are not known to threaten manta rays directly based on information in the listing determination for the species.
[8] LAA = may affect, likely to adversely affect

recreational boaters wishing to view these animals as a result of the project. Construction vessels will also transit through the action area during construction of the marina, which includes dredge vessels. There is a potential for vessel strikes by these vessels. According to US Boats web site there are 6915 registered boats (this includes all vessels) in St. Croix. The 70 new vessels associated with the proposed action represent a 1% increase in the number of registered vessels in St. Croix. The applicant noted that a review of the Sea Turtle Assistance and Rescue (STAR) data, which also records marine mammal strandings, from the last 10-15 years did not contain any reports of interactions between vessels and whales in St. Croix. The applicant also noted that anecdotal information from dive shop operators and boat captains in St. Croix indicates that humpback whales are the most common whale species observed in St. Croix from January-March, but they are usually observed 2-3 miles offshore and are no longer ESA-listed.

As part of the project, the applicant is proposing an education program to inform visitors, residents, and construction personnel of the presence of ESA-listed species and their habitat. This education program will include information for boaters to avoid impacts to ESA-listed whale species. The USACE will also require the implementation of NMFS's *Vessel Strike Avoidance Measures and Reporting for Mariners* (enclosed) and NMFS's *Sea Turtle and Smalltooth Sawfish Construction Conditions* (enclosed). Based on the fact that the proposed action will only increase the number of vessels in St. Croix by 1%, and there have been no reported vessel strikes in St. Croix combined with the increased awareness resulting from the proposed actions education program, NMFS believes that the potential project impacts to ESA-listed whale species will be extremely unlikely to occur.

3.1.2 Loggerhead Sea Turtles

Loggerhead sea turtles may be found in or near the action area. Loggerhead sea turtles are not common in the U.S. Caribbean. However, there are reports of loggerhead sea turtles in waters around St. Croix (including unpublished stranding data from the Virgin Islands DPNR showing one found dead from poaching in the Frederiksted area in 2003) and 2 females have now been documented nesting on Buck Island since 2003 (USNPS 2003, 2004)(USNPS, unpublished data). No nesting of loggerhead sea turtles is reported on the beaches in the project area, although nesting by 3 other species of ESA-listed sea turtles is reported. Loggerheads are sometimes associated with reefs and other natural and artificial hard substrate. Portions of the in-water structures and the created beach will be located over patch reefs and colonized hard bottom. Loggerheads could be present in the area of the reefs and colonized hard ground where the marina channels and jetties and beach construction will take place, although none were observed during any of the marine surveys completed for the project or during site inspections by NMFS's biologists.

The construction of the Amalago Bay project could affect habitat of loggerhead sea turtles through impacts to nearshore reefs and hardgrounds. Loggerhead sea turtles could also be physically injured by construction activities, such as dredging or be struck by construction vessels or private vessels associated with the completed development. Because of the potential presence of sea turtles along the transit routes for work vessels and in the area where the marina and other shoreline construction will occur, as well as potential impacts of upland development to nearshore sea turtle habitat from stormwater and sediment runoff, the applicant detailed measures to avoid and minimize potential impacts of the construction and operation of the

Amalago Bay project to ESA-listed sea turtles in the BA. These measures include (1) the implementation of NMFS's *Sea Turtle and Smalltooth Sawfish Construction Conditions* and pile driving BMPs during construction; (2) the development and implementation of an educational plan for construction personnel, residents, visitors, and employees regarding the presence of ESA-listed species and measures to avoid and minimize impacts to these animals and their habitat; (3) the development and implementation of a lighting plan in coordination with USFWS; (4) the placement of sand to create the new beach during the months of January and February only to avoid peak sea turtle nesting seasons for all ESA-listed sea turtle species; and (5) the use of observers to monitor for sea turtles during construction activities, as well as sea turtle monitoring on beaches prior to, during, and after construction to minimize potential disturbance of nests. Given the low numbers of loggerhead sea turtles reported around St. Croix and the lack of sightings in the project area, as well as the development of an educational plan and the incorporation of NMFS's *Sea Turtle and Smalltooth Sawfish Construction Conditions* in the project, NMFS believes potential impacts of physical injury to loggerhead sea turtles will be extremely unlikely to occur. In terms of the potential impacts to loggerhead sea turtles from habitat loss or degradation associated with the transport of contaminants from in-water construction and in stormwater runoff during project construction and operation, we believe these impacts will be insignificant, because of the species' limited use of the area.

3.1.3 Nassau Grouper

No Nassau grouper were observed during benthic surveys conducted for the project or site inspections in which NMFS's biologists participated. Similarly, the survey of reefs on the west end of St. Croix by Toller (2005) did not result in sightings of this species, although other grouper species were reported. The nearshore benthic habitats that will be affected by the project, including colonized hard bottom, patch reefs, and seagrass, may all serve as refuge and foraging habitat for juvenile Nassau grouper and the coral reefs further offshore could serve as habitat for adults of this species, although surveys in this and other areas around USVI indicate this species continues to be rare likely due to the dramatic population declines associated with overexploitation.

Various life stages of Nassau grouper could be affected by impacts to habitat as a result of the proposed construction and operation of the Amalago Bay project if there are individuals present in the action area. The pile-driving associated with the construction of a temporary trestle pier for marina construction activities could also lead to acoustic impacts to Nassau grouper.

Pile-Driving: Effects to Nassau grouper as a result of noise created by construction activities can physically injure the animals or change animal behavior in the affected areas. Physically injurious effects can occur in 2 ways. First, effects can result from a single noise event's exceeding the threshold for direct physical injury to animals, and these constitute an immediate adverse effect on these animals. Second, effects can result from prolonged exposure to noise levels that exceed the daily cumulative exposure threshold for the animals, and these can constitute adverse effects if animals are exposed to the noise levels for sufficient periods. Behavioral effects can be adverse if they interfere with animals migrating, feeding, resting, or reproducing, for example.

The applicant did not provide details in terms of the size of the piles, number of strikes per pile for installation, or number of piles that will be installed per day. There are currently no established thresholds for injurious or behavioral effects to fish from the use of a vibratory hammer to drive piles. However, we believe it is extremely unlikely that the installation of temporary piles by vibratory hammer will result in injurious or behavioral noise effects to this species. Nassau grouper are rare in the action area based on the lack of observations of this species during surveys conducted for the project and observations of fish during site inspections by NMFS's biologists. Any Nassau grouper that are present are likely to move away from disturbance, including disturbance that may result in temporary sediment plumes as these would interfere with visual cues used by the fish to look for prey and flee from predators. There is uncolonized sand bottom, sand bottom with sparse seagrass, and colonized hard bottom within the footprint of the temporary trestle, which is the same as the marina entrance channel and jetties. Because there are other areas containing seagrass beds and colonized hard bottom in the project area that contain habitat, whichcould be used by various life stages of Nassau grouper, we believe the effects from vibratory pile driving to this species will be insignificant.

NMFS uses a dual-metric criteria to determine the onset of injury to fishes exposed to impact pile driving sound. Specifically, this includes a single-strike peak level (SPL) of 206 dB and a cumulative sound exposure level (cSEL) of 187 dB for fishes two grams or larger, or 183 dB for fish less than two grams. For ESA-listed fishes, if either threshold is exceeded, then physical injury or auditory damage is assumed to occur (e.g., barotrauma and/or temporary threshold shifts in hearing). For sub-injury effects (e.g., behavioral response) of fishes exposed to high levels of underwater sound produced during pile driving, NMFS believes a 150 dB root-mean-square pressure (RMS) threshold for behavioral responses is appropriate in order to establish a sound level where responses of fishes may occur and be a concern.

NMFS assumes the construction of the access trestle for this project will require a maximum size steel pipe pile of 30-inches (in) in diameter, similar to the trestle pile installation used for the Richmond-San Rafael Bridge, San Francisco Bay, San Rafael, CA, and which represents the worse case scenario we were able to identify for similar work. Using sound monitoring data provided in the Compendium of Pile Driving Sound Data (CALTRANS 2015) for 30-in steel pipe piles driven with an impact hammer, NMFS calculated the distance to reach the respective thresholds for onset of injury on fishes for both attenuated (i.e., using a bubble curtain) and unattenuated pile installation. Our calculations assume there will be up to four piles driven per day, requiring 600 strikes per pile for a daily total of 2,400 strikes. For attenuated piles, the distances to reach onset of injury thresholds are: 206 peak dB at 59 ft (18 m), 187 dB cSEL at 1,266 ft (386 m), and 183 dB cSEL at 2,070 ft (631 m). The applicant notes that bubble curtains may be used, which would reduce the distance to reach respective injury thresholds during pile driving activities. NMFS assumes that an effectively implemented bubble curtain can reduce sound levels by approximately 10 dB for impact hammer driven steel piles based on CALTRANS (2015) and observations made by NMFS' biologists during pile driving activities with and without sound reduction measures. Based on this attenuation factor, NMFS calculated the distances to reach acoustic injury thresholds for fish for attenuated pile driving to be: 206 dB peak at 13 ft (4 m), 187 dB cSEL at 272 ft (83 m), and 183 dB cSEL at 446 ft (136 m). These reductions in injurious sound levels only apply to the use of bubble curtains. If double turbidity barriers are used during pile-driving activities instead of bubble curtains (which is another possibility proposed by the applicant), NMFS does not anticipate a reduction in injurious sound levels.

As noted previously, we do not have information to suggest that various life stages of Nassau grouper are common in the action area. Additionally, the applicant proposes the use of dry-firing and ramp-up procedures when using an impact hammer which may cause fish to startle and move away from the work area. We also believeNassau grouper could leave the area of their own volition while pile-driving and other in-water construction activities are taking place as there are no impediments to their movement. Because only 45 piles will be installed, any noise effects associated with pile-driving activities will be short-term. Because we consider Nassau grouper to be uncommon in the area, and we assume any that are present in the action area will be unimpeded and could leave when pile-driving and other in-water construction disturbance occurs, we believe that an animal suffering physical injury from pile driving noise exposure would be extremely unlikely to occur. An animal's behavioral response such as a startle and potential movement away from the pile driving area is discussed below.

The installation of 30-in steel pipe piles using an unattenuated impact hammer could also result in behavioral effects up to a distance of 15,230 ft (4,642 m) from the pile corresponding to the 150 dB RMS threshold for sub-injury for Nassau grouper. If the applicant uses bubble curtains, this could reduce the distance to 3,281 ft (1,000 m). Due to the mobility of Nassau grouper, we expect them to move away from in-water construction disturbance in the open water environment where pile driving will occur. Since installation will only occur during the day, any Nassau grouper that could be in the area would be able to resume normal activities during quiet periods between pile installation and at night. Because there are other areas of colonized hard bottom and seagrass beds in the action area and because of the lack of evidence indicating Nassau grouper are common in the action area, we believe behavioral effects will be insignificant.

3.1.4 Corals

ESA-listed pillar, boulder star, and rough cactus corals could be affected by vessel transit, in particular due to accidental groundings during the in-water construction or transit of vessels to and from the proposed marina. Pillar, boulder star, and rough cactus corals have not been found within the footprint of the proposed marina construction or within the 46-ac area surveyed for the BA that extends along the entire shoreline of the project, and have not been reported in monitoring studies along the shelf edge (Smith et al. 2011a).

ESA-listed pillar, boulder star, and rough cactus coral colonies may be present south of the in-water construction area where additional coral colonized hard bottoms and patch reefs are present, as well as in deeper reefs toward the shelf edge within the action area. These areas were not surveyed as part of the benthic surveys completed for the BA. However, transects from a 2005 DPNR survey (Toller 2005) of the Frederiksted Reef System (from Frederiksted Pier to Sprat Hole) showed that deeper reefs were dominated by *Montastraea/Orbicella* spp. corals but this and other surveys conducted as part of the Territorial Coral Reef Monitoring Program have not reported pillar, boulder star, or rough cactus coral colonies. This lack of reporting could be due to the limited number of stations in the monitoring program rather than an absence of these species from the action area. Because of the potential presence of ESA-listed pillar, boulder star, and rough cactus corals along the transit routes for work vessels, as well as potential impacts of upland development from stormwater and sediment runoff to nearshore habitat, the applicant has proposed the following avoidance and minimization measures to protect corals that will be implemented as part of the construction and operation of the Amalago Bay project:

(1) the construction of the marina basin in the dry and the opening of the navigation and flushing channels only after the basin has been filled, the shoreline stabilized, and sediment levels have returned to naturally occurring levels

(2) the development and implementation of an educational plan for construction personnel, residents, visitors, and employees regarding the presence of ESA-listed species and measures to avoid and minimize impacts to these species and their habitat

(3) the implementation of in-water turbidity controls such as turbidity curtains to minimize the potential for material transport to nearshore habitats outside the construction area

(4) the installation of ATONS to demarcate the navigation channel and jetties to minimize the potential for accidental groundings by vessels transiting to the marina

Based on the information from project surveys and other surveys conducted in the project area indicating that there are no pillar, boulder star, or rough cactus coral colonies in the immediate project area (areas A and B, Figure 5), where the greatest effects to corals are expected, NMFS believes the potential project impacts to any pillar, boulder star, and rough cactus corals that may be present in the larger action area will be extremely unlikely to occur.

For the reasons given above, NMFS has determined that the project may affect, but is not likely to adversely affect, ESA-listed loggerhead sea turtles, pillar, boulder star, and rough cactus coral colonies, and marine mammals.

3.2 Species and Critical Habitat Likely to be Adversely Affected

North and South Atlantic DPSs of green sea turtles, leatherback and hawksbill sea turtles; elkhorn, staghorn, lobed star, and mountainous star corals; and designated critical habitat for elkhorn and staghorn corals are likely to be adversely affected by the proposed action.

The summaries that follow describe the status of the ESA-listed species and their designated critical habitats that occur within the action area and are considered in this Opinion. More detailed information on the status and trends of these listed resources and their biology and ecology can be found in the listing regulations and critical habitat designations published in the Federal Register, status reviews, recovery plans, and on this NMFS website.[9]

3.2.1 Sea Turtles

3.2.1.1 General Threats Faced by All Sea Turtle Species

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

[9] https://www.fisheries.noaa.gov/species-directory/threatened-endangered

turtle species, those identified in this section are discussed in a general sense for all sea turtles. Threat information specific to a particular species are then discussed in the corresponding status sections where appropriate.

Fisheries

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

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

Non-Fishery In-Water Activities

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

Coastal Development and Erosion Control

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

Environmental Contamination

Multiple municipal, industrial, and household sources, as well as atmospheric transport, introduce various pollutants such as pesticides, hydrocarbons, organochlorides (e.g., dichlorodiphenyltrichloroethane [DDT], polychlorinated biphenyls [PCB], and perfluorinated chemicals [PFC]), and others that may cause adverse health effects to sea turtles (Iwata et al. 1993, Grant and Ross 2002, Garrett 2004, Hartwell 2004). Acute exposure to hydrocarbons from petroleum products released into the environment via oil spills and other discharges may directly injure individuals through skin contact with oils (Geraci 1990), inhalation at the water's surface, and ingesting compounds while feeding (Matkin and Saulitis 1997). Hydrocarbons also have the potential to impact prey populations, and therefore may affect listed species indirectly by reducing food availability in the action area.

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

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

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

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

The effects from increased temperatures may be intensified on developed nesting beaches where shoreline armoring and construction have denuded vegetation. Erosion control structures could potentially result in the permanent loss of nesting beach habitat or deter nesting females (NRC 1990). These impacts will be exacerbated by sea level rise. If females nest on the seaward side of the erosion control structures, nests may be exposed to repeated tidal overwash (NMFS and USFWS 2007g). 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 an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Antonelis et al. 2006, Baker et al. 2006).

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

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

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

3.2.1.2 Green Sea Turtle (North and South Atlantic DPS, *Chelonia mydas*)

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

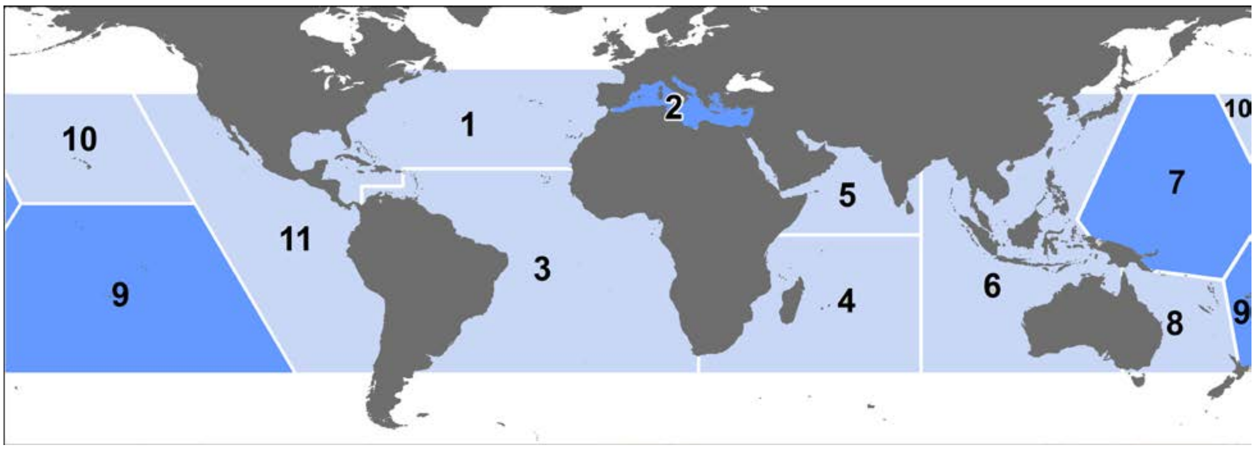

Figure 6. Threatened (light) and endangered (dark) green turtle DPSs: 1. North Atlantic, 2. Mediterranean, 3. South Atlantic, 4. Southwest Indian, 5. North Indian, 6. East Indian-West Pacific, 7. Central West Pacific, 8. Southwest Pacific, 9. Central South Pacific, 10. Central North Pacific, and 11.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

Species Description and Distribution

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

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

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

North Atlantic DPS Distribution

The NA DPS boundary is illustrated in Figure 6. Four regions support nesting concentrations of particular interest in the NA DPS: Costa Rica (Tortuguero), Mexico (Campeche, Yucatan, and Quintana Roo), U.S. (Florida), and Cuba. By far the most important nesting concentration for green turtles in this DPS is Tortuguero, Costa Rica. Nesting also occurs in the Bahamas, Belize, Cayman Islands, Dominican Republic, Haiti, Honduras, Jamaica, Nicaragua, Panama, Puerto Rico, Turks and Caicos Islands, and North Carolina, South Carolina, Georgia, and Texas, U.S.A. In the eastern North Atlantic, nesting has been reported in Mauritania (Fretey 2001).

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

In U.S. Atlantic and Gulf of Mexico waters, green sea turtles are distributed throughout inshore and nearshore waters from Texas to Massachusetts. Principal benthic foraging areas in the southeastern United States include Aransas Bay, Matagorda Bay, Laguna Madre, and the Gulf inlets of Texas (Hildebrand 1982, Doughty 1984, Shaver 1994), the Gulf of Mexico off Florida from Yankeetown to Tarpon Springs (Caldwell and Carr 1957), Florida Bay and the Florida

Keys (Schroeder and Foley 1995), the Indian River Lagoon system in Florida (Ehrhart 1983), and the Atlantic Ocean off Florida from Brevard through Broward Counties (Guseman and Ehrhart 1992, Wershoven and Wershoven 1992a). The summer developmental habitat for green sea turtles also encompasses estuarine and coastal waters from North Carolina to as far north as Long Island Sound (Musick and Limpus 1997). Additional important foraging areas in the western Atlantic include the Culebra archipelago and other Puerto Rico coastal waters, the south coast of Cuba, the Mosquito Coast of Nicaragua, the Caribbean coast of Panama, scattered areas along Colombia and Brazil (Hirth 1971), and the northwestern coast of the Yucatán Peninsula.

South Atlantic DPS Distribution

The SA DPS boundary is shown in Figure 6, and includes the U.S. Virgin Islands in the Caribbean. The SA DPS nesting sites can be roughly divided into four regions: western Africa, Ascension Island, Brazil, and the South Atlantic Caribbean (including Colombia, the Guianas, and Aves Island in addition to the numerous small, island nesting sites).

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

Life History Information

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

Reef in Australia) showing higher survivorship values than nesting sites known to be highly disturbed (e.g., Nicaragua) (Campell and Lagueux 2005, Chaloupka and Limpus 2005).

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

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

Status and Population Dynamics

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

<u>North Atlantic DPS</u>

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

Quintana Roo, Mexico, accounts for approximately 11% of nesting for the DPS (Seminoff et al. 2015). In the early 1980s, approximately 875 nests/year were deposited, but by 2000 this increased to over 1,500 nests/year (NMFS and USFWS 2007h). By 2012, more than 26,000 nests were counted in Quintana Roo (J. Zurita, CIQROO, unpublished data, 2013, in Seminoff et al. 2015).

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

In the continental United States, green sea turtle nesting occurs along the Atlantic coast, primarily along the central and southeast coast of Florida (Meylan et al. 1994, Weishampel et al. 2003b). Occasional nesting has also been documented along the Gulf Coast of Florida (Meylan et al. 1995). Green sea turtle nesting is documented annually on beaches of North Carolina, South Carolina, and Georgia, though nesting is found in low quantities (up to tens of nests) (nesting databases maintained on www.seaturtle.org).

Florida accounts for approximately 5% of nesting for this DPS (Seminoff et al. 2015). In Florida, index beaches were established to standardize data collection methods and effort on key nesting beaches. Since establishment of the index beaches in 1989, the pattern of green sea turtle nesting has generally shown biennial peaks in abundance with a positive trend during the 10 years of regular monitoring (Figure 7). According to data collected from Florida's index nesting beach survey from 1989-2018, green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high of 38,954 in 2017. Two consecutive years of nesting declines in 2008 and 2009 caused some concern, but this was followed by increases in 2010 and 2011, and a return to the trend of biennial peaks in abundance thereafter (Figure 7). Modeling by Chaloupka et al. (2008) using data sets of 25 years or more resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9% at that time. Increases have been even more rapid in recent years.

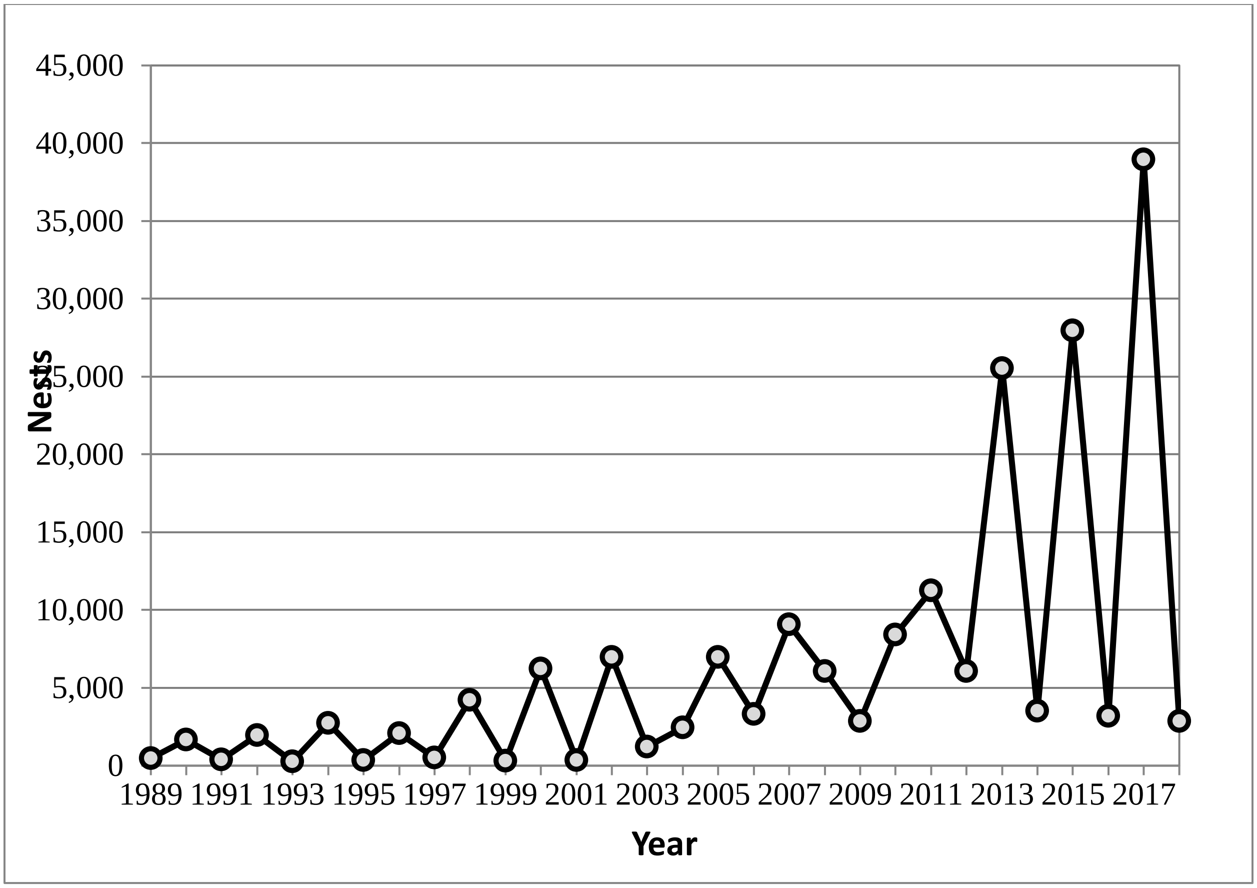

Figure 7. Green sea turtle nesting at Florida index beaches since 1989

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

South Atlantic DPS

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

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

Threats
The principal cause of past declines and extirpations of green sea turtle assemblages has been the overexploitation of the species for food and other products. Although intentional take of green sea turtles and their eggs is not extensive within the southeastern United States, green sea turtles that nest and forage in the region may spend large portions of their life history outside the region and outside U.S. jurisdiction, where exploitation is still a threat. Green sea turtles also face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (e.g., plastics, petroleum products, petrochemicals), ecosystem alterations (e.g., nesting beach development, beach nourishment and shoreline stabilization, vegetation changes), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 3.2.1.1.

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

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

Whereas oil spill impacts are discussed generally for all species in Section 3.2.1.1, specific impacts of the DWH spill on green sea turtles are considered here. Impacts to green sea turtles occurred to offshore small juveniles only. A total of 154,000 small juvenile greens (36.6% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. A large number of small juveniles were removed from the population, as 57,300 small juveniles greens are estimated to have died as a result of the exposure. A total of 4 nests (580 eggs) were also translocated during response efforts, with 455 hatchlings released (the fate of which is unknown) (DWH Trustees 2015). Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources, which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred.

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

3.2.1.3 Leatherback Sea Turtle (*Dermochelys coriacea*)

The leatherback sea turtle was listed as endangered throughout its entire range on June 2, 1970, (35 FR 8491) under the Endangered Species Conservation Act of 1969.

Species Description and Distribution
The leatherback is the largest sea turtle in the world, with a curved carapace length (CCL) that often exceeds 5 ft (150 cm) and front flippers that can span almost 9 ft (270 cm) (NMFS and USFWS 1998b). Mature males and females can reach lengths of over 6 ft (2 m) and weigh close to 2,000 lb (900 kg). The leatherback does not have a bony shell. Instead, its shell is approximately 1.5 in (4 cm) thick and consists of a leathery, oil-saturated connective tissue overlaying loosely interlocking dermal bones. The ridged shell and large flippers help the leatherback during its long-distance trips in search of food.

Unlike other sea turtles, leatherbacks have several unique traits that enable them to live in cold water. For example, leatherbacks have a countercurrent circulatory system (Greer et al. 1973),[10] a thick layer of insulating fat (Goff and Lien 1988, Davenport et al. 1990), gigantothermy (Paladino et al. 1990),[11] and they can increase their body temperature through increased metabolic activity (Southwood et al. 2005, Bostrom and Jones 2007). These adaptations allow leatherbacks to be comfortable in a wide range of temperatures, which helps them to travel further than any other sea turtle species (NMFS and USFWS 1995). For example, a leatherback may swim more than 6,000 miles (10,000 km) in a single year (Eckert 2006, Eckert et al. 2006, Benson et al. 2007a, Benson et al. 2011). They search for food between latitudes 71°N and 47°S in all oceans, and travel extensively to and from their tropical nesting beaches. 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).

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

Genetic analyses using microsatellite markers along with mitochondrial DNA and tagging data indicate there are 7 groups or breeding populations in the Atlantic Ocean: Florida, Northern Caribbean, Western Caribbean, Southern Caribbean/Guianas, West Africa, South Africa, and Brazil (TEWG 2007a). General differences in migration patterns and foraging grounds may occur between the 7 nesting assemblages, although data to support this is limited in most cases.

Life History Information

The leatherback life cycle is broken into several stages: (1) egg/hatchling, (2) post-hatchling, (3) juvenile, (4) subadult, and (5) adult. Leatherbacks are a long-lived species that delay age of maturity, have low and variable survival in the egg and juvenile stages, and have relatively high and constant annual survival in the subadult and adult life stages (Spotila et al. 1996, Crouse 1999, Heppell et al. 1999, Spotila et al. 2000, Chaloupka 2002, Heppell et al. 2003). While a robust estimate of the leatherback sea turtle's life span does not exist, the current best estimate for the maximum age is 43 (Avens et al. 2009). It is still unclear when leatherbacks first become sexually mature. Using skeletochronological data, Avens et al. (2009) estimated that leatherbacks in the western North Atlantic may not reach maturity until 29 years of age, which is longer than earlier estimates of 2-3 years by Pritchard and Trebbau (1984), of 3-6 years by

[10] Countercurrent circulation is a highly efficient means of minimizing heat loss through the skin's surface because heat is recycled. For example, a countercurrent circulation system often has an artery containing warm blood from the heart surrounded by a bundle of veins containing cool blood from the body's surface. As the warm blood flows away from the heart, it passes much of its heat to the colder blood returning to the heart via the veins. This conserves heat by recirculating it back to the body's core.

[11] "Gigantothermy" refers to a condition when an animal has relatively high volume compared to its surface area, and as a result, it loses less heat.

Rhodin (1985), of 13-14 years for females by Zug and Parham (1996), and 12-14 years for leatherbacks nesting in the U.S. Virgin Islands by Dutton et al. (2005). A more recent study that examined leatherback growth rates estimated an age at maturity of 16.1 years (Jones et al. 2011).

The average size of reproductively active females in the Atlantic is generally 5-5.5 ft (150-162 cm) CCL (Hirth et al. 1993, Starbird and Suarez 1994, Benson et al. 2007a). Still, females as small as 3.5-4 ft (105-125 cm) CCL have been observed nesting at various sites (Stewart et al. 2007).

Female leatherbacks typically nest on sandy, tropical beaches at intervals of 2-4 years (McDonald and Dutton 1996, Garcia M. and Sarti 2000, Spotila et al. 2000). Unlike other sea turtle species, female leatherbacks do not always nest at the same beach year after year; some females may even nest at different beaches during the same year (Eckert 1989, Keinath and Musick 1993, Steyermark et al. 1996, Dutton et al. 2005). Individual female leatherbacks have been observed with fertility spans as long as 25 years (Hughes 1996). Females usually lay up to 10 nests during the 3-6 month nesting season (March through July in the United States), typically 8-12 days apart, with 100 eggs or more per nest (Matos 1986, Tucker 1988, Eckert 1989, Maharaj 2004, Stewart and Johnson 2006, Eckert et al. 2012). Yet, up to approximately 30% of the eggs may be infertile (Eckert et al. 1984, Matos 1986, Tucker 1988, Eckert 1989, Maharaj 2004, Stewart and Johnson 2006). The number of leatherback hatchlings that make it out of the nest on to the beach (i.e., emergent success) is approximately 50% worldwide (Eckert et al. 2012), which is lower than the greater than 80% reported for other sea turtle species (Miller 1997). In the United States, the emergent success is higher at 54-72% (Tucker 1988, Eckert and Eckert 1990, Stewart and Johnson 2006). Thus the number of hatchlings in a given year may be less than the total number of eggs produced in a season. Eggs hatch after 60-65 days, and the hatchlings have white striping along the ridges of their backs and on the edges of the flippers. Leatherback hatchlings weigh approximately 1.5-2 oz (40-50 g), and have lengths of approximately 2-3 in (51-76 mm), with fore flippers as long as their bodies. Hatchlings grow rapidly, with reported growth rates for leatherbacks from 2.5-27.6 in (6-70 cm) in length, estimated at 12.6 in (32 cm) per year (Jones et al. 2011).

In the Atlantic, the sex ratio appears to be skewed toward females. The Turtle Expert Working Group (TEWG) reports that nearshore and onshore strandings data from the U.S. Atlantic and Gulf of Mexico coasts indicate that 60% of strandings were females (TEWG 2007a). Those data also show that the proportion of females among adults (57%) and juveniles (61%) was also skewed toward females in these areas (TEWG 2007a). James et al. (2007) collected size and sex data from large subadult and adult leatherbacks off Nova Scotia and also concluded a bias toward females at a rate of 1.86:1.

The survival and mortality rates for leatherbacks are difficult to estimate and vary by location. For example, the annual mortality rate for leatherbacks that nested at Playa Grande, Costa Rica, was estimated to be 34.6% in 1993-1994, and 34.0% in 1994-1995 (Spotila et al. 2000). In contrast, leatherbacks nesting in French Guiana and St. Croix had estimated annual survival rates of 91% (Rivalan et al. 2005) and 89% (Dutton et al. 2005), respectively. For the St. Croix population, the average annual juvenile survival rate was estimated to be approximately 63% and the total survival rate from hatchling to first year of reproduction for a female was estimated to be between 0.4% and 2%, assuming age at first reproduction is between 9-13 years (Eguchi et al. 2006). Spotila et al. (1996) estimated first-year survival rates for leatherbacks at 6.25%.

Migratory routes of leatherbacks are not entirely known; however, recent information from satellite tags have documented long travels between nesting beaches and foraging areas in the Atlantic and Pacific Ocean basins (Ferraroli et al. 2004, Hays et al. 2004, James et al. 2005, Eckert 2006, Eckert et al. 2006, Benson et al. 2007a, Benson et al. 2011). Leatherbacks nesting in Central America and Mexico travel thousands of miles through tropical and temperate waters of the South Pacific (Eckert and Sarti 1997, Shillinger et al. 2008). Data from satellite tagged leatherbacks suggest that they may be traveling in search of seasonal aggregations of jellyfish (Shenker 1984, Starbird et al. 1993, Bowlby et al. 1994, Suchman and Brodeur 2005, Benson et al. 2007b, Graham 2009).

Status and Population Dynamics

The status of the Atlantic leatherback population had been less clear than the Pacific population, which has shown dramatic declines at many nesting sites (Spotila et al. 2000, Santidrián Tomillo et al. 2007, Sarti Martínez et al. 2007). This uncertainty resulted from inconsistent beach and aerial surveys, cycles of erosion, and reformation of nesting beaches in the Guianas (representing the largest nesting area). Leatherbacks also show a lesser degree of nest-site fidelity than occurs with the hardshell sea turtle species. Coordinated efforts of data collection and analyses by the leatherback Turtle Expert Working Group helped to clarify the understanding of the Atlantic population status up through the early 2000’s (TEWG 2007a). However, additional information for the Northwest Atlantic population has more recently shown declines in that population as well, contrary to what earlier information indicated (Northwest Atlantic Leatherback Working Group 2018). A full status review covering leatherback status and trends for all populations worldwide is being finalized (2019).

The Southern Caribbean/Guianas stock is the largest known Atlantic leatherback nesting aggregation (TEWG 2007a). This area includes the Guianas (Guyana, Suriname, and French Guiana), Trinidad, Dominica, and Venezuela, with most of the nesting occurring in the Guianas and Trinidad. The Southern Caribbean/Guianas stock of leatherbacks was designated after genetics studies indicated that animals from the Guianas (and possibly Trinidad) should be viewed as a single population. Using nesting females as a proxy for population, the TEWG (2007a) determined that the Southern Caribbean/Guianas stock had demonstrated a long-term, positive population growth rate. TEWG observed positive growth within major nesting areas for the stock, including Trinidad, Guyana, and the combined beaches of Suriname and French Guiana (TEWG 2007a). More specifically, Tiwari et al. (2013) report an estimated three-generation abundance change of +3%, +20,800%, +1,778%, and +6% in Trinidad, Guyana, Suriname, and French Guiana, respectively. However, subsequent analysis using data up through 2017 has shown decreases in this stock, with an annual geometric mean decline of 10.43% over what they described as the short term (2008-2017) and a long-term (1990-2017) annual geometric mean decline of 5% (Northwest Atlantic Leatherback Working Group 2018).

Researchers believe the cyclical pattern of beach erosion and then reformation has affected leatherback nesting patterns in the Guianas. For example, between 1979 and 1986, the number of leatherback nests in French Guiana had increased by about 15% annually (NMFS 2001). This increase was then followed by a nesting decline of about 15% annually. This decline corresponded with the erosion of beaches in French Guiana and increased nesting in Suriname. This pattern suggests that the declines observed since 1987 might actually be a part of a nesting cycle that coincides with cyclic beach erosion in Guiana (Schulz 1975). Researchers think that the cycle of erosion and reformation of beaches may have changed where leatherbacks nest throughout this region. The idea of shifting nesting beach locations was supported by increased nesting in Suriname,[12] while the number of nests was declining at beaches in Guiana (Hilterman et al. 2003). Though this information suggested the long-term trend for the overall Suriname and French Guiana population was increasing. A more recent cycle of nesting declines from 2008-2017, as high at 31% annual decline in the Awala-Yalimapo area of French Guiana and almost 20% annual declines in Guyana, has changed the long-term nesting trends in the region negative as described above (Northwest Atlantic Leatherback Working Group 2018).

The Western Caribbean stock includes nesting beaches from Honduras to Colombia. Across the Western Caribbean, nesting is most prevalent in Costa Rica, Panama, and the Gulf of Uraba in Colombia (Duque et al. 2000). The Caribbean coastline of Costa Rica and extending through Chiriquí Beach, Panama, represents the fourth largest known leatherback rookery in the world (Troëng et al. 2004). Examination of data from index nesting beaches in Tortuguero, Gandoca, and Pacuaré in Costa Rica indicate that the nesting population likely was not growing over the 1995-2005 time series (TEWG 2007a). Other modeling of the nesting data for Tortuguero indicates a possible 67.8% decline between 1995 and 2006 (Troëng et al. 2007). Tiwari et al. (2013) report an estimated three-generation abundance change of -72%, -24%, and +6% for Tortuguero, Gandoca, and Pacuare, respectively. Further decline of almost 6% annual geometric mean from 2008-2017 reflects declines in nesting beaches throughout this stock (Northwest Atlantic Leatherback Working Group 2018).

Nesting data for the Northern Caribbean stock is available from Puerto Rico, St. Croix (U.S. Virgin Islands), 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 2007a). Tiwari et al. (2013) report an estimated three-generation abundance change of -4% and +5,583% at Culebra and Fajardo, respectively. At the primary nesting beach on St. Croix, the Sandy Point National Wildlife Refuge, nesting has varied 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 2007a). From 2006-2010, Tiwari et al. (2013) report an annual growth rate of +7.5% in St. Croix and a three-generation abundance change of +1,058%. 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 2007a). The nesting trend reversed course later, with an annual geometric mean decline of 10% from 2008-2017 driving the long-term trend (1990-2017) down to a 2% annual decline (Northwest Atlantic Leatherback Working Group 2018).

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

[12] Leatherback nesting in Suriname increased by more than 10,000 nests per year since 1999 with a peak of 30,000 nests in 2001.

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 (2007a) estimated a significant annual nesting growth rate of 1.17% between 1989 and 2005. FWC Index Nesting Beach Survey Data generally indicates biennial peaks in nesting abundance beginning in 2007 (Figure 8 and Table 3). A similar pattern was also observed statewide (Table 3). 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. Overall, the trend showed growth on Florida's east coast beaches. Tiwari et al. (2013) report an annual growth rate of 9.7% and a three-generation abundance change of +1,863%. However, in recent years nesting has declined on Florida beaches, with 2017 hitting a decade-low number, with a partial rebound in 2018. The annual geometric mean trend for Florida has been a decline of almost 7% from 2008-2017, but the long-term trend (1990-2017) remains positive with an annual geometric mean increase of over 9% (Northwest Atlantic Leatherback Working Group 2018).

Table 3. Number of Leatherback Sea Turtle Nests in Florida

Nests Recorded	2011	2012	2013	2014	2015	2016	2017	2018
Index Nesting Beaches	625	515	322	641	489	319	205	316
Statewide	1,653	1,712	896	1,604	1,493	1,054	663	949

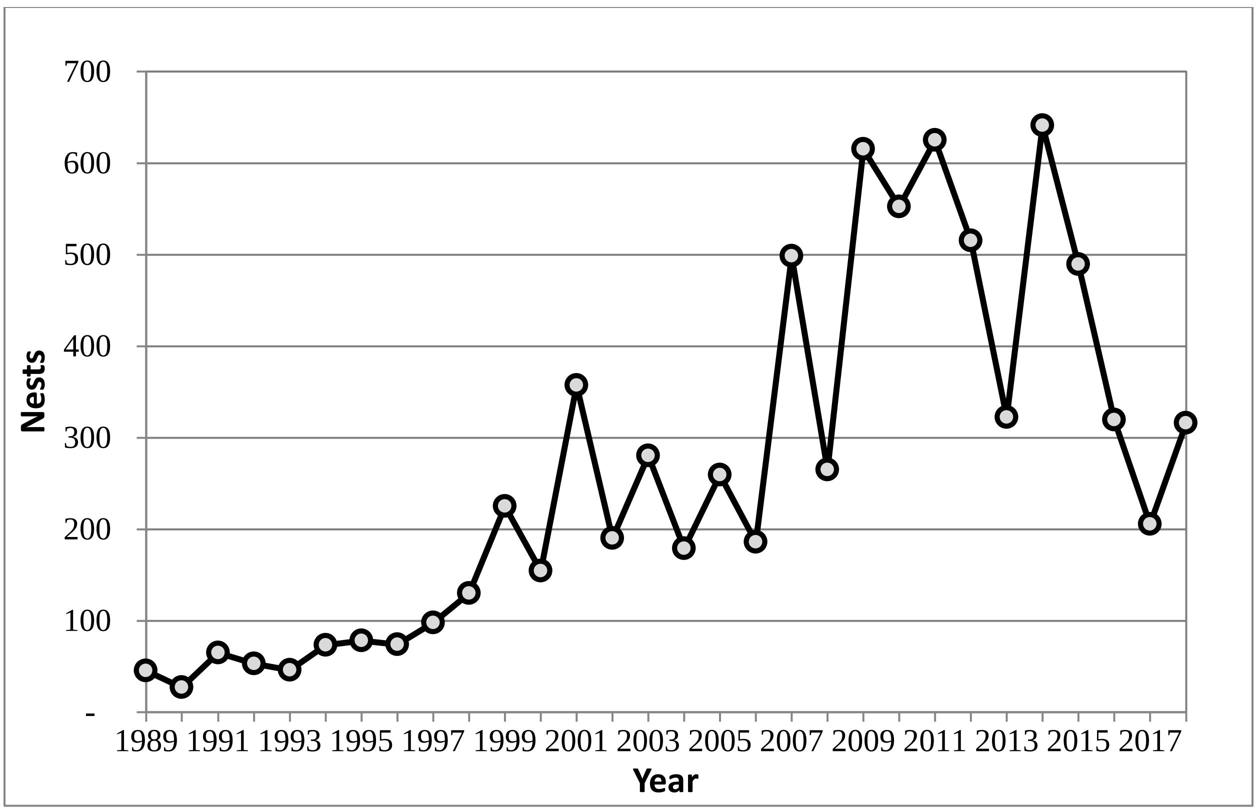

Figure 8. Leatherback sea turtle nesting at Florida index beaches since 1989

The West African nesting stock of leatherbacks is large and important, but it is a 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. Gabon has a very large amount of leatherback nesting, with at least 30,000 nests laid along its coast in a single season

(Fretey et al. 2007). Fretey et al. (2007) 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 2007a).

Two other small but growing stocks nest on the beaches of Brazil and South Africa. Based on the data available, TEWG (2007a) determined that between 1988 and 2003, there was a positive annual average growth rate between 1.07% and 1.08% for the Brazilian stock. TEWG (2007a) estimated an annual average growth rate between 1.04% and 1.06% for the South African stock.

Because the available nesting information is inconsistent, it is difficult to estimate the total population size for Atlantic leatherbacks. Spotila et al. (1996) characterized the entire Western Atlantic population as stable at best and estimated a population of 18,800 nesting females. Spotila et al. (1996) further estimated that the adult female leatherback population for the entire Atlantic basin, including all nesting beaches in the Americas, the Caribbean, and West Africa, was about 27,600 (considering both nesting and interesting females), with an estimated range of 20,082-35,133. This is consistent with the estimate of 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) determined by the TEWG (2007a). The TEWG (2007a) also determined that at the time of their publication, leatherback sea turtle populations in the Atlantic were all stable or increasing with the exception of the Western Caribbean and West Africa populations. A later review by NMFS USFWS (2013) suggested the leatherback nesting population was stable in most nesting regions of the Atlantic Ocean. However, as described earlier, the NW Atlantic population has experienced declines over the near term (2008-2017), often severe enough to reverse the longer term trends to negative where increases had previously been seen (Northwest Atlantic Leatherback Working Group 2018). Given the relatively large size of the NW Atlantic population, it is likely that the overall Atlantic leatherback trend is no longer increasing.

Threats

Leatherbacks face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (plastics, petroleum products, petrochemicals, etc.), ecosystem alterations (nesting beach development, beach nourishment and shoreline stabilization, vegetation changes, etc.), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 3.2.1.1; the remainder of this section will expand on a few of the aforementioned threats and how they may specifically impact leatherback sea turtles.

Of all sea turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear, especially gillnet and pot/trap lines. This vulnerability may be because 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, their method of locomotion, and/or their attraction to the lightsticks used to attract target species in longline fisheries. From 1990-2000, 92 entangled leatherbacks were reported from New York through Maine and many other stranded individuals exhibited evidence of prior entanglement (Dwyer et al. 2003). Zug and Parham (1996) point out that a combination of the loss of long-lived adults in fishery-related mortalities and a lack of recruitment from intense egg harvesting in some areas has caused a sharp decline in leatherback sea turtle populations. This represents a significant threat to survival and recovery of the species worldwide.

Leatherback sea turtles may also be more susceptible to marine debris ingestion than other sea turtle species due to their predominantly pelagic existence and the tendency of floating debris to concentrate in convergence zones that adults and juveniles use for feeding and migratory purposes (Shoop and Kenney 1992, Lutcavage et al. 1997). The stomach contents of leatherback sea turtles revealed that a substantial percentage (33.8% or 138 of 408 cases examined) contained some form of plastic debris (Mrosovsky et al. 2009). Blocking of the gut by plastic to an extent that could have caused death was evident in 8.7% of all leatherbacks that ingested plastic (Mrosovsky et al. 2009). Mrosovsky et al. (2009) also note that in a number of cases, the ingestion of plastic may not cause death outright, but could cause the animal to absorb fewer nutrients from food, eat less in general, etc.– factors which could cause other adverse effects. The presence of plastic in the digestive tract suggests that leatherbacks might not be able to distinguish between prey items and forms of debris such a plastic bags (Mrosovsky et al. 2009). Balazs (1985) speculated that the plastic object might resemble a food item by its shape, color, size, or even movement as it drifts about, and therefore induce a feeding response in leatherbacks.

As discussed in Section 3.2.1.1, global climate change can be expected to have various impacts on all sea turtles, including leatherbacks. Global climate change is likely to also influence the distribution and abundance of jellyfish, the primary prey item of leatherbacks (NMFS and USFWS 2007f). Several studies have shown leatherback distribution is influenced by jellyfish abundance ((Houghton et al. 2006, Witt et al. 2006, Witt et al. 2007); however, more studies need to be done to monitor how changes to prey items affect distribution and foraging success of leatherbacks so population-level effects can be determined.

While oil spill impacts are discussed generally for all species in Section 3.2.1.1, specific impacts of the DWH oil spill on leatherback sea turtles are considered here. Available information indicates leatherback sea turtles (along with hawksbill turtles) were likely directly affected by the oil spill. Leatherbacks were documented in the spill area, but the number of affected leatherbacks was not estimated due to a lack of information compared to other species. But given that the northern Gulf of Mexico is important habitat for leatherback migration and foraging (TEWG 2007), and documentation of leatherbacks in the DWH oil spill zone during the spill period, it was concluded that leatherbacks were exposed to DWH oil, and some portion of those exposed leatherbacks likely died. Potential DWH-related impacts to leatherback sea turtles include direct oiling or contact with dispersants from surface and subsurface oil and dispersants, inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred. Although adverse impacts likely occurred to leatherbacks, the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event may be relatively low. Thus, a population-level impact may not have occurred due to the widespread distribution and nesting location outside of the Gulf of Mexico for this species.

