NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7
BIOLOGICAL OPINION

Title: Biological Opinion on the 2018 Virgin Islands Water Quality Standards

Consultation Conducted By: Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce

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ABRT – *Acropora* Biological Review Team

ATON – Aid to Navigation

BE – Biological Evaluation

CAS – Chemical Abstracts Service

CCC – Criterion Continuous Concentration

CFMC – Caribbean Fishery Management Council

CFU – Colony Forming Units

CITES – Convention on International Trade in Endangered Species of Wild Flora and Fauna

CMC – Criterion Maximum Concentration

CO_2 – Carbon Dioxide

CRCP – Coral Reef Conservation Program

CWA – Clean Water Act

DDT – Dichlorodiphenyltrichloroethane

DEP – Department of Environmental Protection

DO – Dissolved Oxygen

DPS – Distinct Population Segment

DWH – Deepwater Horizon

EC## – Effect Concentration at Which xx Percent of Test Organisms Show an Adverse Response

ECOTOX – Ecotoxicology KnowledgeBase

EEZ – Exclusive Economic Zone

EPA (or USEPA) – U.S. Environmental Protection Agency

ESA – Endangered Species Act

FFWCC – Florida Fish and Wildlife Conservation Commission

FMP – Fishery Management Plan

FP – Fibropapilloma

GHG – Greenhouse Gas

HC5 – Hazardous Concentration for 5 Percent of the Species

HDR_5 – Hazardous Dose Rate

IAEA – International Atomic Energy Agency

ITS – Incidental Take Statement

IUCN – International Union for Conservation of Species

LC50 – Lethal Concentration for 50 Percent of Organisms

LOEC – Lowest Observed Effects Concentration

MATC – Maximum Acceptable Toxic Concentration

MOA – Memorandum of Agreement

MPA – Marine Protected Area

MST – Microbial Source Tracking

N - Nitrogen

NCRMP – National Coral Reef Monitoring Program

NLAA – Not Likely to Adversely Affect

NMFS – National Marine Fisheries Service

NOAA – National Oceanic and Atmospheric Administration

NOEC – No Observed Effects Concentration

NPDES – National Pollutant Discharge Elimination System

NPS – National Park Service

NTU – Nephelometric Turbidity Unit

NYSDEC – New York State Department of Environmental Conservation

OPR – Office of Protected Resources

P - Phosphorus

PAR – Photosynthetically Active Radiation

PCB – Polychlorinated Biphenyls

PFC – Perfluorinated Chemicals

RC – Restoration Center

RPM – Reasonable and Prudent Measure

SE – Standard Error

SERO – Southeast Regional Office

STORET – STOrage and RETrieval

TCRMP – Territorial Coral Reef Monitoring Program

TEWG – Technical Expert Working Group

TMDL – Total Maximum Daily Load

TN – Total Nitrogen

TP – Total Phosphorus

TPDES – Territorial Pollutant Discharge Elimination System

UNSCEAR – United Nations Scientific Committee on the Effects of Atomic Radiation

USACE – U.S. Army Corps of Engineers

USCG – U.S. Coast Guard

USFWS – U.S. Fish and Wildlife Service

USGAO – U.S. Government Accountability Office

USVI – U.S. Virgin Islands

UVI – University of the Virgin Islands

VIDPNR – Virgin Islands Department of Planning and Natural Resources

VIWQS – Virgin Islands Water Quality Standards

VIWQSR – Virgin Islands Water Quality Standards Regulations

WQS – Water Quality Standards

WQSR – Water Quality Standards Regulations

1 INTRODUCTION

The Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. 1531 et seq.) establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat they depend on. Section 7(a) (2) of the ESA requires Federal agencies to insure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitat. Federal agencies must do so in consultation with National Marine Fisheries Service (NMFS) for threatened or endangered species (ESA-listed), or designated critical habitat that may be affected by the action that are under NMFS jurisdiction (50 C.F.R. §402.14(a)). If a Federal action agency determines that an action "may affect, but is not likely to adversely affect" endangered species, threatened species, or designated critical habitat and NMFS concurs with that determination for species under NMFS jurisdiction, consultation concludes informally (50 C.F.R. §402.14(b)).

Section 7(b) (3) of the ESA requires that at the conclusion of consultation, NMFS provides an opinion stating whether the Federal agency's action is likely to jeopardize ESA-listed species or destroy or adversely modify designated critical habitat. If NMFS determines that the action is likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS provides a reasonable and prudent alternative that allows the action to proceed in compliance with section 7(a) (2) of the ESA. If an incidental take is expected, section 7(b) (4) requires NMFS to provide an incidental take statement (ITS) that specifies the impact of any incidental taking and includes reasonable and prudent measures to minimize such impacts and terms and conditions to implement the reasonable and prudent measures (RPMs).

The action agency for this consultation is the U.S. Environmental Protection Agency (EPA) Region 2. The EPA proposes the approval of the 2018 Virgin Islands Water Quality Standards (VIWQS; Appendix A) under Section 303(c) of the Clean Water Act (CWA) as they relate to the protection of water quality, aquatic life, and wildlife uses as set forth in the U.S. Virgin Islands (USVI) Code of Regulations, Title 12 Chapter 7 put forward by the Virgin Islands Department of Planning and Natural Resources (VIDPNR). These standards include all of the water quality standards included in the 2015 USVI's WQS regulations (VIWQSR) and subsequent water quality standards being proposed for adoption during 2018 WQSR triennial review process.

This consultation, biological opinion, and ITS, were completed in accordance with section 7(a) (2) of the statute (16 U.S.C. 1536 (a) (2)), associated implementing regulations (50 C.F.R. §§402.01-402.16), and agency policy and guidance and was conducted by NMFS Office of Protected Resources Endangered Species Act Interagency Cooperation Division (hereafter referred to as "we"). This biological opinion (opinion) and ITS were prepared by NMFS Office of Protected Resources Endangered Species Act Interagency Cooperation Division in accordance with section 7(b) of the ESA and implementing regulations at 50 C.F.R. Part 402.