3.2.2 Hawksbill Sea Turtle (*Eretmochelys imbricata*)

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

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

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

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

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

Life History Information

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

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

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

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

Status and Population Dynamics
There are currently no reliable estimates of population abundance and trends for non-nesting hawksbills at the time of this consultation; therefore, nesting beach data is currently the primary information source for evaluating trends in global abundance. Most hawksbill populations around the globe are either declining, depleted, and/or remnants of larger aggregations (NMFS and USFWS 2007c). The largest nesting population of hawksbills occurs in Australia where approximately 2,000 hawksbills nest off the northwest coast and about 6,000-8,000 nest off the Great Barrier Reef each year (Spotila 2004). Additionally, about 2,000 hawksbills nest each year in Indonesia and 1,000 nest in the Republic of Seychelles (Spotila 2004). In the United States, hawksbills typically laid about 500-1,000 nests on Mona Island, Puerto Rico in the past (Diez and Van Dam 2007), but the numbers appear to be increasing, as the Puerto Rico Department of Natural and Environmental Resources counted nearly 1,600 nests in 2010 (PRDNER nesting data). Another 56-150 nests are typically laid on Buck Island off St. Croix (Meylan 1999c, Mortimer and Donnelly 2008a). Nesting also occurs to a lesser extent on beaches on Culebra Island and Vieques Island in Puerto Rico, the mainland of Puerto Rico, and additional beaches on St. Croix, St. John, and St. Thomas, U.S. Virgin Islands.

Mortimer and Donnelly (2008a) reviewed nesting data for 83 nesting concentrations organized among 10 different ocean regions (i.e., Insular Caribbean, Western Caribbean Mainland, Southwestern Atlantic Ocean, Eastern Atlantic Ocean, Southwestern Indian Ocean, Northwestern Indian Ocean, Central Indian Ocean, Eastern Indian Ocean, Western Pacific Ocean, Central Pacific Ocean, and Eastern Pacific Ocean). They determined historic trends (i.e., 20-100 years ago) for 58 of the 83 sites, and also determined recent abundance trends (i.e., within the past 20 years) for 42 of the 83 sites. Among the 58 sites where historic trends could be determined, all showed a declining trend during the long-term period. Among the 42 sites where recent (past 20 years) trend data were available, 10 appeared to be increasing, 3 appeared to be stable, and 29 appeared to be decreasing. With respect to regional trends, nesting populations in the Atlantic (especially in the Insular Caribbean and Western Caribbean Mainland) are generally doing better than those in the Indo-Pacific regions. For instance, 9 of the 10 sites that showed recent increases are located in the Caribbean. Buck Island and St. Croix's East End beaches support 2 remnant populations of between 17-30 nesting females per season (Hillis and Mackay 1989, Mackay 2006). While the proportion of hawksbills nesting on Buck Island represents a small proportion of the total hawksbill nesting occurring in the greater Caribbean region, Mortimer and Donnelly (2008a) report an increasing trend in nesting at that site based on data collected from 2001-2006. The conservation measures implemented when BIRNM was expanded in 2001 most likely explains this increase.

Nesting concentrations in the Pacific Ocean appear to be performing the worst of all regions despite the fact that the region currently supports more nesting hawksbills than either the Atlantic or Indian Oceans (Mortimer and Donnelly 2008a). While still critically low in numbers, sightings of hawksbills in the eastern Pacific appear to have been increasing since 2007, though some of that increase may be attributable to better observations (Gaos et al. 2010). More information about site-specific trends can be found in the most recent 5-year status review for the species (NMFS and USFWS 2007c).

Threats

Hawksbills are currently subjected to the same suite of threats on both nesting beaches and in the marine environment that affect other sea turtles (e.g., interaction with federal and state fisheries, coastal construction, oil spills, climate change affecting sex ratios) as discussed in Section 3.2.1.1. There are also specific threats that are of special emphasis, or are unique, for hawksbill sea turtles discussed in further detail below.

While oil spill impacts are discussed generally for all species in Section 3.2.1.1, specific impacts of the DWH spill on hawksbill turtles have been estimated. Hawksbills made up 2.2% (8,850) of small juvenile sea turtle (of those that could be identified to species) exposures to oil in offshore areas, with an estimate of 615 to 3,090 individuals dying as a result of the direct exposure (DWH Trustees 2015). No quantification of large benthic juveniles or adults was made. Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred. Although adverse impacts occurred to hawksbills, the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event is relatively low, and thus a population-level impact is not believed to have occurred due to the widespread distribution and nesting location outside of the Gulf of Mexico for this species.

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

The continuing demand for the hawksbills' shells as well as other products derived from the species (e.g., leather, oil, perfume, and cosmetics) represents an ongoing threat to its recovery. The British Virgin Islands, Cayman Islands, Cuba, Haiti, and the Turks and Caicos Islands (United Kingdom) all permit some form of legal take of hawksbill sea turtles. In the northern Caribbean, hawksbills continue to be harvested for their shells, which are often carved into hair clips, combs, jewelry, and other trinkets (Márquez M. 1990, Stapleton and Stapleton 2006).

Additionally, hawksbills are harvested for their eggs and meat, while whole, stuffed sea turtles are sold as curios in the tourist trade. Hawksbill sea turtle products are openly available in the Dominican Republic and Jamaica, despite a prohibition on harvesting hawksbills and their eggs (Fleming 2001). Up to 500 hawksbills per year from 2 harvest sites within Cuba were legally captured each year until 2008 when the Cuban government placed a voluntary moratorium on the sea-turtle fishery (Carillo et al. 1999, Mortimer and Donnelly 2008a). While current nesting trends are unknown, the number of nesting females is suspected to be declining in some areas (Carillo et al. 1999, Moncada et al. 1999). International trade in the shell of this species is prohibited between countries that have signed the Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES), but illegal trade still occurs and remains an ongoing threat to hawksbill survival and recovery throughout its range.

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

3.2.3 Coral

3.2.3.1 General Threats Faced by All Coral Species

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

Several of the most important threats contributing to the extinction risk of corals are related to global climate change. The main concerns regarding impacts of global climate change on coral reefs generally, and on listed corals in particular, are the magnitude and the rapid pace of change in greenhouse gas (GHG) concentrations (e.g., carbon dioxide [CO_2] and methane) and atmospheric warming since the Industrial Revolution in the mid-19th century. These changes are increasing the warming of the global climate system and altering the carbonate chemistry of the ocean (ocean acidification). Ocean acidification affects a number of biological processes in corals, including secretion of their skeletons.

Ocean Warming
Ocean warming is one of the most important threats posing extinction risks to the listed coral species, but individual susceptibility varies among species. The primary observable coral response to ocean warming is bleaching of adult coral colonies, wherein corals expel their

symbiotic algae in response to stress. For many corals, an episodic increase of only 1°C–2°C above the normal local seasonal maximum ocean temperature can induce bleaching. Corals can withstand mild to moderate bleaching; however, severe, repeated, and/or prolonged bleaching can lead to colony death. Coral bleaching patterns are complex, with several species exhibiting seasonal cycles in symbiotic algae density. Thermal stress has led to bleaching and mass mortality in many coral species during the past 25 years.

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

Ocean Acidification

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

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

Diseases

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

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

Trophic Effects of Reef Fishing

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

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

Sedimentation

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

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

Nutrient Enrichment

Elevated nutrient concentrations in seawater affect corals through 2 main mechanisms: direct impacts on coral physiology, and indirect effects through stimulation of other community components (e.g., macroalgal turfs and seaweeds, and filter feeders) that compete with corals for space on the reef. Increased nutrients can decrease calcification; however, nutrients may also enhance linear extension while reducing skeletal density. Either condition results in corals that are more prone to breakage or erosion, but individual species do have varying tolerances to increased nutrients. Anthropogenic nutrients mainly come from point-source discharges (such as rivers or sewage outfalls) and surface runoff from modified watersheds. Natural processes, such as *in situ* nitrogen fixation and delivery of nutrient-rich deep water by internal waves and upwelling, also bring nutrients to coral reefs.

Elkhorn Coral (*Acropora palmata*)

Elkhorn coral was listed as threatened under the ESA in May 2006 (71 FR 26852). In December 2012, NMFS proposed changing its status from threatened to endangered (77 FR 73219). On September 10, 2014, NMFS determined that elkhorn coral should remain listed as threatened (79 FR 53851).

Species Description and Distribution

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

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

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

Life History Information

Relative to other corals, elkhorn coral has a high growth rate allowing acroporid reef growth to keep pace with past changes in sea level (Fairbanks 1989). Growth rates, measured as skeletal extension of the end of branches, range from approximately 2-4 in (4-11 cm) per year (*Acropora* Biological Review Team 2005). However, growth rates in Curaçao have been reported to be slower today than they were several decades ago (Brainard et al. 2011). Annual growth has been found to be dependent on the size of the colony, and new recruits and juveniles typically grow at slower rates. Additionally, stressed colonies and fragments may also exhibit slower growth.

Elkhorn coral is a hermaphroditic broadcast spawning[13] species that reproduces sexually after the full moon of July, August, and/or September, depending on location and timing of the full moon (Acropora Biological Review Team 2005). Split spawning (spawning over a 2 month period) has been reported from the Florida Keys (Fogarty et al. 2012). The estimated size at sexual maturity is approximately 250 in^2 (1,600 cm^2), and growing edges and encrusting base areas are not fertile (Soong and Lang 1992). Larger colonies have higher fecundity per unit area, as do the upper branch surfaces (Soong and Lang 1992). Although self-fertilization is possible, elkhorn coral is largely self-incompatible (Baums et al. 2005a, Fogarty et al. 2012).

Sexual recruitment rates are low, and this species is generally not observed in coral settlement studies in the field. Rates of post-settlement mortality after 9 months are high based on settlement experiments (Szmant and Miller 2006). Laboratory studies have found that certain species of crustose-coralline algae facilitate larval settlement and post-settlement survival (Ritson-Williams et al. 2010). Laboratory experiments have shown that some individuals (i.e., genotypes) are sexually incompatible (Baums et al. 2013) and that the proportion of eggs fertilized increases with higher sperm concentration (Fogarty et al. 2012). Experiments using gametes collected in Florida and Belize showed that Florida corals had lower fertilization rates than those from Belize, possibly due to genotype incompatibilities (Fogarty et al. 2012).

Reproduction occurs primarily through asexual fragmentation that produces multiple colonies that are genetically identical (Bak and Criens 1982, Highsmith 1982, Wallace 1985, Lirman 2000, Miller et al. 2007). Storms can be a method of producing fragments to establish new colonies (Fong and Lirman 1995). Fragmentation is an important mode of reproduction in many

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

reef-building corals, especially for branching species such as elkhorn coral (Highsmith 1982, Wallace 1985, Lirman 2000). However, in the Florida Keys where populations have declined, there have been reports of failure of asexual recruitment due to high fragment mortality after storms (Williams et al. 2008, Williams and Miller 2010, Porter et al. 2012).

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

Status and Population Dynamics

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

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

The western Caribbean is characterized by genetically poor populations with lower densities (0.13 ± 0.08 colonies per m^2). The eastern Caribbean populations are characterized by denser (0.30 ± 0.21 colonies per m^2), genotypically richer stands (Baums et al. 2006a). Baums et al. (2006a) concluded that the western Caribbean had higher rates of asexual recruitment and that the eastern Caribbean had higher rates of sexual recruitment. They postulated these geographic differences in the contribution of reproductive modes to population structure may be related to habitat characteristics, possibly the amount of shelf area available.

Genotypic diversity is highly variable. At two sites in the Florida Keys, only one genotype per site was detected out of 20 colonies sampled at each site (Baums et al. 2005b). In contrast, all 15 colonies sampled in Navassa had unique genotypes (Baums et al. 2006a). Some sites have relatively high genotypic diversity such as in Los Roques, Venezuela (118 unique genotpyes out of 120 samples; Zubillaga et al. 2008) and in Bonaire and Curaçao (18 genotypes of 22 samples and 19 genotypes of 20 samples, respectively; Baums et al. 2006a). In the Bahamas, about one third of the sampled colonies were unique genotypes, and in Panama between 24% and 65% of the sampled colonies had unique genotypes, depending on the site (Baums et al. 2006a).

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

Elkhorn coral was historically one of the dominant species on Caribbean reefs, forming large, monotypic thickets and giving rise to the “elkhorn” zone in classical descriptions of Caribbean reef morphology (Goreau 1959). However, mass mortality, apparently from white-band disease (Aronson and Precht 2001), spread throughout the Caribbean in the mid-1970s to mid-1980s and precipitated widespread and radical changes in reef community structure (Brainard et al. 2011). This mass mortality occurred throughout the range of the species within all Caribbean countries and archipelagos, even on reefs and banks far from localized human influence (Aronson and Precht 2001, Wilkinson 2008). In addition, continuing coral mortality from periodic acute events such as hurricanes, disease outbreaks, and mass bleaching events added to the decline of elkhorn coral (Brainard et al. 2011). In locations where historic quantitative data are available (Florida, Jamaica, U.S. Virgin Islands), there was a reduction of greater than 97% between the 1970s and early 2000s in elkhorn coral populations (Acropora Biological Review Team 2005).

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

There is some density data available for elkhorn corals in Florida, Puerto Rico, the US Virgin Islands, and Cuba. In Florida, elkhorn coral was detected at 0% to 78% of the sites surveyed between 1999 and 2017. Average density ranged from 0.001 to 0.12 colonies per m^2 (NOAA, unpublished data). Elkhorn coral was encountered less frequently during benthic surveys in the US Virgin Islands from 2002 to 2017. It was observed at 0 to 7% of surveyed reefs, and average density ranged from 0.001 to 0.01 colonies per m^2 (NOAA, unpublished data). Maximum elkhorn coral density at ten sites in St. John, U.S. Virgin Islands was 0.18 colonies per m^2 (Muller et al. 2014). In Puerto Rico, average density ranged from 0.002 to 0.09 colonies per m^2 in surveys conducted between 2008 and 2018, and elkhorn coal was observed on 1% to 27% of surveyed sites (NOAA, unpublished data). Density estimates from sites in Cuba range from 0.14 colonies per m^2 (Alcolado et al. 2010) to 0.18 colonies per m^2 (González-Díaz et al. 2010).

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

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

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

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

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

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

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

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

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

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

Threats

A summary of threats to all corals is provided in Section 3.2.3.1 General Threats Faced by All Coral Species. Detailed information on the threats to elkhorn coral can be found in the Final Listing rule (79 FR 53851; September 10, 2014); however, a brief summary is provided here. Elkhorn coral is highly susceptible to ocean warming, disease, ocean acidification, sedimentation, and nutrients, and susceptible to trophic effects of fishing, depensatory population effects from rapid, drastic declines and low sexual recruitment, and anthropogenic and natural abrasion and breakage.

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

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

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

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

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

Summary of Status

The species has undergone substantial population decline and decreases in the extent of occurrence throughout its range due mostly to disease. There is evidence of synergistic effects of threats for this species including disease outbreaks following bleaching events. Elkhorn coral is highly susceptible to a number of threats, and cumulative effects of multiple threats are likely to exacerbate vulnerability to extinction. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because elkhorn coral is limited to an area with high localized human impacts and predicted increasing threats. Elkhorn coral occurs in turbulent water on the back reef, fore reef, reef crest, and spur and groove zone in water ranging from 1 to 30 m in depth. This moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that will, on local and regional scales, experience highly variable thermal regimes and ocean chemistry at any given point in time. Elkhorn coral has low sexual recruitment rates, which exacerbates vulnerability to extinction due to decreased ability to recover from mortality events when all colonies at a site are extirpated. In contrast, its fast growth rates and propensity for formation of clones through asexual fragmentation enables it to expand between rare events of sexual recruitment and increases its potential for local recovery from mortality events, thus moderating vulnerability to extinction. Its abundance and life history characteristics, combined with spatial variability in ocean warming and acidification across the species' range, moderate vulnerability to extinction because the threats are non-uniform. Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time. We anticipate that the population abundance is likely to decrease in the future with increasing threats.

3.2.3.2 Staghorn Coral (*Acropora cervicornis*)

Staghorn coral was listed as threatened under the ESA in May 2006 (71 FR 26852). In December 2012, NMFS proposed changing its status from threatened to endangered (77 FR 73219). On September 10, 2014, NMFS determined that staghorn coral should remain listed as threatened (79 FR 53851).

Species Description and Distribution

Staghorn coral is characterized by antler-like colonies with straight or slightly curved, cylindrical branches. The diameter of branches ranges from 0.1-2 in (0.25-5 cm; Lirman et al. 2010), and linear branch growth rates have been reported to range between 1.2-4.5 in (3-11.5 cm) per year (*Acropora* Biological Review Team 2005). The species can exist as isolated branches, individual colonies up to about 5 ft (1.5 m) diameter, and thickets comprised of multiple colonies that are difficult to distinguish from one another (*Acropora* Biological Review Team 2005).

Staghorn coral is distributed throughout the Caribbean Sea, in the southwestern Gulf of Mexico, and in the western Atlantic Ocean. The fossil record indicates that during the Holocene epoch, staghorn coral was present as far north as Palm Beach County in southeast Florida (Lighty et al. 1978), which is also the northern extent of its current distribution (Goldberg 1973).

Staghorn coral commonly occurs in water ranging from 16 to 65 ft (5 to 20 m) in depth, though it occurs in depths of 16-30 m at the northern extent of its range, and has been rarely found to 60 m in depth. Staghorn coral naturally occurs on spur and groove, bank reef, patch reef, and transitional reef habitats, as well as on limestone ridges, terraces, and hard bottom habitats (Goldberg 1973, Gilmore and Hall 1976, Cairns 1982, Davis 1982, Jaap 1984, Wheaton and Jaap 1988, Miller et al. 2008). Historically it grew in thickets in water ranging from approximately 16-65 ft (5-20 m) in depth; though it has rarely been found to approximately 195 ft (60 m; Davis 1982, Jaap 1984, Schuhmacher and Zibrowius 1985, Wheaton and Jaap 1988, Jaap et al. 1989). At the northern extent of its range, it grows in deeper water (~53-99 ft [16-30 m]; Goldberg 1973). Historically, staghorn coral was one of the primary constructors of mid-depth (approximately 33-50 ft [10-15 m]) reef terraces in the western Caribbean, including Jamaica, the Cayman Islands, Belize, and some reefs along the eastern Yucatan peninsula (Adey 1978). In the Florida Keys, staghorn coral occurs in various habitats but is most prevalent on patch reefs as opposed to their former abundance in deeper fore-reef habitats (i.e., 16-65 ft; Miller et al. 2008). There is no evidence of range constriction, though loss of staghorn coral at the reef level has occurred (*Acropora* Biological Review Team 2005).

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

Life History Information

Relative to other corals, staghorn coral has a high growth rate that have allowed acroporid reef growth to keep pace with past changes in sea level (Fairbanks 1989). Growth rates, measured as skeletal extension of the end of branches, range from approximately 2-4 in (4-11 cm) per year (*Acropora* Biological Review Team 2005). Annual linear extension has been found to be dependent on the size of the colony. New recruits and juveniles typically grow at slower rates. Stressed colonies and fragments may also exhibit slower growth.

Staghorn coral is a hermaphroditic broadcast spawning species[14]. The spawning season occurs several nights after the full moon in July, August, or September depending on location and timing of the full moon, and may be split over the course of more than one lunar cycle (Szmant 1986, Vargas-Angel et al. 2006). The estimated size at sexual maturity is approximately 6 in (17 cm) branch length, and large colonies produce proportionally more gametes than small colonies

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

(Soong and Lang 1992). Basal and branch tip tissue is not fertile (Soong and Lang 1992). Sexual recruitment rates are low, and this species is generally not observed in coral settlement studies. Laboratory studies have found that the presence of certain species of crustose-coralline algae facilitate larval settlement and post-settlement survival (Ritson-Williams et al. 2010).

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

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

Vollmer and Palumbi (2007) examined 22 populations of staghorn coral from 9 regions in the Caribbean (Panama, Belize, Mexico, Florida, Bahamas, Turks and Caicos, Jamaica, Puerto Rico, and Curaçao) and concluded that populations greater than approximately 310 miles (500 km) apart are genetically different from each other with low gene flow across the greater Caribbean. Fine-scale genetic differences have been detected at reefs separated by as little as 1.25 miles (2 km), suggesting that gene flow in staghorn coral may not occur at much smaller spatial scales (Vollmer and Palumbi 2007, Garcia Reyes and Schizas 2010). This fine-scale population structure was greater when considering genes of elkhorn coral were found in staghorn coral due to back-crossing of the hybrid *A. prolifera* with staghorn coral (Vollmer and Palumbi 2007, Garcia Reyes and Schizas 2010). Populations in Florida and Honduras are genetically distinct from each other and other populations in the U.S. Virgin Islands, Puerto Rico, Bahamas, and Navassa (Baums et al. 2010), indicating little to no larval connectivity overall. However, some potential connectivity between the U.S. Virgin Islands and Puerto Rico was detected and also between Navassa and the Bahamas (Baums et al. 2010).

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

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

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

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

A report on the status and trends of Caribbean corals over the last century indicates that the percentage of reefs with staghorn coral present has decreased over time. The frequency of reefs at which staghorn coral was described as the dominant coral has remained stable. The number of reefs with staghorn coral present declined during the 1980s from approximately 50 to 30% of reefs and remained relatively stable at 30% through the 1990s. The number of reefs with staghorn coral present decreased to approximately 20% in 2000-2004 and approximately 10% in 2005-2011 (Jackson et al. 2014).

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

Benthic surveys between 2008 and 2018 in Puerto Rico detected an average density of 0.001 to 0.17 colonies per m^2, and colonies were observed at 4% to 25% of the reefs surveyed (NOAA, unpublished data). Staghorn coral was observed in 21 out of 301 stations between 2011 and 2013 in stratified random surveys designed to detect *Acropora* colonies along the south, southeast, southwest, and west coasts of Puerto Rico (García Sais et al. 2013). Staghorn coral was also observed at 16 sites outside of the surveyed area. The largest colony was 24 in (60 cm) and density ranged from 1-10 colonies per 162 ft^2 (15 m^2; García Sais et al. 2013).

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

Overall, populations appear to consist mostly of isolated colonies or small groups of colonies compared to the vast thickets once prominent throughout its range. Thickets are a prominent feature at only a few known locations. Across the Caribbean, frequency of occurrence has decreased since the 1980s. There are examples of increasing trends in some locations (Dry Tortugas and southeast Florida), but not over larger spatial scales or longer time frames. Population model projections from Honduras at one of the only known remaining thickets indicate the retention of this dense stand under undisturbed conditions. If refuge populations are able to persist, it is unclear whether they would be able to repopulate nearby reefs as observed sexual recruitment is low. Thus, we conclude that the species has undergone substantial population decline and decreases in the extent of occurrence throughout its range. We anticipate that population abundance is likely to decrease in the future with increasing threats.

Threats

A summary of threats to all corals is provided in Section 3.2.2.1 General Threats Faced by All Coral Species. Detailed information on the threats to staghorn coral can be found in the Final Listing rule (79 FR 53851; September 10, 2014); however, a brief summary is provided here. Staghorn coral is highly susceptible to ocean warming, disease, ocean acidification, sedimentation, and nutrients, as well as susceptible to trophic effects of fishing, depensatory population effects from rapid, drastic declines and low sexual recruitment, and anthropogenic and natural abrasion and breakage.

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

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

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

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

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

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

Summary of Status

The species has undergone substantial population decline and decreases in the extent of occurrence throughout its range due mostly to disease. There is evidence of synergistic effects of threats for this species where the effects of increased nutrients are combined with acidification and sedimentation. Staghorn coral is highly susceptible to a number of threats, and cumulative effects of multiple threats are likely to exacerbate vulnerability to extinction. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because staghorn coral is limited to areas with high, localized human impacts and predicted increasing threats. Staghorn coral commonly occurs in water ranging from 5 to 20 m in depth, though it occurs in depths of 16-30 m at the northern extent of its range, and has been rarely found to 60 m in depth. It occurs in spur and groove, bank reef, patch reef, and transitional reef habitats, as well as on limestone ridges, terraces, and hard bottom habitats. This habitat heterogeneity moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef and hard bottom environments that are predicted, on local and regional scales, to experience highly variable thermal regimes and ocean chemistry at any given point in time. Staghorn coral has low sexual recruitment rates, which exacerbates vulnerability to extinction due to decreased ability to recover from mortality events when all colonies at a site are extirpated. In contrast, its fast growth rates and propensity for formation of clones through asexual fragmentation enables it to expand between rare events of sexual recruitment and increases its potential for local recovery from mortality events, thus moderating vulnerability to extinction. Its abundance and life history characteristics, combined with spatial variability in ocean warming and acidification across the species' range, moderate the species' vulnerability to extinction because the threats are non-uniform. Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time. However, we also anticipate that the population abundance is likely to decrease in the future with increasing threats.

3.2.3.3 Lobed and Mountainous Star Coral (*Orbicella annularis, Orbicella faveolata*)

On September 10, 2014, NMFS listed lobed, mountainous and boulder star coral as threatened (79 FR 53851). Lobed star coral (*Orbicella annularis*), mountainous star coral (*Orbicella faveolata*), and boulder star coral (*Orbicella franksi*) are the 3 species in the *Orbicella annularis* star coral complex. These 3 species were formerly in the genus *Montastraea*; however, recent work has reclassified the 3 species in the *annularis* complex to the genus *Orbicella* (Budd et al. 2012). The star coral species complex was historically one of the primary reef framework builders throughout the wider Caribbean. The complex was considered a single species – *Montastraea annularis* – with varying growth forms ranging from columns, to massive boulders, to plates. In the early 1990s, Weil and Knowlton (1994) suggested the partitioning of these growth forms into separate species, resurrecting the previously described taxa, *Montastraea* (now *Orbicella*) *faveolata* and *Montastraea* (now *Orbicella*) *franksi.* The 3 species were differentiated on the basis of morphology, depth range, ecology, and behavior (Weil and Knowton 1994). Subsequent reproductive and genetic studies have supported the partitioning of the *annularis* complex into 3 species.

Some studies report on the star coral species complex rather than individual species since visual distinction can be difficult where colony morphology cannot be discerned (e.g. small colonies or photographic methods). Information from these studies is reported for the species complex. Where species-specific information is available, it is reported. However, information about *Orbicella annularis* published prior to 1994 will be attributed to the species complex since it is dated prior to the split of *Orbicella annularis* into 3 separate species.

Species Description and Distribution

Lobe Star Coral: Lobed star coral colonies grow in columns that exhibit rapid and regular upward growth. In contrast to the other 2 star coral species, margins on the sides of columns are typically dead. Live colony surfaces usually lack ridges or bumps.
Lobed star coral is common throughout the western Atlantic Ocean and greater Caribbean Sea including the Flower Garden Banks, but may be absent from Bermuda. Lobed star coral is reported from most reef environments in depths of approximately 1.5-66 ft (0.5-20 m). The star coral species complex is a common, often dominant component of Caribbean mesophotic (e.g., >100 ft [30 m]) reefs, suggesting the potential for deep refuge across a broader depth range, but lobed star coral is generally described with a shallower distribution.

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

Mountainous Star Coral: Mountainous star coral grows in heads or sheets, the surface of which may be smooth or have keels or bumps. The skeleton is much less dense than in the other 2 star coral species. Colony diameters can reach up to 33 ft (10 m) with heights of 13-16 ft (4-5 m).

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

Life History Information

Lobe Star Coral: The star coral species complex has growth rates ranging from 0.02-0.5 in (0.06-1.2 cm) per year and averaging approximately 0.3 in (1 cm) linear growth per year. The reported growth rate of lobed star coral is 0.4 to 1.2 cm per year (Tomascik 1990, Cruz-Piñón et al. 2003). They grow more slowly in deeper water and in less clear water.

All 3 species of the star coral complex are hermaphroditic broadcast spawners[15], with spawning concentrated on 6-8 nights following the full moon in late August, September, or early October depending on location and timing of the full moon. All 3 species are largely self-incompatible (Knowlton et al. 1997, Szmant et al. 1997). Further, mountainous star coral is largely reproductively incompatible with boulder star coral and lobed star coral, and it spawns about 1-2 hours earlier. Fertilization success measured in the field was generally below 15% for all 3

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

species, as it is closely linked to the number of colonies concurrently spawning. Lobed star coral is reported to have slightly smaller egg size and potentially smaller size/age at first reproduction that the other 2 species of the *Orbicella* genus. In Puerto Rico, minimum size at reproduction for the star coral species complex was 12 in^2 (83 cm^2).

Successful recruitment by the star coral complex species has seemingly always been rare. Only a single recruit of *Orbicella* was observed over 18 years of intensive observation of 130 ft^2 (12 m^2) of reef in Discovery Bay, Jamaica. Many other studies throughout the Caribbean also report negligible to absent recruitment of the species complex.

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

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

Status and Population Dynamics

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

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

Colony density varies by habitat and location, and ranges from less than 0.1 to greater than 1 colony per approximately 100 ft^2 (10 m^2). Benthic surveys along the Florida Reef Tract between 1999 and 2017 recorded an average density of 0.01 to 0.09 colonies per m^2, and lobed star coral was observed at 4% to 16% of surveyed sites (NOAA, unpublished data). Average density of lobed star corals in Puerto Rico ranged from 0.01 to 0.08 colonies per m^2 in surveys conducted between 2008 and 2018 and was observed at 9% to 63% of surveyed sites (NOAA, unpublished data). In the US Virgin Islands, average density ranged from 0.03 to 0.21 colonies per m^2 in benthic surveys conducted between 2002 and 2017, and lobed star coral was observed at 25% to 54% of surveyed sites (NOAA, unpublished data). In the Flower Garden Banks, limited surveys detected lobed star corals at none to 24% of surveyed sites, and density was recorded as 0.1 colonies per m^2 in 2010 and 0.01 colonies per m^2 in 2013 (NOAA, unpublished data). Off southwest Cuba on remote reefs, average lobed star coral density was 0.31 colonies per approximately 108 ft^2 (10 m^2) at 38 reef-crest sites and 1.58 colonies per approximately 108 ft^2 (10 m^2) at 30 reef-front sites. Colonies with partial mortality were far more frequent than those with no partial mortality, which only occurred in the size class less than 40 in (100 cm) (Alcolado et al. 2010).

Recent events have greatly impacted coral populations in Florida and the US Caribbean. An unprecedented, multi-year disease event, which began in 2014, swept through Florida and caused massive mortality from St. Lucie Inlet in Martin County to Looe Key in the lower Florida Keys. The effects of this widespread disease have been severe, causing mortality of millions of coral colonies across several species. At study sites in southeast Florida, prevalence of disease was recorded at 67% of all coral colonies and 81% of colonies of those species susceptible to the disease (Precht et al. 2016). Lobed star coral was one of the species in surveys that showed the highest prevalence of disease, and populations were reduced to < 25% of the initial population size (Precht et al. 2016).

Hurricanes Irma and Maria caused substantial damage in Florida, Puerto Rico, and the US Virgin Islands in 2017. Hurricane impacts included large, overturned and dislodged coral heads and extensive burial and breakage. At 153 survey locations in Puerto Rico, approximately 43-44% of lobed star corals were impacted (NOAA 2018). In Florida, approximately 80% of lobed star corals surveyed at 57 sites were impacted (Florida Fish and Wildlife Conservation Commission, unpublished data). Survey data are not available for the US Virgin Islands, though qualitative observations indicate that damage was also widespread but variable by site.

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

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

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

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

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

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

Lobed star coral was historically considered to be one of the most abundant species in the Caribbean (Weil and Knowton 1994). Percent cover has declined by 37% to 90% over the past several decades at reefs at Jamaica, Belize, Florida Keys, The Bahamas, Bonaire, Cayman Islands, Curaçao, Puerto Rico, U.S. Virgin Islands, and St. Kitts and Nevis. Although star coral remains common in occurrence, abundance has decreased in some areas by 19% to 57%, and shifts to smaller size classes have occurred in locations such as Jamaica, Colombia, The Bahamas, Bonaire, Cayman Islands, Puerto Rico, U.S. Virgin Islands, and St. Kitts and Nevis. At some reefs, a large proportion of the population is comprised of non-fertile or less-reproductive size classes. Several population projections indicate population decline in the future is likely at specific sites, and local extirpation is possible within 25-50 years at conditions of high mortality, low recruitment, and slow growth rates. Although lobed star coral is still common throughout the Caribbean, substantial population decline has occurred. The buffering capacity of lobed star coral's life history strategy that has allowed it to remain abundant has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. Population abundance is likely to decrease in the future with increasing threats.

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

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

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

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

Recent events have greatly impacted coral populations in Florida and the US Caribbean. An unprecedented, multi-year disease event, which began in 2014, swept through Florida and caused massive mortality from St. Lucie Inlet in Martin County to Looe Key in the lower Florida Keys. The effects of this widespread disease have been severe, causing mortality of millions of coral colonies across several species, including mountainous star coral. At study sites in southeast Florida, prevalence of disease was recorded at 67% of all coral colonies and 81% of colonies of those species susceptible to the disease (Precht et al. 2016).

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

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

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

Population trend data exists for several locations. At 9 sites off Mona and Desecheo Islands, Puerto Rico, no species extirpations were noted at any site over 10 years of monitoring between 1998 and 2008 (Bruckner and Hill 2009). Both mountainous star coral and lobed star coral sustained large losses during the period. The number of colonies of mountainous star coral decreased by 36% and 48% at Mona and Desecheo Islands, respectively (Bruckner and Hill 2009). In 1998, 27% of all corals at 6 sites surveyed off Mona Island were mountainous star coral colonies, but this statistic decreased to approximately 11% in 2008 (Bruckner and Hill 2009). At Desecheo Island, 12% of all coral colonies were mountainous star coral in 2000, compared to 7% in 2008.

In a survey of 185 sites in 5 countries (Bahamas, Bonaire, Cayman Islands, Puerto Rico, and St. Kitts and Nevis) between 2010 and 2011, size of mountainous star coral colonies was significantly greater than boulder star coral and lobed star coral. The total mean partial mortality of mountainous star coral at all sites was 38%. The total live area occupied by mountainous star coral declined by a mean of 65%, and mean colony size declined from 43 ft^2 to 15 ft^2 (4005 cm^2 to 1413 cm^2). At the same time, there was a 168% increase in small tissue remnants less than 5 ft^2 (500 cm^2), while the proportion of completely live large (1.6 ft^2 to 32 ft^2 [1,500- 30,000 cm^2]) colonies decreased. Mountainous star coral colonies in Puerto Rico were much larger and sustained higher levels of mortality compared to the other 4 countries. Colonies in Bonaire were also large, but they experienced much lower levels of mortality. Mortality was attributed primarily to outbreaks of white plague and yellow band disease, which emerged as corals began recovering from mass bleaching events. This was followed by increased predation and removal of live tissue by damselfish to cultivate algal lawns (Bruckner 2012).

Overall, it appears that populations of mountainous star coral have been decreasing. Population decline has occurred over the past few decades with a 65% loss in mountainous star coral cover across 5 countries. Losses of mountainous star coral from Mona and Descheo Islands, Puerto Rico include a 36-48% reduction in abundance and a decrease of 42-59% in its relative abundance (i.e., proportion relative to all coral colonies). High partial mortality of colonies has led to smaller colony sizes and a decrease of larger colonies in some locations such as The Bahamas, Bonaire, Puerto Rico, Cayman Islands, and St. Kitts and Nevis. We conclude that mountainous star coral has declined and that the buffering capacity of mountainous star coral's life history strategy, which has allowed it to remain abundant, has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. We also conclude that the population abundance is likely to decrease in the future with increasing threats.

Threats

A summary of threats to all corals is provided in Section 3.2.2.1 General Threats Faced by All Coral Species. Detailed information on the threats to lobed star coral can be found in the Final Listing Rule (79 FR 53851; September 10, 2014); however, a brief summary is provided here. Lobed star and mountainous star coral is highly susceptible to ocean warming, disease, ocean acidification, sedimentation, and nutrients, and susceptible to trophic effects of fishing.

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

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

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

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

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

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

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

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

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

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

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

Summary of Status

Lobe Star Coral: Lobed star coral has undergone major declines mostly due to warming-induced bleaching and disease. Several population projections indicate population decline in the future is likely at specific sites and that local extirpation is possible within 25-50 years at conditions of high mortality, low recruitment, and slow growth rates. There is evidence of synergistic effects of threats for this species, including disease outbreaks following bleaching events and increased disease severity with nutrient enrichment. Lobed star coral is highly susceptible to a number of threats, and cumulative effects of multiple threats have likely contributed to its decline and exacerbate vulnerability to extinction. Despite high declines, the species is still common and remains one of the most abundant species on Caribbean reefs. Its life history characteristics of large colony size and long life span have enabled it to remain relatively persistent despite slow growth and low recruitment rates, thus moderating vulnerability to extinction. However, the buffering capacity of these life history characteristics is expected to decrease as colonies shift to smaller size classes, as has been observed in locations in the species' range. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because lobed star coral is limited to areas with high localized human impacts and predicted increasing threats. Star coral occurs in most reef habitats 0.5-20 m in depth which moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that are predicted, on local and regional scales, to experience high temperature variation and ocean chemistry at any given point in time. However, we anticipate that the population abundance is likely to decrease in the future with increasing threats.

Mountainous Star Coral: Mountainous star coral has undergone major declines mostly due to warming-induced bleaching and disease. There is evidence of synergistic effects of threats for this species including disease outbreaks following bleaching events and reduced thermal tolerance due to chronic local stressors stemming from land-based sources of pollution. Mountainous star coral is highly susceptible to a number of threats, and cumulative effects of multiple threats have likely contributed to its decline and exacerbate its vulnerability to extinction. Despite high declines, the species is still common and remains one of the most abundant species on Caribbean reefs. Its life history characteristics of large colony size and long life span have enabled it to remain relatively persistent despite slow growth and low recruitment rates, thus moderating vulnerability to extinction. The buffering capacity of these life history characteristics, however, is expected to decrease as colonies shift to smaller size classes as has been observed in locations in its range. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because mountainous star coral is limited to an area with high, localized human impacts and predicted increasing threats. Its depth range of 0.5 m to at least 40 m, possibly up to 90 m, moderates vulnerability to extinction over the foreseeable future because deeper areas of its range will usually have lower temperatures than surface waters, and acidification is generally predicted to accelerate most in waters that are deeper and cooler than those in which the species occurs. Mountainous star coral occurs in most reef habitats, including both shallow and mesophotic reefs, which moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that are predicted, on local and regional scales, to experience highly variable temperatures and ocean chemistry at any given point in time. Its abundance, life history characteristics, and depth distribution, combined with spatial variability in ocean warming and acidification across the species' range, moderate vulnerability to extinction because the threats are non-uniform. Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time. However, we anticipate that the population abundance is likely to decrease in the future with increasing threats.

3.2.3.4 Elkhorn and Staghorn Coral Critical Habitat

On November 26, 2008, a Final Rule designating *Acropora* critical habitat was published in the Federal Register. Within the geographical area occupied by a listed species, critical habitat consists of specific areas on which are found those physical or biological features essential to the conservation of the species. The feature essential to the conservation of *Acropora* species (also known as the essential feature) is substrate of suitable quality and availability in water depths from the mean high water line to 30 m in order to support successful larval settlement, recruitment, and reattachment of fragments. "Substrate of suitable quality and availability" means consolidated hard bottom or dead coral skeletons free from fleshy macroalgae or turf algae and sediment cover. Areas containing this feature have been identified in 4 locations within the jurisdiction of the United States: the Florida area, which comprises approximately 1,329 square miles (3,442 sq km) of marine habitat; the Puerto Rico area, which comprises approximately 1,383 square miles (3,582 sq km) of marine habitat; the St. John/St. Thomas area, which comprises approximately 121 square miles (313 sq km) of marine habitat; and the St. Croix area, which comprises approximately 126 square miles (326 sq km) of marine habitat. The total area covered by the designation is thus approximately 2,959 square miles (7,664 sq km).

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

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

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

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

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

Elkhorn and staghorn corals require hard, consolidated substrate, including attached, dead coral skeleton, devoid of turf or fleshy macroalgae for their larvae to settle. Atlantic and Gulf of Mexico Rapid Reef Assessment Program data from 1997-2004 indicate that although the historic range of both species remains intact, the number and size of colonies and percent cover by both species has declined dramatically in comparison to historic levels (Ginsburg and Lang 2003). Monitoring data from the USVI TCRMP indicate that the 2005 coral bleaching event caused the largest documented loss of coral in USVI since coral monitoring data have been available with a decline of at least 50% of coral cover in waters less than 25 m deep (Smith et al. 2011a). Many of the shallow water coral monitoring stations showed at most a 12% recovery of coral cover by 2011, 6 years after the loss of coral cover due to the bleaching event (Smith et al. 2011a). The lack of coral cover has led to increases in algal cover on area hard bottom, including the critical habitat essential feature.

St. Croix Unit

The St. Croix marine unit, which includes the action area for the proposed project, comprises approximately 126 square miles (mi^2) or 80,640 ac of ESA-designated elkhorn and staghorn coral critical habitat (Figure 9). Of this area, approximately 57,600 ac (90 mi^2), or 71%, are likely to contain the essential features of ESA-designated acroporid coral critical habitat, based on the amount of coral, rock reef, colonized hard bottom, and other coralline communities mapped by NOS's Biogeography Program in 2000 (Kendall et al. 2001). The other areas within the St. Croix marine unit are dominated by sand and unconsolidated bottom, seagrass beds with varying densities of coverage, and uncolonized hard bottoms (Kendall et al. 2001). Of the 57,600-ac area in the St. Croix unit, approximately 7,117.7 ac (11.12 mi^2) are within the 0-5 m depth range that is particularly important to elkhorn corals. It should be noted that elkhorn corals can be found in deeper water (up to 30 meters in backreef environments) but maximum depth of framework construction ranges from 3 to 12 m, and colonies generally do not form thickets below a depth of 5 m (Lighty et al. 1982).

The west end of St. Croix, including the Frederiksted Reef System where the action area for the Amalago Bay project is located, contains approximately 732 ac (1.14 mi^2) of reef and hard bottom that are likely to contain the essential feature for acroporid coral critical habitat in depths of 5 m or less. Staghorn corals are typically found in waters with depths greater than 5 m around St. Croix. Smith et al. (2014) found staghorn corals in waters from 6-18 m in depth, but noted that more colonies are likely present in deeper waters. Toller (2005) found staghorn corals in depths up to 35 m within the Frederiksted Reef System.

Toller (2005) reported occasional colonies of elkhorn coral in waters from 0-3 ft in depth and occasional staghorn coral colonies in waters from 18-35 ft in depth within the Frederiksted Reef System (from King's Corner south of the Frederiksted Pier to Sprat Hole to the north). Toller (2005) found good agreement between the NOS benthic habitat maps and their diver survey, indicating that the use of NOS benthic maps to determine the extent of acroporid coral critical habitat is reasonable. Toller (2005) found approximately 13% of all coral habitat within this reef system had suffered mechanical damage from boats (i.e., damage from anchoring and groundings), mainly cruise ships and other commercial vessels using the Frederiksted Pier, but also recreational vessels (as there is a recreational boat ramp and other facilities in the area of the Frederiksted Pier). USVI monitoring of a point on the Sprat Hole shelf edge reef (Figure 10) that is approximately 1,900 ft directly seaward of the Amalago Bay project found the habitat (corals and hard bottom) was impacted by derelict fishing gear and anchoring of dive vessels (Smith et al. 2011a).

In the Frederiksted area, a portion of the nearshore reef system was removed for the construction of the pier and subsequent dredging operations and pier expansion projects. The offshore portion of the reef in this area has been impacted by large vessel anchoring associated with the use of the pier (Toller 2005). Portions of nearshore and offshore reef in the vicinity of Frederiksted have also been affected by land-based sources of pollution from the developed area of Frederiksted and associated declines in water quality (Toller 2005, Smith et al. 2011b). This and other areas around St. Croix were greatly affected by the 2005 bleaching event and few sites have demonstrated significant recovery in terms of coral cover and abundance (Smith et al. 2011a). Lack of coral recovery has led to increased algal cover of the essential feature in this area.

A monitoring station known as Kings Corner south of the Frederiksted Pier is affected by chronic sedimentation due to sand plumes moving around Sandy Point. The area is also affected by fishing, in particular due to derelict gear, which is prevalent on the reef, and recreational diving (Smith et al. 2011a). As part of the Territorial Coral Reef Monitoring Program, Smith et al. (2011a) also have a monitoring station north of the Amalago Bay project they call Sprat Hole (Figure 10). This area is currently affected by snorkel and dive tours and associated anchoring and fishing and related paraphernalia such as a large amount of derelict fishing gear. At this time, the area of the Frederiksted Reef offshore of the project is relatively unaffected by sediment, that could cover the essential feature, but Smith et al. (2011a) note that any development of the watershed would affect the reef because wave energy is usually low and the currents are usually weak, which would lead to settling of terrigenous sediments on the reef. To the north of the Amalago Bay project, surveys have shown that there are numerous elkhorn colonies in the 0-6 m depth zone in Butler Bay and Ham’s Bluff, most of which have over 50% live tissue on each colony, although this is lower than other sites around St. Croix. Smith et al. (2014) theorized that this lower tissue coverage was due to chronic impacts associated with land-based sources of pollution, including erosion of the road in this area due in part to a quarry operation near Ham’s Bay. These land based sources of pollution can affect the essential feature of *Acropora* critical habitat by covering the feature in sediment.

Elkhorn and staghorn corals require hard, consolidated substrate, including attached, dead coral skeleton, devoid of turf or fleshy macroalgae for their larvae to settle. Atlantic and Gulf of Mexico Rapid Reef Assessment Program data from 1997-2004 indicate that although the historic range of both species remains intact, the number and size of colonies and percent cover by both species has declined dramatically in comparison to historic levels (Lang 2003). Monitoring data from the USVI Territorial Coral Reef Monitoring Program indicate that the 2005 coral bleaching event caused the largest documented loss of coral in USVI since coral monitoring data have been available with a decline of at least 50% of coral cover in waters less than 25 m deep (Smith et al. 2011a). Many of the shallow water coral monitoring stations, including areas with elkhorn corals, showed at most a 12% recovery of coral cover by 2011, 6 years after the loss of coral cover due to the bleaching event (Smith et al. 2011a). Lack of coral cover has led to increases in algal cover on area hard bottom.

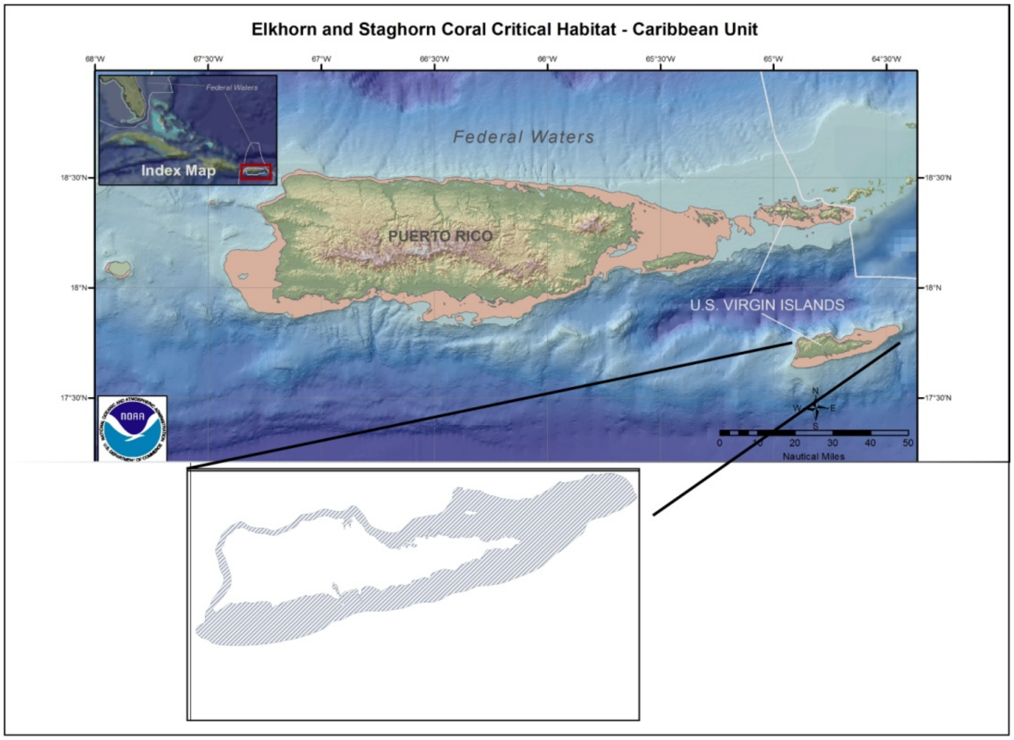

Figure 9. Critical habitat map, with inset of St. Croix unit, for elkhorn and staghorn corals (*Acropora* Critical Habitat map created by NMFS, 2008;

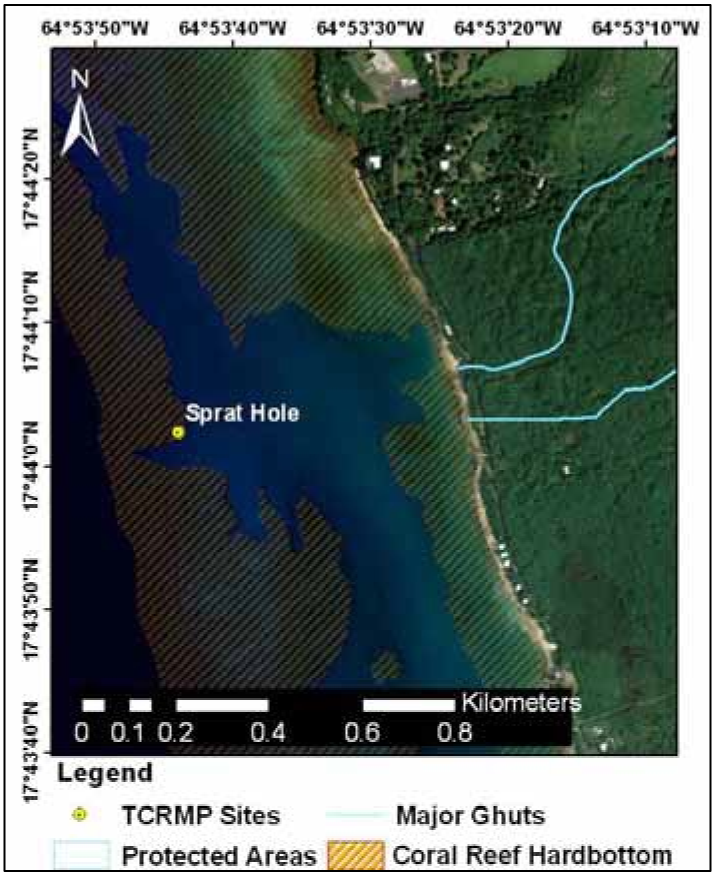

Figure 10. Location of Sprat Hole Territorial Coral Reef Program Sampling Point. The light blue lines on the landward portion of the image represent 2 of the ghuts flowing through the Amalago property (Smith et al. 2011a).

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 by 1-2 orders of magnitude over the past 15-25 years (Rogers et al. 2008a). As noted above, long-term monitoring of a site offshore of the Amalago Bay project indicates that anchoring of recreational vessels and derelict fishing gear are impacting coral habitat (Figure 10; (Smith et al. 2011a)). The monitoring program has also found evidence that land-based sources of pollutants are having negative impacts on nearshore coral reefs by blocking sunlight leading to decreases in photosynthesis and growth of corals, increasing the growth of organisms that compete with corals for space due to increasing nutrient concentrations, and smothering of corals and potential settlement habitat (Smith et al. 2011a). Recent studies from the USVI have found that sediment levels as low as 3 mg per cm^2 per day can cause large increases in the proportion of corals experiencing impairment, partial mortality, and bleaching if sediment is terrigenous in nature (Smith et al. 2013). The majority of nearshore waters around USVI were found to have sediment rates of at least 10 mg per cm^2 per day indicating that the majority of nearshore hard bottoms and reefs around USVI are impacted by sedimentation (Smith et al. 2008). Changes that have affected elkhorn and staghorn corals and led to decreases in the numbers and cover of these species have also affected the essential feature of their critical habitat. Specifically, macroalgal cover has increased (Rogers et al.

2008a) due, in part, to increases in nutrient concentrations (Smith et al. 2001) and sediment cover has increased (Smith et al. 2008). Therefore, we conclude that the essential feature of elkhorn and staghorn coral, which is consolidated hard bottom or dead coral skeletons free from fleshy macroalgae or turf algae and sediment cover, has been adversely affected by land-based sources of pollutants to nearshore waters around the USVI. The impacts have resulted in a fragmented patchwork of habitat containing the essential feature capable of supporting settlement of coral larve, due to the distances between sutiable hardbottom.

McLaughlin et al. (2002) found that when distributions of coral species become isolated because of habitat loss, populations become more vulnerable to climate change and other threats. The loss of habitat patches will affect the availability of areas for coral larvae to settle. Larvae are only viable for a short time so larger distances between areas of suitable habitat for elkhorn corals make settlement and growth less likely. Toller (2005) found that both elkhorn and staghorn corals were sparsely distributed along the entire Frederiksted Reef System with elkhorn most common in shallow waters up to approximately 5 m in depth. At this time, there are elkhorn coral colonies north of the project site (Sprat Hole) but only dead elkhorn coral skeletons on colonized hard bottom in the immediate project area and south of the project site (A. Dempsey, BioImpact, pers. comm. to L. Carrubba, NMFS, April 29, 2008). Smith et al. (2014) concluded that the lack of colonization by elkhorn corals on the west and south coasts of St. Croix likely indicates prior losses of these corals due to disease, hurricanes, habitat degradation, and the limited availability of shallow hard bottom habitat, making the areas on the north west side of St. Croix (where the action area is) important for recovery of the species due to a relative lack of development in the area.

4 ENVIRONMENTAL BASELINE

This section identifies the effects of past and ongoing human and natural factors leading to the current status of the species, their habitat, and ecosystem, within the action area. The environmental baseline includes state, tribal, local, and private actions already affecting the species, or that will occur contemporaneously with the consultation in progress. Unrelated federal actions affecting the same species or critical habitat that have completed formal or informal consultation are also part of the environmental baseline, as are federal and other actions within the action area that may benefit listed species or critical habitat.

The environmental baseline for this Opinion includes several activities that affect the survival and recovery of green, leatherback, and hawksbill sea turtles; and elkhorn, staghorn, lobed star, and mountainous star corals; and the ability of designated acroporid coral critical habitat in the action area to support its intended conservation function for staghorn and elkhorn corals. Hurricanes Irma and Maria passed through the Caribbean in September 2017. While St. Croix was relatively unaffected by Hurricane Irma, Hurricane Maria caused widespread damage to the island. Because the island is still recovering, assessments of in-water impacts to benthic habitats, including coral reefs that are part of the TCRMP have not been completed. Therefore, there is a possibility that the environmental baseline for sea turtles and ESA-listed corals around St. Croix has been degraded from the conditions described here due to impacts from the recent hurricanes.

Certain activities require a Section 7 consultation with NMFS as part of the federal action. As part of the Section 7 process, NMFS will continue to establish conservation measures to ensure that the construction and operation of facilities and other actions with a federal nexus avoid or minimize adverse effects to ESA-listed sea turtles and corals and acroporid coral critical habitat.

4.1 Status of Green, Leatherback, and Hawksbill Sea Turtles within the Action Area

The distribution of green, leatherback, and hawksbill sea turtle nesting activity in St. Croix makes up a majority of the overall nesting activity around the USVI. For the most part, NMFS was not able to obtain much nesting data for the east end of St. Croix to verify the numbers of nesting green, leatherback, and hawksbill sea turtles in those areas. As a result, the following discussion has more information for the west and southwest beaches of St. Croix, as well as beaches on Buck Island. While those locations are not within the action area for proposed action, St. Croix is a relatively small island, and the areas for which we have additional biological information contain the same types of animals and the same habitat types as the action area, all of which face the same types of threats. Thus, the information from other areas is considered reflective of the conditions within the action area, which is why it has been included here.

4.1.1 Green Sea Turtles

Green sea turtle nesting activity is low in USVI and Puerto Rico when compared to other areas in the Caribbean and Atlantic. Nesting in St. Croix consists of sea turtles from the South Atlantic DPS, and is reported from approximately May through November, with the most nests in August-October. There were a total of 42 green sea turtle nests recorded on southern and western beaches of St. Croix, excluding Sandy Point, in 2009 with as few as 1 nest and as many as 8 green sea turtle nests recorded on approximately 12 different beach segments (Enfield Green, Williams Delight, Carlton, Campo Rico, White’s Bay, and 2 pocket beaches at Concordia on south shore; Dorsch-Ramp-Pool, Kenis-Fort, Prosperity, Sprat, ATT-Beresford, and Hams Bay on west shore). The June 2013 BA does not specify the number of green sea turtle nests or nesting activity (number of crawls) in the project area, but does note that fewer than 5 of the total sea turtle nesting activities in the project area were by green and leatherback sea turtles.

Based on our analysis of the 2009 data collected by DPNR, there were 22.6 green sea turtle nests/km on the Prosperity beach segment south of the Amalago Bay property line, 4.3 green sea turtle nests/km on the Prosperity-Sprat beach segment within the property line, and 0 green sea turtle nests/km on the North Sprat beach segment north of the property line (Table 4). Data from 2002 to 2006 in Sandy Point include green sea turtle activities, including dry runs, successful nests, probable lays, and tracks only for both new and remigrant turtles identified by passive integrated transponder (PIT) tags (Table 5) (DPNR 2002, 2003, 2004, 2005, 2006). Another green sea turtle nesting site is on Buck Island National Monument (North Shore, West Beach, South Shore, and Turtle Bay) off the northeast coast of St. Croix where 56 nesting activities were recorded in 2001, 33 in 2002, 75 in 2003, and 103 in 2004 (see Table 5) (USNPS 2003, 2004, 2005).

Green sea turtles are also known to nest at Isaac's Bay, Grapetree Bay, and Grotto Beach at the Buccaneer on the east coast of St. Croix based on unpublished stranding data and sightings reports from DPNR. Sea turtle nesting data from 2005 collected by West Indies Marine Animal Rescue and Conservation Service (WIMARCS) reported 135 nesting activities with 44 confirmed nests at East End Bay, 14 nesting activities at Halfpenny Bay, 129 nesting activities with 42 confirmed nests at Isaac's Bay, 58 nesting activities with 10 confirmed nests at Jack's Bay, 29 nesting activities at Manchenil, 1 nesting activity at Pelican Cove, 47 nesting activities at Prune Bay, and 13 nesting activities at Southgate Pond on the east side of St. Croix for green sea turtles (http://wimarcs.org/STX_SeaTurtleActivity.htm). Of these activities, 19 green sea turtles were responsible for the activities at East End Bay, of which 11 were remigrants; 22 at Isaac's Bay, of which 14 were remigrants; and 13 at Jack's Bay, of which 6 were remigrants (http://wimarcs.org/STX_SeaTurtleActivity.htm).

Green sea turtles were observed in the water during site visits and surveys in the project area; if these were not nesting females, they could have been from either the North Atlantic or the South Atlantic DPS. Green sea turtles are known to be present in waters around St. Croix year-round as evidenced by unpublished stranding and sighting reports from DPNR. Mortalities were due to entanglement in fishing gear, including nets, trap buoy lines, and fishing line, poaching, boat strikes, and shark attacks. In-water sea turtle surveys conducted within BIRNM off the northeast coast of St. Croix have resulted in the capture of 162 turtles in 2012 and 2013, of which 77% were green sea turtles. Of these, 30 of the green sea turtles were recaptures. The investigators captured the majority of the green sea turtles in areas containing seagrass beds or a mix of seagrass and hard bottoms (Hart et al. 2014.). A similar pattern is likely in other areas around St. Croix where there are seagrass beds and colonized hard bottoms, as is the case off the coast where the Amalago Bay project is proposed. Satellite tagging of green sea turtles as part of the in-water and nesting surveys at BIRNM found that the majority of the tagged turtles remained in the general region of the U.S. and British Virgin Islands and Puerto Rico, indicating that many green sea turtles may be longer-term residents of the region (Hart et al. 2014.).

Table 4. Summary of 2009 Nesting Data from DPNR including Estimated Beach Length and Number of Confirmed Nests versus Crawls or Other Activities[16]

		Unknown			Leatherback (Dc)					Hawksbill (Ei)					Green (Cm)					All sp.	
Location	Beach Length (km)	No. of Nests	No. of Crawls	Other	No. of Nests	No. of Crawls	Other	nests/km of beach	% nests	No. of Nests	No. of Crawls	Other	nests/km of beach	% nests	No. of Nests	No. of Crawls	Other	nests/km of beach	% nests	Total Nests on Beach	nests/km of beach
Enfield Green	0.285	1						0	0				0	0	2			7.017544	66.667	3	10.52632
Williams Delight	0.211	4			3			14.21801	23.0769	3	1		14.21801	23.077	3			14.21801	23.077	13	61.61137
Long Point	0.387							0	0	1			2.583979	100				0	0	1	2.583979
Carlton	1.104	1		hatchling unk	30			27.17391	61.2245	12	2		10.86957	24.49	6			5.434783	12.245	**49**	44.38406
Good Hope School East	0.31							0	0	9			29.03226	100				0	0	9	29.03226
Good Hope School West	0.475							0	0	20	1		42.10526	100				0	0	20	42.10526
Hidden Beach	0.15			washed out eggs unk				0	0	8	1		53.33333	100				0	0	8	53.33333
Campo Rico	0.331	1			9			27.19033	50	7	2		21.14804	38.889	1			3.021148	5.5556	18	54.38066
White's Bay	0.225	4						0	0	2			8.888889	20	4			17.77778	40	10	44.44444
Concordia (2 pocket beaches)	0.143							0	0	1	1		6.993007	50	1			6.993007	50	2	13.98601
Dorsch-Ramp-Pool	0.2293	5						0	0	9	2		39.24989	40.909	8			34.88879	36.364	22	95.94418
Kensis-Fort	1.46	7		hatchlings unk				0	0	52	3		35.61644	85.246	2			1.369863	3.2787	61	41.78082
Prosperity (south of Amalago Bay property)	0.221	4						0	0	5	1		22.62443	35.714	5			22.62443	35.714	14	63.34842
Sprat-Prosperity (inside property)	0.925	4	1		12			12.97297	16	55	2		59.45946	73.333	4			4.324324	5.3333	**75**	81.08108
North Sprat (north of property)	0.176							0	0	7	4		39.77273	100				0	0	7	39.77273
Butler Bay	0.15	1						0	0	2	1		13.33333	66.667				0	0	3	20
ATT-Beresford	0.682							0	0	16			23.46041	94.118	1			1.466276	5.8824	17	24.92669
Hams Bay	0.202	5						0	0	6			29.70297	37.5	5			24.75248	31.25	16	79.20792
		37	1	3	54	0	0		0	215	21	0			42	0	0				
	south side beaches (except Sandy Point)																				
	west side beaches																				

[16] Note that, due to the monitoring methods used, mainly confirmed nests are reported rather than other nesting activity in these data.

Table 5. Summary of Sea Turtle Nesting Data from 2001-2004 for Buck Island National Monument and 2002-2006 for Sandy Point National Wildlife Refuge (from U.S. National Park Service and DPNR, respectively)[17]

		Leatherback (Dc)				Hawksbill (Ei)				Green (Cm)				
Location	Beach Length (km)	No. of Nests	Other (Crawls or attempts)	Suspected Nest	nests/km of beach	No. of Nests	Other (Crawls or attempts)	Suspected Nest	nests/km of beach	No. of Nests	Other (Crawls or attempts)	Suspected Nest	nests/km of beach	Total Activities per Site
2001														
Buck Island - North Shore	0.375	0	2	0	0	57	93	31	152	1	1	0	2.66666667	185
Buck Island - West Beach	0.51	6	0	0	11.76471	24	38	13	47.05882	7	6	1	13.7254902	95
Buck Island - South Shore	0.36	0	0	0	0	21	36	11	58.33333	3	2	1	8.33333333	74
Buck Island Turtle Bay	0.255	0	0	0	0	24	39	13	94.11765	18	14	2	70.5882353	110
	Totals	6	2	0		126	206	68		29	23	4		
2002														
Sandy Point	3	583	270	59	194.3333	6	10	2	2	9	36	7	3	982
Buck Island - North Shore	0.375	0	0	0	0	57	120	3	152	1	1	0	2.66666667	182
Buck Island - West Beach	0.51	1	0	3	1.960784	18	39	1	35.29412	4	5	0	7.84313725	71
Buck Island - South Shore	0.36	0	0	0	0	29	63	2	80.55556	0	0	0	0	94
Buck Island Turtle Bay	0.255	0	0	0	0	27	57	1	105.8824	9	12	1	35.2941176	107
	Totals	584	270	62		137	289	9		23	54	8		
2003														
Sandy Point	3	974	398	86	324.6667	7	15	5	2.333333	0	1	1	0	1487
Buck Island - North Shore	0.375	0	0	0	0	102	187	8	272	1	2	0	2.66666667	300
Buck Island - West Beach	0.51	0	3	0	0	43	78	3	84.31373	7	7	1	13.7254902	142
Buck Island - South Shore	0.36	0	0	0	0	53	96	4	147.2222	6	6	1	16.6666667	166
Buck Island Turtle Bay	0.255	0	2	0	0	52	95	4	203.9216	19	21	4	74.5098039	197
	Totals	974	403	86		257	471	24		33	37	7		
2004														
Sandy Point	3	444	205	67	148	4	7	6	1.333333	0	9	6	0	748
Buck Island - North Shore	0.375	2	0	1	5.333333	70	119	8	186.6667	0	1	0	0	201
Buck Island - West Beach	0.51	6	0	4	11.76471	38	64	5	74.5098	7	15	1	13.7254902	140
Buck Island - South Shore	0.36	0	0	0	0	38	65	5	105.5556	4	9	1	11.1111111	122
Buck Island Turtle Bay	0.255	0	0	0	0	19	31	2	74.5098	21	41	3	82.3529412	117
	Totals	452	205	72		169	286	26		32	75	11		
2005														
Sandy Point	3	573	290	60	191	7	32	43	2.333333	7	54	36	2.33333333	1102
2006														
Sandy Point	3	337	149	36	112.3333	8	57	8	2.666667	37	142	65	12.3333333	839

Note that loggerhead nesting at Buck Island has also been reported since 2003, but is not included here because our analysis focuses on leatherback, green, and hawksbill sea turtles.

4.1.2 Leatherback Sea Turtles

In St. Croix, leatherbacks begin nesting as early as January and continue nesting as late as August, with a peak in May. One of the most important leatherback nesting beaches is Sandy Point on the southwest tip of St. Croix. Leatherback and other sea turtle species nest along 3 km of sandy beach around the point, which is a National Wildlife Refuge managed by the U.S. Fish and Wildlife Service. Data from 2002 in Sandy Point report 115 adult leatherback sea turtles, including 45 untagged turtles, 67 remigrants, and 3 turtles tagged on other islands (Culebra and mainland Puerto Rico). A total of 912 nesting activities were recorded in 2002, which is less than the 1,289 activities recorded in 2001 (DPNR 2002), but is consistent with the variation reported from 2002-2006 (Table 5) (DPNR 2002, 2003, 2004, 2005, 2006).

There were a total of 54 leatherback sea turtle nests recorded on southern and western beaches of St. Croix, excluding Sandy Point, in 2009 with as few as 3 and as many as 30 leatherback sea turtle nests recorded on approximately 4 different beach segments (Williams Delight, Carlton, and Campo Rico on southwest shore, and Prosperity-Sprat on west shore). The June 2013 BA does not specify the number of leatherback sea turtle nests or nesting activity (number of crawls), but does note that fewer than 5 of the total sea turtle nesting activities in the project area were by green and leatherback sea turtles. This is consistent with DPNR monitoring data from 2009, which indicated that the area is predominantly used by hawksbill sea turtles for nesting. Based

[17] The table also includes calculations of the number of nests (from confirmed nest numbers only) per km of nesting beach for each sea turtle species.

on our analysis of the 2009 data collected by DPNR, there were 0 leatherback sea turtle nests/km on the Prosperity beach segment south of the Amalago Bay property line, 12.97 leatherback sea turtle nests/km on the Prosperity-Sprat beach segment within the property line, and 0 leatherback sea turtle nests/km on the North Sprat beach segment north of the property line (Table 4).

A few leatherback nests are reported annually on Buck Island off the northeast coast of St. Croix. In 2001, 8 nesting activities were recorded on North Shore and West Beach, 4 nesting activities on West Beach in 2002, 3 nesting activities at West Beach and 2 at Turtle Bay in 2003, and 3 nesting activities on North Shore and 10 at West Beach in 2004 (Table 5) (USNPS 2002, 2003, 2004, 2005). Leatherback sea turtles are also known to nest at Ham's Bay and on beaches along the north shore west of Christiansted based on unpublished stranding from DPNR that recorded disorientation of egg-laying females due to lights in these areas. Sea turtle nesting data from 2005 collected by WIMARCS reported 4 nesting activities at Halfpenny Bay, 25 nesting activities at Manchenil, 2 nesting activities at Pelican Cove, 6 nesting activities at Prune Bay, and 14 nesting activities with 2 confirmed nests at Southgate Pond on the east side of St. Croix for leatherback sea turtles (http://wimarcs.org/STX_SeaTurtleActivity.htm). Strandings of leatherback sea turtles have been reported mainly in April and May, but occasionally in January, February, July, and November (DPNR, unpublished data). Mortalities were due to entanglement in fishing gear, especially nets, poaching, disorientation, and boat strikes.

4.1.3 Hawksbill Sea Turtles

Hawksbill sea turtles nest on St. Croix beaches, often throughout the year, with a peak from approximately July-October. There were a total of 215 hawksbill sea turtle nests recorded on southern and western beaches of St. Croix, excluding Sandy Point, in 2009 with as few as 1 nest and as many as 55 nests recorded on approximately 17 different beach segments (Enfield Green, Williams Delight, Long Point, Carlton, Good Hope School East and West, Hidden Beach, Campo Rico, White's Bay, and 2 pocket beaches at Concordia on south shore; Dorsch-Ramp-Pool, Kenis-Fort, Prosperity, Sprat, North Sprat, Butler Bay, ATT-Beresford, and Hams Bay on west shore). Of these beaches, during 2009 beach monitoring conducted by DPNR, the project area had the most hawksbill nests reported with 55.

The June 2013 BA reports 44 sea turtle nesting activities for Williams Beach and a total of 46 activities for Sprat Hole beach, which borders the north end of Williams Beach, with fewer than 5 of the activities on each beach being from green and leatherback sea turtles. However, it is important to note that these nesting site names do not match those used in the 2009 nesting survey by DPNR. Based on NMFS's analysis of the 2009 data collected by DPNR, there were 22.6 hawksbill sea turtle nests/km on the Prosperity beach segment south of the Amalago Bay property line, 59.4 hawksbill sea turtle nests/km on the Prosperity-Sprat beach segment within the property line, and 39.7 hawksbill sea turtle nests/km on the North Sprat beach segment north of the property line (Table 4).

Data from 2002-2006 in Sandy Point include hawksbill sea turtle activities, including dry runs, successful nests, probable lays, and tracks only for new and remigrant turtles identified by PIT tags (Table 5) (DPNR 2002, 2003, 2004, 2005, 2006). Another hawksbill sea turtle nesting site is on Buck Island National Monument (North Shore, West Beach, South Shore, and Turtle Bay) off the northeast coast of St. Croix where 380 nesting activities were recorded in 2001, 417 in

2002, 725 in 2003, and 464 in 2004 (Table 5) (USNPS 2002, 2003, 2004, 2005). Hawksbill sea turtles are also known to nest on the north coast of St. Croix in the Christiansted area based on unpublished stranding data and sightings reports from DPNR, including reported hatchling disorientation. Sea turtle nesting data from 2005 collected by WIMARCS reported 16 nesting activities with 3 confirmed nests at East End Bay, 10 nesting activities at Halfpenny Bay, 13 nesting activities with 4 confirmed nests at Isaac's Bay, 40 nesting activities with 9 confirmed nests at Jack's Bay, 4 nesting activities at Manchenil, 1 nesting activity with 1 confirmed nest at Pelican Cove, 36 nesting activities with 1 confirmed nest at Prune Bay, and 4 nesting activities at Southgate Pond on the east side of St. Croix for hawksbill sea turtles (http://wimarcs.org/STX_SeaTurtleActivity.htm). Of these activities, 2 hawksbill sea turtles were responsible for the activities at East End Bay of which 2 were remigrants; 5 at Isaac's Bay of which 3 were remigrants; and 11 at Jack's Bay of which 5 were remigrants (http://wimarcs.org/STX_SeaTurtleActivity.htm).

Hawksbill sea turtles were observed on coral reefs or hard bottoms in the water during site visits and surveys in the project area. Hawksbill sea turtles are known to be present in waters around St. Croix year-round as evidenced by unpublished stranding and siting reports from DPNR. Mortalities were due to entanglement in fishing gear, poaching, boat strikes, disorientation (from lights during nesting), and dog attacks during nesting. In-water sea turtle surveys in BIRNM found that 33% of the 162 total turtles captured between 2012 and 2013 were hawksbill sea turtles with the rest being green sea turtles. One hawksbill was captured 4 times (on every in-water trip) and 1 of the 7 recaptured hawksbill sea turtles had been tagged as a foraging juvenile in waters of BIRNM in 2002. The investigators captured the majority of the hawksbill sea turtles in areas containing coral reefs and occasionally in seagrass areas adjacent to coral reefs and hard bottoms (Hart et al. 2014.). A similar pattern is likely in other areas around St. Croix where there are seagrass beds and colonized hard bottoms, as is the case off the coast where the Amalago Bay project is proposed. Satellite tagging of hawksbill sea turtles as part of the in-water and nesting surveys at BIRNM found that most of the turtles traveled to foraging areas in the U.S. and British Virgin Islands, as well as to other Caribbean islands, while some traveled long distances to the northeastern coast of Nicaragua and the Bahamas (Hart et al. 2014.).

4.2 Factors Affecting Green, Leatherback, and Hawksbill Sea Turtles within the Action Area

The activities that shape the environmental baseline in the action area are federal fisheries, effects of vessel operations, private vessel traffic, marine pollution, and natural disturbance.

NMFS completed a number of Section 7 consultations to address the effects of federally permitted fisheries and other federal actions on threatened and endangered sea turtle species, and when appropriate, has authorized the incidental taking of these species. Each of those consultations sought to minimize the adverse impacts of the action on sea turtles through changes to the action as proposed or through reasonable and prudent measures. The summary below includes those federal actions in, or having effects in the action area that have already concluded or are currently undergoing formal Section 7 consultation, as well as state and private activities, and natural disturbances.

Fisheries
Threatened and endangered sea turtles are adversely affected by fishing gears used throughout the continental shelf of the action area. Net, hook-and-line gear, and trap fisheries have all been documented as interacting with sea turtles in USVI based on stranding data from Territorial waters (DPNR unpublished data). Entanglement in nets, trap lines, and fishing line accounted for 27% of reported sea turtle strandings around St. Croix for the period from 1982-2010 with 43% of the turtles entangled in line being greens, 48% hawksbills, and 9% leatherbacks (DPNR unpublished data). Fewer data were available from St. Thomas and St. John, but they reflect similar trends with 40% of strandings caused by entanglement in fishing gear in St. Thomas (of which 88% were greens and 12% were hawksbills) and 22% in St. John (of which 100% were greens) (DPNR unpublished data). The USVI Territorial Coral Reef Monitoring Program found derelict fishing gear in the area of the shelf edge reef off the coast of the proposed Amalago Bay project and indications of fishing pressure at several other permanent monitoring sites around St. Croix (Smith et al. 2011a).

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

Vessel Operations
Potential sources of adverse effects from federal vessel operations in the action area include operations of the USCG. NMFS and the USCG completed a programmatic consultation for the USCG's ATONS program to determine the magnitude of the adverse impacts resulting from ATON operations in portions of Florida, Puerto Rico, and the USVI. The consultation ended on August 5, 2013, and NMFS's Opinion determined that ATON maintenance activities were not likely to adversely affect sea turtles. In addition, NMFS is currently working on a national programmatic consultation that will determine the magnitude of the adverse impacts resulting from all ATON operations nationwide, including those in the U.S. Caribbean. Through the Section 7 process, where applicable, NMFS will continue to establish conservation measures for agency vessel operations to avoid or minimize adverse effect to ESA-listed species.

Commercial and recreational vessel traffic can have adverse effects on sea turtles via propeller and boat-strike injuries. NMFS and the USCG have completed an informal Section 7 consultation for the Caribbean Marine Event Program for all annually occurring marine events in USVI and Puerto Rico. As a result of this consultation, the USCG now includes guidelines to avoid and minimize potential impacts of marine events, especially events involving motorized vessels such as speedboat races, to listed sea turtles and their habitat as permit conditions the event participants must follow.

Stranding data reported known strandings of 77 sea turtles (loggerhead, leatherback, green, and hawksbill) around St. Croix from 2001-2010 (DPNR, unpublished data). Of these, 4 green, 2 leatherback, and 1 unknown species of sea turtle could be confirmed to have been impacted by boats (DPNR unpublished data). Thus, approximately 9% of the reported strandings around St. Croix for which a cause could be identified were caused by boat strikes. The majority of these strikes were fatal resulting in massive injuries to the turtles due to the cutting action of the propeller (DPNR unpublished data). Similarly, 22% of the reported strandings around St. John and 25% of the reported strandings around St. Thomas were caused by boat strikes. Of these, all of the St. John strandings were greens, 4 of the 5 St. Thomas strandings were greens and the other was a hawksbill (DPNR unpublished data). 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 in the USVI, although the majority of these activities have been on the east, north, and south coasts of St. Croix and around St. Thomas and St. John. As part of the Section 7 process for dock, port, and marine construction activities under the jurisdiction of the USACE, NMFS also considers the impacts of the vessel traffic from the operation of these facilities and any measures to avoid and minimize adverse impacts to sea turtles.

ESA Permits

Sea turtles are the focus of research activities authorized by Section 10 permits under the ESA. Section 10(a)(1)(a) of the ESA allows issuance of permits for take of certain ESA-listed species for the purposes of scientific research, and section 10(a)(1)(B) authorizes issuance of permits for take of listed species incidental to other activities under certain conditions. Research activities authorized through ESA permits range from photographing, weighing, and tagging sea turtles incidentally taken in fisheries, to blood sampling, tissue sampling (biopsy), and performing laparoscopy on intentionally captured sea turtles. The number of authorized takes varies widely depending on the research and species involved, but may involve the taking of hundreds of sea turtles annually. Most takes authorized under these permits are expected to be (and are) nonlethal. 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 or USFWS must also be reviewed for compliance with Section 7(a)(2) of the ESA to ensure that issuance of the permit does not result in jeopardy to the species or adverse modification of its critical habitat.

Coastal Development

Sources of pollutants along the coast of St. Croix include atmospheric loading of pollutants such as polychlorinated biphenols (PCBs), stormwater runoff from coastal development into ghuts that empty into the Caribbean Sea, industrial discharges, sewage discharges, and groundwater discharges. Nutrient loading from land-based sources such as coastal community discharges is known to stimulate plankton blooms in closed or semi-closed estuarine systems. The effects on larger embayments are unknown. Although pathological effects of oil spills have been documented in laboratory studies of marine mammals and sea turtles (Vargo et al. 1986), the impacts of many other anthropogenic toxins have not been investigated.

Coastal runoff, marina and dock construction, dredging, industrial operations, increased under water noise, and boat traffic can degrade marine habitats used by sea turtles (Colburn et al. 1996). The development of marinas and docks can negatively impact nearshore habitats. An increase in the number of docks built thereby increases boat and vessel traffic. Fueling and pump-out facilities at marinas can sometimes discharge oil, gas, and sewage into sensitive coastal habitats. Although these contaminant concentrations do not likely affect the more pelagic waters, the species of turtles analyzed in this Opinion travel between nearshore and offshore habitats and various life stages of green and hawksbill sea turtles in particular can be found in nearshore waters of St. Croix year-round. Therefore, the species of turtles analyzed in this Opinion may be exposed to and accumulate terrestrial contaminants that are released into the marine environment during their life cycles.

There are studies on organic contaminants and trace metal accumulation in green, loggerhead, and leatherback sea turtles (Aguirre et al. 1994, Caurant et al. 1999, Mckenzie et al. 1999, Corsolini et al. 2000). Although we have determined that loggerhead sea turtles are not likely to be adversely affected by the Amalago Bay project, we use studies on the impacts of pollutants on this species because similar effects could occur in greens and hawksbills, as well as leatherbacks although the preference of this species for oceanic habitats except when nesting may limit exposure to greens and hawksbills. (Mckenzie et al. 1999) measured concentrations of chlorobiphenyls and organochlorine pesticides in sea turtles tissues from different life stages and eggs collected from the Mediterranean (Cyprus, Greece) and European Atlantic waters (Scotland) between 1994 and 1996. Omnivorous loggerhead turtles had the highest organochlorine contaminant concentrations in all the tissues sampled, including those from green and leatherback turtles (Storelli et al. 2008). It is thought that dietary preferences were likely to be the main differentiating factor among species. Decreasing lipid contaminant burdens with turtle size were observed in green turtles, most likely attributable to a change in diet with age as turtles shift from an omnivorous to an herbivorous diet. (Sakai et al. 1995) found the presence of metal residues occurring in loggerhead turtle organs and eggs. (Storelli et al. 1998) analyzed tissues from 12 loggerhead sea turtles stranded along the Adriatic Sea (Italy) and found that characteristically, mercury accumulates in sea turtle livers while cadmium accumulates in their kidneys, as has been reported for other marine organisms like dolphins, seals, and porpoises (Law et al. 1991). No information on detrimental threshold concentrations is available, and little is known about the consequences of exposure of organochlorine compounds to sea turtles. Research is needed on the short- and long-term health and fecundity effects of chlorobiphenyl, organochlorine, and heavy metal accumulation in sea turtles.

Nutrient loading from land-based sources, such as coastal communities and agricultural operations, are known to stimulate plankton blooms in closed or semi-closed estuarine systems. The effects on larger embayments are unknown. An example is the large area of the Louisiana continental shelf with seasonally-depleted oxygen levels (< 2 mg/L) caused by eutrophication (process by which a water body becomes enriched in dissolved nutrients that stimular the growth of phytoplankton usually resulting in the depletion of dissolved oxygen) 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.” The oxygen depletion, referred to as hypoxia, begins in late spring, reaches a maximum in mid-summer, and disappears in the fall. Water quality monitoring studies by DPNR’s Division of Environmental Protection (DEP) in waters around USVI indicate that surface waters are affected by increasing point and non-point source pollution from failing septic systems, discharges from vessels, failure of BMPs on construction sites, and failure of on-site disposal methods (Rothenberger et al. 2008). These factors result in increased sedimentation and nutrient transport, bacterial contamination, and trash and other debris entering surface and nearshore waters from developed areas. The effects of these water quality declines on species such as sea turtles are unknown. However, it is clear that water quality degradation leads to habitat degradation of coral reefs and other coralline communities, as well as seagrass beds. Thus, at least an indirect effect on green and hawksbill sea turtles due to degradation of foraging habitat quality is expected.

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. 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 1980-1981 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). Thus, coastal development such as the proposed Amalago Bay project, which will be located on slopes of up to 100%, is likely to generate significant sediment loading from transport in runoff and associated degradation of nearshore habitat utilized by sea turtles.

Natural Disturbances

Hurricanes and large coastal storms can significantly modify both nesting and in-water sea turtle habitat. Beach profiles change in response to wave action and storm-induced erosion on the coast, which can also lead to the loss of nests or the loss of nesting habitat for at least a season if not longer depending on the size of the beach and the extent to which the beach profile is altered. Storms also result in breakage of sessile benthic organisms from extreme wave action and storm surges. Intense storms that cover a broad area can eliminate or damage large expanses of reef or result in blowouts and loss of seagrass habitats. Major hurricanes have caused significant losses in coral cover and changes in the physical structure of many reefs in USVI. There have been 10 hurricanes that have affected the reefs of 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 30-40% decline in coral cover along transects and within quadrats in Great Lameshur Bay, St. John (Rogers et al. 2008b). Tropical storms and hurricanes in 2004, 2008, and 2010 also resulted in severe flooding across USVI. This flooding also caused significant sedimentation of areas resulting in additional degredation of reef habitats. In addition to affecting the sessile benthic organisms themselves, these changes in the structure of the reef affect species like sea turtles, in particular greens and hawksbills. In-water habitat for green and hawksbill sea turtles is temporarily lost or temporarily or permanently degraded (depending on the magnitude of the storm). As noted above, in early September 2017 Hurricane Irma had a greater impact on St. Thomas and St. John but Hurricane Maria at the end of September 2017 caused widespread damage from wind, waves, and rain across St. Croix.

Hurricanes Irma and Maria passed through the Caribbean in September 2017. St. Croix was relatively unaffected by Hurricane Irma, which did impact St. Thomas and St. John, and all three islands suffered damage from Hurricane Maria. Because the islands are still recovering, assessments of in-water habitats, including areas that provide nesting habitat to green, hawksbill, and leatherback sea turtles and refuge and foraging habitat to hawksbill and green sea turtles, are still on-going. However, based on assessments that have been completed to date around Puerto Rico and USVI, some coral areas suffered only minor damage (Figure 11), including all the areas surveyed around St. Croix to date. In other areas, triage of affected corals was performed or is ongoing to stabilize colonies affected by the storms. Therefore, while there is a possibility that the environmental baseline described here may have been degraded by hurricane damage, survey results to date indicate that many coral reef sites around the islands were relatively unaffected.

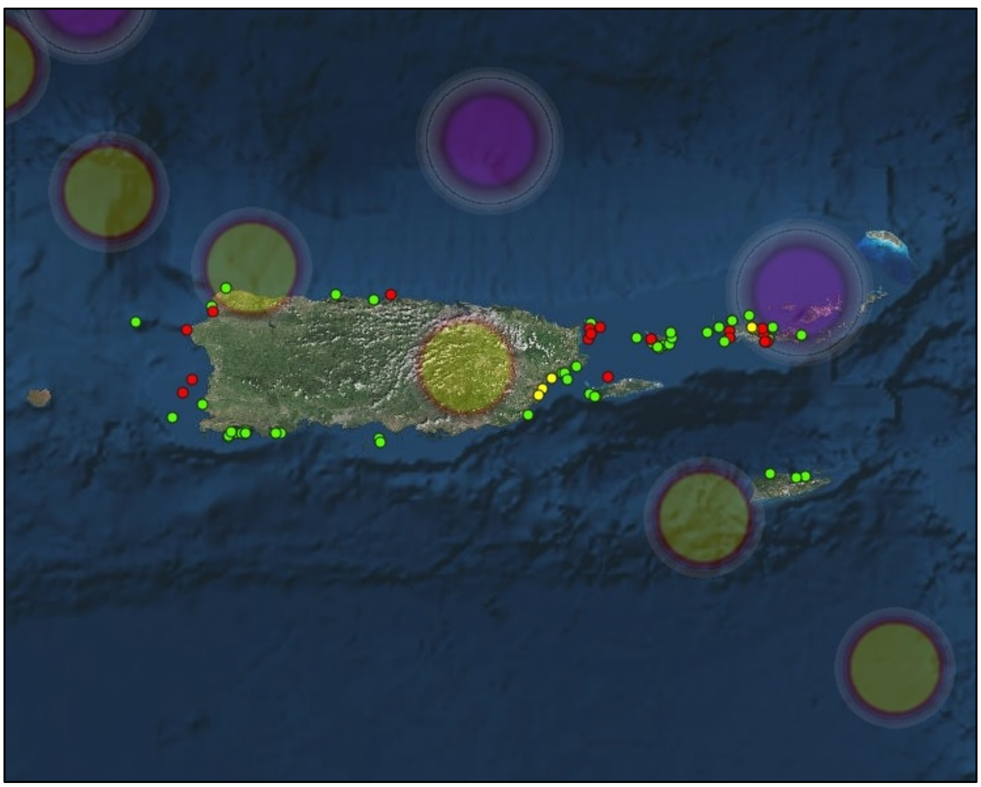

Figure 11. Map showing tracks of Hurricanes Irma (large purple dots) and Maria (large yellow dots) in area where Puerto Rico and USVI are located and results of coral surveys conducted to date. Small green dots indicate areas where coral surveys indicated no triage was needed, red dots indicate areas where triage was needed, and yellow dots indicate areas where the need for triage is still under evaluation (NOAA Restoration Center)

Conservation and Recovery Actions Benefiting Green, Leatherback, and Hawksbill Sea Turtles

NMFS has implemented a series of regulations aimed at reducing the potential for incidental capture and mortality of sea turtles from commercial fisheries in the action area. These include sea turtle release gear requirements for Caribbean fisheries, including long line and trap gears.

Under Section 6 of the ESA, we may enter into cooperative research and conservation agreements with states to assist in recovery actions of listed species. We currently have an agreement with USVI, which was renewed in September of 2018. Any projects conducted under these agreements must be reviewed for compliance with Section 7 of the ESA. The Virgin Islands Department of Planning and Natural Resources (DPNR) conducts research and monitoring under this agreement to support the sea turtle species' conservationand recovery.

Outreach and Education, Sea Turtle Entanglements, and Rehabilitation

NMFS and the USVI have established stranding procedures to rescue and rehabilitate any live stranded sea turtles. The STAR network responds to sea turtle strandings on St. Croix. STAR is a volunteer network composed of local agency personnel, non-governmental organizations, veterinarians, and private individuals. STAR is managed through WIMARCS.

Sea Turtle Handling and Resuscitation Techniques

NMFS has issued regulations (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 turtles caught in fishing or scientific research gear.

A Final Rule (70 FR 42508) published on July 25, 2005, allows any agent or employee of NMFS, USFWS, USCG, 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)].

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

Other Actions

Recovery teams comprised of sea turtle experts have been convened and are currently working towards revising sea turtle recovery plans based upon the latest and best available information.

Five-year status reviews have recently been completed for green, hawksbill, and leatherback sea turtles. These reviews were conducted to comply with the ESA mandate for periodic status evaluation of listed species to ensure that their threatened or endangered listing status remains accurate. These reviews also evaluate whether DPSs should be established for the species. The 2007 5-year status reviews of green, hawksbill and leatherback sea turtles recommended further review of the species data to evaluate whether DPS should be established for these species (NMFS and USFWS 2007a-e). In response to a 2012 petition to identify and delist a Hawaii population of green sea turtles, the Services initiated a status review which led to the 2016 final rule listing 11 DPSs of green sea turtles. The 2013 5-year review of the status of hawksbill sea turtles reiterated that new information indicates the need to evaluate possible DPSs of the species. In response to a 2017 petition to identify and list a Northwest Atlantic DPS of leatherback sea turtles, the Services have initiated a status review to determine whether the petitioned action is warranted and to examine the species globally with regard to application of the DPS Policy.

4.2.1 Summary and Synthesis of Environmental Baseline for Listed Sea Turtles

In summary, several factors adversely affect sea turtles in the action area. These factors are ongoing and are expected to occur contemporaneously with the proposed action. Fisheries in the action area have the greatest adverse impacts on sea turtles due to the prevalence of net fishing around St. Croix. Over the past 5 years, the impacts associated with fisheries may have been reduced through the Section 7 consultation process and regulations implementing effective bycatch reduction strategies, such as the requirement of turtle release gear in some fisheries. However, interactions with commercial and recreational fishing gear are ongoing and are expected to occur contemporaneously with the proposed action. Poaching and boat strikes are the other factors that have the most impact on sea turtles in the action area based on stranding data for St. Croix (DPNR unpublished data). Other environmental impacts including effects of vessel operations, permits allowing take under the ESA, marine pollution, and natural disturbance have also had and continue to have adverse effects on sea turtles in the action area. Based on the information discussed in this section, the environmental baseline for sea turtles in the action area is not pristine and has been degraded by impacts from development, in particular related to the construction of residential, commercial, and tourist facilities. The Amalago Bay project is proposed in an area with minimal development at this time. There is a residential community of approximately 20 homes (along the southern border of the property known as Prosperity) and a few other scattered residences, a restaurant that has since been destroyed by fire, and a few other commercial establishments on the southern border of the beach property line. A few sailboats moor periodically in the water in the action area. Therefore, while the environmental baseline is not pristine, there is a low level of human-caused degradation related to the construction of infrastructure such as a main road and utilities and limited residential and commercial development, especially along the southern border of the property.

4.3 Status of Elkhorn, Staghorn, Lobed Star and Mountainous Star Corals, and Acroporid Coral Critical Habitat within the Action Area

4.3.1 Elkhorn Coral

No elkhorn corals were found within the 46-ac benthic survey area. An elkhorn coral recruit is present on the point of Sprat Hole north of the immediate project area. According to NOS monitoring data, other elkhorn colonies are present in the area of Sprat Hole, as well as south of Frederiksted Pier near the Westend Saltpond. Near the Westend Saltpond, elkhorn corals are found close to shore while staghorn corals are located on the shelf edge reef (S. Pittman, NOS Contractor, pers. comm. to L. Carrubba, NMFS, December 12, 2014). Based on observations during other site inspections NMFS has conducted in the project area, there are also numerous elkhorn coral colonies along various portions of the west coast north of the project area, including the areas of Sprat Hole and Butler Bay. Toller (2005) reported elkhorn corals to be occasional in waters from 0-3 ft in depth within the Frederiksted Reef System.

4.3.2 Staghorn Coral

No staghorn corals were found within the 46-ac benthic survey area. Staghorn corals have been reported on inshore colonized hard bottoms, mid-shelf colonized hard bottoms and patch reefs, and the offshore shelf edge reef within the Frederiksted Reef System, including the reef seaward of the Amalago Bay project, although the number of colonies was not quantified (Toller 2005, Smith et al. 2014). The 2005 DPNR survey (Toller 2005) reported occasional staghorn coral colonies in waters from 18-35 ft in depth within the Frederiksted Reef System. NOS also reports staghorn corals in nearshore areas from Sprat Hole northward and near the Westend Saltpond (S. Pittman, NOS Contractor, pers. comm. to L. Carrubba, NMFS, December 12, 2014). Information from historical and recent studies indicates that acroporid corals are no longer present in high abundance within the Frederiksted Reef System, making them difficult to find without targeted surveys.

4.3.3 Lobed Star and Mountainous Star Corals

The benthic studies conducted for this project note that there are colonies of lobed star (*Orbicella annularis*) and mountainous star (*O. faveolata*) corals within the construction footprint. We also expect that there are additional colonies of each of these corals within the action area. The number of colonies for all coral species observed during the benthic surveys for the project, including those within the in-water construction and dredging footprint was not provided in the survey results.

Based on observations along the study transects for the 2005 DPNR survey (Toller 2005), deeper reefs were dominated by *Montastraea/Orbicella* spp. corals. The study by Toller (2005) found transects in the area of Sprat Hole, had 24.5% coral cover of which 78% was *Orbicella annularis*. Similarly, surveys done by EPA around St. Croix as part of their bioassessment program found *Orbicella annularis* to be the coral with the most coverage (approximately 15%) and *O. faveolata* to be half the cover of *Orbicella annularis* (or 7.5%).

Smith (2013) provided summary information from the USVI's TCRMP for the years 2001-2012. The TCRMP has 33 monitoring sites, including 14 surrounding St. Croix. The Sprat Hole site is north, immediately adjacent to the Amalago Bay development. The coral coverage levels of lobed star coral, *O. annularis*, at Sprat Hole were the highest, significantly so, of all the TCRMP sites in St. Croix, with 8.33% coverage. This was the second highest coverage of lobed star coral across all sites in the USVI. Coverage of mountainous star coral, *O. faveolata*, was 0.5%. It is unclear why these coverage levels differ from Toller's values, perhaps because Smith's data were time-averaged and extended to 2012 and therefore may be more reflective of recent partial mortality impacts on *Orbicella* spp. live tissue cover. Nevertheless, the fact that Sprat Hole contains the highest cover of lobed star coral among St. Croix's TCRMP sites, and Sprat Hole is immediately north of the project area, suggests that the project area likely provides some of the highest quality habitat for this species around St. Croix, and even the entire USVI.

4.3.4 Acropora Critical Habitat

The feature of critical habitat essential to the conservation of elkhorn and staghorn corals is substrate of suitable quality and availability, in water depths of 30 m or less, to support successful recruitment and population growth. This includes areas of exposed hard substrate and dead coral skeleton free of sediment cover and turf and fleshy macroalgae cover. The St. Croix marine unit comprises approximately 126 mi^2 (80,640 ac). Of this area, approximately 90 mi^2 (57,600 ac), or 71%, are most likely to contain the essential physical feature of ESA-designated coral critical habitat, based on the amount of coral, rock reef, colonized hard bottom, and other coralline communities mapped by NOS in 2001. The other areas within the St. Croix marine unit are dominated by sand and unconsolidated bottom, seagrass beds with varying densities of coverage, and uncolonized hard bottoms based on the NOS benthic maps (Kendall et al. 2001).

According to the NOS benthic habitat maps, within the 200-acre area extending from the shoreline of the Amalago Bay project to the shelf edge reef (areas A and B, Figure 5, approximately 2,500 ft from shore), there are approximately 110 ac of habitat containing the essential feature of substrate of suitable quality and availability, in water depths of 30 meters or less, to support successful recruitment and population growth. The benthic surveys completed for the BA covered 46 ac of area along 4,000 ft of shoreline and extending 500 ft (area A, Figure 5) offshore. The benthic surveys found 33.06 ac of colonized hard bottom habitat within the 46-ac survey area (Area A, Figure 5). Elkhorn corals are most often found in water depths 5 m or less (as stated previously this species can occasionally be found in 30 m of water in back reef environments). Using the NOS benthic maps, the available essential feature in water depths 5 m or less is approximately 11.12 mi^2 (7,117.7 ac) in the St. Croix Unit. The west end of St. Croix, including the Frederiksted Reef System where the Amalago Bay project is located, contains approximately 1.14 mi^2 (732 ac) of essential feature in depths of 5 m or less, and the action area contains approximately 33 acres of essential feature in depths of 5 m or less.

Staghorn corals are typically found in waters with depths greater than 5 m around St. Croix. Smith et al. (2014) found staghorn corals in waters from 6-18 m in depth, but noted that more colonies are likely present in deeper waters. Toller (2005) found staghorn corals in depths up to 35 m within the Frederiksted Reef System. As discussed in Section 3.0, the waters within the project area are part of a connected system of coral reefs and colonized pavement and hard grounds interspersed with some areas of sand known as the Frederiksted Reef System that extends from Sprat Hole south to the Westend Saltpond. This is the only reef system on the west end of St. Croix and represents the dominant benthic habitat along most of the west coast of the island. Smith et al. (2014) found that the lack of colonization by elkhorn and staghorn corals on the west and south coasts of St. Croix is likely the result of limited availability of shallow hard bottom habitat in much of the area, as well as erosion of colonies and anthropogenic effects decades before monitoring. Therefore, we believe the action area, with significant hardbottom with very little anthropogenic effects, is an important area for recovery of the species.

Historically, the highest densities of elkhorn corals in St. Croix were north of Rainbow Beach (immediately south of the Amalago Bay project) and extended along the west coast toward Ham's Bluff (on the northwest corner of St. Croix) due, in part, to the sedimentary-sand system around Sandy Point on the southwest corner of St. Croix (K. Amon-Lewis, The Nature Conservancy, pers. comm. to J. Moore, NMFS, September 24, 2014, and T. Smith, UVI, pers. comm. to J. Moore, NMFS, September 25, 2014). NOS has now found colonies of both elkhorn and staghorn corals in the areas of Sprat Hole, Ham's Bluff, and Butler Bay north of the project and south of the Frederiksted Pier near the Westgate Saltpond (S. Pittman, NOS Contractor, pers. comm. to L. Carrubba, NMFS, December 12, 2014) indicating that the habitat in the action area is important expansion and recovery habitat for the species.