This document represents the NMFS opinion on the effects of these actions on blue, fin, sei, and sperm whales; Nassau grouper; scalloped hammerhead and oceanic whitetip shark; giant manta ray; green (North and South Atlantic Distinct Population Segment [DPS]), hawksbill, leatherback, and loggerhead (Northwest Atlantic DPS) sea turtles; elkhorn, staghorn, pillar, rough cactus, lobed star, mountainous star, and boulder star corals; leatherback sea turtle critical habitat; and elkhorn and staghorn coral critical habitat. A complete record of this consultation is on file at the NMFS Office of Protected Resources (OPR) in Silver Spring, Maryland.

1.1 Background

Water quality standards (WQS) established under the CWA are intended to protect public health and welfare; enhance the quality of water; restore and maintain the chemical, physical, and biological integrity of state waters; and provide water quality protection and propagation of fish, shellfish, and wildlife, and recreation in and on the water.

Section 303(c) of the CWA requires that, at least once every three years, states, tribes, and territories review and, when necessary, modify their WQS or adopt new WQS to protect waters under their jurisdiction. As required by Section 303(c) of the CWA and 40 CFR Part 131, EPA reviews state and territorial WQS and these are not considered in effect under the CWA until approved by EPA.

Water quality standards are regulations that include designated uses and narrative or numeric water quality criteria to protect those uses. Narrative water quality criteria describe the desired conditions of a water body as being "free from" certain negative conditions. Numeric water quality criteria are the maximum allowable concentrations of pollutants. The uses designated for state waters inform the narrative and numeric water quality criteria that will apply for each use. Section 303(c) (2) (B) of the CWA requires states adopt numeric criteria for all toxic pollutants for which criteria have been published under Section 304(a). The numeric criteria are also used to determine use attainment of waters through monitoring, permitting limits for point and non-point source discharges to waters, and in setting loading limits to restore pollution-impaired waters.

The goal of the 2001 Memorandum of Agreement (MOA) between the EPA, NMFS, and the U.S. Fish and Wildlife Service (USFWS) is to enhance coordination under the CWA and the ESA for section 7 consultations. EPA consults with the Services on newly proposed and/or revised state aquatic life criteria to ensure any state-adopted aquatic life criteria are protective of ESA-listed species and designated critical habitats that occur in waters under that state's jurisdiction and have a WQS description that includes the protection and propagation of fish, shellfish, and wildlife.

1.2 Consultation History

The most recent revisions to the VIWQS were adopted by the USVI on August 28, 2015. These revisions were approved by EPA on May 2, 2016. EPA’s original request for informal consultation with NMFS was made in 2009 for the 2010 WQSR, which were adopted by the USVI on June 11, 2010 and approved by EPA on June 29, 2010. This 2009 request for informal consultation with NMFS was made for specific sections of the 2010 VIWQSR revisions (including only new and revised water quality standards), which was not completed in part because NMFS informed EPA that consultation was necessary on the entire document rather than specific provisions.

Beginning in April 2010, NMFS Southeast Regional Office (SERO) began providing technical assistance to EPA in anticipation of the initiation of consultation for the 2014 VIWQSR. The 2014 consultation initiation letter from EPA requested informal consultation on the following provisions of the VIWQS:

- §186-1. Definitions (terms relative to protection of aquatic life only)
- §186-2. Classification of Territorial Waters (Water Classes I, A, B, and C)
- §186-3. Legal Limits (Water Classes I, A, B, and C)
- §186-4. Classification of Water Designated Uses (Water Classes I, A, B, and C)
- §186-5. General Water Quality Criteria (Section (a) and (b) (1) (A), Tables I and III)
- §186-6. Thermal Policy
- §186-7. Mixing Zones
- §186-10. Applicability of Standards
- §186-11. Natural Conditions
- §186-13. Site-Specific Criteria (Sections (a) and (b))
- §186-14. Variances

This consultation now includes consultation on new and revised water quality standards and existing standards that were previously adopted by the VIDPNR and were included in the draft 2014 WQSR. As a part of this consultation, EPA submitted the draft Biological Evaluation to NMFS in December of 2015. SERO transferred the consultation to OPR in January 2017. At that time, OPR and EPA decided to change the scope of the consultation to also cover the 2018 draft VIWQS revisions that included the adoption of a new marine criterion for total nitrogen (TN) and a more stringent standard for clarity (applicable in areas where coral reefs are located), in addition to previously revised temperature and turbidity criteria adopted by DPNR in 2015. In a February 28, 2019 letter, in response to NMFS analysis of human-health criteria for fecal coliform bacteria, EPA expressed its position that the new human health-based criterion for fecal bacteria included in the VIWQS did not fall within EPA's scope of action under ESA section 7(a)(2); therefore, EPA was not required to undertake ESA consultation for this criterion.

This opinion is based on information provided by EPA and the VIDPNR, including the *Biological Evaluation for the National Marine Fisheries Service in regards to the U.S. Virgin Islands Water Quality Standards* (EPA 2015) and *Appendix A: Critical Habitats for Corals in the US Virgin Islands* (EPA 2016) prepared by EPA Region 2, Clean Water Division. Our communication with EPA regarding this consultation is summarized as follows:

- **July 30, 2014:** NMFS SERO received the ESA section 7 consultation initiation request and a copy of the draft 2014 VIWQS.