4.4 Factors Affecting Elkhorn, Staghorn, Lobed Star, and Mountainous Star Corals, and Acroporid Coral Critical Habitat within the Action Area

Activities funded, authorized, or carried out by federal agencies, state agencies, and private entities have been identified as threats and may affect critical habitat for staghorn and elkhorn corals and colonies of elkhorn, staghorn, lobed star, and mountainous star corals in the action area. The activities that shape the environmental baseline in the action area of this consultation are fisheries, effects of vessel operations, private vessel traffic, marine pollution, and natural disturbance. Climate change is also likely to play an increasingly important role in determining the abundance of ESA-listed coral species and the conservation value of elkhorn and staghorn coral critical habitat around St. Croix. High thermal stress caused by climate change has been identified as the greatest threat the coral reef ecosystems in USVI (Smith et al. 2011a). The 2005 mass bleaching event caused a 50% decline in coral cover, particularly of the dominant *Orbicella* species complex in waters less than 25 m deep, the largest documented loss of coral in USVI history (Smith et al. 2011a). Recovery has been marginal at most sites since the 2005 bleaching event (Smith et al. 2011a).

Although many regulations exist to protect corals (see Section 4.4 Fisheries), including ESA-listed corals, many of the activities identified as threats still adversely affect ESA-listed coral species and acroporid coral critical habitat. Poor boating and anchoring practices, poor snorkeling and diving techniques, and destructive fishing practices cause physical damage to habitat and ESA-listed coral colonies. Nutrients, contaminants, and sediment from point and non-point sources create an unfavorable environment for reproduction and growth of corals by promoting overgrowth of hard substrate by algae or the buildup of sediment layers that prohibit coral settlement. There are existing developments on the west coast of St. Croix, but none of the existing developments are at the scale and density proposed as part of the Amalago Bay project. The NOAA Reef Prioritization Tool currently in development (an objective, data driven, decision support framework to help resource managers prioritize coral reef ecosystems for conservation investment in the USVI) indicates that the watershed where the Amalago Bay project is located currently poses little threat to nearshore waters, from landbased sediment and contamination due to the low level of development (S. Pittman, NOS Contractor, pers. comm. to L. Carrubba, NMFS, December 12, 2014). Boating and anchoring is currently not the most significant issue impacting the action area, although there is evidence of anchor damage due to recreational diving activities along the shelf edge based on surveys conducted at the Sprat Hole monitoring site as part of the Territorial Coral Reef Monitoring Program (Smith et al. 2011a). South of the action area in the area of Frederiksted Pier and the town of Frederiksted, boating and anchoring are more frequent, including infrequent visits by cruise ships to Frederiksted Pier. There are also scattered sailboat mooring and occasional jet ski operations associated with commercial businesses on the southern boundary of the property in the Rainbow Beach area.

Fisheries

Several types of fishing gears used within the action area may adversely affect acroporid coral critical habitat and coral colonies. The low abundance of important fishery species around St. Croix was noted in the results of the Territorial Coral Reef Monitoring Program. This is also thought to be part of the reason reefs around St. Croix have not recovered following the 2005 bleaching event as the lack of herbivorous fish and invertebrates is thought to have contributed to the colonization of affected reef areas by an abundance of macroalgae and filamentous cyanobacteria, which limit coral regrowth and recruitment (Smith et al. 2011a). Fishing pressure measured by the number of registered commercial fisherman versus shelf areas with less than 64 m depths is approximately 4 times greater on St. Croix than on St. Thomas/St. John, likely because St. Thomas/St. John has more deep shelf area, and shallow waters around St. Croix were found to have more intensive netting and spearfishing (Smith et al. 2011a). A large amount of derelict fishing gear was found at the Sprat Hole monitoring site (where staghorn and lobed star and mountainous star corals are present) directly offshore of the Amalago Bay project over the course of the Territorial Coral Monitoring Program leading to impacts to the shelf edge reef (Smith et al. 2011a).

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

For all fisheries for which there is an FMP or for which any federal action is taken to manage that fishery, impacts are evaluated under Section 7 of the ESA. NMFS reinitiated Section 7 consultations for the Coral, Queen Conch, Reef Fish, and Spiny Lobster FMPs under the jurisdiction of the Caribbean Fishery Management Council (CFMC) when elkhorn and staghorn corals were listed and critical habitat was designated for these corals. NMFS concluded that the implementation of the Coral FMP would have no effect on ESA-listed corals or acroporid coral designated critical habitat. NMFS determined that the Queen Conch FMP is not likely to adversely affect elkhorn and staghorn corals or their designated critical habitat. NMFS determined the Reef Fish and Spiny Lobster FMPs would adversely affect but not jeopardize elkhorn and staghorn corals and would adversely affect but not destroy or modify their designated critical habitat. NMFS reinitiated consultation for the Spiny Lobster and Reef Fish FMPs on September 26, 2016 to consider the potential effects of these fisheries on pillar, rough cactus, lobed star, mountainous star, and boulder star corals. On January 19, 2016, NMFS determined that allowing the continued authorization of fishing under the Spiny Lobster and Reef Fish FMPs was not likely to adversely affect pillar, rough cactus, lobed star, mountainous star, and boulder star corals.

Vessel Operations

Potential sources of adverse effects from federal vessel operations in the action area include operations of the USCG and NOAA. 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 ESA-listed corals and acroporid coral critical habitat. At the present time, however, they present the potential for some level of interaction.

Commercial and recreational vessel traffic can adversely affect ESA-listed coral colonies and acroporid coral critical habitat through propeller scarring, propeller wash, and accidental groundings. Based on information from the NOAA Restoration Center (RC) and NOAA's ResponseLink, reports of accidental groundings are becoming more common in USVI and Puerto Rico, but numerous vessel groundings are likely not reported. There are no reports of vessel groundings in the project area, although there are DPNR reports regarding anchor damage to corals and coral habitat in the area of Frederiksted Pier (DPNR unpublished data). Toller (2005) reported that approximately 13% of the Frederiksted Reef System had been impacted by vessel anchoring, in particular in the area of Frederiksted. At present, there is little vessel traffic in the project area, although Smith et al. (2011a) report that the shelf edge reef is visited by recreational divers and fishers. Smith et al. (2011a) noted impacts to the reef from anchoring of vessels associated with dive and snorkel commercial operations, as well as private vessels, including at their monitoring site directly offshore of the Amalago Bay project. During site visits to the project site in 2008 and 2011, NMFS observed a few sailboats moored in the area, but no motorized vessels. The proliferation of vessels around St. Croix is currently limited to areas with existing marinas, in particular on the north coast of the island in the areas of Salt River and Christiansted. The only area on the west coast where recreational vessel traffic is common is around Frederiksted where there are also public boat ramps and a small marina used largely by fishers. Through the Section 7 process for dock, port, and marine construction activities under the jurisdiction of the USACE, NMFS will continue to establish conservation measures to ensure that the construction and operation of these facilities avoids or minimizes adverse effects to ESA-listed species and critical habitat.

ESA Permits

Section 10(a)(1)(a) of the ESA allows issuance of permits for take of certain ESA-listed species for the purposes of scientific research, and section 10(a)(1)(B) authorizes issuance of permits for take of listed species incidental to other activities under certain conditions. Section 10 permits are not required for research on ESA-listed corals because they are listed as threatened and the

4(d) rule that was promulgated for elkhorn and staghorn corals found that permits from VIDPNR in the USVI were sufficiently protective such that a Section 10 permit was not required from NMFS for these species. The other 5 species of listed corals do not have a 4(d) rule therefore no Section 9 prohibitions apply and a Section 10 permit for directed take of these species is not required at this time.

Coastal Development

Anthropogenic sources of marine pollution, while difficult to attribute to a specific federal, state, local or private action, may indirectly affect coral colonies and coral critical habitat in the action area. Nutrient loading from land-based sources, such as coastal communities, are known to stimulate plankton blooms in closed or semi-closed estuarine systems and algal blooms in these areas, as well as in nearshore waters. As noted previously, water quality monitoring studies by DEP in waters around USVI indicate that surface waters are affected by increasing point and non-point source pollution from failing septic systems, discharges from vessels, failure of BMPs on construction sites, and failure of on-site disposal methods (Rothenberger et al. 2008). These factors result in increased sedimentation and nutrient transport, bacterial contamination, and trash and other debris entering surface and nearshore waters from developed areas. DEP reports that water quality around USVI continues to decline based on monitoring data from around USVI. This is indicated by the designation of 69 areas as impaired in 2006 versus 50 in 2005 (Rothenberger et al. 2008). The 2012 impaired waters list included 98 sites and the 2016 list includes 89 sites throughout USVI, indicating that water quality continues to decline throughout USVI. The 2016 impaired waters list includes 34 sites around St. Croix (https://www.epa.gov/sites/production/files/2017-02/.../2016_usvi_303d_list.pdf). In 2010, 2012, and 2016, Prosperity, nearshore at Rainbow Beach was on the impaired waters list due to high turbidity, apparently associated with multiple non-point sources from businesses located along the beach. This is adjacent to the southern boundary of the Amalago Bay property and is likely due to the Prosperity Homes subdivision, the only subdivision in the area. There were no previous reports of impairment for this area according to the 2010 EPA impaired waters report (http://ofmpub.epa.gov/waters10/attains_impaired_waters.impaired_waters_list?p_state=VI&p_cycle=2012). In the 2016 report, Sprat Hall Beach, north of the project site, is listed as impaired due to dissolved oxygen, phosphorous, turbidity and enterococci. Increases in pollutant levels and sediment loading result in habitat degradation leading to the loss of suitable habitat for coral settlement and growth due to increased algal growth and sedimentation as has been reported for sites around USVI. A study of 3 sites in Puerto Rico showed that resuspension of marine sediments did not significantly affect coral growth but sedimentation by terrigenous sediments in reef areas had a negative effect on coral growth rates (Torres 2001a).

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 in St. John from 1980-1981 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) possibly due in part to the degradation of habitat due to the increased sediment cover on hard substrate as well as physical impacts to the corals themselves. 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 storms was 60%, while it was only 39% between the 10- and 50-year storms (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 ft per second (cfs) of runoff intensity were more likely to contain better-developed nearshore reefs (Hubbard et al. 1987).

Sediment core data from nearshore wetland and coastal embayments around St. Thomas and St. John show that, for the 15-25 years of data analyzed by the researchers, sedimentation rates have increased from 1-2 orders of magnitude (Rogers et al. 2008b). 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. 2008b) (Smith et al. 2008). During a severe rain event, sediment load can increase to > 30 mg per cm^2 per day (Rogers et al. 2008b). 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 15 times, in comparison to undeveloped watersheds, and mean organic matter flux rates by up to 10 times. These changes in sedimentation affect the quality of the acroporid coral critical habitat for supporting coral recruitment and growth. Increased sediment on hard bottom and reefs reduces larval settlement by coral planulae and the survival of coral recruits and juveniles in addition to having lethal and sublethal effects on established coral colonies (Babcock and Smith 2000, Fabricius 2005). Similarly, increases in nutrient loading to nearshore waters can increase algal and phytoplankton growth, affecting coral habitat function related to coral settlement and growth. In USVI excessive inputs of nitrogen and phosphorous, in particular, in highly developed areas are negatively impacting coastal and marine ecosystems and variations in chlorophyll levels (indicative of phytoplankton growth) among and within embayments. This supports the conclusion that nutrient enrichment has increased in areas with greater human development (Smith et al. 2013).

From 2001-2005, 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 mid-shelf reefs and over 45-fold greater than for shelf edge reefs (Smith et al. 2008). The clay and silt fraction is an indicator of terrigenous material content of the sediments due to terrestrial development on steep slopes with poor soils and the transport of eroded soils in stormwater runoff to nearshore waters. 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 Nowlis 2001). Reef sites exposed to average sedimentation rates between 10 and 14 mg per cm^2 per day showed a 38% increase in the number of coral colonies experiencing pigment loss (pigment loss is indicative of poor coral health caused by stressors such as sedimentation or increased temperature, as further described in this paragraph) compared to reef sites exposed to sedimentation rates between 4 and 8 mg per cm^2 per day (Nemeth and Nowlis 2001). The findings of (Nemeth and Nowlis 2001) correspond to those of other studies in the USVI regarding coral tolerance thresholds for sedimentation that result in declines in coral health, as well as habitat degradation (Rogers et al. 1984a, Rogers et al. 2008b)). The tolerance threshold of 10 mg per cm^2 per day suggested by these studies was exceeded during 6 of the 13 sample periods, indicating chronic sediment stress approximately 50 % of the time in the (Nemeth and Nowlis 2001) study associated with a development porject. Bleaching of corals was strongly correlated to sedimentation rate, indicating that bleaching can be a response to sediment stress.

Natural Disturbance

Hurricanes and large coastal storms can also harm coral colonies and acroporid coral critical habitat. 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). Major hurricanes have caused significant losses in coral cover and changes in the physical structure of many reefs in Puerto Rico and the USVI. Based on data from the Caribbean Hurricane Network, there have been a total of 15 hurricanes and tropical storms that have affected Puerto Rico between 1975 and 2010 with 5 hurricanes occurring between 1995 and 1999. Hurricane David in 1979 caused violent sea conditions and flooding and was followed 5 days later by Tropical Storm Frederick which resulted in additional flooding. Tropical Storm Klaus in 1984 affected some parts of USVI. Hurricane Hugo in 1989 led to violent sea conditions and major flooding across the USVI. Hurricanes Marilyn in 1995, Bertha in 1996, Georges in 1998, and Lenny in 1999 led to additional impacts to reefs already suffering damage from Hurricane Hugo. Tropical storms and hurricanes in 2004, 2008, and 2010 also resulted in severe flooding across USVI. Flooding from hurricane events leads to transport of land-based sources of pollutants to reefs, along with an influx of freshwater to nearshore environments that affects water quality, in addition to physical damage caused by the storms themselves. In the action area, tropical storms frequently cause beach erosion, sometimes exposing bedrock along portions of the coast due to heavy surge. Following heavy rain events over the past several years, the amount of sediment on nearshore hard grounds near the 3 natural drainage outlets along the shore of the Amalago Bay project has increased based on NMFS's observations during site inspections in 2008 and 2011.

As discussed in Section 4.2 Disturbances, Hurricanes Irma and Maria passed through the Caribbean in September 2017 with Hurricane Maria having a significant impact on St. Croix. Although St. Croix and other areas of USVI are still recovering, assessments of in-water habitats have not been completed in all areas but information to date indicates that damage in reef areas around St. Croix, including Buck Island, Salt River, and Cane Bay, was limited (Figure 11). Therefore, while there is a possibility that the environmental baseline described here may have been degraded by hurricane damage, survey results to date indicate that many coral reef sites around the islands were relatively unaffected. Given the low levels of development, the fact that much of the damage found in surveys to date indicates that vessel groundings had a large impact on coral habitats and because the action area does not have large numbers of vessels present in nearshore waters, the action area likely continues to provide habitat for ESA-listed corals and the essential feature of elkhorn and staghorn coral critical habitat.

Conservation and Recovery Actions Benefiting ESA-Listed Corals and Coral Critical Habitat

The CFMC has established regulations prohibiting the use of bottom-tending fishing gear in certain areas in the federal waters of the Exclusive Economic Zone (EEZ). These are areas that are either closed to any fishing seasonally or permanently closed to all fishing. The Territory has similar fisheries regulations for both commercial and recreational fishers. In addition to regulations, education and outreach activities as part of the NOAA Coral Reef Conservation Program (CRCP), as well as through NMFS's ESA program, are ongoing through the Southeast Regional Office. NOAA RC has also established a contract position in Puerto Rico to participate in grounding response in Puerto Rico and USVI and carry out restoration activities. The summaries below discuss these measures in more detail.

A recovery team comprised of fishers, scientists, managers, and agency personnel from Florida, Puerto Rico, and USVI, and federal representatives was convened by NMFS and has created a recovery plan based upon the latest and best available information for elkhorn and staghorn corals and their habitat (http://sero.nmfs.noaa.gov/protected_resources/coral/documents/acropora_recovery_plan.pdf).

Regulations Reducing Threats to ESA-Listed Corals

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

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

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

On November 26, 2008, NMFS published a Final Rule which designated critical habitat for ESA-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 is avoided as part of a Section 7 consultation with NMFS for the action. This reduces the threats to elkhorn and staghorn corals by adding a layer of protection to habitat nessesary for the conservation of the species.

Other ESA-Listed Coral and Coral Critical Habitat Conservation Efforts

Restoration

The Final Section 4(d) Rule for elkhorn and staghorn corals allows certain 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, to be conducted promptly without the need for ESA permits. Restoration activities are also carried out to restore damaged critical habitat.

Outreach and Education

The NOAA Coral Reef Conservation Program, through its internal grants, external grants, and grants to the Territory and the CFMC, has provided funding for several activities with an education and outreach component for informing the public about the importance of the coral reef ecosystem of the USVI. The Southeast Regional Office of NMFS has also developed outreach materials regarding the listing of elkhorn and staghorn corals, the proposed listing of 7 other coral species, the ESA Section 4(d) rule for elkhorn and staghorn corals, and the designation of coral critical habitat. These materials have been circulated to constituents during education and outreach activities and public meetings, and as part of other Section 7 consultations, and are readily available on the web at: outreach materials[18] and Southeast Regional Office of NMFS[19].

4.4.1 Summary and Synthesis of Environmental Baseline for Elkhorn, Staghorn, Lobed Star, and Mountainous Star Corals and Acroporid Coral Designated Critical Habitat

In summary, several factors are presently adversely affecting elkhorn, staghorn, lobed star, and mountainous star corals, and acroporid coral critical habitat in the action area. These factors are ongoing and are expected to occur contemporaneously with the proposed action. Marine pollution as a result of coastal development is expected to pose the greatest threat to elkhorn, staghorn, lobed star, and mountainous star coral colonies and acroporid coral critical habitat in the action area based on data from surveys such as Smith et al. (2011a), (Nemeth and Nowlis 2001), (Hubbard et al. 1987), and (Smith et al. 2008). Vessel traffic will also continue to result in damage to acroporid coral critical habitat and abrasion and breakage of lobed star and mountainous star coral colonies and potentially elkhorn and staghorn coral colonies due to

[18] https://www.fisheries.noaa.gov/corals

[19] https://www.fisheries.noaa.gov/region/southeast

accidental groundings and poor anchoring techniques. Smith et al. (2011a) found evidence that recreational boating and anchoring associated with the use of areas immediately offshore of the proposed Amalago Bay development may be impacting acroporid coral critical habitat. Fishing activities, in particular the loss of fishing gear which was found to be causing impacts in the Sprat Hole monitoring station offshore of the proposed project (Smith et al. 2011a), as well as marine operations and natural disturbance, are also expected to continue to result in impacts to ESA-listed coral colonies and acroporid coral critical habitat. The collective ongoing activities that are already impacting ESA-listed corals and acroporid coral critical habitat have affected the continuity of habitat in the Frederiksted Reef System through sedimentation of nearshore reefs and hard bottom areas and physical damage to coral colonies and habitat from boating, fishing, and recreational diving and snorkeling.

These activities are expected to combine to adversely affect the quality and suitability of acroporid coral critical habitat throughout the ranges of elkhorn and staghorn coral, and in the action area. The factors adversely affecting acroporid coral critical habitat around St. Croix have led to a degraded baseline due to sediment and nutrient transport in stormwater runoff. This is evidenced by the fact that the Rainbow Beach area associated with Prosperity homes and commercial businesses (immediately south of the Amalago Bay project) was included on the EPA 2010, 2012 and 2016 Impaired Waters Lists and Sprat Hall Beach was included in the 2016 Impaired Waters List, as noted in Section 4.4 Coastal Development. According to most recent studies, the largest numbers of elkhorn coral colonies on the west coast of St. Croix have been found from Sprat Hole to Ham's Bluff despite some residential development and the impacts of a poorly managed quarry that discharges sediment to Ham's Bay. The area where the Amalago Bay development is proposed is currently largely undeveloped and characterized by good water clarity and the presence of staghorn corals directly offshore, as well as numerous lobed star and mountainous star coral colonies on nearshore hard bottoms and reefs in deeper waters (Smith et al. (2011a); S. Pittman, NOS Contractor, pers. comm. to L. Carrubba, NMFS, December 12, 2014; Toller (2005)). Therefore, it appears that the overall condition of acroporid coral critical habitat in the action area is good compared to other sites around St. Croix and on the west side of the island, meaning the habitat should be able to function to support elkhorn corals in shallow hard bottoms and reefs and staghorn corals in deeper reef and hard bottom areas. Smith et al. (2014) concluded that the existing hard bottom habitats on the west side of St. Croix are important for the recovery of elkhorn and staghorn corals in part due to the low level and extent of development when compared to other areas of St. Croix outside of protected areas.

5 EFFECTS OF THE ACTION

As described below, NMFS believes that the proposed action will adversely affect threatened green sea turtles, endangered leatherback and hawksbill sea turtles, and threatened lobed star, mountainous star, elkhorn, and staghorn corals. As part of the Opinion and because the action will result in adverse effects to ESA-listed sea turtles and corals, NMFS must evaluate whether the action is likely to jeopardize the continued existence of green, leatherback, and hawksbill sea turtles, and lobed star, mountainous star, elkhorn, and staghorn corals and, if so, develop reasonable and prudent alternatives to avoid the likelihood of jeopardy to the species. If NMFS determines the action is not likely to jeopardize the continued existence of these species, NMFS may authorize incidental take, subject to reasonable and prudent measures to minimize the effects of the take.

As described below, NMFS also believes the proposed action will adversely affect designated critical habitat for staghorn and elkhorn coral. When an action will adversely affect critical habitat, NMFS must evaluate whether a proposed action will result in the destruction or adverse modification of critical habitat and if so, develop reasonable and prudent alternatives to avoid destruction or adverse modification.

In the discussions that follow in Sections 5.1 and 5.2, we determine that there will be impacts to ESA-listed sea turtles, ESA-listed corals, and acroporid coral critical habitat as a result of the permanent loss of 2.75 acres of hard bottom, chronic impacts to 30.31 ac of hard bottom, and episodic impacts to 77 ac of hard bottom within the 2500 ft (200-ac) area extending from the property boundaries of the Amalago Bay project to the shelf edge. The 30.31 ac area is contained in the 46 acres within 500-ft from shore that was part of the applicant's benthic survey (Area A, Figure 5). The 77 ac area (Area B, Figure 5) is the reefs and colonized hard bottoms defined by the NOS benthic maps (Kendall et al. 2001). We define chronic impacts to mean adverse impacts that are frequently recurring, at a high to moderate intensity, and are expected to continue for a long time (i.e., decades). We define episodic impacts to mean adverse impacts that occur infrequently, at a low to moderate intensity, and are expected to continue for decades.

5.1 Effects of the Action on Green, Leatherback, and Hawksbill Sea Turtles

In this effects analysis we consider potential impacts to sea turtles in the marine environment within the action area that may result from the proposed project. To determine the exposure to and severity of potential impacts to these species we first estimate the local in-water populations around St. Croix, then compare the in-water estimates to the estimated number of lethal and nonlethal takes of sea turtles from the project, and discuss how that take affects the species beyond the local in-water and nesting assemblage. USFWS completed an Opinion for this project focusing on impacts to sea turtle nesting under their jurisdiction on February 10, 2014 (USFWS 2014).

There are no estimates of resident sea turtle populations, or year-round in-water sea turtle monitoring surveys in the action area. Therefore, we used the estimates of nesting females based on the 2009 DPNR nesting data (Table 4), the average number of nests in a given year based on the NPS Buck Island and USFWS Sandy Point data (Table 5), and WIMARCS nesting data (Section 4.1) to create a minimum estimate of the total adult population (males and females) for each species in waters surrounding St. Croix. In calculating the number of adult female turtles we divided the total number of nests for a given year by the number of times an average female nests per season in order to estimate the total population of female green and hawksbill sea turtles in the project area at least during nesting season if not year-round, as well as the number of female leatherbacks in the project area during nesting season. We assumed a 1:1 sex ratio in the adult population, so we double the estimated number of females to account for males not identified by nesting data. For green sea turtles, adult females are likely to be from the South Atlantic DPS based on genetic analyses of nesting females that was used to create the DPS designations. However, because adult males may not be confined to the South Atlantic verus North Atlantic DPS and because adult females present in USVI waters outside the nesting season may also be from the North Atlantic or South Atlantic DPS, we do not specify a particular DPS for these animals. We do assume that nests and hatchlings are all from the South Atlantic DPS of green sea turtles. We estimated a total population of 219 adult green sea turtles (287 nests in a given year divided by 2.625 [average number of nests per season laid by a single female per season] multiplied by 2 because we are assuming a 1:1 sex ratio), 536 adult leatherback sea turtles during nesting season (1200 nests in a given year divided by 4.475 multiplied by 2), and 441 adult hawksbill sea turtles (538 nests in a given year divided by 2.4425 multiplied by 2) around St. Croix. The number of adult individuals utilizing the waters of the action area is expected to be a fraction of those totals. While no numbers of adult residents in the area are available, the nesting beach in the property boundry represents 1.79% of the total nesting beaches in St. Croix (USFWS 2014), and thus that percentage is likely somewhat representative of the proportion of St. Croix turtles using the nearshore waters of the action area as well.

It is likely that the colonized hard bottom in the action area serves as a foraging and resting habitat for both green and hawksbill juveniles, as well as adults. This is supported by observations of both green and hawksbill sea turtles during site inspections as well as information on sea turtle sightings in the benthic reports prepared for the Amalago Bay project. Nesting females of all three species likely use the nearshore waters of the action area during internesting intervals.

Using the NOS benthic maps for the area, we estimate that, of the 57,600 ac around St. Croix composed of linear reefs, colonized pavement, colonized bedrock, and other coral habitats, approximately 582.5 ac are nearshore colonized hard bottom habitats. As these habitats are preferred by juvenile green and hawksbill sea turtles, we used this acreage to estimate the potential population of juvenile green and hawksbill sea turtles around St. Croix in a given year. Using the estimate of 5 immature green sea turtles per acre from the (Wershoven and Wershoven 1992b) study and between 0.11 and 0.5 immature hawksbills per acre from the (Diez and Dam 2002) study, we estimate that, within the 582.5 ac of nearshore colonized hard bottom habitat around St. Croix, there could be 2,913juvenile greens and between 64 (if there are only 0.11 turtles per acre) and 291 (if there are 0.5 turtles per acre) juvenile hawksbills. Data does not exist that would allow us to separate juvenile green sea turtles into the North or South Atlantic DPS so numbers of juveniles could be from either DPS.

We elected to estimate the numbers of immature green and hawksbill sea turtles using the (Wershoven and Wershoven 1992b) and (Diez and Dam 2002) studies for the following reasons: 1) the mixture of sparse seagrass beds, colonized hardgrounds, and linear reefs in the action area is similar to the mixture of habitats in these studies; 2) observations during site inspections, as well as during in-water surveys conducted for the proposed project indicate that green and hawksbill sea turtles frequent the action area so we can infer that there is refuge and foraging habitat for these turtle species in the action area; and 3) the lack of in-water sea turtle data for St. Croix that requires that we use estimates from other studies. We recognize that the Broward County, Florida, study area in (Wershoven and Wershoven 1992b) and the Mona and Monito Island, Puerto Rico, study areas in (Diez and Dam 2002) are different from the St. Croix action area of the Amalago Bay project in terms of geographic location. However, we believe that the habitats in both studies are similar to the habitats in the action area, making these studies the best available information to enable us to estimate the in-water density of green and hawksbill sea turtles in the action area and around St. Croix.

5.1.1 Marina Facilities and Created Beach

In-water Dredging and Construction: The in-water dredging and construction activities associated with the creation of the beach, the marina and its flushing channels, and jetty construction may result in adverse impacts to adult and hatchling green, leatherback, and hawksbill sea turtles, and juvenile green and hawksbill sea turtles. The applicant has incorporated NMFS's *Sea Turtle and Smalltooth Sawfish Construction Conditions* in the avoidance and minimization measures for in-water construction and dredging (see Section 2.1 for details of the applicant's proposed avoidance and minimization measures). Compliance with these conditions will provide a measure of protection to adult and juvenile sea turtles by requiring work to stop if a sea turtle is seen within 50 feet of operating machinery. Sea turtle observers will also be on-site daily to monitor for sea turtles before, during, and after marine construction activities according to the applicant. The applicant will install a double floating turbidity boom seaward of the areas of dredging and construction, which will serve as a barrier to sea turtles from entering the dredging footprint. The applicant also proposes curtailing dredging operations if sea conditions limit the functional efficiency of the turbidity curtain. Turbidity booms will also be installed seaward of the created beach and will similarly serve as a barrier to sea turtles from entering the construction footprint. Thus, we believe that the risk of injury to sea turtles from in-water equipment operation during dredging and in-water construction will be extremely unlikely to occur.

Both hatchlings and adult sea turtles can also become disoriented by structures along the coast and in nearshore waters. Stranding data from St. Croix documented a female leatherback that become disoriented after nesting in Ham's Bay and became entangled in rip rap. Disorientation and entanglement or concentration along the rock jetties and/or the offshore breakwaters will make hatchlings and female sea turtles more vulnerable to predators such as dogs on land and sharks (in the case of adults – there are several reported cases of shark attacks in the Sandy Point area in particular) and sea birds (in the case of hatchlings) in the water. Hawksbill hatchlings, which are the most likely to be in the area of the created beach because hawksbills are the most common nester in the project area, tend to swim slowly away from the beach and shelter in floating algal mats and other marine detritus (Chung et al. 2009). Depending on current patterns, marine detritus could concentrate along the jetties, which could make the jetties more attractive to hatchlings looking for shelter, defeating the hatchling's attempts to disperse offshore. Studies have shown that hatchling mortality rates range from 30 - 60% as the animals leave the beach and swim toward open water, and only 2.5 in 1,000 reach adulthood (Frazer 1992, Pilcher et al. 1999). Congregation of hatchlings along the jetties would make them more vulnerable to predation by seabirds and other marine organisms. Hatchling sea turtles are preyed upon by large predatory fishes such as jacks, tarpon, barracuda, and grouper as they attempt to reach the open ocean (Stewart and Wyneken 2004, Whelan and Wyneken 2007). (Stewart and Wyneken 2004) reported in-water hatchling survival rates of 95% adjacent to a natural nesting beach versus an area adjacent to a Florida sea turtle hatchery that had in-water hatchling survival rates of only 72%. This difference is likely due to the concentration of hatchlings – and thus their predators -- in waters near the hatchery versus along the natural nesting beach. Studies have shown that predation rates on hatchling sea turtles are also much higher when hatchlings are more concentrated in a particular area (Wyneken et al. 1998, Wyneken et al. 2000). Marine predators are known to learn to wait at locations of increased concentration for the hatchlings (Wyneken et al. 2000). Increases in predation on hatchlings by avian predators are expected as well because the jetties will provide a perch for seabirds. Therefore, we expect that the construction of the jetties and created beach will adversely affect sea turtles, particularly hatchlings due to disorientation and increased susceptibility to predation.

One of the 2 proposed jetties will extend out from the existing shoreline up to approximately 300 ft and the other less than 100 ft. There will be available existing and/or created beach nesting habitat to either side of the jetties. Jetties orient primarily perpendicular to the shoreline. The impact of jetty structures on hatchling sea turtle entrapment and disorientation is expected to be less than that for nearshore structures that run parallel to the shoreline, as hatchling sea turtles typically swim directly perpendicular to the beach, out to open water. Therefore, it is unlikely that hatchlings emerging more than a short distance on either side of a jetty would become entrapped, impinged, or disoriented by the jetty structure. Because of limited information available to determine impacts, and the ESA mandate to err on the side of the species, we take a conservative approach and make assumptions that may overestimate numbers of hatchlings impacted. Despite the tendency to swim directly out to sea perpendicular to the beach, we will assume that any nests within 200 ft of either side of each jetty (thus 800 ft of beach or 0.24 km) have the potential to be impacted by impingement, entrapment, disorientation, and/or predator concentration resulting from the structures. We also use the nesting densities from the 2009 DPNR data (Table 4), which are higher than the estimates in the applicant's 2013 BA. Based on those nesting densities we estimate that hatchlings from as many as 1 green turtle nest (4.3 nests/km x 0.24 km), 3 leatherback nests (12.97 nests/km x 0.24 km), and 14 hawksbill nests (59.4 nests/km x 0.24 km) could be impacted annually.

Based on available data for green sea turtles from nearby Buck Island for 2001-2004, the mean number of eggs in each green sea turtle clutch was 113.72 with a mean emergence success rate of 79.69%. This results in a mean number of 91 green turtle hatchlings per nest. With 1 estimated green turtle nest in the area where hatchlings could be impacted by the jetties, we estimate that 91 green turtle hatchlings from the South Atlantic DPS per year have the potential to be killed as a result of the jetties.

Based on data for leatherback nests from nearby Sandy Point for 2002-2006, the mean number of eggs in each leatherback clutch was 113.24, with a mean of 78.89 yolked eggs, and a mean emergence success rate of 48.38%. This results in a mean number of hatchlings per nest of 38. With up to an estimated 3 leatherback nests in the area where hatchlings could be impacted by the jetties, we derive a total of 114 leatherback hatchlings per year with the potential to be killed as a result of the jetties.

Based on available data for hawksbill sea turtles from nearby Buck Island for 2001-2004, the mean number of eggs in each clutch is 142.67, with a mean emergence success of 60.54%. This results in a mean number of 86 hawksbill turtle hatchlings per nest. With 14 estimated hawksbill turtle nests in the area where hatchlings could be impacted by the jetties, we estimate that 1,204 hawksbill turtle hatchlings per year have the potential to be killed as a result of the jetties.

For all 3 of the species' estimates of impacts from the jetties, it is important to note that the numbers of potentially-impacted hatchlings represent all possible hatchlings from those nests, which likely overestimates the number of hatchlings that would face potential impacts from the jetties. It does not account for natural hatchling mortality on the beach via predation before they even get to the water, or the possibility of nest relocations required by USFWS.

The construction of the marina channels and jetties and the creation of the beach on the island that will be formed from the excavation of the navigation and flushing channels requires the dredging of sand, seagrass, and colonized hard bottom in order to create the navigation and flushing channels and the placement of fill to construct the jetties and beach. Of the area to be altered or lost due to in-water construction and dredging, 0.8 ac is sparse seagrass and 2.75 ac is colonized hard bottom. The applicant will conduct a pre-construction survey to determine the amount of seagrass within the in-water dredging and construction footprint and has proposed the transplant of seagrass outside the footprint. Because the seagrass in the area is *Halodule beaudettei*, which is relatively fragile, we do not believe that transplant attempts are likely to be successful and are unaware of projects that have successfully transplanted this species. Therefore, we expect there will be some loss of seagrass habitat. Similarly, the colonized hard bottom that will be eliminated by dredging represents a loss of sea turtle habitat. Pieces of hard bottom that will be dredged to create the offshore breakwater structures the applicant has proposed as mitigation for direct impacts to hard bottom will lead to other impacts to refuge and foraging habitat for sea turtles as discussed below, leading to a net loss of seagrass and colonized hard bottom habitat. Both resident and transient juvenile hawksbill and green sea turtles currently using the colonized hard bottom and patch reef habitats in the project area will be affected by the loss and degradation of foraging and refuge habitat. The estimated permanent loss of 3.49 ac of sparse seagrass and colonized hard bottom would result in up to 17 resident juvenile green sea turtles (i.e., 5 resident juvenile turtles per acre multiplied by 3.49 ac = 17.45 turtles) and up to 2 resident juvenile hawksbill sea turtles (i.e., 0.5 turtles per acre multiplied by 3.49 ac = 1.745 resident juvenile turtles), being permanently displaced. This could lead to decreased health due to stress and lower foraging opertunities, as well as increased exposure to predators.

There is no information on adult home range densities in the nearshore waters off the project area, though we would expect a much lower density of adult sea turtles compared to juveniles. However, it is known that reproductively mature hawksbill sea turtles (Proietti et al. 2012, Rincon-Diaz et al. 2011) and green turtles (Bresette et al. 2010) tend to establish foraging home ranges in waters further offshore and deeper than do juveniles, so the impact of habitat disruption or degradation from this project is likely to have a lesser effect on mature turtles than juveniles. Based on the habitat preferences of sea turtles according to their life stage, and the reduced importance of the potentially impacted nearshore waters for adult foraging habitat, we conclude that the impact of habitat loss would be insignificant for adult sea turtles.

The use of a spud barge during dredging and construction of the temporary trestle has the potential to result in impacts to sea turtle refuge and foraging habitat. Recent monitoring reports from projects in USVI have documented the impacts of spudding on benthic habitats. Spudding leads to holes up to 10 ft deep in the marine bottom depending on the substrate. If these holes are not back-filled once the use of the spud barge is complete, they remain indefinitely. The applicant has proposed to survey the area prior to spudding to ensure spuds are not dropped in areas containing coral habitats. This practice will protect hard bottoms and patch reefs from impacts that would lessen their utility for sea turtles but will not protect seagrass beds and will not prevent the creation of holes from the use of spuds. Although this impact represents a permanent loss in a localized area (each spud hole), given the extent of seagrass and other soft bottom habitats in the action area, we do not expect it to have a measurable impact to sea turtles associated with the loss of habitat in spud holes. Therefore, we believe that this potential spudding effect will be insignificant.

Pile-driving: Pile driving can have acoustic impacts to sea turtles. Acoustic effects to sea turtles as a result of noise created by construction activities can physically injure animals or change animal behavior in the affected areas. Injurious effects can occur in two ways. First, effects can result from a single noise event exceeding the threshold for direct physical injury to animals, and these constitute an immediate adverse effect on these animals. Second, effects can result from prolonged exposure to noise levels that exceed the daily cumulative exposure threshold for the animals, and these can constitute adverse effects, if animals are exposed to the noise levels for sufficient periods. Behavioral effects can be adverse if such effects interfere with animals' migration, feeding, resting, or reproducing, for example. Acoustic impacts to sea turtles in the water are likely to occur from the use of an impact hammer to set the piles as part of the installation of approximately 45 steel piles for the construction of the temporary trestle pier for the crane that will be used during marina construction.

There are currently no established thresholds for injurious or behavioral effects to sea turtles from the use of a vibratory hammer to drive piles. However, we believe it is extremely unlike that the installation of temporary piles by vibratory hammer will result in injurious or behavioral noise effects to these animals. Given the mobility of sea turtles, we expect them to move away from noise disturbances. Because there are benthic habitats that may be used by green and hawksbill sea turtles outside the footprint of proposed pile-driving activities, we believe animals will use other refuge and foraging habitats rather than stay in the area of vibratory pile-driving and other construction activities. Similarly, if piles are installed during nesting season, the beach area that is not immediately adjacent to pile driving activities will still be accessible to nesting females because pile driving will not occur at night. If an individual choses to remain within the behavioral response zone, it could be exposed to behavioral noise impacts during pile installation. Since installation will occur only during the day, leatherback, green, and hawksbill sea turtles will be able to resume normal activities during quiet periods between pile installations and at night. Therefore, we anticipate any behavioral effects to sea turtles from vibratory pile-driving activities for the installation of temporary piles will be insignificant.

The applicant did not provide details in terms of the size of the piles, number of strikes per pile for installation, or number of piles that will be installed per day. Therefore, we assumed that the applicant will be using cast-in-steel shell (CISS) piles with a diameter of 30 in because the pier will be used to support a large crane during construction. To be conservative, we have used thresholds accepted by NMFS, the Navy, and the Air Force in recent consultations[20]. We use

[20] NMFS' Biological Opinion for the U.S. Navy's Atlantic Fleet Training and Testing Activities from November 2013 through November 2018 and the National Marine Fisheries Services' Promulgation of Regulations and Issuance of Letters of Authorization Pursuant to the Marine Mammal Protection Act for the U.S. Navy to "Take"

these thresholds rather than the default marine mammal thresholds that have been used in the past because they are suitably protective of sea turtles. These thresholds are 226 dB and 232 dB for peak pressure for the onset of temporary and permanent effects, respectively, and cumulative SELs of 189 dB and 204 dB for temporary and permanent effects, respectively.

We assume that up to 4 30-in diameter steel piles will be driven per day and up to 600 strikes will be required for each pile for a daily total of 2,400 strikes. Based on our noise calculations, the peak pressure threshold for injurious noise effects would be exceeded immediately adjacent to the source for temporary and permanent effects, respectively, to sea turtles. The cumulative sound exposure level of multiple pile strikes over the course of a day would cause injury to sea turtles up to 932 ft (284 m) and 93 ft (28 m) from the pile for temporary and permanent effects, respectively. The applicant notes that bubble curtains may be used during pile-driving activities associated with the consultruction of the temporary trestle. NMFS assumes that an effectively implemented bubble curtain can reduce sound levels by up to 10 dB when using an impact hammer to drive steel piles. We base these sound reductions on CALTRANS (2015) and observations made by NMFS' biologists during pile driving activities with and without sound reduction measures. NMFS calculated the distance to reach the thresholds for temporary and permanent injurious sound impacts for attenuated pile driving assuming a 10 dB reduction in sound levels. This means that the cumulative sound exposure would be reduced to 201 ft (61 m) and 20 ft (6 m) away from the pile for temporary and permanent effects, respectively. These reductions in sound only apply to the use of bubble curtains. If double turbidity barriers are used around pile-driving activities instead, based on the applicant noting the possible use of one or the other but not both around pile-driving activities, NMFS does not anticipate any reduction in injurious sound levels.

In terms of potential behavioral impacts to green and hawksbill sea turtles associated with pile driving, we use the metric of root mean square (RMS) to determine the extent of potential behavioral noise disturbance. The RMS threshold for behavioral disturbance is assumed to be 160 dB RMS for sea turtles. The installation of 30-in diameter steel pipe piles using an impact hammer could result in behavioral impacts up to 3,281 ft (1,000 m) from the pile for sea turtles. This effect is reduced to a radius of 707 ft (215 m) if bubble curtains are effectively used to reduce sound levels by 10 dB.

The use of double turbidity barriers in the immediate area where pile driving is taking place (which the applicant notes is a possibility) will help to keep sea turtles further away from the area where piles are being driven, reducing the likelihood of injurious effects by excluding sea

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Marine Mammals Incidental to Atlantic Fleet Training and Testing Activities from November 2013 through November 2018 (2013), FPR-2012-9025; NMFS' Biological Opinion for the U.S. Navy's Atlantic Fleet Training and Testing Activities in the Hawaii-Southern California Training and Testing Study from December 2013 through December 2018, the National Marine Fisheries Services' Promulgation of Regulations Pursuant to the Marine Mammal Protection Act for the U.S. Navy to "Take" Marine Mammals Incidental to Training and Testing Activities in the Hawaii-Southern California Training and Testing Study Area from December 2013 through December 2018, and the National Marine Fisheries Services' Issuance of Two Letters of Authorization Pursuant to Regulations under the Marine Mammal Protection Act to "Take" Marine Mammals Incidental to Training Exercises and Testing Activities in the Hawaii-Southern California Training and Testing Study Area from December 2013 through December 2018 (2013), FPR-2012-9026; and NMFS' Biological Opinion for the Ongoing Eglin Gulf Testing and Training Range Activities by the United States Air Force (2017), FPR-2016-9151 turtles from the area as turbidity curtains are not expected to result in any reductions in injurious sound levels as noted above. The project area contains seagrass and hard bottom habitats utilized by green and hawksbill sea turtles and nesting habitat for green, hawksbill, and leatherback sea turtles, and therefore the likelihood of turtles being close enough to the pile driving activities to result in behavioral impacts still remains. The applicant proposes to use a sea turtle monitor during marina and shoreline construction, and will be following the sea turtle and sawfish construction guidelines. With the use of turbidity barriers and the sea turtle monitors we believe that the risk of effects of a single strike during pile driving will be insignificant. We also believe that with the use of turbidity barriers and sea turtle monitors, along with the fact that sea turtles would likely not remain in the area long enough for the cumulative acoustic effects to become injurious, the likelihood of daily cumulative noise exposure injuries from the pile driving is extremely unlikely to occur. Behavioral acoustic impacts, which extend to much greater distances, are likely to affect hawksbill and green sea turtles, as well as leatherbacks if the pile installation for the temporary trestle occurs during leatherback nesting season. All turtles (adults during nesting season and resident juveniles), as calculated above, would experience possible behavioral modifications due to pile driving activities. The likely scope of changes in behavior (such as startle effects, vacating the foraging area, altering trajectory of movement) are expected to be temporary, and the pile driving itself is a short-term activity limited to approximately 45 piles to be driven at a maximum of 4 per day (for a total of approximately 12 days), and there are similar habitats that sea turtles can use adjacent to the project area. Pile driving activities will take place during the day so sea turtles are expected to return to normal behaviors during breaks between pile driving at night, including nesting activities if pile driving occurs during the nesting season of one or more of the 3 species documented to nest in the action area (green, leatherback, and hawksbill sea turtles). Therefore, we believe these effects will be insignificant.

5.1.2 Marina Operation

Vessel Strikes: The need for turtles to move out of the project area to avoid construction activities and pile driving noise, in particular, will increase the risk of exposure to vessel traffic. The use of a spud barge and other work vessels during in-water construction and dredging constitutes a risk to sea turtles of vessel strikes as vessels transit to and from the project area. The applicant has proposed the use of sea turtle observers during all marina and shoreline construction activities, though there is no information indicating that the applicant has incorporated NMFS's *Vessel Strike Avoidance Measures and Reporting for Mariners* into the proposed action or developed similar avoidance measures for work vessel operation associated with the dredging and in-water construction activities. The applicant does propose the implementation of an education program to incorporate information regarding the proper operation of vessels in areas containing ESA-listed sea turtles during construction and operation of the project but the only information in the BA regarding an education program was related to sea turtle nesting. Turtles were found to flee approximately 60% of the time from slow-moving vessels (2.5 mph), but infrequently (22% of the time) when vessels were moving at moderate speeds (6.8 mph) and rarely (4% of the time) when vessels were moving fast (11.8 mph) (Hazel et al. 2007). Given that work vessels are expected to transit at slow speeds, we conclude that vessel strikes from work vessels is extremely unlikely to occur. The same is true for work vessel use during project operation given the applicant's expressed need to conduct maintenance dredging activities following storm events, which will require one month of in-water work per event.

The supplemental information provided by the applicant in October 2012 included a Marina Market Analysis by Economics Research Associates (ERA 2006). This analysis estimated that there are approximately 6915 registered vessels around St. Croix. Data from marina studies in Puerto Rico (because the ERA analysis did not include information on trips) indicate that approximately 2 months of the year, when there are summer holidays and long weekends, 37% of vessels transit to and from the marina per week, with the rest of the vessels remaining moored in the marina. Approximately 14% of vessels transit to and from the marinas per week the rest of the year. The Amalago Bay marina will hold 70 vessels, which is a 1% increase in the current number of vessels estimated to be present around St. Croix.

Based on unpublished stranding data from 2001-2010 from DPNR, 9% of strandings around St. Croix (7 of 77) are due to boat strikes. Of these, 57% (4 animals) were green sea turtles, 29% (2 animals) were leatherback, and 14% (1 animal) could not be identified to species due to the condition of the carcass. With less than 1 documented mortality per year in St. Croix attributable to boat strikes, we believe that the increased risk of boat strikes from a 1% increase in vessels due to the proposed Amalago Bay project is extremely unlikely to result in additional boat strikes.

Fueling Facility and Accidental Spills: The marina will have a fueling facility. Spills from the facility could affect various life stages of sea turtles. Because the inland marina is designed with a flushing channel for flow-through and short residence times within the basin, petroleum spills could result in the transport of petroleum products out of the marina basin and along the shoreline to nesting beaches. However, according to the information provided by the applicant, the marina will be operated under requirements for containment and clean up plans (developed by the applicant), operational limitations to prevent spills, sewage and vessel maintenance guidelines and other measures. Based on this information, we believe it is extremely unlikely that a large-scale, acute petroleum spills from the marina operation that would be severe enough to produce adverse effects to sea turtles. Any differences in the operation from what is presented by the applicant that are likely to result in greater likelihood of fuel spills or other such sources of contamination would constitute new information and grounds for reinitiation of consultation.

Recreational Activities: Studies such as that of (Scales et al. 2011) found that protected areas within their study area had larger concentrations of turtles. Studies of nesting habitat have found similar patterns that show preference for habitats further from developed areas (Weishampel et al. 2003a). Thus, there may be more sea turtles in the project area at this time than further south in the area of Frederiksted where human activity, including boating, is concentrated along the west coast of St. Croix. At the underwater snorkel trail in Buck Island Reef National Monument (BUINM), green and hawksbill sea turtles appear to have altered their use of refuge and foraging habitat so as to avoid the peak times of day when many visitors are at the trail (Z. Hillis-Starr, NPS, pers. comm. to L. Carrubba, NMFS, July 2, 2014). This shift in habitat use around the underwater trail likely has little impact to turtles in BUINM given the extent of refuge and foraging habitat areas and the small size of the area occupied by the snorkeling trail. In the area of the Amalago Bay project, an increase in the concentration of recreational uses in the area could lead to a similar habitat use shift by green and hawksbill juvenile and adult sea turtles. As with BUINM, we expect that if any impacts occurred, they would be so minimal as to be insignificant given the extent of seagrass beds, colonized hard bottom, and coral reefs in the action area that would still provide refuge and foraging habitat.