- **November 25, 2014:** NMFS SERO sent a request for additional information to EPA.
- **December 29, 2015:** NMFS SERO received the Biological Evaluation (BE) prepared by EPA in response to our request for additional information.
- **April 8, 2016:** NMFS participated in a water quality meeting at EPA Region 2 and discussed the BE and remaining information needs to move forward with the consultation with the EPA point-of-contact.
- **May 6, 2016:** EPA sent an appendix to the BE that addresses potential impacts of the VIWQS on elkhorn and staghorn coral critical habitat.
- **January 2017:** SERO closed out the consultation for the 2014 VIWQS and transferred consultation responsibilities for the VIWQS to NMFS OPR.
- **May 17, 2017:** OPR and EPA discussed updating the consultation request to address the 2018 VIWQS revisions because the 2014 triennial WQSR review process had already been completed. NMFS sent an email follow up to the discussion with comments on the proposed VIWQS.
- **May 18, 2017:** EPA sent an updated version of the 2018 draft VIWQS to NMFS.
- **May 30, 2017:** VIDPNR formed a nutrient criteria working group for the VIWQS revisions and invited NMFS to participate via email. VIDPNR also sent a summary of marine numeric nutrient criteria that have been adopted by other states and territories and approved by EPA and a copy of the *Analysis of Existing Water Quality Data from the United States Virgin Islands under Nutrient Scientific Technical Exchange Partnership Support (N-STEPS)* report prepared by Tetra Tech Inc. (2017) for EPA. The report provided a detailed explanation of the derivation process for the TN criterion proposed for adoption in 2018.
- **June 13, 2017:** NMFS sent a consultation initiation letter to EPA.
- **June 15, July 12, and September 11, 2017:** NMFS participated in a nutrient criteria working group call to discuss the proposed 2018 VIWQS nutrient standards for nitrogen and reevaluation of the existing phosphorus standard.
- **July 10 and 27, 2017:** NMFS received responses from EPA and VIDPNR to comments regarding water quality criteria adjustments to 2018 VIWQS that will be more stringent than existing criteria and the 2017 N-STEPS report related to the new TN criterion.
- **July 11, 2017:** EPA sent a letter confirming the change in consultation scope to address the 2018 VIWQS.
- **October 26, 2017:** NMFS and EPA had a call to discuss the consultation time line in light of changes to the schedule due to the hurricanes, which changed the dates anticipated for opening a public comment period for the VIWQS.
- **December 8, 2017:** NMFS received an updated version of the 2018 VIWQS and proposed work plan for VIDPNR to further analyze data and add water quality monitoring to coral reef monitoring programs in order to determine whether and what

additional changes to the VIWQS might be needed and should be evaluated by the VIDPNR during the next triennial WQS review process scheduled for 2021.

- **March 6, 2018:** NMFS received an email from the VIDPNR requesting any final comments on the VIWQS because they were about to release them for public comment. NMFS responded to request on March 7, 2018, via email.
- **July 31, 2018:** NMFS received an email notification from EPA that the 60-day public comment period started that day and would close October 4, 2018.
- **December 11, 2018:** EPA sent NMFS a marked up copy of the draft opinion and an additional document with further discussion of their comments regarding the RPMs included in the ITS via email. EPA also sent the revised 2018 VIWQS incorporating the minor revisions made in response to the limited public comments VIDPNR received during the public comment period.
- **January 28, 2019:** Consultation resumed on this day after being held in abeyance for 38 days due to a lapse in appropriations that resulted in a partial government shutdown.
- **February 5, 2019:** NMFS, EPA and VIDPNR held a conference call to discuss EPA and VIDPNR's comments on the draft opinion and NMFS responses.
- **February 13, 2019:** NMFS and EPA held a conference call to continue the discussion regarding EPA's comments on human health criteria for fecal bacteria and NMFS analysis of these criteria in the draft opinion.
- **February 28, 2019:** EPA sent a letter to NMFS via email expressing EPA's position that human-health-based criteria for fecal bacteria are not part of this consultation.
- **March 8, 2019:** NMFS sent EPA our written responses to their comments on the draft opinion via email.
- **March 22, 2019:** EPA sent an email to NMFS regarding our comment responses indicating their agreement with most of the responses and requesting clarification on one of the responses related to an RPM.
- **March 26, 2019:** NMFS, EPA and VIDPNR held a conference call to discuss the RPM for which EPA had requested additional clarification.
- **March 27, 2019:** NMFS sent EPA an email with revisions to the language of the RPM as discussed during the March 26 conference call.
- **March 28, 2019:** EPA sent NMFS an email stating that they accepted the revisions to the RPM language.

2 THE ASSESSMENT FRAMEWORK

Section 7(a) (2) of the ESA requires Federal agencies, in consultation with NMFS, to ensure that their actions are not likely to jeopardize the continued existence of endangered or threatened species; or adversely modify or destroy their designated critical habitat.

"*Jeopardize the continued existence of*" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of an ESA-listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 C.F.R. §402.02).

"Destruction or adverse modification" means a direct or indirect alteration that appreciably diminishes the value of designated critical habitat for the conservation of an ESA-listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features (50 C.F.R. §402.02).

This ESA section 7 assessment involves the following steps:

Description of the Proposed Action (Section 3): In the case of this consultation, we provide a general description of the VIWQS and actions expected to be implemented in the future that incorporate the VIWQS.

Interrelated and Interdependent Actions (Section 4): We identify any interrelated and interdependent actions associated with EPA's approval of the 2018 VIWQS.

Action Area (Section 5): We define the action area based on the spatial extent of potential effects or stressors from the proposed action.

Stressors Associated with the Proposed Action (Section 6): We discuss the potential stressors we expect to result from the proposed action.

Status of Endangered Species Act Protected Resources (Section 7): We identify the ESA-listed species and designated critical habitat that are likely to co-occur with those stressors in space and time and evaluate the status of those species and habitat. In this Section, we also identify those *Species and Designated Critical Habitat Not Likely to be Adversely Affected* (Section 7.1), and those *Species and Designated Critical Habitat Likely to be Adversely Affected* (Section 7.2).

Environmental Baseline (Section 8): We describe the environmental baseline in the action area including: past and present impacts of Federal, state, or private actions and other human activities in the action area; anticipated impacts of proposed Federal projects that have already undergone formal or early section 7 consultation, and impacts of state or private actions that are contemporaneous with the consultation in process.

Effects of the Action (Section 9): In Sections 9.1 and 9.3 (*Exposure and Response Analysis*), respectively, we identify the number, age (or life stage), and gender of ESA-listed individuals that are likely to be exposed to the stressors and the populations or subpopulations to which those individuals belong. We also consider whether the action "may affect" designated critical habitat. We include a section (9.2) for stressors that are not likely to adversely affect those species and critical habitat for which other stressors were determined likely to adversely affect. We evaluate the available evidence, using data from surrogate species when necessary, to determine how individuals of those ESA-listed species are likely to respond given their probable exposure and consider how the action may affect designated critical habitat, which is our exposure analysis. We also determine the responses of ESA-listed species and critical habitat to exposure to stressors. In Section 9.4 (*Risk Analysis*), we assess the consequences of these responses of individuals that are likely to be exposed to the populations those individuals represent, and the species those populations comprise. The risk analysis also considers the impacts of the proposed action on the essential habitat features and conservation value of designated critical habitat.

Specifically, for this action, we are evaluating whether adverse effects will occur in ESA-protected species and essential features of designated critical habitats affected by water quality conditions resulting from application of existing and proposed water quality standards to USVI jurisdictional waters. This includes exposure to the maximum allowable pollutant concentrations and acceptable range of physical water quality parameters (e.g., temperature, pH) under the proposed and existing VIDPNR criteria. The only exposure factor that influences the evaluation is the manner and extent to which widely ranging ESA-protected species rely on affected waters. Exposure estimates are not relevant to this evaluation because the criteria specify the pollutant exposure concentrations and range in physical water quality characteristics that require evaluation. The probability of exposure based on monitoring data and current pollutant sources is not relevant to this evaluation because these criteria will be in effect indefinitely. Meanwhile exposure probability in waters subject to the criteria will change as populations, technologies, economies, land use, and chemical regulations change. For example, if a criterion for plutonium were set at a concentration that would adversely affect aquatic life, NMFS could make a determination that EPA's approval of the criterion may affect, but is not likely to adversely affect (NLAA) ESA-listed species if we determined that exposure was extremely unlikely to occur. However, future industries using plutonium to make pacemakers or thermoelectric generators could legally discharge plutonium at harmful levels until the VIDPNR revises the criterion.

Cumulative Effects (Section 10): Cumulative effects are the effects to ESA-listed species and designated critical habitat of future state or private activities that are reasonably certain to occur within the action area (50 C.F.R. §402.02). Effects from future Federal actions that are unrelated to the proposed action are not considered because they require separate ESA section 7 compliance.

Integration and Synthesis (Section 11): In this section, we integrate the analyses of *Effects of the Action* (Section 9), the *Environmental Baseline* (Section 8), and the *Cumulative Effects* (Section 10) to formulate the agency's biological opinion as to whether the action is likely to appreciably reduce the likelihood of survival and recovery of an ESA-listed species in the wild or reduce the conservation value of designated critical habitat.

Conclusion (Section 12): With full consideration of the status of the species and the designated critical habitat, we consider the effects of the action within the action area on populations or subpopulations and on essential habitat features when added to the environmental baseline and the cumulative effects to determine whether the action could reasonably be expected to:

- Reduce appreciably the likelihood of survival and recovery of ESA-listed species in the wild by reducing its numbers, reproduction, or distribution, and state our conclusion as to whether the action is likely to jeopardize the continued existence of such species; or
- Appreciably diminish the value of designated critical habitat for the conservation of an ESA-listed species, and state our conclusion as to whether the action is likely to destroy or adversely modify designated critical habitat.

If, in completing the last step in the analysis, we determine that the action under consultation is likely to jeopardize the continued existence of ESA-listed species or destroy or adversely modify designated critical habitat, then we must identify reasonable and prudent alternative(s) to the action, if any, or indicate that to the best of our knowledge there are no reasonable and prudent alternatives. See 50 C.F.R. §402.14.

In addition, we include an *Incidental Take Statement* (Section 13) that specifies the impact of the take, reasonable and prudent measures to minimize the impact of the take, and terms and conditions to implement the reasonable and prudent measures. ESA section 7(b)(4); 50 C.F.R. §402.14 (i). We also provide discretionary *Conservation Recommendations* (Section 13) that may be implemented by the action agency; 50 C.F.R. §402.14(j). Finally, we identify the circumstances in which *Reinitiation of Consultation* is required (Section 15); 50 C.F.R. §402.16.

2.1 Framework for Evaluating Aquatic Life Criteria for Toxic Pollutants

For the aquatic life criteria for toxicants, we first describe how these criteria are developed in order to explain how criteria development can lead to setting a criterion at concentration that may be harmful to ESA-listed species. For stressors that cause toxic effects, exposure intensity (the concentration, duration, and frequency of exposure) determines whether effects occur, and the severity of those effects. For this reason, criteria for the protection of aquatic life are based on exposure concentrations over a specified duration and frequency at and below which ecologically relevant effects are not expected to occur. Aquatic life criteria for toxic pollutants consist of two concentrations – the criterion maximum concentration (CMC, highest concentration of a chemical in water that aquatic organisms can be exposed to acutely without causing an adverse effect) that is intended to protect against severe acute effects during short-term exposure, and the criterion continuous concentration (CCC, highest concentration of a chemical in water that aquatic organisms can be exposed to indefinitely without resulting in an adverse effect) that is intended to protect against effects on survival, growth, and reproduction over longer exposures. The EPA applies restrictions to the types of data that may be used in deriving criteria. Data that are not acceptable for criteria development include those from tests that did not contain a control treatment, tests in which too many organisms in the control treatment died or showed signs of stress or disease, or data from tests using species that do not have reproducing wild populations in North American waters. Most of the criteria are developed consistent with the 1985 EPA water quality guidelines that are based on Stephen et al.’s paper

(Stephen et al. 1985) using endpoints identified through toxicity tests exposing laboratory-reared organisms to toxicants over a range of concentrations.

The endpoints that may be used in criteria development include:

- the concentration at which half of the exposed organisms die (lethal concentration for 50 percent of organisms, LC50)
- the lowest exposure at which a given effect did not differ from controls (no observed effects concentration, NOEC)
- the lowest exposure at which the effect differed significantly from controls (lowest observed effects concentration, LOEC)
- the effect concentration (EC) at which a certain proportion of an effect was observed (EC##, such as EC10 = concentration at which 10 percent of test organisms show an adverse response).
- the maximum acceptable toxic concentration (MATC), which is typically the geometric mean of the LOEC and NOEC, but other calculations have been used.

The NOECs and LOECs are not ideal measures of sublethal effects because they are influenced by study design. Depending on the number and distribution of exposures tested and underlying variability in responses, a NOEC could actually represent a 35 percent difference in response from controls in a poorly designed study. An EC## reflecting a biologically significant sublethal response threshold (i.e., 1 percent, 5 percent) would be a more suitable sublethal endpoint. Unfortunately, rigorously derived EC## data are rare and EC## calculated from dose-response relationships of existing toxicity tests are usually accompanied by very broad confidence intervals.