5.1.3 In-Water Dredging and Construction of Roadways, Utilities, Residential and Hotel Units, and Other Amenities

Runoff of Land-Based Sources of Pollutants: In addition to the permanent habitat loss from in-water construction and dredging of 2.75 ac of colonized hard bottom and 0.8 ac of sparse seagrass that likely serve as green and hawksbill sea turtle refuge and foraging habitat, NMFS expects that the rest of the 6.35 ac of mixed shoal grass and sand and 30.31 ac of colonized hard bottom in Area A (Figure 5) will be affected by dredging and stormwater runoff and associated contaminant transport to nearshore waters during the construction and operation of the project. As noted in Section 3.0, total hard bottom, reef, or colonized pavement in the action area includes approximately 77 ac in Area B (Figure 5) out of the total 200 ac extending from the property boundaries out to the shelf edge. Those 77 ac include approximately 12 ac of reef/colonized bedrock along the shore and 65 ac of reef/colonized pavement in waters with depths of 10 ft or more, based on NOS benthic habitat maps.

Dredging activities associated with construction and maintenance of the project will affect the resident and transient sea turtles utilizing the project area. The applicant has proposed water quality monitoring (see Appendix B of the BA) and has established threshold turbidity (measured in nephelometric turbidity units or NTUs) and total suspended solids (TSS measured in mg/L) values based on water quality monitoring data from DEP from the action area. These values are 6 NTUs and 50 mg/L. Based on a review of our project files for previous dredging projects in USVI, the use of a floating turbidity boom should be adequate to reduce the levels of turbidity and TSS from sediment resuspension and transport outside the dredging and construction footprint except during rain events. Because the material to be dredged consists mainly of sand and some calcareous rock, resuspension and transport should be minimal and these larger materials will be effectively trapped by the turbidity barrier. However, rain events will lead to stormwater runoff from land, contributing finer materials such as silts and clays that are not as effectively trapped by these barriers. As a result, these finer sediments will be transported in stormwater runoff and will be resuspended during the initial and subsequent maintenance dredging activities, thus resulting in greater turbidity. The applicant has incorporated a condition to temporarily curtail dredging if sea conditions do not permit the effective use of the turbidity barrier, but rain events are not necessarily accompanied by heavy waves that would affect the stability of the turbidity barriers and lead to a suspension of dredging activities. Therefore, we believe there will be chronic impacts of sediment resuspension and transport during both initial construction and maintenance dredging activities associated with the introduction of terrestrial sediments to nearshore waters. This will lessen the utility of these areas for sea turtles, making turtles find new areas for foraging and refuge, which can lead to increased predation and decreased health.

Chronic impacts to nearshore habitats are produced by frequently recurring storm events of low intensity (larger than 1 cm of rainfall corresponding to 2- to 10-year storm events).[21] These

[21] An "*x*-year storm event" refers to the probability of a specific amount of rain falling in a given place in a given year; it does not refer to storm frequency. For example, the amount of rain associated with a 10-year storm event

events have been shown to account for approximately half of the total annual precipitation in St. John, for example, but produce about 90% of the total runoff and sediment yield (Ramos-Sharron and MacDonald 2007b), since St Croix does not have this data St. John is being used as a surrogate. Information provided by the USACE indicates that storm surge and stormwater runoff during storms could impact areas up to 1200 ft from the shoreline, which would potentially affect much of the 77 ac of colonized bedrock and pavement in the immediate project area and potentially other areas of seagrass, reef, and colonized hard bottom in the action area. We believe that impacts to the colonized bedrock and pavement, lying between 500 ft from shore and the shelf-edge reef that likely serves as sea turtle refuge and foraging habitat from large storms will be infrequent but may still lead to a long-term degradation of the habitat.

Land-based sources of sediment and other contaminants, as well as nutrients in coral reef and hard bottom areas in USVI, were found to affect coral areas up to 984 ft from shore depending on the location of each survey station in relation to the shoreline. During large storms, this increased up to 1 km from shore (Nemeth and Nowlis 2001); (Smith et al. 2008); T. Smith, UVI, pers. comm. to L. Carrubba, NMFS, December 12, 2014). The alteration of natural drainage patterns, intensity of development on areas with rocky soils and steep slopes, and associated degradation of nearshore water quality due to the transport of sediments and other contaminants in runoff will lead to impacts to sea turtles, in particular resident and transient juveniles and adults that use nearshore areas as forage and refuge habitat especially during nesting seasons. The transport of contaminants to the marine environment has been shown to affect sea turtles in the water, in particular species such as greens and hawksbills that frequent nearshore areas (Aguirre et al. 1994, Caurant et al. 1999, Mckenzie et al. 1999, Corsolini et al. 2000) (Sakai et al. 1995) (Storelli et al. 1998) through the contamination of food items and direct ingestion of substances that are toxic to turtles.

Information provided in the applicant's May 2016 submission included an assessment of the ability of the erosion and sediment control plan for the project to reduce sediment loading by 80% from an unspecified baseline. The information also included a sediment basin the applicant indicated had been added to the plan to further control sediment in the northeastern corner of the property from the development of single family homes. NMFS requested a limited review of the erosion and sediment control information developed by William and Punch LLC, including the May 2016 submission, by the HWG. Based on the HWG review and NMFS's previous analyses, we believe that chronic impacts to the 30.31-ac area immediately adjacent to the Amalago Bay project (i.e., hard-bottoms and seagrass areas that were surveyed, out to 500 ft from shore) are reasonably certain to occur over the construction and operational lifetime of the project. Some of the BMPs for sediment and erosion control presented by the applicant include the use of silt fences, hydroseeding, and liquid polymer emulsions to capture eroded soils and stabilize slopes, road cuts, and construction areas. Studies in USVI have shown that, despite similar measures, cleared areas on steep slopes and roads that are under construction or have not been completely stabilized prior to the rainy season are major contributors of land-based sources of pollution in

means the rainfall associated with that storm had a 1-in-10 chance (10%) of falling in that location during a given year. A 2-year storm event refers to amount of rainfall with a 1-in-2 chance (50%) of falling at a given location in a given year. Since a 2- or 10-year storm refers to a probability and not frequency, the likelihood of a storm of that magnitude occurring again in the same location remains the same the in preceding days. It does not mean a storm of that magnitude will not occur again for 2 or 10 years.

runoff during periods of rainfall (Nemeth and Nowlis 2001); (Smith et al. 2008); (Ramos-Sharron and MacDonald 2007b, a). Rainfall intensity of 2 cm per hour (0.79 in per hour) is all that is required to generate runoff and lead to an increase in water turbidity based on a study on the East End of St. Croix (Reale-Munroe et al. 2014). Based on tidal currents and wave patterns, transport of land-based pollutants will be from nearshore to offshore with some net transport of sediment to the southeast due to wave refraction (M. Canals, UPRM, pers. comm. to L. Carrubba, NMFS, December 21, 2014). Chronic impacts associated with the transport of land-based sources of pollution to the 30.31-ac area (Area A from Figure 5) are expected to include increases in sedimentation of seagrass and coral habitats utilized as refuge and foraging habitats by sea turtles, increases in nutrients that could lead to increases in algal growth, decreases in seagrass and sponges that are food items for green and hawksbill sea turtles, respectively, and increases in other contaminants such as pesticides that could affect sea turtles through ingestion of contaminated prey items. Based on this, we believe the degraded function of this area will lead to a permanent loss of this habitat to resident turtles over time.

We also believe that episodic impacts to the additional 77 acres (Area B from Figure 5) of reef and colonized hard bottom within the area from the shoreline to the shelf edge will result in declines in the quality of refuge and foraging habitat for green and hawksbill sea turtles over time. However, due to the distance from shore and the episodic nature of the impacts, we believe the areas will still provide refuge and foraging habitat for sea turtles in the 77 acre area, and effects will be insignificant. Both resident and transient juvenile and adult sea turtles currently using the colonized hard bottom and patch reef habitats in Area A of the project area will be affected by the loss and degradation of foraging and refuge habitat as a result of land-based sources of pollutants from the construction and operation of the terrestrial portion of the Amalago Bay project. Effects include short-term disorientation and decreased health and long-term displacement to other nearshore foraging and resting areas outside of the action area for those juvenile and adult turtles that are part of the resident population of green and hawksbill sea turtles. These individuals may also be more susceptible to predation (compared to juvenile green and hawksbill sea turtles that have adequate foraging and resting habitat and in consequence often also have smaller home ranges) due to the need to find alternate habitat and possibly expand their home range if turtles move to areas with habitat of lesser quality. The action area is part of an extensive nearshore shallow hard bottom habitat system that extends along the west coast of St. Croix. We believe that transient green and hawksbill juveniles and adults will also be affected by the loss and degradation of refuge and foraging habitat because these turtles will not be able to utilize the action area as a stopping point, as the project will contribute to fragmentation of the Fredriksted Reef System. For resident juvenile and adult sea turtles, the inability to utilize the action area due to declines in habitat quality and quantity could make these animals more susceptible to predation and decrease their fitness by having to spend more time swimming between areas of suitable refuge and foraging habitat.

Tag return data for immature hawksbill sea turtles indicate that juveniles tend to remain in the same developmental habitats for long periods (Meylan 1999b). Data from Magens Bay, St. Thomas, showed that, over an approximately 15-year period, 30% of the immature hawksbill turtles that were tagged in the bay and recaptured stayed in the area while approximately 12% were found in countries 46-720 km away (Meylan 1999b). Data from Buck Island showed that, over an approximately 5-year period, 32% of the immature hawksbills that were tagged at Buck

Island were re-sighted locally while none were re-sighted internationally (Meylan 1999b). In addition to foraging efficiency achieved by concentrating efforts in the same area, turtles may better escape predators and shelter from environmental extremes due to familiarity with the physical environment within their home ranges (Bailey 1984, Alcock 2001). The loss or degradation of habitat could also lead to a decrease in fitness. According to (Diez and Dam 2002) the major apparent factor affecting turtle growth is that of location, which suggests that substantial differences in habitat quality exist between Monito Island and the rest of the study area (which includes Mona Island). (Diez and Dam 2002) found that the Monito Island aggregation of immature hawksbills had a higher body condition index and reached maturity less than 14.7 years after recruiting to the aggregation in comparison to the Mona Island aggregation (for which no estimate was provided, but was expected to take much longer to reach maturity) due to the 2.1-times-faster growth rate observed for turtles around Monito.

Both hawksbill and green turtles exhibit strong habitat preferences and site fidelity for foraging and refuge habitat. Data from in-water sea turtle surveys at Buck Island, St. Croix, indicate that the foraging grounds for juvenile and adult hawksbill sea turtles are spatially distinct (USNPS 2003, 2004) based on sizes of turtles captured that were all less than those of nesting hawksbills. (Makowski et al. 2006) found considerable overlap between refuge and foraging sites for green sea turtles with the entire home range of each turtle concentrated over the algal-rich nearshore worm reef where immature green sea turtles were shown to eat macroalgae and sponges as the dominant components of their diet. Green sea turtles were also found to have 1-2 distinct nocturnal resting sites within their home ranges that were not shared with another turtle, although foraging habitats of turtles did overlap (Makowski et al. 2006). Overlap in home ranges suggests the areas provide sufficient resources to be shared by neighboring green sea turtles. (Wershoven and Wershoven 1992b) found that green sea turtles using the reef tract in Broward County over a 5-year study period included resident juvenile green sea turtles (based on re-encounter and recaptures) and other juvenile turtles that likely utilized the area for foraging and resting during some part of the year but did not appear to remain in the area year-round. Similarly, (Witt et al. 2010) found that habitat structure influenced site fidelity for juvenile hawksbills in the British Virgin Islands and (Cuevas et al. 2007) found that juvenile hawksbills in Yucatan, Mexico, showed a difference in habitat preference during the day and night. Because of the strong tendency toward site fidelity for foraging and refuge habitat, the loss or degradation of habitat within a turtle's home range can potentially have significant negative consequences. For both green and hawksbills, maintaining fidelity to home ranges may result in resource deficiencies due to the reduced habitat value, potentially decreasing growth rates, increasing age to maturity, and possibly even reducing annual survivorship rates. Similar consequences may occur if individuals are compelled to leave their established home ranges in search of more suitable and available habitat.

As noted previously, in order to determine the number of resident juvenile green and hawksbill sea turtles that will be affected by the permanent loss of habitat, we used the numbers from (Wershoven and Wershoven 1992b) green sea turtle study and (Diez and Dam 2002) hawksbill sea turtle study as proxies. The (Wershoven and Wershoven 1992b) study area encompassed 7.9 ac and assumed that re-encounter indicated residency, meaning the 37 juvenile re-encountered green sea turtles were believed to be residents (which equals approximately 5 resident juvenile green sea turtles per acre). The (Diez and Dam 2002) study calculated an aggregation index by dividing the total number of hawksbill recaptures in a particular habitat type (coral reef versus cliff wall) by the surface area of the habitat. If we convert the estimates of (Diez and Dam 2002) (in number of turtles per km^2) to the number of turtles per acre, we get an average of 0.11 juvenile hawksbill sea turtles per acre at Mona Island and 0.5 juvenile hawksbill sea turtles per acre at Monito Islands.

As with the analysis for permanent habitat loss above, we estimated the impact of potential habitat displacement due to habitat degradation. We again used the density numbers from the (Wershoven and Wershoven 1992b) green sea turtle study and the (Diez and Dam 2002) hawksbill sea turtle study results. Habitat degradation impacts were calculated for chronic impacts from stormwater discharges and flushing events leading to the transport of land-based pollutants to nearshore waters. Chronic impacts from stormwater discharges and flushing of the inland marina basin into nearshore waters due to storm events larger than 1 cm of rainfall (corresponding to 2- to 10-year storm events) are expected to impact 30.31 ac of colonized hard bottom. We believe these impacts will lead to a permanent loss of this habitat to resident juvenile and adult turtles. The loss of this habitat will impact, via displacement up to 151 green sea turtles, total (i.e., 5 turtles per acre multiplied by 30.31 ac =150.5 turtles) and 15 hawksbill sea turtles, total (i.e., 0.5 turtles per acre multiplied by 30.31 ac = 15.2).

We do not have any studies related to adult resident population densities that we can use as proxies as we did for juvenile green and hawksbill sea turtles. However, it is known that reproductively mature hawksbill sea turtles (Proietti et al. 2012, Rincon-Diaz et al. 2011) and green turtles (Bresette et al. 2010) tend to establish foraging home ranges in waters further offshore and deeper than do juveniles, so the impact of habitat disruption or degradation from this project is likely to have a lesser impact on mature turtles than juveniles. Based on the habitat preferences by age class, the much lower density that would be expected for adult sea turtles compared to juveniles, and the reduced importance of the potentially impacted nearshore waters for adult foraging habitat, and the lower magnitude of impacts to habitats further offshore, we determine that the impact of habitat degradation would be insignificant for adult sea turtles.

Lighting: Artificial lighting can also be an issue for sea turtles. Disorientation of adult and hatchling sea turtles on the beach is common in developed areas due to lights from cars and other vehicles, lighting of navigational aids, and lighting of pathways, roadways, and buildings. Even with lighting plans, the glow from developed areas can lead to disorientation. The applicant proposes the design and implementation of a lighting plan in coordination with the USFWS, as well as post-construction lighting inspections to determine whether the lighting plan is effective in preventing adult and hatchling sea turtle disorientation on the beach. The plan has no specific mention of preventing disorientation of sea turtles in the water due to lights on land and on the marina channel jetties. A recent project in St. John found that many of the lights going up the hillside, as well as lights near the shore, were not evident from the nesting beach but were very evident from the water (Springline Architects 2013), which could affect adult and hatchling sea turtles. The project lighting plan was redesigned to ensure that all lights were designed to minimize luminar contamination of both the nesting beach and nearshore waters in the project area. While lighting disorientation is by far a bigger issue for turtles on the beach, nesting females at times may choose not to nest on a particular beach because of light levels visible from the water. Because the Amalago Bay project does not contemplate ensuring the lighting plan protects sea turtles in the water from disorientation, we believe that there is the possibility of adverse impacts to green (South Atlantic DPS), hawksbill, and leatherback sea turtles where nesting females may abandon attempts to nest as they approach the beach and see excessive lighting. Given the lighting plan to be coordinated with USFWS, this will likely be a rare circumstance but we cannot conclude that it is extremely unlikely to occur. Therefore, we conservatively estimate that up to 1 nesting event per year per species will not be attempted because the females will not leave the water as a result of lighting from the proposed project. This will result in the female nesting at another location or abandoning nesting attempts. Any on-the-beach lighting impacts fall under USFWS jurisdiction and are considered in their Opinion for this project.

5.2 Effects of the Action on Elkhorn, Staghorn, Lobed Star and Mountainous Star Corals and ESA-Designated Acroporid Coral Critical Habitat

The benthic surveys for the project were confined to the shallow habitats within 500 ft of the shoreline. No elkhorn or staghorn corals were reported in the 46-ac survey area. However, Toller (2005) reported occasional colonies of elkhorn coral in waters from 0-3 ft in depth and occasional staghorn coral colonies in waters from 18-35 ft in depth within the Frederiksted Reef System (from King's Corner south of the Frederiksted Pier to Sprat Hole to the north). Also, elkhorn coral, including recruits, have been found adjacent to the northern boundary of the project. Elkhorn coral skeletons have also been found in the project area, indicating the area can support this species. Other studies conducted in the action area, including surveys of the shelf edge reef directly offshore of the Amalago Bay project, included information regarding the presence of staghorn corals (Toller 2005, Smith et al. 2014). None of the studies conducted in the Frederiksted Reef System included estimates of the numbers of staghorn coral colonies present. Studies did indicate that corals in the *Orbicella annularis* complex, including lobed star and mountainous star corals, are dominant in deep and shallow reef areas. Lobed star and mountainous star corals were found within the 46-ac benthic survey area for the Amalago Bay project, including within the in-water construction footprint, but no quantification of the number of colonies was provided.

Construction activities can have significant direct and indirect adverse effects on listed coral species as well as designated Acropora critical habitat. These effects can range from breakage of colonies, to destruction of reef habitat and associated corals, to sedimentation impacts due to construction and runoff from developed slopes. The effects of most concern from this project will result from sedimentation and contaminant impacts during the construction of the beach, marina, and jetties, during future maintenance dredging, and from storm water runoff from land and through flushing from the marina, in rainfall events and during large storms. Adverse sedimentation and contaminant effects from runoff will be chronic in nature in area A (Figure 5) of the action area and are expected to be episodic in area B (Figure 5) of the action area. In addition to the sediment and contaminant impacts, corals and critical habitat may be adversely affected due to accidental groundings of construction vessels, spudding of construction vessels, recreational boating and diving impacts to corals and hard bottom, and boating-related water quality impacts in the marina and nearshore.

ESA-listed coral species and critical habitat may be affected in area C (Figure 5) of the action area via construction vessel transit through accidental groundings. This project will also result in effects from the activities of third parties using the completed development, including through recreational boating and diving impacts to corals and hard bottom. These impacts within Areas A and B will occur to the same corals and critical habitat that will be affected by the overarching sedimentation impacts. We will note those additive impacts especially to corals outside of areas A and B in our analysis below.

5.2.1 Proposed Mitigation Actions

The project proposal includes compensatory mitigation elements for coral and hardbottom impacts. We believe the proposed measures are not reasonably certain to be effective at preventing losses of ESA-listed corals or critical habitat. The applicant proposes the relocation of all corals with a diameter of 7 cm (or a size of 153.94 cm^2) or more outside the construction area so any colonies of lobed star or mountainous star coral having at least this diameter will be moved outside the construction footprint prior to the start of dredging and construction work. We believe this mitigation measure will be impractical because there are no coral nurseries established in the area to use for staging and stabilization of corals, and thus colonies would have to be immediately relocated to other reef areas. In addition, the proposal to transplant corals to artificial mitigation reefs (discussed below) would not be effective, as these reefs will be within the 30.31 acre area NMFS believes will be chronically impacted by sedimentation, which is expected to result in killing or severely debilitating the transplanted corals.

The applicant proposes the construction of large pyramidal structures as artificial reef offshore of the marina to serve as mitigation for habitat loss and a transplant site for corals that will be impacted by the proposed in-water construction. The applicant also proposes 3-years of funding for an existing coral farming operation in St. Croix to fund the outplant of 1,000 elkhorn and staghorn coral colonies per year (for a total of 3,000 outplanted colonies at the end of the 3-year period). We believe the proposed mitigation will not compensate for the loss of habitat for ESA-listed corals because the artificial reefs would provide only 1.45 ac of habitat. The design of the artificial reefs may affect transplant success due to the physical differences in the structure of the habitat currently in the area versus the proposed structures, which do not mimic the natural hard bottom features that will be removed as a result of the Amalago Bay project. The BA also reveals that the amount and timing of the proposed mitigation are insufficient to compensate for lost coral functions in the 2.75 ac of lost hard bottom within the lifetime of the project; this deficit would be far greater given the impacts from sedimentation that will occur in the area proposed for mitigation, discussed below. The applicant performed a Habitat Equivalency Analysis (HEA) for the project and determined that, based on the average size of lobed star coral in the project area, the published growth rate for the species, and the calculated recovery time, 4.1 ac of migitation is required to compensate for impacts to coral habitat (Dial Cordy and Associates 2010). Thus, the artificial reefs are not adequate to reach this acreage and the proposed outplants rely on a supply of coral fragments that may not exist following the 2017 hurricanes, which are known to have damaged all in-water coral farming operations in the USVI and Puerto Rico. In addition, the HEA was performed using lobed star coral and finger coral (*Porites porites*), the latter of which is not listed. It is not clear from the HEA that the proposed outplanting of elkhorn and staghorn corals would benefit other ESA-listed coral species or whether the acreage of mitigation needed to ensure recovery of different ESA-listed coral species in the action area would differ.

NMFS believes that these structures, due to their size and placement, will serve as offshore breakwaters as currently proposed, resulting in additional impacts to acroporid coral critical habitat when wave action against the seaward side of structures leads to scour and when the leeward side accumulates sand. Diminished wave action behind the structures will interrupt natural longshore sediment transport and allow sediment to accumulate between the breakwater and land (Thomas-Blate 2010). The structures can also affect beach erosion and accretion processes in the area (Johnson et al. 2008, Thomas-Blate 2010). An example of this effect can be found in front of the Embassy Suites Hotel in Dorado, Puerto Rico, where the construction of an offshore breakwater structure, similar to that proposed in the Amalago Bay mitigation plan, has led to the creation of a tombolo (sand bar extending from shoreward of the breakwater to the shore) between the breakwater and shore, burying a large area of seagrass. The structure also suffers from scour along its seaward portions, which has affected the colonized hard bottom immediately adjacent to the structure through sand abrasion and deposition (L. Carrubba, NMFS, pers. obs., June 10, 2009).

Therefore, in the effects analyses that follow, we do not discount any of the effects predictions due to potential benefits from proposed mitigation.

5.2.2 Permanent Destruction of Corals and Critical Habitat from Construction of the Marina, Beach, and Jetties

The construction of the marina channels and jetties and the creation of the beach on the island that will be formed from excavation of the navigation and flushing channels will require the dredging of colonized hard bottom in order to create the navigation and flushing channels, and the placement of fill to construct the jetties and beach. Of the area to be lost due to in-water construction and dredging, 2.75 ac is colonized hard bottom and 2.69 ac of this area contains the essential feature of elkhorn and staghorn coral critical habitat. A portion of the colonized hard bottom will be removed by dredging and a portion will be buried under the jetties and newly created beach. This area is in the 46 ac area surveyed by the applicant and is contained in area A of the action area (Figure 5).

5.2.2.1 Lobed Star and Mountainous Star Corals

The benthic studies conducted for this project note that there are known colonies of both species in the footprint of the in-water construction. However, the locations and numbers of these colonies are not provided.

We therefore must calculate estimates for the number of colonies that would be directly destroyed by the project’s in-water construction and destruction of 2.75 acres of colonized hard-bottom. We have estimated the approximate number of each of these species to be affected based on the study by Toller (2005) that found the transects in the area of Sprat Hole, near to the project area, had 24.5% coral cover of which 78% was *Orbicella annularis*. Surveys done by EPA around St. Croix as part of their bioassessment program found lobed star coral to be the coral with the most coverage due to its surface area (approximately 15%) and mountainous star coral to be half the cover of lobed star coral (or 7.5% (Toller 2005, Fore et al. 2006); A. Dempsey, BioImpact, pers. comm. to L. Carrubba, NMFS, April 23, 2014]). Deriving percentages from the (Fore et al. 2006) survey results and using percent cover estimates for each of the species from benthic surveys for projects in USVI and Toller (2005), we expect 24.5% of the hard bottom in Area A to contain corals of which 15% would be lobed star coral and 7.5% would be mountainous star coral, with the remainder comprised by other coral species such as brain corals that were identified as part of the benthic surveys conducted for the Amalago Bay project. Thus, of the 2.75 acres of colonized hard bottom to be lost due to the construction of the in-water portions of the project, approximately 0.67 acre (24.5% of 2.75 acres) is expected to have coral cover, of which 0.1 acre (15% of 0.67 acre) would be covered by lobed star coral and 0.05 acre (7.5% of 0.67 acre) would be covered by mountainous star coral.

Converting acres to square centimeters (1 ac = 40,468,564.224 cm^2), this would be 4,046,856.4 cm^2 of lobed star coral cover (0.1 ac * 40,468,564.224 cm^2) and 2,023,428.2 cm^2 mountainous of star coral cover (0.05 ac * 40,468,564.224 cm^2). (Edmunds and Elahi 2007) reported 4 size categories for lobed star coral from their study area in St. John. Using these 4 categories, their percentages, and their size ranges we were able to determine an approximate average size per lobed star colony of 38.75 cm^2. We believe this estimate of average colony size is also a good estimate for the average colony size for mountainous star coral as they are morphologically similar. Based on this information we estimate that there will be 104,435 colonies of lobed star coral and 52,218 colonies of mountainous star coral lost by the project's nearshore habitat conversion (4,046,856.4 cm^2 of lobed star coral cover divided by 38.75 cm^2 per colony, and 2,023,428.2 cm^2 of mountainous star coral cover divided by 38.75 cm^2 per colony).

5.2.2.2 Elkhorn Coral

There are reports of elkhorn coral skeletons in the project area indicating that the project area supported elkhorn corals prior to mortality events, largely from bleaching, over the last 20 years, though no skeletons were reported in the benthic surveys conducted as part of the Amalago Bay project (A. Dempsey, BioImpact, pers. comm. to L. Carrubba, NMFS, April 29, 2008). We believe that there are no effects of in-water construction activites to existing elkhorn resources in Area A, because no live elkhorn coral colonies are reported in area A of the action area. However, since elkhorn coral skeletons were observed within the project area, and since elkhorn coral populations are present in waters along the west of side of St. Croix as discussed in the Sections 4.3.1 – 4.3.3, it is reasonable to expect that this area could support healthy elkhorn colonies in the absence of additional stressors. Therefore, the loss of 2.69 acres of critical habitat due to in-water construction could reasonably preclude elkhorn coral gametes or fragments from settling and recruiting in this area.

The Acropora Recovery Plan identifies a recovery criterion for abundance of elkhorn coral: "Thickets are present throughout approximately 10 percent of consolidated reef habitat in 1 to 5 m water depth within the forereef zone. Thickets are defined as either a) colonies ≥ 1 m diameter in size at a density of 0.25 colonies per m^2 or b) live elkhorn coral benthic cover of approximately 60 percent. Populations with these characteristics should be present throughout the range and maintained for 20 years." Based on a recovered population of elkhorn coral (per the recovery plan) this area would be expected to support 272 elkhorn coral colonies (1ac = 4,046.8564224 m^2; 2.69 ac = 10886.0437763 m^2; 10% of 10,886.0437763 m^2= 1,088.60437763 m^2; x 0.25 colonies per m^2 = 272 colonies). This loss will reduce future reproduction potential by eliminating settlement habitat for larvae and recruits and any possible reproduction of a possible 272 colonies.

5.2.2.3 Staghorn Coral

Smith et al. (2011a) reported staghorn coral colonies in the shelf edge reef in the action area (Area B). We believe that there are no effects of in-water construction activities to existing staghorn resources in the Area A because no live staghorn coral colonies are reported in the immediate construction area (Area A). The loss of 2.69 acres of habitat due to in-water construction will not npreclude staghorn coral gametes or fragments from settling and recruiting in this area. This species has a much greater depth distribution in St. Croix, is not present in the 0-5 m depth zones, and has not been observed to successfully recruit in this area.

5.2.2.4 Acropora Critical Habitat

The construction of the marina, beach and jetties will dredge and cover 2.75 ac of hard bottom in area A (Figure 5) of which 2.69 ac is considered critical habitat. Therefore, 2.69 ac of Acropora critical habitat will be permanently lost from this aspect of the project. As discussed in Sections 3.2.3 and 4.3, at this time, the portion of the Frederiksted Reef System in the action area is relatively unaffected by sediment, particularly terrigenous sediments, that could cover the essential feature and is, therefore, an important area for expansion and recovery of elkhorn and staghorn corals. The majority of near shore reefs in St. Croix are affected by sedimentation with sediments that are terrigenous in nature, due to coastal development, fragmenting the essential feature throughout the unit. Loss of this area to foster elkhorn reproduction further fragments the St. Croix Unit.

Sedimentation and Contaminants Impacts

Sediment and Contaminant Impacts from Project Construction and Future Maintenance Dredging

During the construction of the project, dredging will be on-going for a number of months. The information provided by the USACE indicates that the design of the jetties will cause waves to penetrate from the westerly quadrant, which is the predominant direction of wave action. Thus, maintenance dredging will be required following large storm events, due to transport of marine sediments into the mouths of the channels as well as terrestrial sediment into the inland marina that serves as the main catchment for all stormwater runoff for the project. The applicant does not define large storms but we assume this means hurricanes and tropical storms. Based on hurricane and tropical storm track data from NOAA, St. Croix may experience these events approximately every 2 years. Dredging will also be needed to maintain water depths in the navigation channel at least every 10 years and in the flushing channel at least every 5 years (Moffatt & Nichols 2013). The applicant notes that at least 1 month of work will be required per maintenance dredging activity depending on the amount of material to be removed.

NMFS believes that the levels of turbidity and suspended sediments associated with the project's dredging will be injurious to corals. While the applicant has proposed to conduct water quality monitoring during dredging (see Appendix B of the BA), the applicant has set threshold levels of 6 NTUs and 50 mg/L TSS, based on water quality monitoring data from DEP from the action area. These threshold levels are above the tolerance threshold for corals and prolonged exposure to these levels would lead to physiological injury to corals. Background turbidity levels in healthy coral reef environments are 0-2 NTUs (Telesnicki and Goldberg 1995) and recent changes to USVI Water Quality Standards reflect this as allowed turbidity levels in coral areas are proposed as 1 NTU maximum (DPNR, draft document, 2014). Reefs not subject to stresses from human activities in the USVI had mean sediment rates of less than 1 mg per cm^2 per day and TSS concentrations less than 10 mg/L (Rogers 1990). (Smith et al. 2013) found that nearshore sites around USVI affected by human activities had TSS concentrations as high as 25 mg/L. Thus, the threshold levels that the applicant will monitor against are at least 5 times higher than values associated with healthy coral reefs and are double the levels associated with impacted reefs. Therefore, we believe that the dredging activities associated with the Amalago Bay project will result in physiological injury to ESA-listed corals through sedimentation associated with construction activities.

The use of a floating turbidity boom should reduce the levels of turbidity and TSS outside the dredging and construction footprint if the sediment particles are largely sand and other coarse-grained sediment. The floating turbidity boom will not reduce all sediment transport outside the in-water construction area and will be less effective if in-water construction continues during rain events. Rain events (discussed in detail below) will lead to stormwater runoff from land, contributing finer materials such as silts and clays that are not as effectively trapped by in-water turbidity barriers. The applicant has incorporated a condition to temporarily curtail dredging if sea conditions do not permit the effective use of the turbidity barrier, but rain events are not necessarily accompanied by heavy waves that would affect the stability of the turbidity barriers and lead to a suspension of dredging activities. As noted in Section 4.3., the area where the Amalago Bay project is proposed is currently characterized by good water clarity and the presence of staghorn, lobed and mountainous star corals in the 200-acre area extending from the shoreline properties boundaries out to the shelf edge. Thus, we believe project construction and future maintenance dredging activities will lead to the degradation of water quality and adversely affect ESA-listed corals and acroporid coral critical habitat in Area A. Because the levels proposed as allowable for turbidity and TSS are higher than those known to affect corals and because the dominant transport of material is mainly onshore to offshore (M. Canals, UPRM, pers. comm. to L. Carrubba, NMFS, December 21, 2014), we anticipate that resuspended sediments will be transported to areas containing staghorn coral colonies and additional lobed and mountainous star colonies (e.g., Area B). While we anticipate that the sediment concentration in these outer areas will be less than the 50 mg/L proposed for inside the construction footprint, we believe that it will still be higher than the 10 mg/L reported to be within the tolerance level for healthy corals (Rogers 1990) given that impacts from sediments have been observed up to 1 km from shore during large storms (T. Smith, UVI, pers. comm. to L. Carrubba, NMFS, December 12, 2014) and up to 300 m from shore during monitoring surveys in USVI (Smith et al. 2008). All of these dredging-related sedimentation impacts will occur to the same corals and coral habitat and be additive to the chronic runoff and storm distribution impacts discussed below.

Sediment and Contaminant Impacts from Runoff and Storm Distribution
(Ramos-Sharron and MacDonald 2007b, a) found that chronic sediment impacts were produced by frequently recurring storm events of low intensity, characterized as events with rainfall greater than 1 cm (0.395 in per hour). These types of rainfall events account for almost half the total precipitation for a year but produce about 90% of the runoff and sediment yield at these

authors' St. John study sites (Ramos-Sharron and MacDonald 2007b). Studies from the east end of St. Croix also indicate a realtively common rainfall intensity for the area of 2 cm per hour (0.79 in per hour) is all that is required to generate runoff and lead to increased in-water turbidity and sedimentation (Reale-Munroe et al. 2014).

In the USVI, the rates of runoff into adjacent waters have a major impact on the health of corals. Shoreline segments with less than 20 cubic feet per second of runoff intensity were found to be more likely to contain well-developed nearshore reefs than shoreline segments with more intense runoff rates (Hubbard et al. 1987). The difference in reef development was related to the funneling of runoff to nearshore areas, as well as declines in coral growth related to the transport of land-based sources of sediments and contaminants in runoff (Hubbard et al. 1987). Flooding and overland flow resulting in nearshore contamination by land-based sources of pollution are likely during periods of rain based on other studies in USVI ((Nemeth and Nowlis 2001); (Ramos-Sharron and MacDonald 2007b, a); (Reale-Munroe et al. 2014).

Inspections at the Amalago Bay site indicate that this process of runoff of land-based sources of sediment and contaminants, particularly nutrients, is already acting at this location, although in only a very limited area near the current drainage points into the sea. The first project site inspection was in April 2008. Another site inspection was conducted in December 2011. The 2 inspections revealed significant differences in sediment loading from the 3 main drainage points to the sea, due only to heavy precipitation in 2010 and 2011. In 2008, sediment cover and macroalgal growth on hardbottom near the mouths of these drainages was very limited and water clarity was very high. In 2011, on the other hand, concentrations of fine sediments and dense growth of macroalgae were observed at the outfall points of these discharges. Given that significant differences in sediment loading from uplands to nearshore waters were visible in the absence of development and with the natural vegetative cover present on most of the site, NMFS believes that the impacts of stormwater transport of sediments and other contaminants to nearshore waters will be significant with the major changes in land-use that the project will introduce.

The project's changes in land-use will cause a dramatic increase in the frequency of runoff events into the Caribbean. We calculated the changes in expected stormwater runoff from normal rainfall events using an EPA stormwater calculator (U.S. EPA 2014; available at: EPA stormwater calculator[22]) and the soil survey information for the site. A large change in the amount of runoff is predicted due to the change from a largely forested area (91%) to an area with approximately 35% impervious cover. The volume of runoff is predicted to increase approximately 570% annually.[23] More dramatically, and of greater impact to corals, the number of runoff events per year will increase by more than 17 times: to 34 days per year of runoff generated by normal rainfall versus the current estimate of less than 2 days per year of runoff.

Several other sources confirm the vulnerability of reefs in the Caribbean region to runoff impacts from coastal development. For example, NOS's Summit to Sea project characterizing coastal watersheds in Puerto Rico and USVI showed that nearshore benthic habitats are at high risk for

[22] http://www2.epa.gov/water-research/national-stormwater-calculator
[23] Pre-construction runoff estimated at 2.08 inches per year, post-constuction runoff estimated at 13.96 inches per year.

sediment impacts due to the high potential for sediment and other land-based contaminants being delivered to nearshore areas because of steep slopes, peak rainfall amounts, and the erodibility of the soils on the site (Figure 12). Any construction work done in the rainy season (peak from December-March) or during the hurricane season (July-November) is reasonably certain to result in significant transport of sediment to nearshore waters during rain-generating events.

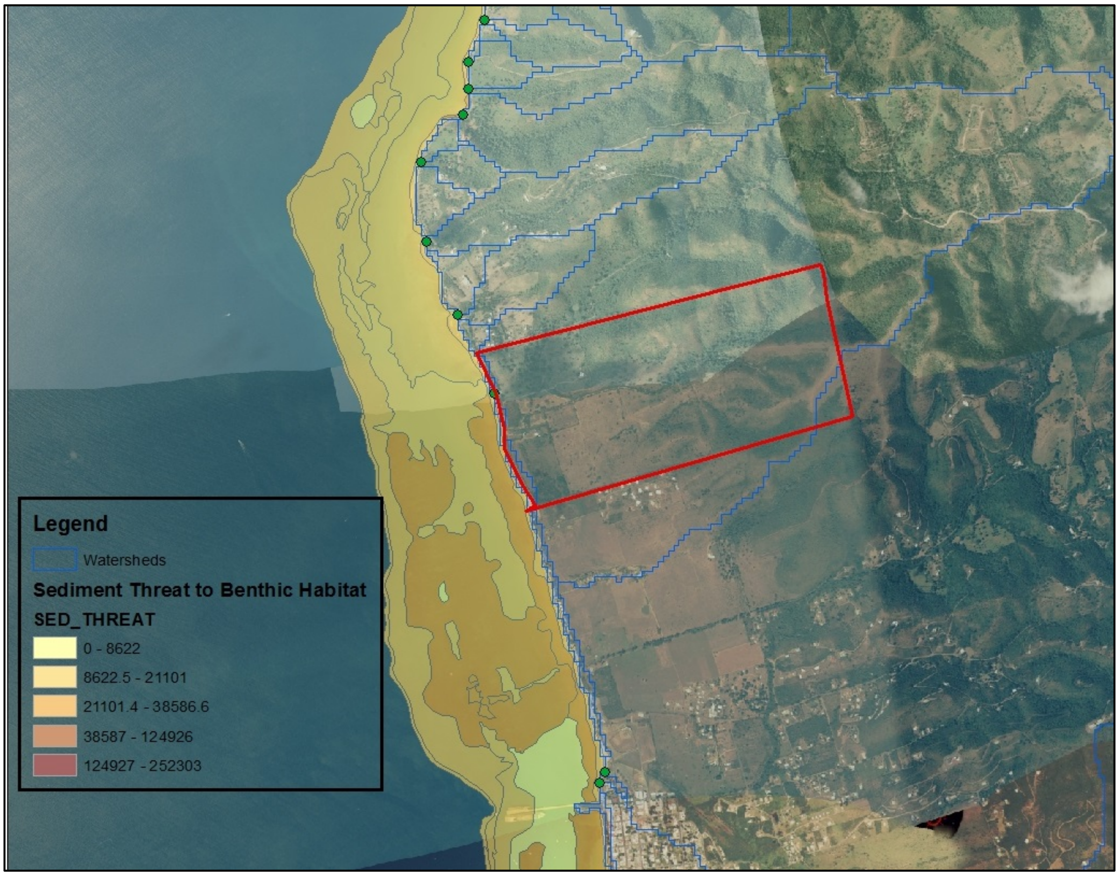

Figure 12. Map of sediment threat to benthic habitats in the project area—approximate project boundaries are shown in red and watersheds are delimited in blue. Sediment threat to benthic habitat was determined by calculating the sediment delivery potential at outflow points to the sea (shown as green dots) and a radius of 1 km from shore for USVI. The colors represent increasing levels of sediment threat based on calculated sediment loading.

As noted previously, as part of the May 2016 submission, William and Punch LLC submitted information regarding proposed sediment loading reduction based on the the project's sediment and erosion control plan. NMFS analyzed this information and requested a limited technical review from HWG of the May 2016 stormwater treatment calculations, as well as other sediment and erosion control information submitted by the applicant. Based on our analysis and technical comments from HWG, NMFS does not believe the information provided by the applicant supports the assertions that there is significant sediment loading from the undeveloped property where the Amalago Bay project is proposed and that the applicant will be able to achieve an 80% reduction in the current sediment loading rate during construction and operation of the project. The calculations done by the applicant do not indicate the amount of sediment being generated or discharged from the overall project compared to the existing undisturbed conditions. The applicant provided water quality monitoring data from DPNR for 4 sampling events in 2003, 4 in 2004, and 1 in 2005 for a station that appears to be in nearshore waters of the project area. In December 2003, turbidity of 3.78 was recorded following a rain event of less than 1-in based on historic weather data from NOAA, although the weater station is not in the project area. In September and November 2004, turbidities of 5.39 and 10.3 NTU, respectively, were recorded, apparently corresponding to runoff generated by storm events of over 2-in prior to sampling events. These high turbidity levels appear to have been reached as stormwater flow crested because the increase in turbidity was gradual based on three measurements taken during each sampling event. The applicant did not provide other water quality data that would indicate sediment loading from the property is significant and the results of our previous site inspections also do not support this assertion as discussed previously in this document. Further, HWG determined that even with the sediment control measures proposed by the applicant, in one of the basins (18A) that will have approximately 50% of its area disturbed due to the proposed project, sediment loading would be approximately 26 times existing conditions. In the other basin (1) that will have all of its area disturbed, sediment loading will be approximately 48 times existing conditions using the same assumptions regarding annual erosion rates for construction and wooded areas used by the applicant. Additionally, the applicant's assertion that they can achieve an 80% reduction in the currect sediment loading including during construction is unusual. This 80% reduction measure is typically applied only to post-construction stormwater and erosion control measures. Earth movement will always generate more sediment and site conditions, such as the steep slopes and unstable soils in the action area, often make it extremely difficult to control sediment transport downhill overall but even less when there are storm events. The applicant's calculations used values that were from Rhode Island rather than USVI, which HWG found could lead to dramatic differences in the actual erosion rate during construction (up to 15 times more). The discussion that follows further details the effects of sediment loading to ESA-listed corals and elkhorn and staghorn coral critical habitat we expect to occur as a result of the project.

Areas Expected to Be Impacted by Runoff

The distance sediments and other contaminants will be transported into the Caribbean will depend on the size of the rainfall event and local oceanographic patterns. The distance of transport will also affect the severity of impacts, with sediments and contaminants more likely to settle out nearer to shore, or become diluted farther from shore. Based on tidal currents and wave patterns, transport of land-based sediments and contaminants are generally expected to be from nearshore to offshore with some net transport of sediment to the southeast due to wave refraction (M. Canals, UPRM, pers. comm. to L. Carrubba, NMFS, December 21, 2014). Land-based sources of sediment and other contaminants, as well as nutrients (using chlorophyll concentrations as indicators of in-water nutrient levels) in coral reef and hard bottom areas in USVI were found to affect coral areas up to 984 ft (during normal rain fall) from shore depending on the location of each survey station in relation to the shoreline and up to 1 km (3281 ft) from shore during large storms (Nemeth and Nowlis 2001); (Smith et al. 2008); T. Smith, UVI, pers. comm. to L. Carrubba, NMFS, December 12, 2014).

During larger storm events (tropical systems that occur about once every 2 years), when the dominant onshore-offshore transport pattern and southeastern transport pattern due to wave refraction will be stronger (M. Canals, University of Puerto Rico-Mayagüez Campus [UPRM], pers. comm. to L. Carrubba, NMFS, December 21, 2014), Smith noted that effects of sediment plumes from larger storms on corals can be observed up to 1 km offshore in USVI (T. Smith, UVI, pers. comm. to L. Carrubba, NMFS, December 12, 2014). Information provided by the USACE indicates that storm surge and stormwater runoff during storms could impact areas up to 1,200 ft from the shoreline, which would affect the 77 ac of colonized bedrock and pavement in Area B of the action area and potentially other areas of reef and colonized hard bottom in Area C of the action area (Figure 5). NMFS believes that adverse impacts to the additional 77 acres of colonized bedrock and pavement that contains ESA-listed corals and the essential feature of elkhorn and staghorn coral critical habitat from large storms will occur episodically (Figure 5 Area B).

Coral Impacts from Sediment and Contaminant Runoff from Development
Studies in Puerto Rico showed that resuspension of marine sediments did not significantly affect coral growth whereas sedimentation by terrigenous sediments in reef areas had a negative effect on coral growth rates (Torres 2001a). Sediment core data from nearshore wetland and coastal embayments around St. Thomas and St. John show that over the past 15-25 years, sedimentation rates have increased up to 2 orders of magnitude (Rogers et al. 2008b). Sediment deposition maps created for the Coral Bay watershed in St. John indicate that the current runoff and sediment deposition rates are 7 times greater than in the past 5,000 years with the layer of terrigenous material that is currently found on marine bottoms in the bay likely having been deposited over the past 40 years as development has increased in the watershed (Thomas and Devine 2005). Sediments accumulate on dead and living corals and exposed hard bottom reducing the available substrate for larval settlement and affecting coral growth. Sedimentation leads to degradation of the quality of coral habitat for coral recruitment and growth, as well as impacts to coral colonies such as abrasion and excess mucous production that then affects the coral's ability to produce enough energy for survival, growth, and sexual reproduction. Reef sites exposed to average sedimentation rates between 10-14 mg per cm^2 per day showed a 38% increase in the number of coral colonies experiencing pigment loss compared to reef sites exposed to sedimentation rates between 4-8 mg per cm^2 per day (Nemeth and Nowlis 2001). The (Nemeth and Nowlis 2001) study was associated with development in the Caret Bay watershed, but other studies in developed watersheds with and without on-going construction activities around St. Thomas and St. John have found that nearshore waters adjacent to highly developed watersheds typically average over 10 mg per cm^2 per day whereas offshore reefs typically average less than 0.5 mg per cm^2 per day (Rogers et al. 2008b); (Smith et al. 2008). During a severe rain event, sediment load can increase to more than 30 mg per cm^2 per day (Rogers et al. 2008b). Several studies have established a level of 10 mg per cm^2 per day as the tolerance threshold for corals before sediment stress from land-based sources becomes too severe for colonies to recover (Nemeth and Nowlis 2001); (Rogers et al. 2002); (Rogers et al. 2008b) (Smith et al. 2008).

As discussed in Section 4.3 Coastal Development, several studies in St. Thomas and St. John have demonstrated that development of watersheds leads to dramatic increases in sedimentation associated with higher runoff intensities and increases in impervious surfaces and unstable slopes in areas with poor soils and steep slopes (Hubbard et al. 1987); (Nemeth and Nowlis 2001); (Ramos-Sharron and MacDonald 2007b, a); (Gray et al. 2008); (Rogers et al. 2008b) (Smith et al. 2008). A study by (Smith et al. 2013) in areas of Coral Bay, Fish Bay, and Lameshur Bay, St.

John, and Teague Bay, St. Croix, as well as offshore sites found that turbidity and chlorophyll showed variation within and among bays consistent with inputs of land-based sources of pollution. (Smith et al. 2013) found chlorophyll concentrations to be indicative of nutrient enrichment leading to increased concentrations of phytoplankton. (Smith et al. 2013) concluded that heavily developed catchments (areas of Fish Bay and Coral Harbor in Coral Bay), even in the absence of active construction, had the greatest chlorophyll concentrations and turbidity levels, especially nearshore. Studies linking development to sedimentation and coral growth have found that coral growth was affected for several years following development, including development at the headwaters of large watersheds (Hubbard et al. 1987). In watersheds where development continues, coral growth has been affected permanently, likely due to damage to the corals themselves as well as changes in habitat quality (Smith et al. 2008).

From 2001-2005, 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 mid-shelf reefs and over 45-fold greater than for shelf edge reefs (Smith et al. 2008). The clay and silt fraction is an indicator of terrigenous material content of the sediments due to terrestrial development on steep slopes with poor soils and the transport of eroded soils in stormwater runoff to nearshore waters. 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 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 pigment loss compared to reef sites exposed to sedimentation rates between 4-8 mg per cm^2 per day (Nemeth and Nowlis 2001), which corresponds to findings of other studies in the USVI regarding coral tolerance thresholds for sedimentation to result in declines in coral health, as well as habitat degradation (Rogers et al. 1984b); (Rogers et al. 2008b).

Based on the above, chronic impacts associated with the transport of land-based (terrigenous) sediments and contaminants to Area A are expected to include increases in sedimentation of ESA-listed corals and elkhorn and staghorn coral critical habitat, increases in nutrients that could lead to increased algal growth and decreases in coral growth or smothering of corals by algae, and increases in other contaminants such as pesticides that could affect corals through accumulation in coral tissue leading to impacts to survival, growth, and reproduction. Coral colonies chronically affected by stressors such as pollutants suffer more severe effects from warming-induced bleaching events and associated disease outbreaks as well. Coral growth rates have been found to decline in areas with increased terrestrial sediment inputs (Torres 2001a); (Hubbard et al. 1987). Nutrients can also affect the skeletal density and ability of corals to calcify making colonies more prone to breakage and erosion. Chronic impacts will be frequently recurring, at a high to moderate intensity, and are expected to continue for a long time (i.e. decades). Episodic adverse impacts from storm redistribution of sediments and contaminants will occur infrequently, at a low to moderate intensity, and are expected to continue indefinitely. In particular, the transport of land-based sources of pollution to the area from 500 ft offshore of the Amalago Bay property to the shelf edge associated with large storm events is expected to occur every 2 years, on average.