For the CMC, the cumulative distribution of species average LC50s from at least eight taxonomic families is used to identify the concentration at which 5 percent of species would experience 50 percent or greater mortality. This is also referred to as the final acute value, the 5th percentile, or the HC5 (hazardous concentration for five percent of the species) of the “species sensitivity distribution.” The CMC is set at half of the HC5 and is expected to fall below a concentration where any acute adverse effects to most organisms occur.

If suitable LOECs, NOECS, and EC##s for chronic exposure responses are available for a toxicant, they may be used to derive a CCC. The EPA requires that the types of effects applied be related to organism-level consequences. This often excludes behavioral effects that influence survival in the wild such as swimming, predator evasion, and nest tending. Sufficient data for chronic toxicity are rarely available for deriving criteria as described for CMC. Flaws in the NOECs and LOECs used to derive criteria has been estimated to result in a level of protection closer to 90 percent rather than 95 percent of species as specified by the guidelines (Crane and Newman 2000; NMFS 2012a). The EPA guidelines provide several approaches for deriving CCC depending on the quantity and quality of the available data. These approaches can include extrapolations from acute toxicity data. In some cases, the available data are so sparse a CCC cannot be derived.

The EPA guidelines are designed to arrive at CMC and CCC values that, when applied as discharge limits, monitoring thresholds, and restoration goals, will achieve water quality that provides for the protection and propagation of fish, shellfish, and wildlife and provide for recreation in and on the water as stated in Section 1.1. The EPA guidelines state:

> *"Because aquatic ecosystems can tolerate some stress and occasional adverse effects, protection of all species at all times and places it is not deemed necessary for the derivation of a standard. ...[given adequate data]... a reasonable level of protection will probably be provided if all except a small fraction of the taxa are protected, unless a commercially or recreationally important species is very sensitive."*

Because the criteria developed using the 1985 water quality guidelines are not expected to protect all species under all circumstances, waters compliant with the criteria may result in pollutant exposures that cause adverse effects in threatened and endangered species. Studies comparing the sensitivities of threatened and endangered species relative to species commonly used in laboratory toxicity tests suggest that multiplying the National Recommended Water Quality Criteria developed using the EPA guidelines by a generic adjustment factor of about 0.5 provides a limit that would protect ESA-listed species (Dwyer et al. 2005; Sappington et al. 2001; Besser et al. 2005). However, this adjustment is based on differences in the physiological sensitivity of freshwater bony fish to a toxic pollutant. This factor may not be appropriate for important behavioral effects, other types of stressors like sediment or dissolved oxygen (DO), or other species groups, like corals or cartilaginous fish (i.e., sharks, rays). In addition, when assessing risk to an ESA-listed species, the vulnerability of an imperiled population of that species to the loss of an individual, or key individuals, amplifies the fundamental threat posed by a toxic pollutant.

The underlying assumptions in the methods used to arrive at criteria affect how well ESA-listed species and designated critical habitat are protected. These assumptions include:

- Effects that occur on a species exposed to a toxicant in laboratory tests will generally be the same for the same species exposed to that toxicant under field conditions (i.e., effects are not influenced by predation, competition, disease, exposure to other stressors in the field, and fluctuations in natural water quality parameters).
- Collections of single-species laboratory toxicity test data used to derive criteria reflect communities in natural ecosystems.
- Data on severely toxic effects from short-term "acute" toxicity tests used to derive CMC can be extrapolated to less severe effects that would be expected to occur in long-term "chronic" exposures to derive CCC.
- Loss of a small number of species will not affect the propagation of fish, shellfish, and wildlife.

- Loss of a small number of species will not result in incidental loss of any "economically or recreationally valuable species" for which data were not available.
- Sensitive species and life stages are adequately represented such that criteria are not biased.
- Derivation of criterion for a single chemical in isolation without regard to the potential for additive toxicity or other chemical or biological interactions is acceptable despite chemicals typically occurring in mixtures in the environment.
- When applied to National Pollutant Discharge Elimination System (NPDES) permits, unless the waters are already identified as impaired by a pollutant, the waters are free from that pollutant.
- Accumulation of chemicals in tissues and along the food web does not result in ecologically significant latent toxicity or toxic exposures for predators.

There are also concerns about the underlying data used in the derivation of criteria including:

- Data sets for sublethal responses are usually small and have gaps such that sensitive species and life stages are under-represented.
- Variability within and among species used in calculating an HC5 may be substantial, but this variability is not reflected in the final HC5 estimate used to derive a CMC.
- The NOEC for one type of sublethal response (e.g., growth) may be the LOEC for a different type of sublethal response (e.g., brood size) in the same study.

2.2 Evidence Available for the Consultation

To comply with our obligation to use the best scientific and commercial data available, we collected information identified through searches of Google Scholar, Web of Science, Peer reviewed articles and their literature cited sections, species listing documentation, and reports published by government and private entities. This opinion is based on our review and analysis of various information sources, including:

- Information submitted by EPA and the applicant, including water quality monitoring data
- Water quality monitoring reports generated from permitted construction projects in USVI
- Government reports (including NMFS biological opinions and stock assessment reports)
- NOAA technical memos
- Peer-reviewed scientific literature
- Toxicity data from peer-reviewed literature and the EPA Ecotoxicology KnowledgeBase (ECOTOX)

These resources were used to identify information relevant to the potential stressors and responses of ESA-listed species and designated critical habitat under NMFS jurisdiction that may be affected by the proposed action to draw conclusions on risks the action may pose to the continued existence of these species and the value of designated critical habitat for the conservation of ESA-listed species.

2.2.1 Selecting Data for Evaluating Toxics Criteria

It is important to note that EPA's use of data for criteria derivation and associated regulatory actions is not the same as NMFS use of data for this consultation. For an ESA consultation, NMFS is required to use "the best scientific and commercial data available" (ESA § 7(a)(2)). The data guidelines for criteria development in Stephen et al. (1985) were arrived at with freshwater criteria in mind and an overwhelming majority of toxicity data are for freshwater species. However, the guidelines in Stephen et al. (1985) do allow for the use of "Other Data." NMFS interprets this section of the guidelines to indicate that our use of other data is appropriate for evaluating the applicability of the criteria to our species. The guidelines in Stephen et al. (1985) state that data used to derive criteria should include only species that are native to waters of the US. That means data on effects to corals of the same coral genus as our listed corals that occur only in foreign waters, possibly elsewhere in the Caribbean, would be excluded. Exclusion of such data is not appropriate for this opinion. All data needs to be considered in order to address the data gaps for marine species and to meet the ESA requirement of using best available data to analyze potential effects to listed species.

The BE provided by EPA includes summaries of toxicity data that were used to evaluate whether proposed criteria may result in harm to ESA-listed species and designated critical habitat. Our assessment considers these summaries, but also uses other data found in EPA's ECOTOX, the open literature, particularly data published since the criteria were developed[1] to include data that provide insight on effects for ESA-listed species, but were otherwise not available or considered suitable for the derivation of criteria. Use of additional data when vetting the criteria for effects to ESA-listed species is consistent with EPA's guidelines and the requirement under the ESA that determinations be made based on the best available data.

Our data extraction from ECOTOX used the Chemical Abstracts Service (CAS) numbers provided in the VIWQS documentation to identify data that were relatable to effects on survival, fitness, and growth for saltwater exposures that had acceptable controls and were collected using organisms of known origins and prior toxicant exposures (e.g., generational studies). Additional CAS numbers were added to the query to identify all data for the chemicals of concern. For example, data for mixtures of α- and β-endosulfan were included because both forms have identical criteria.

Where necessary, metal concentrations were corrected to dissolved form using EPA's recommended conversion factors. We excluded data for static and renewal toxicity tests of highly volatile toxicants when concentrations in the test chamber were not confirmed. This eliminated studies where the actual exposure concentrations were likely much lower than the reported concentrations due to volatilization from the test chamber. We also excluded data for metals toxicity where only the free ion concentration was reported because the metals criteria are

[1] Some of the criteria, such as those for lead, silver, and DDT were developed as long ago as 1980 and more recent data for marine species are now available.

based on the dissolved fraction of the metal (i.e., the sample fraction that will pass through a 0.45-micron filter).

There are no data for sea turtles and whales, toxicity data for other marine species is sparse, and there are few data for the taxonomic families of the ESA-listed species in USVI waters. Given the sparseness of saltwater toxicity data and the need to be protective of ESA-listed species, we apply a comprehensive perspective that considers all data with the expectation that mechanisms of effect are similar to effects in the ESA-listed species based on fundamental physiological functions (e.g., ionic homeostasis, antioxidant defense, nerve function, calcification). While the size difference between adult ESA-listed Nassau grouper and fish species commonly used in toxicity tests is considerable, the common laboratory subjects are comparable in size to the nearshore life stages of Nassau grouper, which are most likely to be exposed to land-sourced pollutants and are expected to be the most sensitive life stage. For these reasons, the species in the body of data applied to this assessment are relied on as surrogates providing evidence supporting effect determinations. Larger datasets with diverse species provide more confidence in extrapolating for effects determinations to ESA-listed species by demonstrating the breadth of expected response thresholds and the degree of variability within and among studies.

We are concerned with Caribbean species so any data on responses of species with reproducing populations in the Caribbean Sea would be particularly applicable to this assessment. Meanwhile, data for processes that do not occur in the species assessed in this opinion, such as instar molting or smoltification would not be applicable. Generally, responses of species that occur in tropical waters worldwide would also be informative. For example, data on foreign coral species of the genus *Acropora* that occur in Pacific waters would be useful to this assessment. Where data are absent for coral, data for invertebrates (mollusks, crustaceans, worms, and other invertebrates) are used to interpret potential effects.

3 DESCRIPTION OF THE PROPOSED ACTION

"Action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by federal agencies. The action is EPA Region 2's approval of the 2018 VIWQS under Section 303(c) of the CWA as they relate to the protection of water quality, aquatic life, and wildlife uses as set forth in the USVI Code of Regulations, Title 12 Chapter 7 put forward by the VIDPNR. Specifically, EPA requested consultation on the following provisions of the VIWQS:

- §186-1. Definitions (terms relative to protection of aquatic life only)
- §186-2. Classification of Territorial Waters (Water Classes I, A, B, and C)
- §186-3. Legal Limits (Water Classes I, A, B, and C)
- §186-4. Classification of Water Designated Uses (Water Classes I, A, B, and C)
- §186-5. General Water Quality Criteria (Section (a) and (b) (1) (A), Tables I and III)
- §186-6. Thermal Policy
- §186-7. Mixing Zones

- §186-10. Applicability of Standards
- §186-11. Natural Conditions
- §186-13. Site-Specific Criteria (Sections (a) and (b))
- §186-14. Variances

In its initiation letter, EPA requested NMFS concurrence on its determination that the proposed VIWQS may affect but are not likely to adversely affect ESA-listed species or designated critical habitat in the USVI. EPA asserted that the proposed VIWQS provide expanded protection to aquatic resources in the Territory and will therefore have a beneficial effect on ESA-listed species. EPA requested that the consultation be limited to the portions of the proposed 2018 VIWQS for which EPA has relevant discretion or control to introduce considerations of impacts on federally listed species or their habitats into its actions. This excludes the following sections from the consultation request: Antidegradation Policy and Implementation Procedures (§186-8), Analytical Procedures (§186-9), Schedules of Compliance (§186-12, Public Review Process (§186-15), and Enforcement (§186-16).

EPA's analysis and conclusions for each of the provisions for which they requested consultation are outlined in their *Biological Evaluation for the National Marine Fisheries Service in Regards to the U.S. Virgin Islands Water Quality Standards* (EPA 2015) and *Biological Evaluation for the National Marine Fisheries Service in Regards to the U.S. Virgin Islands Water Quality Standards, Appendix A: Critical Habitats for Corals in the US Virgin Islands* (EPA 2016), as well as documents prepared to respond to comments from NMFS regarding proposed nutrient criteria, the draft 2018 VIWQS (Appendix A), and the Basis and Background for the 2018 Water Quality Standards Revision for the U.S. Virgin Islands (February 2018; Appendix B) prepared by the VIDPNR.

In the 2014/2015 revisions to the VIWQS, VIDPNR changed the standards for temperature and turbidity to make them more restrictive in areas where corals are present. In the 2018 revisions, VIDPNR has included a numeric standard for total nitrogen applicable to all of USVI's marine and coastal waters, as well as a more stringent criterion for water clarity, applicable to areas where coral are present. Water quality standards vary based on the classification of territorial waters into inland or marine and coastal waters. Marine and coastal waters are further separated into Class A, Class B, and Class C. Therefore, our discussion of the proposed action follows this classification and is taken from the 2018 VIWQS (Appendix A).