Effects of Project Design Elements in Preventing or Minimizing Runoff
There are two significant categories of concern with these elements of the proposed action: they do not meet the ESA's requirements that harm minimizing aspects of an action must be reasonably certain to occur, and they do not seem capable of rendering any meaningful protections to listed resources. In the relatively arid and undeveloped east end of St. Croix, (Reale-Munroe et al. 2014) found elevated erosion rates just from the erosion of a foot trail to the beach and concluded that this suggests the need for carefully planned development and BMPs in steeper and wetter areas of St. Croix, such as the west end where the Amalago Bay project is proposed. Given that many of the areas proposed for development as part of the Amalago Bay project have 100% slopes, coupled with the density of development and the fact that development will occur over several years, it is unlikely that the proposed construction and operational BMPs can be effective in preventing the transport of land-based sources of pollutants to nearshore waters, for a number of reasons discussed below.

The ESA requires that species protection measures be reasonably specific, certain to occur, and capable of implementation; they must be subject to deadlines or otherwise-enforceable obligations. Species protection or harm mitigation measures must have some form of measurable goals, action measures, and a certain implementation schedule; i.e., mitigation measures must incorporate some definite and certain requirements that ensure needed mitigation measures will be implemented. Biological opinions have been invalidated by courts where the measures required to avoid jeopardy were to be identified in a plan to be developed in the future. Further, proposed measures to avoid or mitigate harm have been found unlawfully uncertain because their implementation was not within the control and enforcement of the relevant federal agency parties to the consultation and instead were uncertain and vaguely defined actions of third parties.

Any construction work done in the rainy season (peak from December-March) or during the hurricane season (July-November) could result in significant transport of sediment to the shelf edge reef during large storms. The applicant did not indicate any restrictions on construction during the rainy season in the project documents.

The project will be constructed over the course of several years, but no completed development plan, including paving, grading, and landscaping, is available for upland portions of the proposed development. The development and stormwater management plans are not complete because the applicant has indicated that the majority of the development will be the responsibility of others to build. With the majority of the development being the responsibility of others to build, the project could take longer than currently projected by the applicant based on similar projects with master plans, such as the Botany Bay development in St. Thomas. A master plan was created for a 365-ac area within the Botany Bay development in 2002 for a resort and high-end residential development. This plan was part of the larger Botany Bay development that has been underway for about 40 years with several owners and several development projects completed, as well as many of the road cuts throughout the entire property. The development of a portion of the acreage in the 2002 plan beginning in October 2005 led to a marked increase in sedimentation rates in nearshore waters of the bay (Rothenberger et al. 2008). The TCRMP has a monitoring site in the northern portion of Botany Bay and has found that the decades-long residential development of the area and associated decreases in vegetative cover in the watershed and increases in land-based sources of pollutants have led to declines in coral cover and increases in coral disease (Smith et al. 2011a). Because many of the subdivision components are simply the division of properties into lots with the individual owners responsible for development and construction on their property, land clearing and associated impacts to nearshore waters related to the transport of land-based sediments and contaminants have been on-going for decades. We are concerned actions proposed for this Amalago Bay project to avoid and mitigate impacts to ESA-listed corals and acroporid coral critical habitat will not be binding and enforceable because the actual development of much of the residential project will be the responsibility of third parties—similar to what has occurred in Botany Bay. The *Declaration of Covenants, Conditions, Restrictions and Easements for the Residences at Estates William & Punch* (Declaration) notes that an Architectural Control Committee (Committee), consisting of 3 individuals designated by William & Punch LLC until 60% of the lots have been sold, will be responsible for reviewing and approving development proposed by individual lot owners. After 60% of the lots have been sold, the owners will select the members of the Committee. The Declaration repeats some of the proposed stormwater, erosion and sediment control measures that, as discussed previously, NMFS believes will not be adequate to minimize sediment transport to nearshore waters based on the site conditions, NMFS' review, and the technical review HWG prepared. There are no requirements related to the qualifications of the members of the Committee that would guarantee the members are knowledgeable about construction inspections (which they are authorized to do in the Declaration), BMPs and their implementation, drawing and reading building plans, or other subjects relevant to their role as the approvers, inspectors, and enforcers of the Declaration. The Botany Bay development included requirements to use construction methods that would minimize runoff to protect nearshore coral habitats but the TCRMP has found that, even with these requirements and as individual homeowners continue to develop their lots, there have been adverse impacts to coral areas attributed to land-based sediment transport. Therefore, we believe the measures such as those in the *Declaration of Covenants, Conditions, Restrictions and Easements for the Residences at Estates William & Punch* cannot be relied upon as reasonably certain to provide protection to ESA-listed corals and their habitats.

The BMPs for sediment and erosion control presented by the applicant include the use of silt fences, hydroseeding, and liquid polymer emulsions to capture eroded soils and stabilize slopes, road cuts, and construction areas. Studies in USVI have shown that, despite similar BMPs as those proposed by the applicant, roads that are under construction, roads that have not been completely stabilized prior to the rainy season, and cleared areas on steep slopes are major contributors of land-based sources of pollution in runoff during periods of rainfall (Nemeth and Nowlis 2001); (Smith et al. 2008); (Ramos-Sharron and MacDonald 2007b, a). Liquid polymer breaks down very quickly when driven on so it does not remain effective in preventing erosion of road surfaces for long periods of time. The degradation of liquid polymer emulsions from slopes and road surfaces will lead to the transport of nutrients to nearshore waters. These nutrients could lead to algal growth on hard bottom and phytoplankton blooms in the water column, resulting in impacts to ESA-listed corals from the loss of available space for recruitment, smothering by algae, and declines in light availability due to increased phytoplankton in the water column and associated impacts to photosynthesis and growth and to acoporid coral critical habitat from the degradation of the function of the essential feature for recruitment and settlement. Chronic elevations in nutrient concentrations can produce bleaching and partial mortality in lobed star corals (Kuntz et al. 2005) and staghorn and elkhorn corals have also been found to show susceptibility to bleaching and decreases in growth due to increases in nutrient concentrations (see Section 3.2.3). Hydroseeding is also a source of nutrients if it is used during the rainy season and washed downslope to nearshore waters. The timing of the use of hydroseeding is not provided in the project information, so there is no indication that it will not be used during periods of heavy rains.

The stormwater management and sediment controls proposed for during and after development all require maintenance to remove accumulated sediment. Maintenance is proposed, but it includes the stockpiling of sediment removed from sediment and erosion control structures for potential future use as material for landscaping during the construction of the project only. Inspections are proposed quarterly and maintenance up to twice per year. Thus, in addition to the potential erosion of stockpiled sediment from cleaning of sediment and erosion control structures, control measures may fail at times when storms are frequent and the combination of infrequent maintenance and inspections will lead to sediment transport to nearshore waters.

The use of the inland marina as the main receiver of all stormwater from the project will concentrate terrestrial sediment deposition in this area and will also lead to the transport of sediment and other contaminants to waters in the action area given that the marina design contemplates a maximum 4-day flushing time for the entire basin. The marina entrance and flushing channel along with the natural drainage through the mangrove wetland, which will remain as part of the Amalago Bay project but will receive more runoff than it does currently, will transport land-based sediments and contaminants to ESA-listed corals and acroporid coral critical habitat.

Flooding and overland flow resulting in nearshore contamination by land-based sediments and contaminants are also likely from the project because structures such as culverts are designed for 50-year storms (with rainfall of 12.4 – 20.5 inches over 24 hours for St. Croix; NOAA's Hydrometeorological Design Studies Center.[24] However, the rainfall recorded during hurricanes and tropical storms is consistent with a "100-year storm" and storms of that intensity may occur as frequently as every 2 years in St. Croix.[25] The EAR indicates that, when 100-year storms occur, runoff will flow over the golf course and other areas of the development.

Two of the storage stormwater diversion basins are part of the golf course. A summary of studies examining the effectiveness of various stormwater and water quality controls found that detention basins were able to reduce the levels of total suspended solids transported in stormwater, but not total or dissolved phosphorus or nitrogen concentrations (Leisenring et al.

[24]The term "50-year storm" refers to a storm with a 1-in-50 probability (2%) of dropping 12.4 – 20.5 inches over 24 hours for St. Croix during a given year. Thus, in any given year, there is a 2% chance a storm will drop that much rain in 24 hours for St. Croix. This does not mean that a storm of this magnitude will only occur once every 50 years. http://hdsc.nws.noaa.gov/hdsc/pfds/pfds_map_pr.html

[25]A "100-year storm" refers to a storm with a 1-in-100 probability (1%) of dropping 14.5 – 24.3 inches over 24 hours for St. Croix.

2013). Further, the summary found that basins with fast emptying times, such as those proposed in the Amalago Bay development, are not as effective in removing sediments and associated contaminants like total phosphorus, which binds to clay sediments in particular (Leisenring et al. 2013). Thus, the probability of contaminant transport to nearshore waters, including pesticides and fertilizers, during large storms is high given that the diversion basins are the main components of the proposed stormwater management system.

Last, concerns raised about insufficient sewer capacity for the project suggest an additional route of contamination that could have equally serious impacts as the stormwater runoff. As noted by the EPA in their July 13, 2012, letter, the proposed development may also result in wastewater discharges as the existing system is likely not adequate to handle the increased wastewater load even with the proposed improvements to the Lagoon Street Pump Station in Frederiksted. EPA also noted that the sediment and erosion control information and new effluent limitation requirements were not met by the proposed project plan. The EPA concluded this, along with stormwater runoff from the steep slopes within the property, is likely to lead to water quality degradation.

5.2.3 Partial Destruction of Corals and Coral Critical Habitat from Indirect Impacts

NMFS expects that indirect impacts will result outside of the footprint of of the marina channels and jetties and the creation of the beach on the island. NMFS expects that stormwater runoff and terrigenous sediments will affect an additional 30.31 ac in Area A, and that land-based sources of pulltants deroived from storm event will affect 77 ac in Area B. These indirect impacts are considered below.

5.2.3.1 Lobed and Mountainous Star Coral

In addition to the direct impacts to lobed star and mountainous star corals within the marine dredging and construction footprint, NMFS expects the proposed action will result in full or partial mortality of an additional 1,151,064 lobed star coral colonies (104,435 colonies impacted in 2.75 ac (calculated in Section 5.2.2); 30.31 ac ÷ 2.75 ac x 104,435 colonies = 1,151,063.58182 colonies) and 575,537 mountainous star coral colonies (52,218 colonies impacted in 2.75 ac (calculated in Section 5.2.2); 30.31 ac ÷ 2.75 ac x 52,218 colonies = 575,537.301818 colonies) in the 30.31 ac area immediately offshore of the Amalago Bay project (Figure 5, Area A) we believe will be chronically affected by stormwater runoff and associated terrigenous sediments and contaminants. Lobed and mountainous star coral colonies within other portions of the action area (77 ac area, Figure 5, Area B) will also be adversely affected episodically by the transport of land-based sources of pollutants to the larger area during storms and discussed below.

Elevated nitrogen levels were found to reduce calcification in lobed star and mountainous star corals (Marubini and Davies 1996). Elevated nutrient levels were also found to increase the rate of tissue loss in mountainous star corals that were affected by yellow band disease (Bruno et al. 2003). Lobed star corals exhibited declines in their growth on reefs impacted by terrigenous sediment in Puerto Rico (Torres 2001b) and (Eakin et al. 1994) found declines in lobed star coral linear growth during periods of construction in Aruba which they associated with sediment stress. NMFS believes that the lobed star and mountainous star corals in Area A that will not be killed by sediment impacts will not be able to successfully reproduce. There was no demographic survey conducted for this project so we use information from a limited demographic survey completed for a project in Puerto Rico that estimated approximately 9% of the *Orbicella* spp. colonies were sexually mature based on observed colony sizes. Thus, 402 lobed star and 200 mountainous star coral colonies that will be affected by the project are likely to be sexually mature and would be expected to suffer reproductive failure due to the stress caused by sediment impacts. Fertilization success for lobed star and mountainous star corals has been found to be low and highly linked to the number of colonies available to spawn at the same time (Levitan et al. 2004). The spatial distribution of colonies may also influence reproductive success on a reef (Villinski 2003). Further, successful recruitment by *Orbicella* species is thought to have always been rare and mortality rates are correlated to size with larger sizes being more likely to survive (Smith and Aronson 2006). Lobed and mountainous star corals are sensitive to stressors such as sediment and nutrients, and the corals will have to dedicate energy to trying to remove sediment cover and will not have energy to dedicate to sexual reproduction while tissue loss and declines in growth and potentially increases in disease are occurring. Therefore, it is extremely unlikely that successful reproduction or recruitment of these species would occur in Area A as a result of the project.

Lobed and mountainous star coral colonies are expected to be adversely affected within the 77 ac zone (Figure 5 Area B) where episodic impacts from large storms will result in the transport of land-based sediments and contaminants to the shelf edge. Approximately 2,955,250 lobed star (77 acres x 24.5% coral cover = 18.865 acres coral; 18.865 ac x 15% lobed star coral = 2.82975 ac lobed star coral; 1 ac = 40,468,564.224 cm^2; 2.82975 ac x 40,468,564.224 cm^2 = 114,515,919.613 cm^2 lobed star coral; 114,515,919.613 cm^2 ÷ 38.75 cm2 per colony = 2,955,249.5 colonies, rounded to 2,955,250 colonies lobed star coral) and 1,477,625 mountainous star coral colonies (77 acres x 24.5% coral cover = 18.865 acres coral; 18.865 ac x 7.5% mountainous star coral = 1.41488 ac mountainous star coral; 1 ac = 40,468,564.224 cm^2; 1.41488 ac x 40,468,564.224 cm^2 = 57,257,959.806 cm^2 mountainous star coral; 57,257,959.806 cm^2 ÷ 38.75 cm2 per colony = 1,477,624.8 colonies, rounded to 1,477,625 colonies mountainous star coral) at the shelf edge are expected to experience sediment stress. These impacts will result in declines in colony health due to increases in sedimentation, nutrients, and other contaminants that can be absorbed into coral tissue and affect colony health and could lead to mortality. Sexual reproduction by these corals is likely to be reduced as the corals will dedicate energy to trying to remove sediment cover from their tissue rather than to sexual reproduction. Lobed and mountainous star corals usually spawn in August or September, which is during hurricane season, so it is possible that sexual reproduction will be affected due to the transport of land-based pollutants from the Amalago Bay project prior to spawning.

5.2.3.2 Elkhorn Coral

As noted previously, while no elkhorn coral colonies are present in Area A, there are elkhorn corals in the northern part of the action area (Sprat Hole). Historically elkhorn corals have been reported in the nearshore hard bottom areas up to 5 m depth within the Frederiksted Reef System (Toller 2005). As previously stated the action area is part of this integrated system. NMFS expects that the 30.31 ac of colonized hard bottom (Area A) will be chronically adversely affected by stormwater runoff and associated sediment and contaminant deposition due to the project. This means that elkhorn coral larvae that would settle within this 30.31-ac space would not survive, either due to the impacts of sedimentation and other contaminants to the larvae themselves, or due to the impacts to the essential feature of acroporid coral critical habitat, specifically the covering of the essential feature by sediments and macroalgae.

Based on a recovered population of elkhorn coral (per the recovery plan) this area would be expected to support 3,065 elkhorn coral colonies (10% cover of 30.31 acres = 3.03 acres x 4046.86 m^2 [1 acre = 4046.86 m^2] = 12,261.98 m^2 x 0.25 colonies per m^{2} = 3,065 colonies). This loss will reduce future reproduction potential by eliminating settlement habitat for larve and recruits and any possible reproduction of a possible 3,065 colonies.

Elkhorn coral's predominant habitat is reef crests and shallow fore-reefs less than 12 m depth. It also occurs in back-reef environments and in depths up to 30 m, however, as stated above, within the Frederiksted Reef System elkhorn coral has been historically found in hard bottom up to 5 m in depth, and the action area is not a back-reef environment. Based on this information we do not believe that sediments and contaminants will impact elkhorn colonies or larvae in the deeper waters within the 77 ac zone (Area B) where episodic impacts from large storms will lead to the transport of land-based pollutants to the shelf edge.

5.2.3.3 Staghorn Coral

Staghorn corals have been reported on inshore colonized hard bottoms, mid-shelf colonized hard bottoms and patch reefs, and the offshore shelf edge reef within the Frederiksted Reef System, including the reef seaward of the Amalago Bay project, although the number of colonies was not quantified (Toller 2005, Smith et al. 2014). Staghorn coral has also been documented in nearshore areas from Sprat Hole northward and near the Westend Saltpond (S. Pittman, NOS Contractor, pers. comm. to L. Carrubba, NMFS, December 12, 2014).

We do not predict that the loss of the 30.31 ac of colonized hard bottom in Area A will result in mortality of staghorn coral larvae and fragments. While staghorn could settle in this area, we do not believe they would successfully recruit, given this species' much deeper depth distribution in St. Croix, and likely outcompetition in these shallower depths by species such as the star corals.

We anticipate that storms of the magnitude that leads to the transport of land-based pollutants to the shelf edge (the 77 ac area, Figure 5, Area B) will occur approximately every 2 years on average during the construction and operation of the project. The staghorn coral colonies at the shelf edge can experience sediment stress. These impacts can result in declines in staghorn colony health due to increases in sedimentation, nutrients, and other contaminants that can be absorbed into coral tissue and affect colony health and could lead to mortality. Sexual reproduction by these corals can be reduced as the corals will dedicate energy to trying to remove sediment cover from their tissue rather than to sexual reproduction. We do not have an exact number in terms of staghorn coral colonies that may be affected because the TCRMP only reported the presence of this species in the monitoring station within the action area. Similarly, none of the studies conducted within the Frederiksted Reef System included quantification of the number of staghorn coral colonies present in the area. Therefore, in order to estimate the potential number of colonies that could be affected by the transport of sediments and contaminants to the shelf-edge, we calculated the density based on the NOAA National Coral Reef Monitoring Program (NCRMP) from 2015 surveys. NCRMP surveys were conducted at 6 locations around St. Croix. One of the stations on the northwest coast of St. Croix is closest to the action area and the only station where staghorn coral was observed. Two staghorn coral colonies were reported in 1 of 15 10m^2 transects in deep and shallow waters (similar to depths in the action area), which yields a density of 0.013 colonies per m^2. Thus, there could be as many as 4,051 staghorn coral colonies within the 77 ac area at the shelf edge (1 acre = 4046.86 m^2 x 77 acres = 311608.22 m^2 x .013 colonies per m^2 = 4050.91 colonies) that will be affected by land-based pollutants from project construction and operation. This is likely a conservative estimate because the distribution of staghorn coral colonies is not uniform, as evidenced by the fact that 2 colonies were observed in only 1 of 15 transects, but this estimate is based on the best available information.

5.2.3.4 Acropora Critical Habitat

NMFS expects that the 30.31 ac of colonized hard bottom within Area A that will not be directly destroyed by project construction will be chronically impacted by stormwater runoff and associated sediment and contaminant deposition, given the increase in runoff events and volumes expected due to the project. Chronic impacts will be frequently recurring, at a high to moderate intensity, and are expected to continue for a long time (i.e., decades). In particular, the increases in normal rainfall runoff events to at least 34 days instead of the current 2 days will be related to the loss of vegetative cover on the Amalago Bay project site and associated transport of land-based sources of pollution. This represents a significant increase. These events are expected to lead to long-term impacts of continuous sediment covering of the essential feature, and continuous promotion of macroalgal growth that will further impede any coral larval settlement and recruitment. Thus, 30.31 ac of the essential feature necessary for elkhorn coral recovery will not be functional in the nearshore area up to 500 ft offshore of the project (Figure 5, Area A). As discussed in Sections 3.2.3, 4.3.4 and again in Section 5.2.2, the portion of the Frederiksted Reef System in the action area is relatively unaffected by sediment, particularly terrigenous sediments, that could cover the essential feature and is therefore an important area for expansion and recovery of elkhorn coral. The majority of near shore reefs in St. Croix are affected by sedimentation with sediments that are terrigenous in nature, due to coastal development, fragmenting the essential feature throughout the unit. McLaughlin et al. (2002) found that when distributions of coral species become isolated because of habitat loss, populations become more vulnerable to climate change and other threats. The loss of habitat patches will affect the availability of areas for coral larvae to settle. Larvae are only viable for a short time so larger distances between areas of suitable habitat for elkhorn corals make settlement and growth less likely. Also, elkhorn coral's primary mode of reproduction in the USVI is through fragmentation. A branch of elkhorn coral may be carried by waves and currents away from the parent colony, and fragments cleaved from the colony may grow into new colonies (Highsmith et al. 1980, Bak and Criens 1982, Highsmith 1982, Rogers et al. 1982). Genetically identical clones have been found separated by distances that range from 0.1 to 100 m (0.3 to 328 ft), but usually less than 30 m (98 ft) (Baums et al. 2006a). The horizontal length of area A is 4,000 ft; therefore, the loss of this area will make reproduction more difficult as larve and fragments will have to travel further.

Material transported from the developed areas of the project and the terrigenous sediments that will accumulate in the 30.31 ac area will be transported seaward during large storms and will result in episodic impacts to the essential feature in the 77 ac in Area B (Figure 5). However, these impacts will be less frequent and less intense. We believe that these impacts will cause degradation of the essential feature of critical habitat over time, but the essential feature will remain functionally established, because sediments will not cover the hardbottom areas long enough or deeply enough to preclude successful recruitment.

5.2.4 Additive Effects to ESA-Listed Corals and Acroporid Critical Habitat from other Routes of Effects

As discussed above, the proposed project has numerous routes of potential adverse effects to listed corals and designated critical habitat. In addition to the sediment and contaminant impacts discussed above, corals and critical habitat may be adversely affected due to accidental groundings of construction vessels, spudding of construction vessels, recreational boating and diving impacts to corals and hard bottom, and boating-related water quality impacts in the marina and nearshore.

Lobed and mountainous star corals are known to be located in Areas A, B and C, staghorn coral colonies are known to be located in areas B and C, and elkhorn coral colonies are known to be in area C, and could be crushed or broken by accidental groundings by work vessels transiting to and from the construction site. Acroporid critical habitat is in all three areas and could also be broken by groundings. Given the depth at which staghorn corals are reported in the project area (from 6-18 m), we believe that accidental groundings of work vessels impacting staghorn coral colonies are extremely unlikely to occur. Lobed and mountainous star colonies in Area A can be affected by work vessels, however, these are the same corals expected to be killed or severely injured due to chronic sedimentation impacts described above. Lobed and mountainous star corals and staghorn corals can be found in areas B and C and and elkhorn coral can be found in area C, and could be affected by transiting work vessels. Work vessels are expected to follow marked navigation channels. The applicant will also require an education program be implemented incorporating information regarding the proper operation of vessels in areas containing ESA-listed sea turtles and corals during construction and operation of the project. Based on this information we believe that adverse effects in areas B and C due to transiting work vessels are extremely unlikely to occur. Spudding of construction vessels will only happen in area A where construction will be taking place, however the applicant will perform pre-spudding surveys to ensure spuds are not placed on coral habitat. Moreover, the corals in this area are the same corals expected to be killed or severely injured due to chronic sedimentation impacts. The effects of work vessel transit and spudding on elkhorn and staghorn critcal habitat will be the same as for the corals, as explained in this paragraph, for the same reasons.

Toller (2005) found approximately 13% of all coral habitat within this reef system had suffered mechanical damage from boats (i.e., damage from anchoring and groundings), mainly cruise ships and other commercial vessels using the Frederiksted Pier, but also recreational vessels (as a recreational boat ramp and other facilities are in the area of the Frederiksted Pier). USVI monitoring of a point on the Sprat Hole shelf edge found the habitat (corals and hard bottom) was impacted by derelict fishing gear and anchoring of dive vessels (Smith et al. 2011a).

Up to 70 vessels will be able to use the marina and these vessels may range in size from 40-100 ft, on average. According to Marine Title[26], there are 4,479 registered boats (this includes all registered vessels) in the U.S. Virgin Islands. The 70 new vessels associated with the proposed action represent a 1.6% increase in the of registered vessels in U.S. Virgin Islands. The vessels that will be using the marina will be able to travel anywhere they want and can cause damage anyplace they go; however, we believe that the majority of any damage done by these vessels will be done in the Frederiksted reef system. As stated above, the main source of the 13% damage to this reef system is from commercial vessels using the Frederiksted Pier. Corals within the action area may also be impacted by the anticipated increase in recreational diving on reefs in the action area. This reef system has approximately 14 popular dive sites where anchoring and damage from divers takes place. Potential impacts from increased vessel use of the action area could include accidental groundings, anchor damage, release of contaminants such as PAHs, and breakage and abrasion of colonies and habitat by divers. We can expect more impacts to areas containing acroporid coral critical habitat adjacent to the project site because vessel owners will want to be close to the Amalago Bay facilities and will anchor closer to shore where colonized hard bottom containing the essential feature of acroporid coral critical habitat is present. Impacts in Area A would be additive to the same corals and critical habitat expected to be lost due to chronic sedimentation impacts. This project will not increase the number of dive sites in the area. Also, the applicant proposes a number of measures to help lessen the effects of the vessels associated with this project: 1) 15 mooring buoys will be installed between Frederiksted and the project site to minimize potential anchoring impacts from vessels associated with the Amalago Bay development; 2) use of the mooring buoys and marina will be controlled by an on-site harbormaster who will also oversee the boat-related educational plan; and 3) the jetties and associated channels will be clearly marked with aids to navigation (ATONS) in coordination with the U.S. Coast Guard (USCG) to minimize the potential for accidental groundings. Based on this information we believe that any damage caused by the 70 new vessels associated with the marina will not be detectable when compared to the damage being caused currently by recreational and commercial vessels and the damage expected from land-based sources of pollution.

The marina will have a fueling facility. Spills from the facility could affect corals. Because the inland marina is designed with a flushing channel for flow-through and short residence times within the basin, petroleum spills could result in the transport of petroleum products out of the marina basin and along the shoreline. However, according to the information provided by the applicant, the marina will be operated under requirements for containment and clean up plans (developed by the applicant), operational limitations to prevent spills, sewage and vessel maintenance guidelines and other measures. Based on this information, we believe it is extremely unlikely that a large-scale, acute petroleum spills from the marina operation that would be severe enough to produce adverse effects to corals. Any differences in the operation from what is presented by the applicant that are likely to result in greater likelihood of fuel spills, use of response techniques that are harmful to corals, or other such sources of contamination would constitute new information and would require reinitiation of consultation.

[26] Marine Title, https://www.marinetitle.com/boat-registration/VI-Virgin-Islands.htm

5.2.5 Summary of the Effects of the Action on ESA-Listed Sea Turtles, Corals, and Acroporid Coral Designated Critical Habitat

As discussed in Section 5.1, we determined that a variety of sources of take are expected from the proposed project (Table 6). Implementation of the proposed action will have adverse effects on different life stages of hawksbill and green sea turtles due to the elimination and degradation of in-water foraging and refuge habitat, the construction of jetties, and lighting impacts. The jetties have the potential for lethal and/or non-lethal take of hatchling green, leatherback, and hawksbill sea turtles that emerge from their nests in the vicinity of the structures. Permanent habitat loss, as well as chronic and episodic habitat degradation, is expected to impact juvenile green and hawksbill turtles via displacement or need to expand home ranges. Finally, onshore lighting visible from the nearshore waters may impact nesting females of all 3 species by deterring them from leaving the water to nest.

Table 6. Summary of Expected Take of Sea Turtles

	Source of Take			
Species	*Jetties*	*Habitat Loss by Construction*	*Habitat Loss by Sedimentation*	*Lighting*
Green	91 hatchlings/ year killed, South Atlantic DPS	17 juveniles	151 juveniles	1 nest abandonment per year, South Atlantic DPS
Leatherback	114 hatchlings/ year killed	NA	NA	1 nest abandonment per year
Hawksbill	1,204 hatchlings/ year killed	2 juveniles	15 juveniles	1 nest abandonment per year
Notes: Additional information provided in Section 5.1				Impact consists of in-water female that was going to nest on a stretch of beach being deterred by lighting visible from nearshore waters.

The direct destruction of nearshore habitats proposed for this project will have permanent adverse effects on lobed star and mountainous star corals and acroporid coral designated critical habitat due to the loss of 2.75 ac of hard bottom habitat within the in-water construction footprint of the marina channels and jetties and created beach (within Area A in Figure 5). We anticipate that the loss of 2.75 ac of hard bottom due to the construction of the in-water portions of the project will result in direct impacts to approximately 405 lobed star corals and 202 mountainous star corals. It will result in the loss of future recruitment and growth habitat for 272 elkhorn coral planulae.

Construction and operation of the project will also have adverse effects on elkhorn, staghorn, lobed star, and mountainous star coral colonies in the adjacent waters that will not be directly affected by construction and operation. The impacts of greatest concern are associated with the transport of pollutants, including sediments and contaminants, from dredging and in runoff. NMFS estimates that, at a minimum, 30.31 ac of colonized hard bottom and patch reefs within 500 ft (within Area A in Figure 5) of the shoreline of the project will be adversely affected by chronic impacts from land-based sources of pollution. These areas will also be adversely affected by the additive effects of accidental groundings, vessel and diver breakage, and water quality impacts due to vessel use. Impacts to these 30.31 ac of hard bottom would kill or seriously injure an additional 1,151,064 lobed star corals and 575,537 mountainous star corals. Impacts to this hard bottom habitat will also result in the take of 3,065 future elkhorn coral planulae and recruits as the expected level of sedimentation and contamination in the area will prohibit the settlement of elkhorn corals in this area, which is acroporid critical habitat, and inhibit the growth of any recruits that do try and settle.

Information provided by the USACE indicates that large storm events could lead to project impacts up to 1,200 ft from the shore (within Area B in Figure 5), which would could result in additional episodic impacts to lobed star, mountainous star, and staghorn coral colonies, which are the dominant coral species in waters over 10 ft in depth along the west end of St. Croix (Toller 2005, Smith et al. 2014), as well as staghorn coral colonies, which are reported on the shelf edge reef in the action area (Toller 2005, Smith et al. 2014). Appoximately 2,955,250 lobed star corals, 1,477,625 mountainous star corals, and an unquantifiable number of staghorn coral colonies at the shelf edge are expected to experience sediment stress leading to reduced reproduction from these episodic impacts.

The essential feature of acroporid coral critical habitat is substrate of suitable quality and availability, meaning consolidated hard bottom or dead coral skeletons free from fleshy macroalgal and sediment cover, in water depths from the mean high water line to 30 m, to support successful larval settlement, recruitment, and reattachment of fragments. The combined chronic and episodic effects of in-water dredging and construction, land-based sources of pollution, and operation of a fueling facility in the marina, accidental spilling, vessel use, and recreational use, will result in the loss of function of the essential feature of acroporid coral critical habitat. The proposed project will result in the loss of function of the essential feature in 33 ac (2.69 plus 30.31) of acroporid coral critical habitat from the Frederiksted Reef System and will therefore affect the connectivity of the nearshore habitat. The loss of the essential feature along the section of the coast where the project is located will lead to further fragmentation of this habitat. The essential feature will be degraded in an additional 77 ac in deeper waters due to episodic sedimentation impacts, but the feature will remain functionally established.

6 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, or local private actions that are reasonably certain to occur in 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.

Cumulative effects from unrelated, non-federal actions occurring around St. Croix that may affect green, leatherback, and hawksbill sea turtles, and their habitats, elkhorn, staghorn and lobed star and mountainous star corals, and elkhorn and staghorn coral critical habitat include the continuation of activities described in the environmental baseline. NMFS is not aware of any other future state, tribal or local private activities that are reasonably certain to occur and have effects to the environmental baseline. Stranding data indicate that human activities lead to sea turtle mortality in waters around St. Croix. Human activities known to kill sea turtles include incidental capture in state fisheries, ingestion of and/or entanglement in debris, vessel strikes, and poaching. The cause of death of many stranded sea turtles is unknown. Many activities affecting sea turtles and coral critical habitat 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 around USVI that has been shown to affect water quality (in particular through increases in sedimentation rates) does not require federal authorization. Development often has no federal nexus if the project is located on uplands and is small in size. Depending on the number and location of these developments, sediment and nutrient loading to nearshore waters could become a chronic stressor. Indeed, information from EPA's list of impaired waterways in the USVI for 2010 and 2012 indicates that there were 204 instances where a pollutant caused impairment of the waterway's designated use (http://ofmpub.epa.gov/tmdl_waters10/attains_state.control?p_state=VI&p_cycle=&p_report_type=T). There were 196 instances in 2014 and 206 instances in 2016 of which 34% were due to turbidity (https://www.epa.gov/tmdl/us-virgin-islands-impaired-waters-list). In 2016, of the 32 reported impairments in St. Croix alone, 24 of them were due to turbidity. The most common pollutants causing impairment included turbidity, oxygen enrichment/depletion, pathogens (including coliform bacteria), pH/acidity/caustic conditions, and nutrients. The pattern of water quality degradation in USVI actually accelerated up to 2012 with 3 impairments reported in 2003 and 2004, 5 in 2005, 1 in 2006, 12 in 2007, 37 in 2010, and 90 in 2012. In 2016, 83 impairments were reported.

The fisheries occurring within the action area are expected to continue into the foreseeable future. Numerous fisheries in territorial waters have been known to adversely affect threatened and endangered sea turtles. NMFS is not aware of any proposed or anticipated changes in these fisheries that would substantially change the impacts each fishery has on the sea turtles, ESA-listed corals, and acroporid coral critical habitat covered by this Opinion.

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., over-abundance of land or sea predators, changes in oceanic conditions) that would substantially change the impacts that each threat has on the sea turtles, ESA-listed corals, and acroporid coral critical habitat covered by this Opinion. Therefore, other than expected increases in impacts from development, NMFS expects that the levels of interactions with sea turtles, elkhorn, staghorn and lobed star and mountainous star coral colonies, and acroporid coral critical habitat described for each of the fisheries and non-fisheries will continue at similar levels into the foreseeable future.

7 ANALYSIS OF DESTRUCTION OR ADVERSE MODIFICATION OF DESIGNATED CRITICAL HABITAT FOR ELKHORN AND STAGHORN CORALS

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

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

The critical habitat rule for elkhorn and staghorn corals identified specific areas where the feature essential to the conservation of Atlantic *Acropora* species occurs in 4 units within the jurisdiction of the United States: Florida, Puerto Rico, St. Thomas/St. John, and St. Croix. The St. Croix marine unit includes the action area for the proposed Amalago Bay project. The action area is on the west end of St. Croix in the Frederiksted Reef System that runs from north of the project in Sprat Hole to south of the Frederiksted Pier offshore of the Westgate Saltpond.

The St. Croix marine unit comprises approximately 126 mi^2 (80,640 ac). Of this area, approximately 90 mi^2 (57,600 ac) are likely to contain the essential feature of ESA-designated coral critical habitat, based on the amount of coral, rock reef, colonized hard bottom, and other coralline communities mapped by NOAA's NOS Biogeography Program in 2000 (Kendall et al. 2001). The key objective for the conservation and recovery of Atlantic acroporid corals that is the basis for the critical habitat designation is the facilitation of an increase in the incidence of sexual and asexual reproduction. Recovery cannot occur without protecting the essential feature of coral critical habitat from destruction or adverse modification because the quality and quantity of suitable substrate for ESA-listed corals affects their reproductive success. As noted in the rule designating acroporid coral critical habitat (73 FR 72210, November 26, 2008), the loss of suitable habitat is one of the greatest threats to the recovery of elkhorn and staghorn coral populations. Human-caused stressors have the greatest impact on habitat quality for elkhorn and staghorn corals.

The loss of the essential feature or a diminution in the function of the essential feature affects the reproductive success of elkhorn and staghorn corals because substrate for sexual recruits to settle is lost or unavailable. 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 (decline in individual fitness at low population size or density that can result in critical population thresholds below which populations crash to extinction), lower effective density (of genetically distinct adults required for sexual reproduction), and a reduced source of fragments for asexual reproduction and recruitment. In other words, colonies may be separated by too much distance for successful sexual reproduction to occur. Fragmentation and degradation of settlement habitat clearly exacerbates this problem.

Therefore, the key conservation objective of designated Acropora critical habitat is to increase the potential for sexual and asexual reproduction to be successful, which in turn facilitates increases in the species' abundance, distribution, and genetic diversity. To this end, our analysis seeks to determine whether or not the proposed action is likely to destroy or adversely modify designated critical habitat, in the context of the Status of Critical Habitat (Section 3.2.3), the Environmental Baseline (Section 4.3.4), the Effects of the Action (Section 5.2), and Cumulative Effects (Section 7). Ultimately, we seek to determine if critical habitat would remain functional to serve the intended conservation role for the species with the implementation of the proposed action, or whether the conservation function and value of critical habitat is appreciably diminished through alterations to the physical or biological features essential to the conservation of a species. The first step in this analysis is to evaluate the project's expected effects on the species' ability to meet identified recovery objectives relevant to the key conservation objective of critical habitat, given the effects of the proposed action.

The most directly relevant recovery objective in the *Acropora* Recovery Plan[27] related to the impacts of the Amalago Bay project on critical habitat is Criterion 6, and it applies to both elkhorn and staghorn corals:

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[27] Acropora Recovery Plan

Criterion 6: Loss of Recruitment Habitat (Listing Factor A)
Abundance (Criterion 1) addresses the threat of Loss of Recruitment Habitat because the criterion specifies the amount of habitat occupied by the two species. If [Abundance] Criterion 1 is met, then this threat is sufficiently abated;

Or

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

As indicated below, Abundance Criterion 1 is not expected to be met, so this analysis focuses on the proposed action's effects on the second, alternative prong of Criterion 6. The proposed action is expected to eliminate 33 acres of the essential feature (consolidated reef substrate). Although they can be found at greater depths, especially in back reef areas, elkhorn corals are predominantly found in 5 m or less of depth in St. Croix. The majority of the 33 acre habitat that will be lost is 5 m or less in depth. As discussed above, the Fredricksted Reef System is the only reef habitat on the west end of St. Croix, and within this system, we estimate that only 723 acres of essential feature can be found in the 0-5 m depth range. The loss of 33 acres thus represents a 4.5% (33 ac divided by 732 ac of essential feature in 5 m or less times 100) reduction in reef habitat in 5 m or less on a broad reef scale in the Frederiksted Reef System. However, much of this 732 acres is already impacted by development, especially in the southern portion of the unit but also north of the immediate project area. Portions of the nearshore Frederiksted Reef System were removed for the construction of the pier and subsequent dredging operations and pier expansion projects. The offshore portion of the reef in this area has been impacted by large vessel anchoring associated with the use of the pier (Toller 2005). Portions of nearshore and offshore reef in the vicinity of Frederiksted have also been affected by land-based sources of pollution from the developed area of Frederiksted and associated declines in water quality (Toller 2005, Smith et al. 2011b). To the north of the Amalago Bay project, elkhorn colonies in the 0-6 m depth zone in Butler Bay and Ham's Bluff exhibit 50% live tissue on each colony, which is lower than other sites around St. Croix. Smith et al. (2014) heorized that this lower tissue coverage was due to chronic impacts associated with land-based sources of pollution, including erosion of the road in this area due in part to a quarry operation near Ham's Bay. Recent complaints residents in the area of the quarry have made to NMFS indicate that sediment loading from the quarry has increased as quarry operations have expanded, resulting in the burial of consolidated reef habitat in nearshore portions of Ham's Bay (C. Johnson, private citizen, pers. comm. to L. Carrubba, NMFS, January 20, 2017).

On a regional scale, sedimentation rates around USVI have increased 1-2 orders of magnitude over the last 15-25 years. Recent studies from the USVI have found that sediment levels as low as 3 mg per cm^2 per day can cause large increases in the proportion of corals experiencing impairment, partial mortality, and bleaching if sediment is terrigenous in nature (Smith et al. 2013). The majority of nearshore waters around USVI exhibit sedimentation rates of at least 10 mg per cm^2 per day, due to development, indicating that the majority of nearshore hard bottoms and reefs around USVI are impacted by sedimentation that is deleterious to coral survival and recovery (Smith et al. 2008). Macroalgal cover affecting the essential feature has also increased throughout USVI (Rogers et al. 2008b) due in part to increases in nutrient concentrations (Smith et al. 2001). In addition, the 2005 bleaching event resulted in at least 50% decrease in coral cover around USVI in depths less than 25 m. The maximum recovery observed from this bleaching and coral mortality was 12% by 2011. The lack of recovery and return of coral cover leads to increased macroalgal cover.

The section of the Frederiksted Reef System where the action area is located is relatively free of development with good water quality, which is why Smith et al. (2014) concluded that the nearshore hard bottom habitats and shelf edge reefs along the west coast of St. Croix will play an important role in the recovery of elkhorn and staghorn corals. The loss of this 33 acres will fragment an area with little sedimentation impact, unlike much of the St. Croix unit and the greater USVI.

Based on the current information, the essential feature in the St. Croix unit generally has been significantly affected by development, sedimentation and increased macroalgal cover. Recent hurricanes, particularly Hurricane Maria in September 2017 likely led to impacts related to breakage and input of land-based pollutants due to the heavy rainfall and flooding as a result of the 2017 hurricanes. However, as discussed in Section 4.1.1, surveys completed to date in coral areas around Puerto Rico and the USVI indicate that many coral habitats did not require triage or interventions because storm impacts were not significant. None of the 3 sites surveyed in St. Croix required intervention in the form of restoration, meaning impacts were small and natural recovery of any damage that did occur as a result of the hurricanes is expected. Reports from surveys of coral areas following the 2017 hurricanes in South Florida and the U.S. Caribbean did not find significant impacts due to sediment plumes but instead due largely to debris, including grounded vessels. The proposed action will cause a permanent loss of 33 acres of the essential feature in 0-5 m and chronic episodic degradation of another 77 acres in deeper water and impede natural recovery from any recent hurricane impacts. The loss of 33 acres from the project would comprise about one half a percent of total hardbottom for the entire island of St. Croix and a 4.5% loss of hardbottom in the 0-5 m depth range important to elkhorn coral, from the Frederiksted Reef System. Actual percent losses of the essential feature are higher, given the impacted baseline of the hardbottom around St. Croix from sedimentation (especially terrigenous sediments) and macroalgal cover. Based on this we believe that the proposed action will appreciably reduce the St. Croix unit's ability to reach recovery Criterion 6 (the unit will not have at least 40 percent of consolidated reef substrate in 1-20 m depth within the forereef zone free of sediment and macoalgal cover) for elkhorn coral. Staghorn coral are predominantly distributed deeper than 5 m, so the project's primary impacts on this species' habitat is in the 77 acre area (Area B, Figure 5) of 5-20 m deep waters where we expect episodic impacts to the essential feature. We have determined that the essential feature in the 77 acre area will remain functional despite the episodic effects of the proposed action. Staghorn coral colonies have been noted on the shelf edge in this area. The proposed action will not add to the fragmentation of this deeper habitat.

The effects of the proposed action on acroporid recruitment habitat will also affect the essential feature's ability to support recovery criteria 1 and 3. These objectives encompass recruitment and abundance increases that are the key conservation objective for designated critical habitat.

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

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

Criterion 3: Recruitment (for elkhorn and staghorn**)**
Observe recruitment rates necessary to achieve Criteria 1 and 2 (Genotypic Diversity) over approximately 20 years;
and
Observe effective sexual recruitment (i.e., establishment of new larval derived colonies and survival to sexual maturity) in each species' population across their geographic range.

The proposed action will result in the loss of 33 acres of essential feature in depths of 5 m or less, will further fragment hardbottom habitat on the west end of St. Croix, and will degrade 77 acres of essential feature in deeper waters, in an area identified as having an important role in the recovery of elkhorn coral. McLaughlin et al. (2002) found that when distributions of coral species become isolated because of habitat loss, populations become more vulnerable to climate change and other threats. The loss of habitat patches will affect the availability of areas for coral larvae to settle. Information on current movement from the Caribbean Coastal Ocean Observing System indicates that wind-driven transport to the southeast dominates and tidal currents move material on- and offshore in the project area. This means that overall we expect larval and fragment transport from the Sprat Hole area southward into the project area. Larvae are only viable for a short time so larger distances between areas of suitable habitat for elkhorn and staghorn corals make settlement and recruitment less likely. In addition, elkhorn and staghorn corals' primary mode of reproduction in the USVI is through fragmentation. A branch of elkhorn or staghorn coral may be carried by waves and currents away from the parent colony, and fragments cleaved from the colony may grow into new colonies (Highsmith et al. 1980, Bak and Criens 1982, Highsmith 1982, Rogers et al. 1982). Genetically identical clones have been found separated by distances that range from 0.1 to 100 m (0.3 to 328 ft), but usually less than 30 m (98 ft) (Baums et al. 2006a). The horizontal length of area A is 4,000 ft; therefore, the loss of this area will make reproduction more difficult as larvae and fragments will have to survive a further travel distance, assuming some currents will transport them beyond the project area. However, recruitment habitat in areas south of the project area is currently degraded and fragmented due to construction, development, sedimentation, macroalgal cover, and vessel and recreational use impacts.

The impacts of the project will cause a significant decrease and possibly eliminate the ability of elkhorn coral north of the action area to successfully reproduce, particularly through sexual reproduction, and expand their population to the south, given the permanent loss of the 33 acres of essential feature. As discussed above, the loss of 33 acres from the project would comprise about one half a percent of hardbottom containing the essential feature for the entire island of St. Croix (St. Croix unit); however, it comprises a 4.5% loss from the Frederiksted Reef System, in the 0-5 m range important to elkhorn coral. Actual loss of habitat is likely to be higher, given the impacted baseline of the hardbottom around St. Croix from sedimentation (especially terrigenous sediments) and macroalgae in areas with residential, commercial, and industrial development. The proposed action will reduce areas for effective recruitment and population growth due to lack of settlement habitat, thereby reducing the chances of achieving recover criterion 3.

The effects of sedimentation and macroalgal growth on acroporid recruitment habitat discussed above and in our analysis of Criterion 6, indicates the second prong of abundance Criterion 1 for elkhorn coral is clearly less likely to be met as a result of the proposed action (i.e., live elkhorn coral benthic cover will be less likely to achieve 60 percent for coral populations on the west end of St. Croix). Loss of suitable recruitment habitat in the 0-5 m range will also decrease the likelihood that elkhorn corals will develop thickets over 10 percent of consolidated reef habitat; as discussed above, elkhorn do not generally form thickets below 5 m. Further, the impacts of the project in increasing the fragmentation of suitable settling substrate and increasing the patchiness of reef habitat, will make it difficult for elkhorn to attain the recovery densities of thickets in Criterion 1; observed elkhorn colony densities in USVI are currently already much below the recovery density. Therefore, we believe that the proposed action will appreciably reduce the chances of the St. Croix unit achieving recovery Criteria 1 and 3 for elkhorn coral.

Criteria 1 for staghorn coral calls for thickets present throughout approximately 5 percent of consolidated reef habitat in 5 to 20 m water depth within the forereef zone or live staghorn coral benthic cover of approximately 25 percent. The proposed action will cause the loss of 33 acres of essential feature in the 0-5 m range and episodic impacts to an additional 77 acres where waters depths of 5-20 m can be found. We have determined that the essential feature in the 77 acre area will be degraded but will remain functional due to the episodic effects of the proposed action. Staghorn coral colonies have been noted on the shelf edge in this area. The proposed action will not contribute to further fragmentation of this deeper habitat and therefore, it will not have an appreciable effect on recruitment and population growth due to habitat limitation. Therefore, the proposed action will not appreciably reduce the chances of the St. Croix unit achieving recovery Criterion 1 and 3 for staghorn coral.

Based on the above, we conclude that due to the effects of the proposed action the conservation value of designated critical habitat in the St. Croix unit will be appreciably diminished for elkhorn coral. We also conclude that these same effects will not appreciably diminish the conservation value of designated critical habitat in the St. Croix unit for staghorn coral.

Whether the effects of the action will appreciably diminish the conservation value of critical habitat depends on the impacts on designated critical habitat as a whole, not just in the area where the action takes place. The question we must ask is whether the adverse effects in that one part of the critical habitat will diminish the conservation value of the critical habitat overall in such a manner that we can discern a difference in the recovery prospects of the species due to the effects of the project. For example, if we conclude that the effects of the proposed action on designated critical habitat will delay recovery, or make recovery more difficult or less likely, we will conclude the effects of the project will appreciably diminish the value of the critical habitat for the conservation of the species, and thus the project is likely to destroy or adversely modify designated critical habitat.

In the status of the species section, we document that there has been a significant decline of elkhorn coral throughout its range, with recent population stability at low percent coverage. We also concluded that absolute abundance is at least hundreds of thousands of colonies, but likely to decrease in the future with projected increases in threats. The above analysis has shown that the proposed action combined with increasing sedimentation and macroalgal cover in St. Croix has appreciably diminished this unit's conservation value. The critical habitat designation for elkhorn and staghorn corals identified 4 units within the jurisdiction of the United States where the physical feature essential to the species' conservation can be protected from destruction or adverse modification: Florida, Puerto Rico, St. Thomas/St. John, and St. Croix. Given the extremely low current abundance of elkhorn coral and characteristics of its sexual reproduction (e.g., limited success over long ranges), we determined that protecting the essential feature throughout the species' range and throughout each of the four specific areas is extremely important for conservation of the species. As discussed above, the best evidence of recovery would come from scientific evidence showing an increase in the overall amount of living tissue of this species, growth of existing colonies, and an increase in the number of small corals arising from sexual recruitment. None of these trends are currently observed or expected to be promoted by the proposed action. Thus, recovery of the species on St. Croix will be delayed and more difficult as a result of the proposed action. We believe the appreciable reduction in one of only four units' conservation value for elkhorn coral and the adverse impacts on the species' recovery at the island level represent an appreciable reduction in the designated critical habitat's conservation value rangewide. Therefore, we believe that the proposed action will destroy or adversely modify designated *Acropora* critical habitat for elkhorn coral.