The 2014/2015 revisions to the VIWQS also incorporated the EPA's national recommended Clean Water Act section 304(a) water quality criteria for the protection of aquatic life as pollutant criteria for all USVI water classifications (see Appendix A Section 186-5, *General Water Quality Criteria*, tables for aquatic life criteria in saltwater). These criteria were further revised and updated in the 2018 draft WQSR. Other VIWQS include the same criteria regardless of the classification of inland surface, coastal and marine waters. Criteria that fall into this group and could affect aquatic life are human health-based criteria for fecal bacteria for which the 30-

day geometric mean for enterococci shall not exceed 30 colony forming unit per 100 milliliters (CFU/100 ml) and no more than 10 percent of the samples collected in the same 30 days shall exceed 110 CFU/100 ml. By letter dated February 28, 2019, EPA expressed its position that the fecal bacteria criteria were derived and adopted to protect human health and not aquatic life, thus should not be included in this consultation. However, NMFS did include an analysis of the criterion for enterococci because of the scientific data indicating human sewage has health consequences for corals (Sutherland et al. 2010; Lipp et al. 2002; Bonkosky et al. 2009).

Because the ESA requires that we look at all the potential effects of the proposed action on ESA-listed species and designated critical habitat, we consider the act of setting water quality standards and examine the application of the standards in USVI. Once the standards are approved by EPA and implemented by the VIDPNR, pollutants would be allowed to enter USVI waters up to the approved standards, which could lead to effects to ESA-listed species and designated critical habitat. Therefore, as part of the ESA section 7 consultation for the 2018 VIWQS, NMFS must examine the potential effects of the pollutants included in the standards and their discharge to USVI waters in concentrations up to the approved standards, on ESA-listed species and designated critical habitat.

3.1 §186-1 Definitions

EPA's 2014 consultation initiation letter requested that NMFS review the definitions related to protection of aquatic life. NMFS provided comments on some of the definitions relevant to ESA-listed species and their habitat and the VIDPNR revised the definitions per our comments. The definitions and their application by VIDPNR will not result in effects to ESA-listed species or designated critical habitat. Therefore, this section of the VIWQS is not discussed further in this opinion.

3.2 §186-2 Classification of Territorial Waters

Territorial waters are classified as either inland waters or coastal and marine waters, as described in subparagraphs 3.2.1 and 3.2.2, respectively.

3.2.1 Inland waters

Inland waters include groundwater and surface waters that may be fresh or brackish. Freshwater surface waters include streams, guts, and freshwater wetlands, specifically freshwater ponds. Brackish or saline surface waters include inland estuaries and brackish or saline wetlands, specifically swamps, salt flats, salt ponds, mangrove wetlands, and marshes. Groundwater, particularly in wells, is also included in this classification.

3.2.2 Coastal and Marine Waters

All coastal and marine waters are either embayments, open coastal, or oceanic waters including all contiguous saline bays, inlets, coastal estuaries, and harbors in territorial waters. These waters are divided into:

- Class A Waters – Outstanding Natural Resource Waters
- Class B – All other coastal or marine waters not classified as Class A or Class C
- Class C – All other coastal or marine waters not classified as Class A or B

While this section was included in EPA's consultation request, the classification of waters by VIDPNR will not result in effects to ESA-listed species or designated critical habitat. Therefore, this section of the VIWQS is not discussed further in this opinion except when the class of waters is noted because of differences in proposed criteria for some waters such as the turbidity criteria.

3.3 §186-3 Legal Limits

This section provides a legal description and boundaries for Territorial waters, as described in subsections 3.3.1, 3.3.2, 3.3.3, and 3.3.4, respectively.

3.3.1 Class I, Inland Waters

Surface waters whether fresh or saline/brackish are designated for use in the maintenance and propagation of desirable species of wildlife and aquatic life including ESA-listed species and species listed as threatened or endangered under the VI Code, primary contact recreation and as potable water sources.

3.3.2 Class A, Outstanding National Resource Waters

These waters consist of the waters within 0.5 miles of the boundaries of Buck Island Reef National Monument, St. Croix (Figure 1) and Trunk Bay, St. John (Figure 2), which is within the Virgin Islands Coral Reef Monument and National Park.

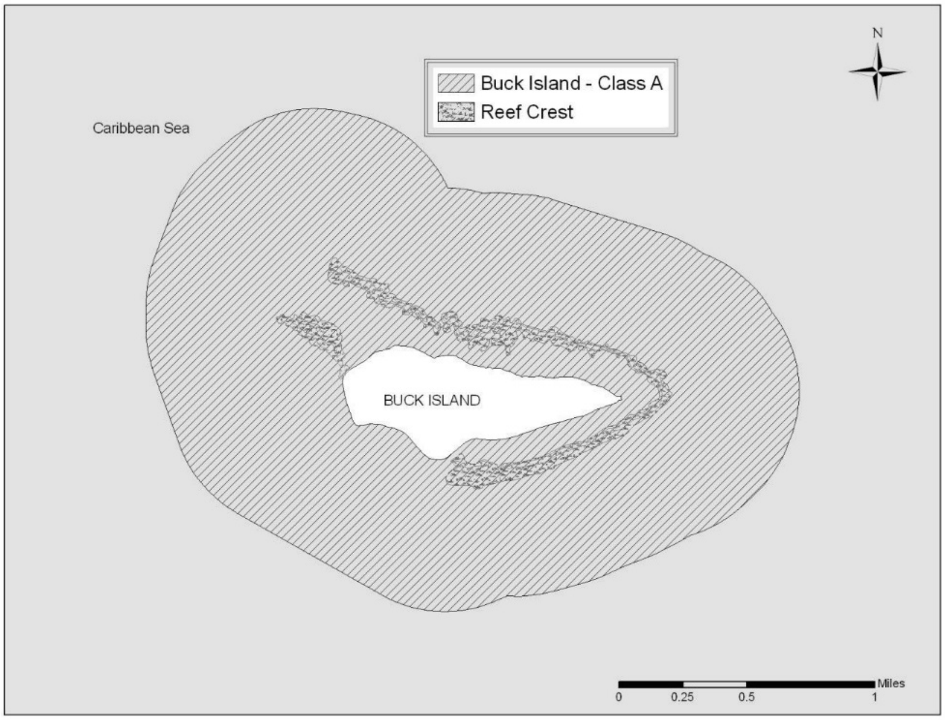

Figure 1. Class A waters around Buck Island, St. Croix (from 2018 VIWQS)