8 JEOPARDY ANALYSIS

This section considers the likelihood that the proposed action will jeopardize the continued existence of elkhorn, staghorn, and lobed star and mountainous star corals, and the South Atlantic and North Atlantic DPSs of green sea turtles, and leatherback and hawksbill sea turtles in the wild. To *jeopardize the continued existence of* is defined as "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). The Effects of the Action section (Section 5.0) describes the effects resulting from the proposed action on green (North and South Atlantic DPS), leatherback, and hawksbill sea turtles; and elkhorn, staghorn, lobed star and mountainous star corals. Sections 4.0 and 6.0 inform the context of these effects, by considering the environmental baseline and cumulative effects relevant to the action area of the proposed project. The following jeopardy analysis first considers the effects of the action to determine if we would reasonably expect the action to result in reductions in reproduction, numbers, or distribution of these listed species. The analysis next considers whether any such reduction would, in turn, result in an appreciable reduction in the likelihood of survival and recovery of green, leatherback, and hawksbill sea turtles, and elkhorn, staghorn, and lobed star and mountainous star corals in the wild.

In the following analyses, we find that some reduction in numbers and reproduction is expected for elkhorn, staghorn, lobed star and mountainous star corals as a result of the proposed Amalago Bay development. We also find that some reduction in numbers and reproduction is expected for green, leatherback, and hawksbill sea turtles as a result of anticipated take of hatchlings, juveniles, and adults of these species due to the project.

8.1 Elkhorn Corals

As noted in Section 5.2, elkhorn coral is expected to be adversely affected by the proposed action, in particular due to the loss of future recruitment and growth habitat (designated critical habitat) for elkhorn coral planulae and recruits due to the chronic effects of land-based sediments and contaminants from the project. The 33-acre area of critical habitat that is expected to be lost should be able to support corals at a level that promotes recovery. As discussed below, the recovery plan calls for 10% cover of consolidated hardbottm with elkhorn coral thickets. Using the numbers in the recovery plan this area should be able to support (10% of 33 acres = 3.3 acres or 13,354.63 m^2 x 0.25 colonies per m^2) 3,337 colonies. Therefore, this project may lead to the potential loss of 3,337 future elkhorn colonies.

Elkhorn coral was first listed as threatened under the ESA in May 2006. In December 2012, NMFS proposed changing its status from threatened to endangered, but in September 2014, NMFS determined that elkhorn coral should remain listed as threatened. The species has undergone a substantial population decline and decreases in occurrence to low levels of coverage throughout its range. Elkhorn coral is highly susceptible to a number of threats, and cumulative and synergistic effects of multiple threats are likely to exacerbate vulnerability to extinction. Localized mortality events have continued to occur, but percent benthic cover and proportion of reefs where elkhorn coral is dominant have remained stable over its range since the mid-1980s. The species retains a large number of islands and environments in its range, but its vulnerability to extinction is exacerbated because elkhorn coral's distribution is limited to an area with high localized human impacts and predicted increasing threats. The species' abundance is at least hundreds of thousands of colonies, but likely to decrease in the future with increasing threats. Elkhorn coral's low sexual recruitment rates exacerbate vulnerability to extinction due to decreased ability to recover from mortality events when all colonies at a site are extirpated, but its fast growth rates and propensity for formation of clones through asexual fragmentation enables it to expand between rare events of sexual recruitment and increases its potential for local recovery from mortality events, thus moderating vulnerability to extinction.

Existing elkhorn coral colonies are not expected to be lost as a result of this project so there will be no direct reduction in numbers of colonies. NMFS does expect that the chronic impacts to nearshore hard bottom habitat immediately adjacent to the Amalago Bay project will make 33 ac of future settlement habitat unavailable, accounting for the loss of up to 3,337 future elkhorn colonies. This loss will reduce future reproduction potential by eliminating settlement habitat for larvae and recruits and any possible reproduction of a possible 3,337 colonies. The loss will also make the survival of elkhorn coral larvae and recruits settling in the action area from areas such as Sprat hole less likely due to the need for larvae and recruits to travel further (fragments can be carried by waves and current) to encounter appropriate settlement habitat beyond the project area. Although larvae can travel large distances, elkhorn coral's primary mode of reproduction in the USVI is through fragmentation. A branch of elkhorn coral may be carried by waves and currents away from the parent colony, and fragments cleaved from the colony may grow into new colonies (Highsmith et al. 1980, Bak and Criens 1982, Highsmith 1982, Rogers et al. 1982). Genetically identical clones have been found separated by distances that range from 0.1 to 100 m (0.3 to 328 ft), but usually less than 30 m (98 ft) (Baums et al. 2006a). Thus, the loss of this 33 acres from the Frederiksted Reef System will lead to a locally significant gap in appropriate settlement habitat (areas for successful settlement and attachment), but we do not expect the proposed action to alter the larger geographic range for the species, and we do not expect that the proposed action will result in a reduction in the overall distribution of the species.

Whether the expected reduction in reproduction of the species would appreciably reduce its likelihood of survival depends on the probable effect the changes in reproduction would have relative to the current population levels and trends. Based on best available population estimates there are at least hundreds of thousands of elkhorn coral colonies present in both the Florida Keys and St. Croix, U.S. Virgin Islands. Absolute abundance is higher than estimates from these locations alone given the presence of this species in many other locations throughout its range. In the status of the species section we conclude there has been a significant decline of elkhorn coral throughout its range, with recent population stability at low percent coverage. We also conclude that abundance is likely to decrease in the future with increasing threats.

Elkhorn coral has low sexual recruitment rates, meaning that genetic heterogeneity is low. However, its fast growth rates and propensity for formation of clones through asexual fragmentation enables it to expand between rare events of sexual recruitment and increases its potential for local recovery from mortality events, thus moderating vulnerability to extinction. Also, given elkhorn coral's estimated abundance, (hundreds of thousands of colonies throughout Florida and the USVI) the loss of the reproductive potential represented by the loss of 3,337 future colonies in the action area will not measurably impact the species' abundance in USVI or throughout the species' range. Therefore, we believe the potential loss of the reproductive potential afforded by the loss of 33 acres of habitat due to the chronic effects of land-based pollutants will not appreciably reduce elkhorn coral's likelihood of survival in the wild.

Now we evaluate whether the expected reduction in reproduction will appreciably reduce the likelihood of the species' recovery in the wild. A recovery plan for staghorn and elkhorn was published March 5, 2015. The recovery plan notes that elkhorn and staghorn corals continue to decline and are at only a small percentage of their abundance throughout their ranges. The recovery plan outlines a recovery strategy for the species: Elkhorn and staghorn coral populations should be large enough so that successfully reproducing individuals comprise numerous populations across the historical ranges of these species and are large enough to protect their genetic diversity and maintain their ecosystem functions. Threats to these species and their habitat must be sufficiently abated to ensure a high probability of survival into the future. The recovery plan established 3 recovery criteria associated with the objective of ensuring population viability and 7 recovery criteria associated with the objective of eliminating or sufficiently abating global, regional, and local threats that contribute to the species status. The best available information indicates that all recovery objectives must be met for elkhorn and staghorn corals achieve recovery. The most relevant criteria to the impacts expected from the Amalago Bay project include:

> ***Criterion 1: Abundance***
> **Elkhorn coral:** Thickets are present throughout approximately 10 percent of consolidated reef habitat in 1 to 5 m water depth within the forereef zone. Thickets are defined as either a) colonies ≥ 1 m diameter in size at a density of 0.25 colonies per m^2 or b) live elkhorn coral benthic cover of approximately 60 percent. Populations with these characteristics should be present throughout the range and maintained for 20 years.
>
> ***Criterion 3: Recruitment***
> Observe recruitment rates necessary to achieve Criteria 1 and 2 (Genotypic Diversity) over approximately 20 years;
> and
> Observe effective sexual recruitment (i.e., establishment of new larval derived colonies and survival to sexual maturity) in each species' population across their geographic range.
>
> ***Criterion 6: Loss of Recruitment Habitat (Listing Factor A)***
> Abundance (Criterion 1 above) addresses the threat of Loss of Recruitment Habitat because the criterion specifies the amount of habitat occupied by the two species. If Criterion 1 is met, then this threat is sufficiently abated;
> or
> Throughout the ranges of these two species, at least 40 percent of the consolidated reef substrate in 1-20 m depth within the forereef zone remains free of sediment and macroalgal cover as measured on a broad reef to regional spatial scale.

In our Destruction and Adverse Modification analysis for Acropora designated critical habitat (Section 7) we determined that the proposed action would destroy or adversely modify Acropora designated critical habitat for elkhorn coral, due to an appreciable reduction in the conservation value of the designated critical habitat and its impacts on recovery. Here we determine whether the proposed action will appreciably reduce the likelihood of the species' recovery in the wild. Our analysis of the project's impacts on critical habitat concluded that the proposed action would affect the essential feature of designated critical habitat to an extent that would delay recovery and make recovery more difficult for elkhorn coral, based on the project's effects on this species' ability to meet recovery criteria 1, 3 and 6. We also conclude that these impacts to the species' recovery prospects constitute an appreciable reduction in the likelihood of elkhorn coral's recovery. Not all cases of a project resulting in destruction and adverse modification of critical habitat would also result in an appreciable reduction in a species' likelihood of recovery in the wild. In this case the designated critical habitat is only a relatively small portion of elkhorn coral's overall range; however, we believe the appreciable reduction in the prospects of recovery at the island level represents an appreciable reduction in the likelihood of recovery rangewide for this species. The recovery criteria for elkhorn coral require levels of recruitment and abundance to be achieved throughout the species' range and throughout areas of consolidated hard bottom habitat. As we have determined, the proposed action will appreciably reduce the chances of the

St. Croix unit achieving recovery Criterion 1 and 3 for elkhorn coral, we also conclude that the project will appreciably reduce the likelihood of elkhorn coral recovering in the wild.

As previously discussed, to jeopardize means to cause an appreciable reduction in both the survival and recovery of a species. We determined the proposed action will not appreciably reduce the likelihood of survival. In rare circumstances, an appreciable reduction in the likelihood of recovery alone can jeopardize a species' continued existence. In our judgment such a circumstance would involve severe impacts to the prospects for recovery, if not preclusion of recovery, to an extent that measurably increases a species' risk of extinction. While the proposed action will appreciably reduce the likelihood of recovery of elkhorn coral in the wild, the impacts to recovery alone will not jeopardize the species' continued existence. As stated above, elkhorn coral has fast growth rates and a propensity for formation of clones through asexual fragmentation enabling it to expand between rare events of sexual recruitment and increases its potential for local recovery from mortality events, thus moderating vulnerability to extinction. Also, elkhorn coral has a sufficiently large population (hundreds of thousands of colonies throughout Florida and the USVI alone). We do not believe that the appreciable reduction in recovery expected to result from the proposed action will measurably increase the extinction risk of the species. Based on this we conclude that the proposed action is not likely to jeopardize the continued existence of elkhorn coral in the wild.

8.2 Staghorn Corals

As noted in Section 5.2, staghorn coral is expected to be adversely affected by the proposed action, in particular due to impacts associated with the transport of land-based pollutants from the Amalago Bay project to 77 acres of reefs and colonized hard bottoms in waters between 5 and 20 m deep in the action area. We expect these impacts to degrade the health and reduce the reproductive output of staghorn colonies. As we discuss in Section 5.2.3, based on NOAA NCRMP 2015 data, we estimate that up to 4,051 colonies of staghorn coral within the 77 ac area could be affected by the proposed action. The 2 staghorn coral colonies observed during NCRMP surveys in 2015 had diameters of 50 and 73 cm. As discussed in the status of the species section for staghorn coral (Section 3.2.3), staghorn corals with branch lengths of 17 cm are sexually mature. Thus, because the staghorn coral colonies observed during the NCRMP surveys were both sexually mature, we assume that all of the colonies of staghorn coral that may be within the 77 ac area are sexually mature.

Staghorn coral was first listed as threatened under the ESA in May 2006. In December 2012, NMFS proposed changing its status from threatened to endangered, but in September 2014, NMFS determined that staghorn coral should remain listed as threatened. The species has undergone a substantial population decline and decreases in occurrence to low levels of coverage throughout its range. Staghorn coral is highly susceptible to a number of threats, and cumulative and synergistic effects of multiple threats are likely to exacerbate vulnerability to extinction. Localized mortality events have continued to occur, but percent benthic cover and proportion of reefs where staghorn coral is dominant have remained stable over its range since the mid-1980s. The species retains a large number of islands and environments in its range, but its vulnerability to extinction is exacerbated because staghorn coral's distribution is limited to an area with high localized human impacts and predicted increasing threats. Staghorn corals occupy a broad range of depths and multiple, heterogeneous habitat types, which moderates the species' vulnerability to extinction over the foreseeable future. The species' abundance is at least tens of millions of colonies, but likely to decrease in the future with increasing threats. Staghorn coral's low sexual recruitment rates exacerbate vulnerability to extinction due to decreased ability to recover from mortality events when all colonies at a site are extirpated, but its fast growth rates and propensity for formation of clones through asexual fragmentation enables it to expand between rare events of sexual recruitment and increases its potential for local recovery from mortality events, thus moderating vulnerability to extinction.

Staghorn corals are identified as being occasional in distribution in the Frederiksted Reef System (Toller 2005) and are present in depths ranging from 5-30 m, on average (Smith et al. 2014). Smith et al. (2014) reported that the staghorn coral population along the north and east coasts of St. Croix, including BIRNM was 21,439 colonies, but stated that this was a low estimate because they did not survey areas deeper than 18 m. In the action area staghorn coral can be found on the shelf edge in area B, where they are likely to experience episodic, less intense sedimentation and contamination impacts.

Based on the above information, we do not expect the proposed action to alter the geographic range for the species, thus we do not expect that the proposed action will result in a reduction in distribution of the species.

Whether the reduction in reproduction of the species due to the episodic impacts to Area B would appreciably reduce its likelihood of survival depends on the probable effect the changes in reproduction would have relative to current population. Staghorn corals occur throughout the Caribbean basin and the corals in the action area account for a very small portion of the total numbers of or area occupied by staghorn coral. The species' absolute abundance is at least tens of millions of colonies, based on estimates from only two locations. Given this, the loss of reproductive potential represented by the loss of up to 4,051 colonies in the action area will not measurably impact the species' abundance in USVI or throughout the species' range. Therefore, we believe the potential lost reproduction from the episodic effects of land-based pollutants will not have any measurable effect on the overall populations and is not likely to reduce the species likelihood of survival in the wild.

Now we evaluate whether the expected reduction in reproduction will appreciably reduce the likelihood of the species' recovery in the wild. A recovery plan for staghorn and elkhorn was published March 5, 2015. The recovery plan and its recovery strategy are discussed above in Section 8.1, discussing elkhorn coral. The recovery plan includes a slightly different Criterion 1 for staghorn coral and the same Criteria 3 and 6 discussed for elkhorn coral, that are relevant to the impacts of the proposed action:

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

In our Destruction and Adverse Modification analysis for Acropora designated critical habitat (Section 7) we determined that the proposed action would destroy or adversely modify Acropora designated critical habitat for elkhorn coral due to an appreciable reduction in the conservation value of the designated critical habitat. Our analysis looked at the effects of the proposed action in relation to these recovery criteria. We determined that the proposed action will not appreciably reduce the chances of the St. Croix unit achieving recovery Criterion 1, 3 or 6 for staghorn coral, and thus there would be no discernible negative impact on staghorn coral's prospects for recovery due to the impacts of the proposed action. Based on the analysis in Section 7 (incorporated here by reference) we determine that the proposed action will not cause an appreciable reduction in staghorn coral's ability to recover. Therefore, we determine that the proposed action is not likely to jeopardize the continued existence of staghorn coral.

8.3 Lobed Star and Mountainous Star Corals

Lobed and mountainous star corals were only recently listed as threatened, and we do not have an extensive consultation history. We can, however, assess the effects of the proposed action on lobed star and mountainous star coral populations in the context of our knowledge of the status of each species, their environmental baselines, and the extinction risk analyses in the listing rule. The final listing rule identifies these species' abundance, life history characteristics, and depth distribution, threat vulnerabilities and characteristics that moderate extinction risk. Combined with spatial variability in ocean warming and acidification across the species' ranges, these species' extinction risk is moderated due to their absolute abundances and their habitat heterogeneity, because the threats affecting them are non-uniform, and there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time. These species can be found in wide depth ranges, from 0.5 m to 90 m.

The proposed action will result in a reduction in numbers of lobed star and mountainous star coral colonies. We anticipate that approximately 1,151,064 lobed star and 575,537 mountainous star coral colonies will be lost as a result of impacts from the proposed action. The proposed action will also result in a reduction of reproduction due to the loss of the reproductive potential of the lost colonies and the loss of 33 acres of hardbottom settlement and recruitment habitat. Approximately 2,955,250 lobed star corals and 1,477,625 mountainous star corals colonies at the shelf edge are expected to experience non-lethal sediment stress leading to reduced reproduction from episodic sedimentation impacts.

Even given the large numbers of colonies expected to be lost, the proposed action will not affect either species' current geographic range. Toller (2005) and (Fore et al. 2006) found these species, lobed star coral in particular, to be dominant along the entire west coast of St. Croix. The mortality caused by the proposed action would not result in changes to the overall distribution pattern of the species in St. Croix and the species will still be common throughout the Frederiksted Reef System. The species' distribution throughout the wider Caribbean will not be impacted. Therefore, we believe that the proposed action will not result in a reduction in the distribution of lobed star and mountainous star corals.

Whether the reduction in numbers and reproduction of these species would appreciably reduce their likelihoods of survival depends on the probable effect these changes would have relative to current population levels and trends. The distribution and cover of lobed star and mountainous star corals throughout the Frederiksted Reef System, even after the loss of coral cover associated with the 2005 bleaching event (Smith et al. 2011a), show the common abundance of these species in areas greater than 10 ft in depth, including along the shelf edge. Lobed star and mountainous star corals are still among the most dominant hard coral species in the USVI based on more recent benthic surveys and the EPA survey conducted around all of St. Croix (Fore et al. 2006) done after the mass bleaching. Lobed star coral's absolute abundance has been estimated as at least tens of millions of colonies in the Florida Keys and Dry Tortugas combined. Mountainous star coral's absolute population abundance has been estimated as at least tens of millions of colonies in each of several locations including the Florida Keys, Dry Tortugas, and the U.S. Virgin Islands. Both species' rangewide abundances are higher than the estimate from these three locations due to the occurrence of the species in many other areas throughout their ranges. With tens of millions of colonies of each species in one or several locations, the loss of colonies due to the proposed action would represent a small percentage of the total populations rangewide.

The loss of colonies and habitat for larval settlement due to the proposed action will cause a loss of reproduction in part of the action area (shallow Area A). Although much of the hardbottom habitat in St. Croix is fragmented and degraded there are still areas of colonized hard bottom and patch reefs (including in area B) along the west coast of St. Croix outside the project boundaries. The common abundance of these species on this reef system allows for greater success during sexual reproduction because more larvae have opportunities to encounter suitable settlement habitat because the parent colonies are not so highly isolated as are elkhorn coral colonies in particular. In addition, the wide depth ranges that these species can inhabit (.5 m to 90 m) means that corals and larvae will be found in deeper waters that will not be adversely affected by the proposed action. Based on this we believe that their ability to expand to uncolonized hard bottom will not be significantly affected. Therefore, we do not believe that the loss of reproductive colonies and larval settlement habitat due to the proposed action will affect the overall reproduction by these corals in the western reef system of St. Croix nor throughout their range. Based on this information we do not believe the effects of the proposed action will appreciably reduce the likelihood of lobed star and mountainous star corals' survival in the wild.

As stated above, these species were only recently listed and at this time there is no recovery plan for these species. However, NMFS has developed a recovery outline for these species (available on our webpage at http://sero.nmfs.noaa.gov/protected_resources/coral/index.html). The outline is meant to serve as an interim guidance document to direct recovery efforts, including recovery planning, until a full recovery plan is developed and approved. A preliminary strategy for recovery of the species is presented, as are recommended high priority actions to stabilize and recover the species. The outline is intended to guide recovery-planning efforts and provide information for ESA Section 7 consultations. The Summary Assessment in the recovery outline concludes that overall, available data indicate *Orbicella* coral populations are on the decline and that recovery will depend on successful sexual reproduction and reducing mortality of extant populations. The key challenges will be moderating the impacts of ocean warming associated with climate change and decreasing susceptibility to disease which may be furthered through reduction of local stressors. The recovery of these species will require an ecosystem approach including habitat protection measures, a reduction in threats caused by human activity, additional research, and time. The recovery vision statement in the outline states that populations of *O.*

annularis and O. faveolata should be present across their historical ranges, with populations large enough and genetically diverse enough to support successful reproduction and recovery from mortality events and dense enough to maintain ecosystem function. Given that many of the important threats to the recovery of *O. annularis* and *O. faveolata* are not directly manageable, the recovery strategy must pursue actions both in the short and long term to address both global and local threats. The initial focus of the recovery action plan will be to protect extant populations and the species' habitat through reduction of threats. Specific actions identified for early in the recovery process are reducing locally-manageable stress and mortality sources (e.g., acute sedimentation, nutrients, contaminants, over-fishing).

Therefore, to determine if proposed action will appreciably reduce the likelihood of these species' recovery we will evaluate the proposed action's impacts, if any, on the key elements of the recovery outline discussed above. These species' life history characteristics of large colony size and long life span have enabled them to remain relatively persistent despite slow growth and low recruitment rates, thus moderating vulnerability to extinction. However, the buffering capacity of these life history characteristics is expected to decrease as colonies shift to smaller size classes as has been observed in locations throughout their ranges. The proposed action will not affect these life history vulnerabilities. The listing rule states that the major threats faced by these corals are high vulnerability to ocean warming, disease, acidification, sedimentation, and nutrient enrichment, and the latter has been shown to exacerbate disease. The proposed action will not increase the magnitude of or the species' vulnerability to ocean warming, disease, or acidification; however, the proposed action will cause an increase in sedimentation and nutrient enrichment in the action area. The vast majority of these effects will be in the 33-acre shallow area and will cause the loss of all of the corals in this area (see above for the actual numbers), and the loss of the reproductive potential of the lost colonies and the loss of larval attachment substrate. In addition, reproductive potential of both species will be diminished at the shelf edge. However, the area affected is a small portion of these species' ranges and as stated in the listing rule, the absolute abundance and habitat heterogeneity of these species allows for variation in the responses of individuals to threats to play a role in moderating vulnerability to extinction. The proposed action will cause mortality to high numbers of colonies; however, these species are still common in St. Croix and the losses will not affect overall density and distribution of the species, or impede sexual reproduction. Therefore, we believe that the increased sedimentation and nutrient enrichment resulting from the proposed action will not increase the magnitude of these threats rangewide to levels that will appreciably reduce these species' ability to recover in the wild. We conclude the proposed action is not likely to jeopardize the continued existence of lobed star coral and mountainous star coral.

8.4 Sea Turtles

All sea turtle life stages are important to the survival and recovery of the species; however, it is important to note that individuals of one life stage are not equivalent to those of other life stages. For example, the take of male juveniles may affect survivorship and recruitment rates into the reproductive population in any given year, and yet not significantly reduce the reproductive potential of the population. For sea turtles, a very low percent of hatchlings is typically expected to survive to reproductive age; therefore, the loss of hatchlings from a population level standpoint is not as significant with respect to the survivial and recovery of the species as the loss of older life stages. The death of mature, breeding females can have an immediate effect on the reproductive rate of the species. Sublethal effects on adult females may also reduce reproduction by hindering foraging success, as sufficient energy reserves are probably necessary for producing multiple clutches of eggs in a breeding year. Different age classes may experience varying rates of mortality and resilience. However, based on recent sea turtle population modeling efforts, the reduction of mortality in early age classes is likely to positively affect population dynamics by increasing cohort size (Mazaris et al. 2005). Thus, hatchling protection could act as a short-term preventive factor against abrupt population decline, providing time for population recovery (Mazaris et al. 2005) if other population stressors are addressed. Population modeling shows that the probability of first-year survival (combined as hatchling and hatchling emergence success) is significantly lower than survival probabilities of other life stages. Therefore, even minor changes in the survival of the first year cohort affects the number moving to the next age class compared to other year cohorts (Mazaris et al. 2005). In addition, because existing breeding populations may be 2 orders of magnitude or more below pre-exploitation levels, human perturbations may induce life-history changes that alter results of sea turtle studies (Bowen et al. 2007). (Bowen et al. 2007) found a significant correlation between nesting population size and contribution to juvenile feeding areas for hawksbill sea turtles that likely holds for green sea turtles as well. Based on our effects analysis in Section 5.1, we determined that hatchling and adult green, leatherback, and hawksbill, and juvenile green and hawksbill sea turtles are reasonably certain to suffer lethal and nonlethal take as a result of the Amalago Bay project.

8.4.1 Green Sea Turtles (North and South Atlantic DPS)

As detailed in Section 5, we estimated various non-lethal and lethal takes of green turtles in the marine environment from the proposed project. The potential take of up to 91 green turtle hatchlings per year via impacts from the construction of jetties represents a reduction in numbers. It is important to note that with insufficient information to more accurately parse out the likelihood of lethal (lethal entrapment and increased predation effects) and non-lethal (disorientation) effects we are erring on the side of the species and making the assumption that all of the takes are lethal. We are also assuming that all of the hatchlings emerging from nests on the beaches near the jetties will interact with the jetties, despite the fact that jetties are primarily oriented seaward from the shore, and hatchlings typically take a perpendicular path out to open water from the beach. Overall this results in a conservative estimate erring on the side of the species. In addition, because green sea turtles could be from the North or South Atlantic DPS with the exception of nesting females and hatchlings as discussed in Section 5, we will conduct a jeopardy analysis for each DPS that assumes all the impacts to turtles other than nesting females and hatchlings could be experienced by either DPS.

North Atlantic DPS

No reduction in the distribution of green sea turtles from the North Atlantic DPS is expected from this take as green turtles will continue to be present throughout waters of the action area.

Whether the potential reduction in numbers if take is lethal or due to impacts to reproductive output would appreciably reduce the likelihood of survival of green sea turtles from the North Atlantic DPS depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends. The North Atlantic DPS is the largest of the 11 green turtle DPSs with an estimated abundance of over 167,000 adult females from 73 nesting sites. All major nesting populations demonstrate long-term increases in abundance (Seminoff et al. 2015). Based on genetic analyses that were used to designate the North and South Atlantic DPSs, females from the North Atlantic DPS are not expected to nest in St. Croix. Therefore, there will be no loss of reproductive potential to the green sea turtle North Atlantic DPS as a result of this project.

Non-lethal take, including the effects of habitat loss (168 juveniles permanently displaced), could result in a reduction in reproduction. Such non-lethal takes can cause individuals to expend more energy seeking suitable habitat or moving around more to extract the necessary resources from the degraded habitat. As detailed in section 5.1, we estimated 2,912 juvenile green turtles have established home ranges in the nearshore waters of St. Croix. Therefore, up to 168 juveniles every year experiencing displacement would equate to about 6% of the St. Croix resident juvenile green turtles. This can result in reduced growth rates, older age to maturity, and lower lifetime fecundity. Thus, these impacts could also result in a reduction in reproduction.

Whether the reductions in numbers and reproduction of North Atlantic DPS green turtles would appreciably reduce the species' likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends. The 2014 status review for green turtles, produced in support of the DPS listing rule determined that there were over 167,000 nesting females in the North Atlantic DPS (Seminoff et al. 2015). Those estimates did not include multiple smaller sites for which nesting data were not available. The general trend for most nesting sites was stable or increasing, with overall trends showing an increase.

We believe the proposed action is not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of the North Atlantic DPS in the wild. Although the anticipated mortalities would result in an instantaneous reduction in absolute population numbers, the populations of green sea turtles in the North Atlantic DPS would not be appreciably affected. Likewise, the reduction in reproduction that could occur from the lethal and non-lethal takes would not appreciably affect reproductive output in the North Atlantic. For a population to remain stable, sea turtles must replace themselves through successful reproduction at least once over the course of their reproductive lives, and at least 1 offspring must survive to reproduce itself. If the hatchling survival rate to maturity is greater than the mortality rate of the population, the loss of breeding individuals would be exceeded through recruitment of new breeding individuals from successful reproduction of non-taken sea turtles. Because the abundance trend information for green sea turtles is increasing, we believe the anticipated takes attributed to the proposed action will not have any measurable effect on that trend.

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

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

There are no reliable estimates of the number of immature green sea turtles that inhabit coastal areas (where they come to forage) of the southeastern United States and U.S. Caribbean. Since 2000, sea turtle surveys in Culebra have resulted in the capture of 553 green sea turtles and all have been either juveniles or subadults based on size and testosterone levels suggesting Culebra is an important developmental habitat (Diez and Dam 2002). The largest remaining green turtle population in the Atlantic that provides resident and transient juveniles to the Puerto Rican (North Atlantic DPS) population (based on genetic data) is potentially threatened by the resurgence of the commercial artisanal green turtle fishery in Nicaragua (Campbell and Lagueux 2005). Nicaragua is the site of the principal feeding grounds for adult sea turtles from the Tortuguero, Costa Rica rookery (Campbell and Lagueux 2005). (Campbell and Lagueux 2005) found that survival rate estimates of females tagged at nesting beaches and juveniles and adults tagged at Nicaragua fishing sites may be too low to sustain the population. Similarly, (Troëng and Rankin 2005b) concluded that events and policy decisions in Costa Rica, Nicaragua, and Panama (the main nesting, feeding, and mating grounds for green sea turtles in the North Atlantic DPS) greatly influence survivorship. (Troëng and Rankin 2005b) found that, while protections are in place in Costa Rica and to varying degrees in the other 2 countries, the capture levels in Nicaragua are believed to be higher than ever. However, it is important to note that in the years following that research green turtle nesting in Tortuguero (and elsewhere throughout the Caribbean and Atlantic) has continued to increase, and it is likely that numbers on foraging grounds have increased similarly.

In conclusion, the anticipated lethal and non-lethal green sea turtle takes that could be from the North Atlantic DPS expected to result from the proposed action are not likely to reduce population numbers over time given current population sizes and expected recruitment. Thus, the proposed action is not likely to impede the recovery objective above and will not result in an appreciable reduction in the likelihood of green sea turtles' recovery in the wild. In conclusion, we believe that the effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of survival and recovery of green sea turtles from the North Atlantic DPS in the wild.

<u>South Atlantic DPS</u>
No reduction in the distribution of green sea turtles from the South Atlantic DPS is expected from this take as it will not affect the presence of green sea turtles around St. Croix.

Whether the potential reduction in numbers if take is lethal or due to impacts to reproductive output would appreciably reduce the likelihood of survival of green sea turtles from the South Atlantic DPS depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends. The South Atlantic DPS is large, estimated at over 63,000 nesting females, but data availability is poor with 37 of the 51 identified nesting sites not having sufficient data to estimate number of nesters or trends (Seminoff et al. 2015). While the lack of data was a concern due to increased uncertainty, the overall trend of the South Atlantic DPS was not considered to be a major concern as some of the largest nesting beaches such as Ascension Island and Aves Island in Venezuela and Galibi in Suriname appear to be increasing with others (Trindade, Brazil; Atol das Rocas, Brazil; Poilão and the rest of Guinea-Bissau) appearing to be stable. In the U.S., nesting of green sea turtles occurs in the South

Atlantic DPS on beaches of the U.S. Virgin Islands, primarily on Buck Island and Sandy Beach, St. Croix, although there are not enough data to establish a trend.

While we do not have information on green turtle hatchlings produced for all of St. Croix, the average annual green turtle hatchling production from just Buck Island from 2001-2004 was 2,269. Thus, 91 hatchlings lost would equate to approximately 4% of only the nesting found on one part of St. Croix, Buck Island. The loss relative to the overall percentage of total green turtle hatchling production throughout St. Croix would, therefore, be less than 4%. The nesting population, and thus hatchling production, of green turtles on St. Croix is relatively low compared to other nesting assemblages throughout the Caribbean and Atlantic, including the nesting assemblages of the South Atlantic DPS. This lethal take of hatchlings would also result in a future reduction in reproduction as a result of lost reproductive potential; if any of the takes are female sea turtles that would have survived other threats and reproduced in the future, this take would eliminate those females' individual contributions to future generations. For example, an adult green sea turtle can lay 1-7 clutches (usually 2-3) of eggs every 2-4 years, with 110-115 eggs/nest, of which a small percentage is expected to survive to sexual maturity.

Non-lethal take, including the effects of habitat loss (168 juveniles permanently displaced) and 1 adult female per year abandoning a nesting attempt due to onshore light visible from the nearshore waters, could result in a reduction in reproduction. Such non-lethal takes can cause individuals to expend more energy seeking suitable habitat or moving around more to extract the necessary resources from the degraded habitat. Nesting females deterred by lighting may also have a reduced reproductive output. As detailed in section 5.1, we estimated 2,912 juvenile green turtles have established home ranges in the nearshore waters of St. Croix. Therefore, up to 168 juveniles every year experiencing displacement would equate to about 6% of the St. Croix resident juvenile green turtles. This can result in reduced growth rates, older age to maturity, and lower lifetime fecundity. Thus, these impacts could also result in a reduction in reproduction.

Whether the reductions in numbers and reproduction of green turtles would appreciably reduce the species' likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends. The 2007 5-year status review for green sea turtles states that of the 7 green sea turtle nesting concentrations in the Atlantic Basin for which abundance trend information is available, all were determined to be either stable or increasing (NMFS and USFWS 2007a). Additionally, the 2014 status review for green turtles determined that there were over 63,000 nesting females in the South Atlantic DPS (Seminoff et al. 2015). This estimate did not include multiple smaller sites for which nesting data were not available. The general trend for most nesting sites in both the North and South Atlantic DPS was stable or increasing, with overall trends showing an increase.

We believe the proposed action is not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of the green sea turtle in the wild. Although the anticipated mortalities would result in an instantaneous reduction in absolute population numbers, the populations of green sea turtles in the South Atlantic DPS would not be appreciably affected. Likewise, the reduction in reproduction that could occur from the lethal and non-lethal takes would not appreciably affect reproductive output in the South Atlantic. For a population to remain stable, sea turtles must replace themselves through successful reproduction at least once over the course of their reproductive lives, and at least 1 offspring must survive to reproduce itself. If the hatchling survival rate to maturity is greater than the mortality rate of the population, the loss of breeding individuals would be exceeded through recruitment of new breeding individuals from successful reproduction of non-taken sea turtles. Since the abundance trend information for green sea turtles is increasing, we believe the anticipated takes attributed to the proposed action will not have any measurable effect on that trend.

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

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

There are no reliable estimates of the number of immature green sea turtles that inhabit coastal areas (where they come to forage) of the southeastern United States and the U.S. Caribbean. Juvenile greens from multiple rookeries frequently utilize the nearshore waters off Brazil as foraging grounds and juvenile and adult green turtles utilize foraging areas throughout the Caribbean areas of the south Atlantic based on captures in fisheries (Dow and Eckert 2007, Marcovaldi et al. 2009a, Lima et al. 2010a, López-Barrera et al. 2012). Culebra Island, which is on the border between the North and South Atlantic DPSs, is an important developmental habitat based on capture data from 2000 – 2006 of juveniles and subadults (Diez et al. 2007).

In conclusion, the anticipated lethal and non-lethal green sea turtle takes that could be from the South Atlantic DPS expected to result from the proposed action are not likely to reduce population numbers over time given current population sizes and expected recruitment. Thus, the proposed action is not likely to impede the recovery objective above and will not result in an appreciable reduction in the likelihood of green sea turtles' recovery in the wild. In conclusion, we believe that the effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of survival and recovery of green sea turtles from the South Atlantic DPS in the wild.

8.4.2 Leatherback Sea Turtles

As detailed in Section 5, we estimated various non-lethal and lethal takes of leatherback turtles in the marine environment from the proposed project. The potential take of up to 114 leatherback turtle hatchlings per year via impacts from the construction of jetties represents a reduction in numbers. As discussed for green turtles, our assessment is based on conservative assumptions to err on the side of the species.

While we do not have information on leatherback turtle hatchlings produced for all of St. Croix, the average annual leatherback turtle hatchling production from just Sandy Point from 2002-2006 was 21,966. Thus, 114 hatchlings lost would equate to approximately 0.5% of only the nesting found on one part of St. Croix, Sandy Point. The loss relative to the overall percentage of total leatherback turtle hatchling production throughout St. Croix would, therefore, be less than 0.5%. This lethal take would also result in a future reduction in reproduction as a result of lost reproductive potential; if any of the takes are female sea turtles that would have survived other threats and reproduced in the future, this take would eliminate those females' individual contributions to future generations.

Non-lethal take, including 1 adult female per year abandoning a nesting attempt due to onshore light visible from the nearshore waters, could result in a reduction in reproduction. Nesting females deterred by lighting may have a reduced reproductive output.

Given these sea turtles generally have large ranges in which they disperse, no reduction in the distribution of leatherback sea turtles is expected from the proposed action.

Whether the estimated reduction in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in number and reproduction would have relative to current population sizes and trends. The Leatherback TEWG estimates there are between 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) in the North Atlantic. Of the 5 leatherback populations or groups of populations in the North Atlantic, 3 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 2 populations, there is not enough information available on the West African population to conduct a trend analysis, and, for the Western Caribbean, a slight decline in annual population growth rate was detected (TEWG 2007b). An annual growth rate of 1.0 is considered a stable population; the growth rates of 2 nesting populations in the Western Caribbean were 0.98 and 0.96 (TEWG 2007b).

We believe the proposed action is not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of leatherback sea turtles in the wild. Although the anticipated mortality of up to 114 leatherback sea turtle hatchlings per year would result in a reduction in absolute population numbers, and the lethal and non-lethal impacts would reduce reproduction, it is not likely this 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 sea turtles unaffected by the proposed action. 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 proposed action is not likely to have any measurable effect on overall population trends.

The Atlantic recovery plan for the U.S. population of the leatherback sea turtles (NMFS and USFWS 1992b) 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.

We believe the proposed action is not likely to impede the recovery objective above and will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild. 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 469-882 nests recorded each year between 2000 and 2005. Annual growth rate in nesting was estimated to be 1.1 with a growth rate interval between 1.04 and 1.12, using nest numbers between 1978 and 2005 (NMFS and USFWS 2007e). 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 growth rate was calculated as approximately 1.10 (with an estimated interval of 1.07 to 1.13) (NMFS and USFWS 2007e). 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 growth rate was approximately 1.18 (with an estimated 95% interval of 1.1 to 1.21) (NMFS and USFWS 2007e). The numbers stayed over 500 until 2013 when they dipped to near 300; however, in 2014 they reached a record 641 nests. Thus, even with this newer information the annual growth rate of the Florida nesting population is still within the 95% confidence intervals estimated in the 2007 status review.

The potential lethal and non-lethal take of leatherback sea turtles from the proposed action is not likely to reduce population numbers over time given current population sizes and expected recruitment. Thus, the proposed action is not likely to impede the recovery objective above and will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild. In conclusion, we believe that the effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of survival and recovery of leatherback sea turtles in the wild.

8.4.3 Hawksbill Sea Turtles

As detailed in Section 5, we estimated various non-lethal and lethal takes of hawksbill turtles in the marine environment from the proposed project. The potential take of up to 1,204 hawksbill turtle hatchlings per year via impacts from the construction of jetties represents a reduction in numbers. As discussed for green sea turtles, we based this analysis on conservative assumptions that err on the side of the species.

Using 2009 DPNR nesting data (Table 4), the average number of nests in a given year based on the NPS Buck Island, USFWS Sandy Point data (Table 5), and WIMARCS nesting data (Section 4.1), we determined that there are approximately 538 nests per year on St. Croix. Using the Buck Island data from 2001-2004 we determined that about 86 hatchlings emerge per nest, multiplied by 538 nests per year, the approximate hatchling production for St. Croix is 46,268 hatchlings. Thus, 1,204 hatchlings lost would equate to approximately 2.6% of St. Croix hawksbill hatchlings. The beach within the action area has the highest number of hawksbill nests for beaches around St. Croix, surpassed only by the beaches on Buck Island. This lethal take would also result in a future reduction in reproduction as a result of lost reproductive potential; if any of the takes are female sea turtles that would have survived other threats and reproduced in the future, they would eliminate those females' individual contributions to future generations. Based on survival rates we would expect 3 of the 1,204 hatchlings to make it to adulthood, therefore the loss of 1,204 hatchlings per year would result in 3 adult turtles per year not recruiting into the population starting at approximately 7 to 8 years, the estimated age of maturity (Hawkes 2014).

Non-lethal take, including the effects of habitat loss (17 juveniles permanently displaced) and 1 adult female per year abandoning a nesting attempt due to onshore light visible from the nearshore waters, could result in a reduction in reproduction. Such non-lethal takes can cause individuals to expend more energy seeking suitable habitat or moving around more to extract the necessary resources from the degraded habitat. This can result in reduced growth rates, older age to maturity, and lower lifetime fecundity. Nesting females deterred by lighting may also have a reduced reproductive output. As detailed in section 5.1, we estimated 291 juvenile hawksbill turtles with established home ranges in the nearshore waters of St. Croix. Therefore, up to 17 juveniles every year experiencing displacement would equate to about 6% of the St. Croix resident juvenile hawksbill turtles, and these impacts could result in reductions in future reproduction.

No reduction in the distribution of hawksbill sea turtles is expected from this take, as hawksbill turtles will continue to be present throughout most waters surrounding St. Croix.

Whether the reductions in numbers and reproduction of hawksbill turtles would appreciably reduce their likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

There are currently no reliable estimates of population abundance and trends for non-nesting hawksbills at the time of this consultation; therefore, nesting beach data is currently the primary information source for evaluating trends in abundance. Mortimer and Donnelly (2008b) found that for nesting populations in the Atlantic (especially in the Insular Caribbean and Western Caribbean Mainland), 9 of the 10 sites with recent data (within past 20 years) that show nesting increases were located in the Caribbean. With increasing nesting trends in the Caribbean we believe the losses expected due to the proposed action will be replaced due to increased nest production. Therefore, we believe the reduction in numbers and reproduction will not appreciably reduce hawksbill turtle’s survival in the wild.

The Recovery Plan for the population of the hawksbill sea turtles (NMFS and USFWS 1993a) 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

- The numbers of adults, subadults, and juveniles are increasing, as evidenced by a statistically significant trend on at least five key foraging areas within Puerto Rico, U.S. Virgin Islands, and Florida.

Of the hawksbill sea turtle rookeries regularly monitored—Jumby Bay (Antigua/Barbuda), Barbados, Mona Island (Puerto Rico), and Buck Island Reef National Monument (U.S. Virgin

Islands), all show increasing trends in the annual number of nests (NMFS and USFWS 2007b). 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 loss of 1,204 hatchlings annually, the non-lethal take of 17 juveniles permanently displaced, and the abandonment of one nesting attempt each year on hawksbill sea turtles from the proposed action are not likely to reduce overall population numbers over time due to expected recruitment based on the increasing trends in nesting. With increased nesting in the Caribbean the proposed action is not expected to affect the numbers of adult females recruiting into the population nor the numbers of adults, subadults, and juveniles. Therefore, we believe the proposed action is not likely to impede the recovery objectives above and will not result in an appreciable reduction in the likelihood of hawksbill sea turtles' recovery in the wild. In conclusion, we believe that the effects associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of survival and recovery of hawksbill sea turtles in the wild.

9 CONCLUSION

NMFS has analyzed the best available data, the current status of the species and critical habitat, the environmental baseline with the understanding that recent hurricanes may have degraded the baseline, effects of the proposed action, and cumulative effects to determine whether the proposed action is likely to jeopardize the continued existence of elkhorn, staghorn, lobed star and mountainous star corals or green (North and South Atlantic DPS), leatherback, and hawksbill sea turtles, or result in the destruction or adverse modification of critical habitat for elkhorn and staghorn corals. It is our Opinion that the construction and operation of the Amalago Bay project:

- is *not* likely to jeopardize the continued existence of lobed star (*Orbicella annularis*), mountainous star (*Orbicella faveolata*) , elkhorn (*Acropora palmata*) and staghorn (*Acropora cervicornis*) corals or leatherback (*Dermochelys coriacea*), green (*Chelonia mydas*; North and South Atlantic DPS), or hawksbill (*Eretmochelys imbricata*) sea turtles;
- *is* likely to result in the destruction or adverse modification of designated critical habitat for elkhorn coral (*Acropora palmata*), which is the same critical habitat designated.for staghorn coral (*Acropora cervicornis*)

10 REASONABLE AND PRUDENT ALTERNATIVE

We have determined that the proposed action is likely to result in the destruction or adverse modification of designated critical habitat for elkhorn coral (*Acropora palmate*), which is the same critical habitat designated for staghorn coral (*Acropora cervicornis*).Therefore, we are providing an RPA to the action as proposed that the USACE and William and Punch LLC can implement that will avoid violation of ESA Section 7(a)(2) (50 CFR § 402.14). An RPA is an alternative to the action as proposed, identified during formal consultation that meets the following criteria: (1) can be implemented in a manner consistent with the intended purpose of the action; (2) can be implemented consistent with the scope of the federal agency's legal authority and jurisdiction; (3) is economically and technologically feasible; and (4) we believe would avoid the likelihood of jeopardizing the continued existence of listed species or resulting in the destruction or adverse modification of critical habitat (50 CFR §402.02). This section presents the USACE and the applicant with an RPA that we believe can be implemented to avoid DAM while meeting each of the requirements listed above.

The potential for nearshore loss of 33 acres of the essential feature of elkhorn coral critical habitat on the west end of St. Croix due to sediment loading during project construction and operation, in-water construction footprints, and operation of marine facilities affects the sexual and asexual reproductive success of elkhorn corals. Elkhorn coral population sizes have suffered declines across the species' range. Critical habitat was designated due in part to the existing low population sizes and the likelihood that further declines in population sizes could lead to threshold levels making the chances of recovery low. Colonies may become separated by too much distance for sexual reproduction to occur, and lower population sizes and greater distances between populations also lead to a reduced source of fragments for asexual reproduction and recruitment. These problems are exacerbated by fragmentation and degradation of settlement habitat. There is no scientific evidence such as an increase in the overall amount of living tissue, growth of existing colonies, or an increase in sexual recruits, indicating that the species is recovering. This may be due in part to degradation of the essential feature of critical habitat by stressors such as land-based sources of pollution making conservation of habitat critical for the recovery of the species.

The RPA outlined below consists of modifications to the project that will eliminate the in-water construction footprint in elkhorn coral critical habitat and reduce land-based sources of pollution, particularly sediment loading, to nearshore consolidated hard bottom. Reductions in land-based pollutant loading will minimize the potential for loss of 33 acres of the essential feature of elkhorn coral critical habitat on the west end of St. Croix. The anticipated reduction in sediment loading to nearshore waters will allow successful sexual and asexual reproduction of elkhorn coral by preserving acreage of consolidated coral habitat for settlement and growth of larvae and recruits in a portion of the St. Croix unit (the Frederiksted Reef System) that is anticipated to play an important role in the recovery of elkhorn corals. The RPA focuses on stopping or minimizing erosion from disturbed areas, controlling the erosive impacts of increased or concentrated runoff, minimizing opportunities for sediments to be transported to streams and coastal waters, and minimizing disturbance to benthic habitats from in-water construction and land-based pollutant loading.

The following RPA, with 3 elements, must be implemented by the USACE or the applicant to avoid the destruction or adverse modification of elkhorn coral critical habitat. All of the general design criteria and the mandatory components identified for RPA Elements 1 and 2 must be incorporated in the redesign of the project and development of site specific BMPs. Considerations for meeting the general design criteria that are not mandatory components are included for each element of the RPA. These considerations are not meant to be an exhaustive list for each element that will enable achievement of the performance criteria but instead to serve as guidelines for the redesign of the project and development and implementation of site specific BMPs. The most critical aspect of the RPA is the achievement of the specified performance criteria identified below, which will be determined through a technical peer review of the redesigned project, as detailed in RPA Element 3.

The RPA has been designed based on work being done to revise building standards, stormwater controls and BMPs specific to the USVI, including the development of a training course to guide green building for agency staff, developers, and construction contractors sponsored by NOAA in coordination with DPNR and the Island Green Living Association of St. John called *Our Islands Our Future: Guide to Green Building in the USVI* (see some of the course content)[28]. The RPA also incorporated information from cities and counties in the US and other countries such as Lehigh Valley Planning Commission (Lehigh Valley Planning Commission)[29], Pennsylvania Land Trust Association (Pennsylvania Land Trust Association)[30], Gold Coast, Australia (Gold Coast, Australia)[31], and the Caribbean Environment Program of the United Nations for the Insular Caribbean (Caribbean Environment Program of the United Nations for the Insular Caribbean)[32], among others. Compilations of information regarding building requirements and BMPs were also consulted in the development of the RPA such as *Stormwater Management in Pacific and Caribbean Islands: A Practitioner's Guide to Implementing LID* (Stormwater Management in Pacific and Caribbean Islands: A Practitioner's Guide to Implementing LID)[33] and the International Stormwater BMP Database (International Stormwater BMP Database)[34].

By modifying the project design, impacts to the marine environment, such as sedimentation and the loss of benthic habitat associated with the construction and operation of the project would be greatly reduced and eliminate the destruction or adverse modification of acroporid coral critical habitat. The RPA is designed to reduce the effects of the USACE issuing a permit for the construction of the Amalago Bay Resort and Residential Community on the west end of St. Croix and the subsequent effects of the operation of the project to such a degree that the effects are not expected to appreciably reduce the chances of the St. Croix unit achieving recovery Criteria 6, 1 and 3; thus ensuring the conservation value of coral critical habitat is not appreciably diminished. The USACE shall ensure the applicant incorporates the following changes in the design and implementation of the project that are part of RPA Elements 1 and 2. The USACE shall also ensure that a technical review process is conducted to ensure engineering standards and other technical aspects are adequate to meet the performance standards described below that are part of RPA Element 3.

Pursuant to 50 CFR § 402.15, the USACE shall determine whether and in what manner to proceed with the issuance of a permit for the project and notify NMFS of its final decision.

If the USACE elects to permit the project, the USACE must ensure that the permitted project is consistent with all elements of this RPA and that the reasonable and prudent measures (RPMs)

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28 https://repository.library.noaa.gov/view/noaa/850
29 https://lvpc.org/e-guides---model-regs.html
30 https://conservationtools.org/guides/59-steep-slope-ordinance
31 https://www.goldcoast.qld.gov.au/planning-and-building/gold-coast-planning-scheme-483.html
32 https://www.unenvironment.org/cep/sedimentation-and-erosion
33 https://horsleywitten.com/pdf/Feb2014_IslandBMPGuide_wAppendix.pdf
34 http://bmpdatabase.org/

and the terms and conditions of the Opinion's incidental take statement (ITS) are implemented. The USACE shall be the repository of reports, project plans, and other documents generated to comply with the RPA and the terms and conditions of this Opinion and shall provide copies of such documents to NMFS within 30 days of their receipt.

10.1 RPA Element 1 – Incorporate General Design Criteria in Project Design to Avoid Direct Physical Impacts and Minimize Pollutant Impacts to Elkhorn Coral Critical Habitat

Background
The USACE shall require that William and Punch LLC redesign the project to avoid direct impacts to elkhorn coral critical habitat from in-water construction. The USACE shall also require the redesign to minimize impacts to coral critical habitat from the operation of marine facilities and the potential for erosion and subsequent transport of sediment to nearshore waters associated with marine facility operation and from the construction and operation of other facilities that include the modification of natural water courses. The redesign of the project will incorporate significant site features in the development in order to preserve the area's existing natural terrain and erosion potential, including through the exclusion of steeper portions of the site from the developable area and minimizing grading to reduce erosion and subsequent sediment transport, limit runoff from new development to reduce land-based pollutant transport and maintain ecosystem integrity.

Action
Redesign the terrestrial and marine components of the Amalago Bay Resort and Residential Community project and generate new project plans.

Project Redesign General Design Criteria
The primary purpose of the general design criteria are to: a) reduce impacts to the essential feature of elkhorn coral critical habitat as a result of project construction and operation by eliminating development of steep slopes to reduce erosion and subsequent sediment transport, b) ensure effective management of stormwater runoff to reduce sources of land-based pollutants, c) minimize transport of land-based pollutants to nearshore waters, and d) eliminate in-water construction footprints in coral critical habitat and minimize operational impacts associated with marine facilities. As required by RPA Element 3, the plans for the redesigned project are to be reviewed by third-party experts and the final project plans shall be approved by NMFS. The general design criteria that apply to this project are as follows:

1. The project is designed to fit the existing topography, soil characteristics, waterways, and natural vegetation, and site disturbance is minimal.
2. Development in natural drainageways is avoided.
3. The smallest practical area of land is exposed for the shortest time possible during project construction, erosive flows are safely conveyed, and sediment is trapped on-site.
4. Natural in-water features, particularly colonized hard bottom, are preserved including through the elimination of direct impacts to these features from in-water project design.

Specifically, the following mandatory elements must be reflected in the redesign of the project to incorporate the general design criteria listed above. All of these elements are required in order to ensure the project design adequately addresses the general design criteria in terms of the project incorporating the existing physical characteristics of the site including topography, soils, vegetation, and drainage patterns in order to minimize stormwater runoff and associated land-based pollutant transport to nearshore waters and in-water construction to be protective of coral critical habitat:

1. A site design that is at least 75% complete and includes all project phasing must be provided as part of the permitting process to allow for an adequate assessment of the effectiveness of project redesign and BMPs (see RPA Element 2) in achieving the required performance criteria (see RPA Element 3). A design level of 75% or greater allows for the development of site plans that include BMP sizing, drainage routing, and other details that are essential to confirm that performance criteria can be met through an assessment of the effectiveness of erosion control and stormwater management. This design level also allows for a detailed phasing plan such that the timing of site preparation activities including road cutting and clearing to allow site access versus the timing of BMP construction can be evaluated to ensure control measures will be effective.
2. The plan for development on hillsides must indicate current drainage routing for minor and major storm events and indicate how the redesigned project will alter these patterns. Storm recharge areas (also known as infiltration basins or galleries) shall be incorporated in the redesign. Onsite stormwater recharge shall be maximized at all scales from individual lots to the basin catchment area. Mitigative measures must be incorporated in project design to address unavoidable changes in stormwater patterns that could result in changes to natural resources onsite and off. Natural stormwater recharge areas and stormwater and groundwater routes should be integrated in the site design and must not be altered. These details are essential for assessing the effectiveness of the site design in avoiding the transport of stormwater runoff containing land-based pollutants into nearshore waters.
3. The clearing of steep slopes (20% or greater) must be avoided and modification to the existing topographic condition such as for road or utility crossings will be minimal in these areas. In areas with slopes above 10%, the following is required to ensure site constraints have been properly addressed in the design of buildings: a) retention of the majority of vegetation onsite, and b) adequate engineering and site management techniques determined based on an evaluation by an engineer experienced in slope stability. Due to the erosion potential of steep slopes, particularly when coupled with the types of soils present on the site, avoidance of construction on slopes of 20% or greater and implementation of additional controls for slopes of 10-20% to ensure slope stability, minimize erosion including landslips, and manage surface drainage and groundwater infiltration is necessary in order to prevent the transport land-based pollutants to nearshore waters.
4. Cutting of tops of slope to create straight, linear tops is prohibited as are sharp cuts and the creation of uniform grades on long or wide slopes and the creation of flat terraces on hillsides. Building pad areas must be created as part of lot grading such that structural retaining walls or cut and fill are not required (where, for example, cuts would be greater than 2 ft in depth, create a cut slope greater than 5 ft high measured vertically from toe to top of slope, or be greater than 50 cubic yards [yds^3]; or fill would be greater than 1 ft in depth on slopes flatter than 1 vertical [V] to 5 horizontal [H], greater than 3 ft deep on other slopes, or greater than 50 yds^3). These requirements will minimize erosion by

retaining natural slope peaks and minimizing the need for engineering measures to retain soil by designing with natural slopes. Development of smaller terraces for building pads and minimal rear yard areas is acceptable provided the toe of the structural slope is located within the lot and native slope and vegetation is retained as much as possible at the edge of required grading works in order to prevent soil from moving offshite during rainfall events.

5. The area disturbed by roads must be kept to a minimum. Roads and points of road discharge must be located to avoid slopes greater than 12.5%, and the grade of the road should not exceed 12.5% (1V:8H). Roads must be located to maintain a buffer distance from stream crossings, flood plains and shorelines. Buffer strips should be at least 25 ft wide on either side of natural drainageways, wetlands and shorelines if the topography of the site allows and as required by DPNR regulations. Existing vegetation in floodplain and wetland areas along all natural drainageways and within buffer strips if not included in these areas must be preserved where present. Where not present, buffer strips should be planted with native vegetation appropriate to the soil conditions, slopes, and anticipated stormwater flows.
6. In-water dredge, fill and construction footprints must avoid all areas containing elkhorn coral critical habitat in order to avoid direct impacts to the essential feature of critical habitat.

The following additional elements are provided as examples of considerations that should be applied in the redesign of the project but are not part of the required elements, which are those specified above:

1. Larger estate lots (typically one acre or more) are the most disruptive housing type on hillsides due to the large setback and lot width resulting in low yield per linear meter of road, if structure placement and the extent of site disturbance allowed on each lot are not strictly controlled. Hillside development should restrict the use of large lots to locations where slopes are gradual (10% or less) unless strict restrictions on the placement of structures and extent of land clearing on each lot are implemented and covenants established to prohibit future regrading of slopes.
2. Cluster development locations that minimize cut and fill should be selected. Higher and mixed density clusters of development should be used to protect steep slopes and downstream resources. Clustered development should strive to provide a minimum of 20% of the gross developable site area as permanent open space within each cluster. The location and size of open space to be retained, including to preserve undeveloped buffers, should be considered when deciding the appropriate mix of building forms.
3. Where mixed density development is used, undeveloped buffers between sensitive areas such as natural drainages, wetlands, and coastlines should be larger than the 25 ft buffer size required by DPNR due to the increased density of development. Open space should provide a natural corridor through and around the property and connect open spaces between clusters.
4. Large, irregularly-shaped lots may be considered as a means of utilizing areas where road frontage is limited provided undisturbed areas are maximized and protected on the lot by a restrictive covenant. Panhandle lots should be used to minimize cut and fill and provide access to buildable areas that are too high or low to be directly accessed from the road.

5. The use of retaining walls should be restricted to the preservation of native undisturbed areas or addressing unstable native slopes or rock faces.
6. A split-level building form should be used on sloping sites.
7. Driveways internal to sloping sites should not be steeper than 25% (1V:4H).
8. Stormwater runoff should be collected and used onsite to establish and maintain plantings during construction and as landscape irrigation during operations. Collecting stormwater from impervious surfaces (i.e., roads and buildings) should be incorporated into the project's construction and operating plans. Minimizing offsite transport of stormwater runoff is preferred and should be assumed as part of site grading. Utilizing pervious surfaces (e.g., pervious parking lots) to encourage immediate infiltration of stormwater should be incorporated into the project's construction and operating plans.
9. Irrigation is supported only as a means of re-establishing vegetation used to stabilize cleared areas and after construction is complete and landscaping has been done.
10. The ability of the existing soils and ground cover to withstand overland flows and peak flow rates and limit concentrations of water that can cause erosion should be considered in the design. Eliminating the potential for erosion is more cost effective than cleaning up after an erosion event.
11. Stormwater should not flow to road right-of-ways and stormwater flow off property should be minimal.

10.2 RPA Element 2 – Best Management Practices (BMPs)

Background
The USACE shall require that William and Punch LLC develop and implement BMPs to control erosion, sediment and other land-based sources of pollution, and stormwater runoff and to maintain marine water quality including during in-water construction and operation of facilities.

Action
Develop site-specific BMPs to control erosion and transport of land-based sources of pollution, particularly sediment, manage runoff, and maintain marine water quality to be implemented during project construction and operation.

Best Management Practices
A detailed stormwater, sediment, and erosion control plan or plans that includes the full maintenance schedule for all erosion, sediment, and stormwater control measures must be completed as part of the BMPs. All control measures for stormwater, erosion, and sediment must be included on plan drawings along with the phasing plan for implementation of controls. The mandatory BMPs that apply to this project are as follows:

1. Site specific soil erosion control practices must be applied as a first line of defense against off-site damage.

2. Sediment control practices must be applied as a second line of defense against offsite damage.

3. A maintenance program for stormwater and sediment control measures must be developed and implemented before, during, and after construction operations to continue during project operation.

4. Site specific stormwater management practices must be applied to limit runoff from all building lot footprints.
5. Site specific in-water water quality controls in compliance with proposed 2018 USVI standards for turbidity and with baseline monitoring of pre-construction conditions at the site must be developed and implemented before, during, and after project construction, and during project operation.

The following elements are provided as examples of considerations that should be applied to the development and implementation of site-specific BMPs as appropriate to the project's construction and operation to ensure erosion and sediment control, stormwater management, and water quality controls and maintenance of these controls:

1. Disturbance of soils by heavy equipment such as bulldozers should be minimized and existing vegetation and native ground cover should be maintained to the maximum extent possible.
2. All disturbed areas should be seeded and fertilized within 1 week of completing any phased contruction activites or construction activities on any sublot, and before the removal of sediment/erosion control measures. Fast-growing native grasses may be established to provide ground cover when full landscaping will be delayed. Permanent seeding and planting should be used to stabilize all exposed areas as soon as construction is complete, particularly on steeper slopes and in swales and other vegetated stormwater runoff areas.
3. Any access roads that are no longer needed should be restored to the original grade and planted within 3 days of stoppage of use of these roads. Soil stockpiles should be temporarily seeded or stabilized within 3 days of suspending active use of the stockpile.
4. Culverts should be properly sized for storm events in the area based on the frequency of large storms. Recent NOAA rainfall data should be used for all culvert sizing, determinations of flow diversions between watersheds (if applicable), and effective width of temporary sediment basin calculations. Width formula for sediment basins should incorporate flow from the 25-yr storm event due to the sensitivity of downstream resources. Culverts and culvert outlets should be installed in accordance with the updated 2014 USVI Stormwater Standards.
5. Because of weather patterns in the Caribbean, land clearing should be restricted to late winter or early spring to take advantage of April and May rains and enable vegetative cover to develop prior to the fall when high intensity rains are more common.
6. In order to retain soil moisture until vegetation is established on cleared areas, mulching and matting using materials such as cut grass, wood chips, wood fibers, or straw should be used. Mulch is spread uniformly over the soil such that no more than 25% of the ground surface is visible and anchored in place using matting stapled over the mulch or a liquid tackifier. Matting is preferable on steeper slopes and in channels to be used to convey runoff.
7. Perimeter dikes/swales should be established prior to any major soil disturbing activity, defined as grading for construction of structures such as buildings and roads. Dikes should be compacted using construction equipment to the design height plus 10% to

allow for settlement. If dikes will remain in place for longer than 10 days, they should be stabilized using vegetation, filter fabric, or other materials. Diverted water should be directed to a sediment trap or other sediment treatment area. The use of perimeter dikes/swales should be limited to drainage areas of no more than 2 acres with gently sloping terrain.

8. Channels lined with grass or riprap (on steeper slopes with a foundation of filter fabric or gravel) should be excavated where it will be necessary to carry concentrated runoff to a stable outlet without causing erosion or flooding and to protect downstream resources to allow runoff to infiltrate into surrounding soil as much as possible.

9. In areas where there will be a cut or fill slope, temporary slope drains should be used to convey runoff water down the face of the cut or fill slope to minimize erosion until the slope can be stabilized and revegetated at which time the slope drain is removed. Discharge from a slope drain should be to a sediment trap, sediment basin, or other stabilized outlet.

10. Temporary or permanent check dams may be installed across drainage ditches, swales or small channels to reduce the velocity of runoff, reduce the erosion of the channel, and allow larger sediments to settle out. Dams should be spaced so that the toe of the upstream dam is the same elevation as the top of the downstream dam, or as close as possible to this in steeper channels. Check dams should only be used in small open channels that will not be overtopped by flow once the dams are built. Check dams should not be built in natural stream channels. The sizing of check dams has to take into consideration the capacity needed for the channel to transmit storm runoff.

11. Outlet protection should be installed early in construction at all drainage outlets where flow velocity and quantity may cause erosion.

12. The maximum drainage area to a silt fence should not exceed 0.5 ac per 98 ft (30 m) of fence. A silt fence should only be used for small disturbed areas and is not appropriate for use in channels, gullies, or other locations of concentrated runoff. Sediment fences should be used in conjunction with other practices and are only applicable for certain slope distances. In areas with slopes of 10% or greater, 25 ft or less of slope distance above the fence is required which means the use of sediment fences is inadequate to control erosion, particularly during periods of rainfall and other erosion control and stormwater management measures specific to the site and slopes will be needed.

13. Brush barriers consisting of trimmings, limbs and brush piled in long rows with a filter cloth placed over the row, attached to the brush with staples or other means, and buried in a small trench on the uphill side of the barrier should be used to slow runoff and filter sediment and placed around the perimeter of a disturbed area. Brush barriers are constructed in conjunction with any vegetation clearing and the cleared vegetation is used to create the berm. Brush barriers should not be used where concentrated flow is anticipated and must be replaced with another control measure as vegetation rots or compresses.

14. Sediment basins may be designed as temporary or permanent site features. Sediment basins should be large enough to retain runoff for a period sufficient to allow most sediment to settle out. Runoff should enter the basin as far from the outlet as possible to

provide maximum retention time. Sediment basins should be built as close to the sediment source as conditions allow and are best located in low areas where minimal clearing and grading is needed. The location of sediment basins should be selected to intercept the largest amounts of flow possible and reduce the possibility of embankment failure and to be accessible for maintenance activities. The basin should have a principal spillway or a pipe-and-gravel outlet designed to safely release excess runoff.

15. Sediment traps can be installed prior to grading and filling and typically have an effective lifetime of 24 months. Typically a minimum storage capacity of 250 m^3 per acre disturbed is recommended for the excavation of a sediment trap. Embankments should not exceed 5 ft (1.5 m) in height and should have a minimum top width of 4 ft (1.2 m). Side slopes should be 2:1 or flatter. All embankments and disturbed areas associated with the construction of sediment traps should be stabilized with vegetation. A riprapped spillway or other outlet must be provided for stormwater release. Sediment traps are suitable for drainage areas of 5 ac or less.
16. Stormwater, erosion, and sediment control devices should be inspected within 24 hours after each rainstorm for sediment and debris accumulation and cleaned or repaired within 3 days as needed. All stormwater inspections should be documented in accordance with the Stormwater Pollution Prevention Plan (SWPPP) that includes the erosion and sediment control measures along with any required corrective actions.
17. Turbidity barriers or other in-water controls to minimize sediment resuspension and transport during any in-water construction should be designed and placed in accordance with the oceanographic and physical conditions of the site such as predominant wind and wave direction and speed, current patterns, water depths, and bottom types. Controls must be appropriate to the conditions at the site to ensure they can be securely anchored to prevent movement that could result in damage to ESA-listed coral colonies and avoid impacts to elkhorn coral critical habitat.

10.3 RPA Element 3 - Performance Criteria

The following performance criteria to ensure the goal of reducing land-based pollutant loading and in-water impacts to ESA-listed corals and elkhorn coral critical habitat during project construction and operation is met are required as part of RPA Elements 1 and 2 development and implementation. It is the responsibility of the USACE to lead the technical review detailed below to ensure the achievement of these performance criteria.

1. Normal rainfall events, defined as storms as small as 1 cm per hour (2 to 10-year storms) have been found to account for up to 90% of total runoff and sediment yield in a watershed despite accounting for only about 50% of total annual precipitation (Ramos-Sharron and MacDonald 2007b). The project stormwater management measures shall be designed to ensure the current estimated 2 runoff events per year leading to runoff to nearshore waters during normal rainfall events is not exceeded (estimate obtained by NMFS using the EPA stormwater calculator) under these rainfall conditions and to minimize stormwater runoff to nearshore waters during larger events.

2. Maximum runoff intensity where drainages enter nearshore waters shall not exceed 20 cfs under any storm conditions in order to minimize transport of land-based pollutants and potential scouring impacts to nearshore benthic habitats.
3. The natural and created drainage system on the site has the capacity to convey the increased flows that will result from project construction and operation.
4. During project construction, sedimentation rates of >= 10 mg per cm2 per day due to transport of sediments from terrestrial construction areas shall not be present in nearshore waters for more than 6 consecutive days. Sedimentation events with >= 10 mg per cm^2 per day lasting longer than 1 day may only result from extreme rainfall events (> 10 year storm).
5. During project operation, sedimentation rates shall not exceed 3 mg per cm2 per day following normal rainfall events, defined as storms up to and including 10-year storms (i.e., up to approximately 25 cm per day). Following extreme rainfall events (> 10 year storm), sedimentation rates >= 3 mg per cm2 per day, shall not be present in nearshore waters for more than 6 consecutive days.
6. No direct impacts to nearshore hard bottom habitat such as in-water dredge, fill or construction footprints and creation of temporary or permanent vessel anchor areas or locations for spudding of barges will occur as a result of the project. Indirect impacts as a result of terrestrial construction will be limited as noted in the performance criteria above.
7. During project construction, turbidity values (measured in NTUs) shall never exceed the highest observed pre-construction turbidity values measured during pre-construction monitoring outside any in-water turbidity controls. In addition, turbidities shall not remain elevated above natural levels measured following storm events during pre-construction monitoring for longer than the amount of time over which natural increases in turbidity were observed. In other words, during project construction, elkhorn coral critical habitat should not be exposed to turbidity levels greater than or for durations longer than those that occur during normal rainfall events.During project operation once all construction is complete, turbidity values shall not exceed 1 NTU in nearshore waters in accordance with the proposed revisions to the USVI Water Quality Standards [Title 12, Chapter 7, Subchapter 186, Section 186-4:(c)(2)(B)(ix)(b) for Class B waters in areas where coral reef ecosystems are located], except following an extreme rainfall event (> 10 year storm).

10.4 Technical Review

The technical review process must be used to validate whether the redesigned project and site specific BMPs will be successful in reducing erosion and subsequent sediment and land-based pollutant loading to nearshore waters from terrestrial construction and during project operation and in preserving water quality for in-water construction and operation of the project and must conclude that all the performance criteria are expected to be met.

The technical review process:

1. Requires that the USACE select a professional engineer or engineers with the qualifications necessary to review the redesigned project and certify that the requirements

of RPA Elements 1 and 2 are met. The USACE must provide information regarding qualifications of the engineer or engineers, including examples of similar projects on which the person or persons have worked, the agency is considering to NMFS for approval prior to making a final selection. Alteratively, the USACE may choose to form an evaluation team that includes NMFS to evaluate candidates and make a final selection.

2. The selected engineer(s) will review the redesigned project and site specific BMPS to determine whether (1) the redesign reflects the design requirements and considerations of RPA Elements 1 and 2, and (2) the performance criteria of RPA Element 3 are expected to be met throughout project construction and operation.
3. Once the review period is complete, the technical reviewer(s) will submit a report to USACE and NMFS detailing whether the redesigned project and site specific BMPs will meet the performance criteria. The report should include but not be limited to the technical items detailed below.
4. The USACE and NMFS will have 30 days to review the report and request clarification if needed prior to accepting the results of the technical review.

The technical review report must include but is not limited to:

1. An overall assessment regarding slope stability and construction measures proposed in areas with slopes between 10-20%; adequacy of drainage to minimize land slips and less severe erosion and overall adequacy of the site drainage plan and grading plan in order to assess the effectiveness of the redesigned project and site specific BMPs in meeting the performance criteria.
2. A determination of sediment yield and a calculation of erosion rates. The values from the 2014 updates to the USVI Stormwater Standards for the Virgin Islands Environmental Protection Handbook related to soil characteristics, rainfall, local soil erosivity, slopes, and other physical characteristics specific to the USVI must be used when determining sediment yield regardless of the method used to perform the calculations. In addition, calculated erosion rates must account for ghut erosion.
3. A summary table showing drainage basin areas and land cover assumptions and corresponding basin contributing area maps will be provided.

10.5 Findings on the RPA

Section 7(b)(3)(A) of the ESA provides that

> *If jeopardy or adverse modification is found, the Secretary shall suggest those reasonable and prudent alternatives which he believes would not violate subsection (a)(2) of this section and can be taken by the Federal agency or applicant in implementing the agency action.* 16 U.S.C. § 1536(b)(3)(A).

Our regulations (50 CFR 402.02) further provide that an RPA as:

> *"alternative actions identified during formal consultation that can be implemented in a manner consistent with the intended purpose of the action, that can be implemented*

consistent with the scope of the Federal agency's legal authority and jurisdiction, that is economically and technologically feasible, and that the Director believes would avoid the likelihood of jeopardizing the continued existence of listed species or resulting in the destruction or adverse modification of critical habitat."

Courts have held that avoiding jeopardy or adverse modification is the paramount requirement of an RPA, and a biological opinion must demonstrate that the identified RPA avoids these outcomes. Courts have split on the degree to which a biological opinion is required to evaluate the other elements identified in the regulatory definition. Below, we have provided an element-by-element discussion of the RPA above. The following section analyzes the RPA to ensure it meets this regulation.

The alternative avoids destruction or adverse modification of elkhorn coral critical habitat
Issuing a permit for the Amalago Bay Tourist and Residential Community without the RPA will lead to the destruction or adverse modification of the elkhorn coral St. Croix critical habitat unit and adversely impact the recovery of elkhorn corals. The loss or diminution of the function of the essential feature affects the sexual and asexual reproductive success of elkhorn corals because substrate for sexual and asexual recruits to settle is lost or unavailable. As discussed in Sections 5 and 7, the proposed action is expected to eliminate 33 ac of the essential feature (consolidated reef substrate) predominantly in depths of 5 m or less where elkhorn corals are predominantly found. The loss of 33 acres if the project is authorized by the USACE without the implementation of the RPA will appreciably reduce the St. Croix unit's ability to achieve at least 40% of consolidated reef substrate in 1-20 m depths within the forereef zone free of sediment and macroalgal cover (recovery Criterion 6) for elkhorn coral. As noted previously, this will result in further fragmentation of hard bottom habitat on the west end of St. Croix within the Frederiksted Reef System. The loss of habitat and resulting habitat fragmentation will affect the availability of areas for coral larval settlement due to the increase in distance over which coral larvae will need to travel to find suitable settlement habitat. In addition, the increased habitat fragmentation that will result from the proposed action will affect elkhorn and staghorn corals' primary mode of reproduction in USVI, which is through asexual fragmentation because fragments will also have to survive over greater distances prior to settling and growing. The reduced conservation benefit of the essential feature of elkhorn and staghorn coral critical habitat resulting from the issuance of a permit for the project without implementation of the RPA will delay the recovery of elkhorn corals around St. Croix and make recovery more difficult.

The issuance of a permit for the Amalago Bay project with the RPA will allow elkhorn coral around St. Croix to fully perform their natural life cycle and persist. The amount of land-based sources of pollutant loading, particularly sediment, to nearshore waters containing the essential feature of elkhorn coral critical habitat would not increase significantly and marine water quality would be maintained if the RPA is implemented, meaning sediment and macroalgal cover on consolidated reef substrate would not increase. We base this conclusion on the work done by HWG under contract to NOAA and EPA to draft new USVI-specific requirements for stormwater control developed using findings from the implementation of watershed management measures, specifically aimed at the control of sediment transport to nearshore waters, in St. Thomas and St. Croix, as well as islands in the Pacific. We also base our conclusion on findings regarding the effectiveness of BMP design and implementation from the International

Stormwater BMP Database related to the performance of BMPs meant to remove sediment from stormwater. Finally, we base our conclusion on the information compiled for our green building course developed for the USVI with information from designers, builders, and agency personnel who work in the USVI and have completed projects that incorporated the concepts presented in the course, including many of those that are part of the RPA related to minimizing the transport of land-based pollutants to nearshore waters by designing projects in order to incorporate the natural characteristics of a site with minimal alteration of existing slopes, vegetation, and drainage patterns. The implementation of the RPA is expected to maintain the condition of existing areas of consolidated reef substrate in the action area offshore of the property because, if the performance criteria are achieved, only short-term temporary effects to elkhorn coral critical habitat would occur as a result of project construction during large storm events. During project operation no measurable change above baseline to water quality or pollutant loading to nearshore waters is expected. The lack of habitat fragmentation in the Frederiksted Reef System on the west end of St. Croix will ensure the function of the essential feature of elkhorn coral critical habitat is retained. Connectivity between consolidated reef substrate on the west end of St. Croix will be maintained, allowing larvae produced by sexual reproduction to settle and grow to maturity because there will be no fragmentation of settlement habitat that would affect larval viability. Connectivity within the hard bottom habitat would also ensure that asexual fragmentation, which is currently the predominant way in which elkhorn corals expand their populations in USVI, is successful as fragments would have adequate substrate available for growth.

Issuance of a permit by the USACE to William and Punch LLC with full implementation of the RPA will not appreciably reduce the St. Croix elkhorn coral critical habitat unit's ability to reach recovery Criterion 6 (the unit will have at least 40% of consolidated reef substrate in 1-20 m depth within the forereef zone free of sediment and macroalgal cover); Criterion 1 (thickets present throughout approximately 10% of consolidated reef habitat in 1-5 m water depth within the forereef zone defined as colonized greater than or equal to 1 m diameter at a density of 0.25 colonies per m^2 or live elkhorn coral benthic cover of approximately 60%); or Criterion 3 (2^{nd} prong, effective sexual recruitment observed defined as establishment of new larval-derived colonies and survival to sexual maturity) for elkhorn coral.

The Frederiksted Reef System is the only reef habitat on the west end of St. Croix and we estimate that 723 acres containing the essential feature of elkhorn and staghorn coral critical habitat can be found in the 0-5 m depth range within this reef system. Much of this acreage is already impacted by development, especially in the southern portion of the reef system due to the presence of the town of Frederiksted and associated development, including the cruise ship pier, and to the north of the project area associated with residential development and a quarry that discharges stormwater and sediment directly into Caledonia Ghut and thus into Ham's Bay. To the north of the proposed Amalago Bay project, elkhorn coral colonies in 0-6 m depths in Butler and Ham's Bays were found to have only 50% live tissue cover, which is much lower than colonies in other areas around St. Croix. Smith et al. (2014) theorized that this lower percentage of live tissue cover was due to chronic impacts associated with land-based sources of pollution, including erosion from roads and quarry operations. The section of the Frederiksted Reef System where the Amalago Bay project is located is relatively free of development with good water quality, which is why Smith et al. (2014) concluded that the nearshore hard bottom and shelf edge reefs along the west coast of St. Croix will play an important role in recovery of elkhorn corals. Thus, actions that address conservation of the essential feature of elkhorn coral critical habitat toward recovery Criterion 6 include reducing land-based pollutant loading and maintaining marine water quality, which will also support recovery Criteria 1 and 3.

Conclusion on Avoidance of DAM

Based on these considerations, we find that issuance of a permit for the Amalago Bay Tourist and Residential Community with the RPA is not likely to result in the destruction or adverse modification of elkhorn and staghorn coral critical habitat and meets the criteria stated at 50 CFR 402.02. Implementation of the RPA of this Opinion will ensure that construction and operation of the Amalago Bay project will not appreciably diminish the value of critical habitat for the conservation of elkhorn coral by avoiding and minimizing adverse effects to the essential feature of its critical habitat.

The alternative is consistent with the intended purpose of the USACE permit and within the scope of the USACE's authority.

The primary purpose of the action is to create a mixed residential/tourism development on the west end of St. Croix. The applicant identified a resort core, beach access, residential opportunities, golf course, and marina as key design criteria for the project. The RPA still allows for this type of development, including in-water access to the site but with the replacement of the inland marina with a pier for water access. When issuing Clean Water Act Section 404 and Rivers and Harbors Act Section 10 permits, the USACE must ensure that the waters of the U.S., including wetlands, are protected and impacts to these resources associated with the placement of structures, dredging and filling are avoided and minimized to the maximum extent practicable, and unavoidable impacts compensated, while also complying with the ESA. When developing the RPA, we considered the purpose of the action and the USACE's authority for permitting the project under Section 10 of the Rivers and Harbors Act for temporary and permanent in-water structures that do not require fill placement and under Section 404 of the Clean Water Act for temporary and permanent placement of fill in waters of the U.S.

The alternative is economically and technically feasible.

The alternative is economically and technically feasible for the USACE to implement, as it would involve routine permitting oversight functions. In terms of economic and technical feasibility for the applicant, as noted at the beginning of this section (Section 10), we used information from existing building regulations, guidelines, and technical publications related to site design, construction and maintenance from the Caribbean, the USVI, as well as locations in the U.S. and Canada, and information from previous consultations (such as Veteran's Drive, SER-2013-12200) to develop the RPA. Based on the information we compiled, the construction of the proposed RPA would be less costly compared to the action as proposed due to the reduced need for expensive engineering controls for construction works in areas with steep slopes, as well as over the long term because of reduced maintenance costs, including those associated with the cleanup of sediment transported downslope and restoration of natural areas affected by this sediment transport. The reduced in-water construction footprint would also be less costly to build and maintain and would eliminate the need for frequent maintenance dredging. The proposed general design criteria and BMPs are standard practice in many areas with steep slopes and many areas have incorporated them into building regulations. We believe the RPA is both economically and technically feasible. Specific references for each element of the RPA are provided below:

RPA Element 1 – Incorporate General Design Criteria in Project Design to Avoid Direct Physical Impacts and Minimize Pollutant Impacts to Elkhorn Coral Critical Habitat

In terms of on-site alternatives, the applicant previously analyzed alternatives including reductions in size of the terrestrial project footprint, reductions in the size of the marina, construction of an exterior marina, and the construction of a single fixed pier to service the project rather than a marina as part of the Clean Water Act Section 404 analysis performed for the project. However, the applicant did not analyze the difference in cost between these alternatives. Given the reduction in costs expected from implementation of the RPA due to the elimination of development in areas with steep slopes that would require additional engineering measures as well as incurring additional costs associated with the use of more construction materials due to the need to stabilize slopes from cuts and fill, we believe the estimated costs needed to realize the project as currently proposed, of which the majority is for construction costs according to the applicant, would be significantly reduced. A 2007 estimate of project costs indicated that \$637.6 million was needed to construct the project with \$200.3 million of that being construction labor and \$300.4 million construction materials. Of this, waterfront construction would cost approximately \$69.7 million and marina construction approximately \$19.4 million. The general design criteria incorporated in the RPA are taken from information on building codes, steep slope regulations, and sediment control guidelines from around the world including island nations that face challenges similar to those of St. Croix related to allowing economic development while protecting natural resources on which the economy also depends and programs such as the Caribbean Environment Programme of the United Nations. All of the required and recommended criteria associated with RPA Element 1 were taken from locations where residential, commercial, and tourist developments must comply with similar criteria (see sources listed at the beginning of this section) and from information provided by architects, engineers, and construction contractors who participated in the development of the USVI green construction course materials, including those associated with the Island Green Living Association of St. John. The design, construction and maintenance information included in the USVI green construction course was taken from and has been incorporated in a number of projects developed in USVI. Therefore, we conclude that RPA Element 1 is both economically and technologically feasible.

RPA Element 2 – Best Management Practices to Control Erosion, Sediment and Other Land-Based Sources of Pollution, and Stormwater Runoff and to Maintain Marine Water Quality During In-Water Construction and Operation of Facilities.

The applicant had incorporated some sediment and erosion control practices, stormwater management measures, and in-water turbidity controls in the proposed project, as well as maintenance activities for these. Based on our review of BMPs for construction and maintenance of erosion and sediment control measures and stormwater management, as well as previous construction projects for which NMFS has completed consultations that incorporated similar measures and in-water sediment controls and maintenance, we do not believe the cost of implementation of the RPA would increase over the costs currently associated with the incorporation of measures previously proposed by the applicant. As noted previously, the changes to the project design and the incorporation of BMPs that are appropriate to the characteristics of the site will result in the elimination of some aspects of the project, reducing project costs by millions of dollars. The costs of the BMPs themselves are not expected to exceed this cost reduction based on the detailed costs of similar BMPs contained in the *International Stormwater BMP Database* (http://www.bmpdatabase.org/download-master.html) developed with support from the EPA in part to aid municipalities and others in designing and implementing the most site appropriate and cost effective BMPs. In addition, due to the implementation of RPA Element 1, BMPs associated with erosion and sediment controls and stormwater management during construction and sediment management for in-water construction and maintenance of same are expected to be less costly over the long-term. With respect to the technological feasibility, the BMPs in RPA Element 2 are taken from steep slope and sediment, erosion and runoff control guidelines from around the world but particularly from island nations, the links to which are provided at the beginning of this section and have been tested in terms of their economic and technical feasibility including through studies, some of which are available through the *International Stormwater BMP Database*. Therefore, we conclude that RPA Element 2 is both economically and technologically feasible.

RPA Element 3 – Performance Criteria to Ensure RPA Elements 1 and 2 are Met and Technical Review to Evaluate Effectiveness of Redesigned Project in Meeting Performance Criteria

Based on our review of steep slope building requirements in other jurisdictions, information on engineering firms specializing in the design of commercial and residential projects to minimize potential environmental impacts, and processes that have been used in other jurisdictions and by private entities to develop, implement, and test the effectiveness of BMPs in managing stormwater and controlling erosion and subsequent transport of pollutants outside a development project footprint, as well as based on the expertise within the USACE, we believe the performance criteria and technical review element of the RPA is technologically feasible (see information regarding requirements reviewed as part of the development of the RPA included in this section of our Opinion). We have not estimated costs associated with the additional administrative responsibilities that will be incurred by the USACE in particular in order to provide third-party peer review of the redesigned project plans and site specific BMPs, but the USACE has implemented these types of reviews for engineering and design of large projects such as the Discovery Bay Resort and Marina project proposed in Aguada and Aguadilla, Puerto Rico due to the location of the proposed marina in relation to a proposed USACE flood control project. We conclude that RPA Element 3 is economically and technologically feasible.

Conclusion on Economic and Technical Feasibility

As discussed above, the measures included in the RPA have been addressed in scientific and technical literature on the subject and have been incorporated in building regulations in some jurisdictions. The costs associated with the project are expected to decline with implementation of the RPA based on information from other locations, as well as information from the Green Building Program developed by NMFS in coordination with DPNR and technical experts including from the Island Green Living Association in St. John with funding from NOAA's CRCP for the USVI. Other costs will be incurred by the USACE in particular related to providing third-party peer-review of the redesigned project plans and site-specific BMPs to ensure required performance criteria can be met. We conclude that the RPA meets the requirement of being economically and technologically feasible.

11 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and federal regulations issued pursuant to Section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively. NMFS has not issued 4(d) regulations prohibiting the take of threatened star coral species. Section 7(b)(4) of the ESA requires NMFS to issue a statement specifying the amount and impact of any incidental take on listed species, which results from an agency action otherwise found to comply with Section 7(a)(2) of the ESA, and at least one court has held that the statement must include prohibited as well as non-prohibited take. 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 the purpose of, the carrying out of an otherwise lawful activity. Incidental take statements serve a number of functions, including providing reinitiation triggers for all anticipated take, providing exemptions from Section 9 liability for prohibited take, and identifying reasonable and prudent measures that will minimize the impact of anticipated incidental take.

We must estimate the extent of take expected to occur from implementation of the RPA so as to frame the limits of the take exemption provided in the incidental take statement (ITS). These limits set thresholds which, if exceeded, would be the basis for reinitiating consultation.

11.1 Amount or Extent of Take

Implementation of the RPA will reduce take of green, leatherback, and hawksbill sea turtle hatchlings through reductions in the in-water structure footprint thus reducing the potential for disorientation or entanglement or concentration along rock jetties or artificial reef structures that will function as offshore breakwaters. However, we anticipate that the project redesign will include an in-water structure such as a large dock or pier in lieu of the inland marina that will still lead to some concentration of hatchlings in the area of the structure. Using the methods described in Section 5.1 to calculate potential effects to sea turtle hatchlings, we estimate that approximately 0.12 km of nesting habitat (assuming 200 ft to either side of a structure such as a dock) could be affected. Thus, we anticipate that:

- up to 1 green sea turtle nest, or 91 hatchlings, per year from the South Atlantic DPS
- up to 2 leatherback sea turtle nests, or up to 76 hatchlings per year
- up to 7 hawksbill sea turtle nests, or up to 602 hatchlings per year

could become disoriented and concentrate around the in-water structure leading to an increased probability of mortality from stressors such as predation and an inability to find food. Implementation of the RPA would not eliminate the possibility of nesting female sea turtles abandoning a nesting effort due to lighting. We anticipate that up to 1 nest abandonment will occur per year for hawksbill, green (South Atlantic DPS), and leatherback sea turtles.

Because we do not anticipate any permanent losses of the essential feature of elkhorn and staghorn coral critical habitat or the degradation of the feature such that it will not support settlement of sexual and asexual recruits of elkhorn coral with the implementation of the RPA, we do not antitipate any take of future colonies of elkhorn coral.

If there are in-water structures contemplated as part of the project redesign required as part of the implementation of the RPA, such as a pier, then lobed and mountainous star coral colonies could be affected by impacts of temporary construction activities that cause sediment resuspension and transport. In Section 5.2, the number of impacted corals were derived using percent coral cover (24.5%), percent makeup by each species (lobed star 15%, and mountainous star 7.5%), and average colony size (38.75 cm2). Using these methods, to cacluate the potential number of lobed and mountainous star coral colonies that could be present in nearshore hard bottom in water depths from 0.5 m, we estimate that:

- up to 819 lobed star coral colonies
- up to 205 mountainous star coral colonies

could be affected temporarily by sediment resuspension and transport associated with in-water construction activities.

11.2 Effects of the Take

In the accompanying Biological Opinion, NMFS determined that this level of anticipated take from implementing the RPA is not likely to jeopardize the continued existence of the species identified above.

12 Reasonable and Prudent Measures

Section 7(b)(4) of the ESA requires NMFS to issue to any agency whose proposed action is found to comply with Section 7(a)(2) of the ESA, but may incidentally take individuals of listed species, a statement specifying the impact of that taking. Further, RPMs necessary or appropriate to minimize the impacts of incidental take from agency actions and the terms and conditions to implement those measures must be provided and followed. Only incidental taking by the federal agency or applicant identified in the ITS and in compliance with the specified terms and conditions is authorized.

The RPMs and terms and conditions are specified as required by 50 CFR 402.12 (i)(1)(ii) and (iv) to document the incidental take by the proposed action and to minimize the impact of that take on ESA-listed species. These measures and terms and conditions are non-discretionary, and must be implemented by the USACE. The USACE has a continuing duty to regulate the activity covered by this ITS. To monitor the impact of the incidental take, the USACE or the applicant, as applicable, must report the progress of the action and its impact on the species to NMFS as specified in this ITS [50 CFR 402.12(i)(3)].

NMFS has determined that the following RPMs are necessary and appropriate to minimize impacts of the incidental take of lobed star and mountainous star coral colonies; green, leatherback, and hawksbill hatchlings; green and hawksbill juveniles; and nesting female green, hawksbill, and leatherback sea turtles during the proposed action. The following RPMs and associated terms and conditions are established to implement these measures, and to document incidental takes.

1. An environmental monitoring plan to include water quality, habitat condition, and condition of ESA-listed corals shall be developed in coordination with NMFS and implemented prior to commencement of any construction activities.
2. An in-water sea turtle monitoring plan shall be developed in coordination with NMFS to supplement USFWS requirements for nesting sea turtles prior to commencement of any construction activities.
3. No nighttime activities shall be present along the shoreline and all upland development shall be done in accordance with a sea turtle lighting plan approved by the USFWS and reviewed by NMFS. If a dock is constructed as part of the redesigned project and the USCG requires lighting, the lights shall be in a sea turtle safe wavelength (450 nanometers, see http://seaturtlelighting.net/) to minimize potential hatchling and adult sea turtle disorientation.
4. All in-water structures shall be sited to avoid impacts to coral reefs and colonized hard bottom. No mooring buoys or other structures shall be used for mooring vessels for more than 2 consecutive days.
5. An education program shall be designed and implemented in coordination with NMFS and USFWS for construction personnel, visitors, and residents with information about ESA-listed species and their habitat.
6. If the redesigned project includes an in-water structure component (e.g., pier or dock) in an area containing colonized hard bottom and ESA-listed coral colonies are located within the in-water construction footprint, reinitiation of consultation may be required.
7. The USACE must provide NMFS with all data collected and all reports related to any additional benthic surveys conducted prior to construction and associated with the implementation of the required monitoring plans.

12.1 Terms and Conditions

"Reasonable and prudent measures" are nondiscretionary measures to minimize the amount or extent of incidental take (50 CFR 402.02). "Terms and conditions" implement the RPMs (50 CFR 402.14). The following terms and conditions implement the RPMs listed above:

1. The environmental monitoring plan shall be finalized in coordination with NMFS and the USACE. The plan shall be implemented prior to the commencement of any construction activities. The plan shall include pre- and post-construction determinations of the condition of benthic habitat utilized by ESA-listed corals and sea turtles including colonized hard bottom, coral reef, and seagrass and colonies of ESA-listed corals (that will not be transplanted). Monitoring of water quality shall also be part of the plan and include sampling before, during, and after construction. The monitoring plan shall specify variables to be tested and how, when, and where samples will be collected, including plotting sample locations on a map of the action area. Sampling schedules shall also be part of the plan. For water quality monitoring, sample collection during and immediately following rainfall, as

well as under non-storm conditions shall be included in pre-construction monitoring in order to determine existing concentrations of pollutants such as sediments present in the area under different weather conditions. Permanent transects or quadrats shall be established to determine whether benthic habitats within temporary and permanent in-water construction footprints recover naturally following construction of in-water structures such as a dock under the redesigned project or whether reinitiation of consultation is required to address unanticipated adverse effects. The transects or quadrats shall also be used to monitor the condition of ESA-listed corals, elkhorn and staghorn coral critical habitat, and green and hawksbill sea turtle refuge and foraging habitat to determine whether the construction and operation of the project is meeting the performance criteria of the RPA and avoiding impacts to these resources. The monitoring plan shall include the operational component of the project to determine the extent to which and stormwater runoff and associated transport of land-based pollutants to nearshore waters affect ESA-listed corals, sea turtles, and their habitat. (RPM No. 1)

2. The sea turtle monitoring plan shall include monitoring for different life stages of green, hawksbill, and leatherback sea turtles during the nesting season of each species, as well as in-water monitoring for green and hawksbill sea turtles. The plan shall include measures to be taken to minimize impacts to hatchlings if they are found to concentrate around in-water structures (if included in the redesigned project) such as collection of hatchlings and their transport to deeper waters away from coastal structures. (RPM No. 2)
3. The lighting plan shall be finalized in coordination with NMFS and USFWS prior to commencement of any construction activities. In order to develop the lighting plan, pre-and post-construction lighting inspections should be done from the shoreline and from the water to assess existing lighting along the shoreline and to evaluate the effectiveness of the lighting plan in minimizing impacts to sea turtles once the project construction is complete. (RPM No. 3)
4. The location of any in-water structures included in the redesigned project shall be selected to avoid impacts to ESA-listed corals and elkhorn and staghorn coral critical habitat within the footprint of the structure. (RPM No. 4)
5. The educational program shall include temporary signage during project construction and permanent signage for visitors and residents with information regarding ESA-listed species and their habitats and measures to be taken to avoid and minimize impacts to ESA resources associated with construction activities, boating, in-water recreational activities, and landscaping and property maintenance. The educational program shall include a training program for construction employees and inspectors and a program to provide residents and visitors with information about ESA resources. A training program for residents to assist in monitoring activities may also be included in the training program. The content of signage and other aspects of the training program shall be developed in coordination with NMFS and USFWS. (RPM No. 5)
6. RPM No. 7 assumes that there will be an in-water component to the redesigned project. If ESA-listed coral colonies are within any temporary in-water construction footprints and could be affected by construction operations such as barge spudding, dredging and other sediment-generating activities, reinitiation of consultation may be required. (RPM No. 6)

7. The USACE must provide NMFS with all data collected as part of additional pre-construction benthic surveys and the implementation of monitoring plans. This information can be submitted to nmfs.ser.esa.consultations@noaa.gov with copy to xxx (xxx@noaa.gov). The information should be submitted within 30 days of completion of surveys and monitoring events. (RPM No. 7)

The RPMs, with their implementing terms and conditions, are designed to minimize the impact of incidental take that might otherwise result from the implementation of the RPA. If, during the course of the action, this level of incidental take is exceeded, such incidental take represents new information requiring reinitiation of consultation and review of the RPMs provided. The USACE must immediately provide an explanation of the causes of the taking and review with NMFS the need for possible modification of the RPMs.

13 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs federal agencies to, in consultation with the Services, use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Conservation recommendations identified in Biological Opinions can assist action agencies in implementing their responsibilities under Section 7(a)(1). Conservation recommendations are discretionary activities designed 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 conservation recommendations are discretionary measures that NMFS believes are consistent with this obligation and therefore should be carried out by the federal action agency:

1. We recommend that NMFS's *Sea Turtle and Smalltooth Sawfish Construction Conditions* and NMFS's *Vessel Strike Avoidance Measures and Injured or Dead Protected Species Reporting* be included as special conditions of any permit issued for the project in order to minimize the potential impacts to all ESA-listed sea turtle species during construction and operation of the project.
2. We recommend that the measures developed by the applicant to protect ESA-listed whale species, with any modifications necessary based on implementation of the RPA, be included as special conditions of any permit issued for the project in order to minimize the potential impacts of vessel traffic associated with project construction and operation on ESA-listed whale species.
3. We recommend that pre, during and post-construction surveys include surveys for Nassau grouper and that any sighting of this species be reported to NMFS so that we can update information related to the presence of the species throughout its range.
4. Assuming there will be an in-water component to the redesigned project, we recommend that the applicant mark all navigation routes to and from the project area on nautical charts and post these within the project site, as well as providing them on laminated sheets for boaters and construction personnel. These routes should be included as a special condition of any permit issued for the project and should be selected to minimize the potential for accidental

groundings in areas containing ESA-listed corals and coral critical habitat, in particular along the west coast of St. Croix.

5. We recommend that the covenants and restrictions proposed by the applicant be edited to reflect project changes associated with the implementation of the RPA and that the covenants ensure protection of all green space in the redesigned project.
6. We recommend that acoustic minimization measures such as ramp-up and soft start of pile driving equipment and the use of bubble curtains depending on the type of piles and method to be used to drive the piles be included as a special permit requirement if there will be an in-water component to the redesigned project.
7. If the project will have an in-water component, we recommend that ATONS be installed to notify boaters of the presence of shallow coral reef and colonized hard bottom in order to minimize the potential for accidental groundings.

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

14 REINITIATION OF CONSULTATION

This concludes NMFS's formal consultation on the proposed actions. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary federal action agency involvement or control over the action has been retained, or is authorized by law, and if:

(1) the amount or extent of incidental take is exceeded,

(2) new information reveals effects of the agency action on listed species or designated critical habitat in a manner or to an extent not considered in this Opinion,

(3) the agency action is subsequently modified in a manner that causes an effect on the listed species or critical habitat not considered in this Opinion,

(4) a new species is listed or critical habitat designated that may be affected by the action, or

(5) the redesigned project includes an in-water component that will result in take of ESA-listed corals or damage to elkhorn and staghorn coral critical habitat.

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