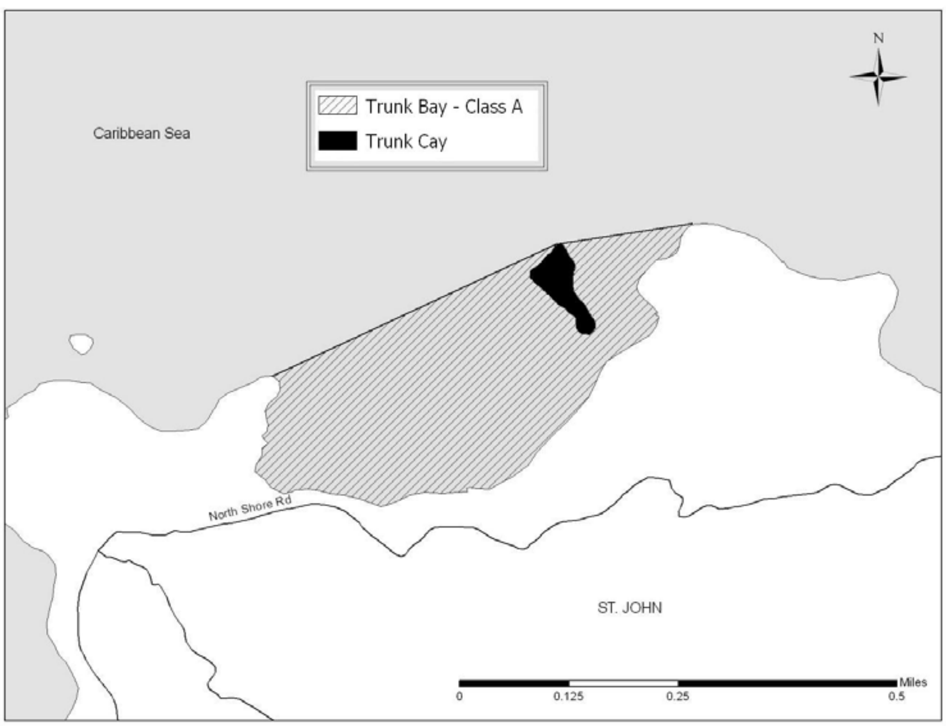

Figure 2. Class A - Trunk Bay, St. John (from 2018 VIWQS)

3.3.3 Class B

These waters include all coastal and marine waters not classified as Class A or Class C.

3.3.4 Class C

This classification is for waters in harbors and industrial areas in USVI.

For St. Thomas, these areas are:

- St. Thomas Harbor beginning at Rupert Rock and extending to Haulover Cut (Figure 3)
- Crown Bay enclosed by a line from Hassel Island at Haulover Cut to Regis Point at West Gregorie Channel (Figure 3)
- Krum Bay (Figure 3)

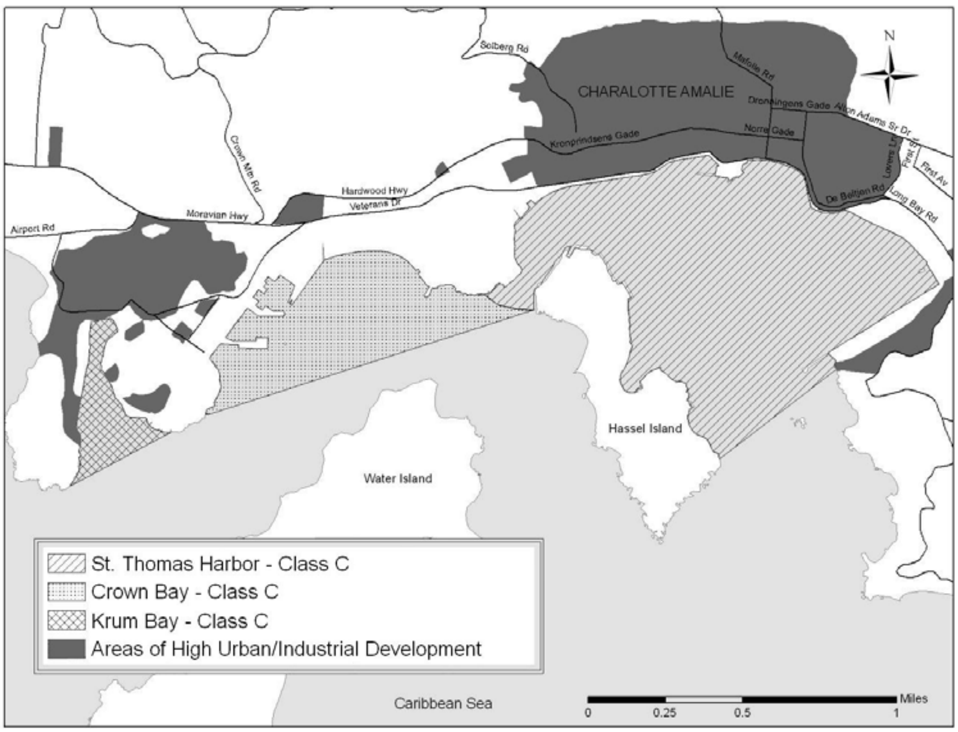

Figure 3. Class C waters off St. Thomas

For St. Croix, Class C waters are:

- Christiansted Harbor from Fort Louise Augusta to Golden Rock, along the waterfront and seaward to include the navigation channels and mooring areas (Figure 4)
- Frederiksted Harbor from La Grange to Fisher Street and seaward to the end of the Frederiksted Pier (Figure 5)
- Hess Oil Virgin Islands Harbor (also called HOVENSA Harbor; Figure 6)
- Martin-Marietta Alumina Harbor (also called Port Alucroix or St. Croix Renaissance Group Harbor; Figure 6)

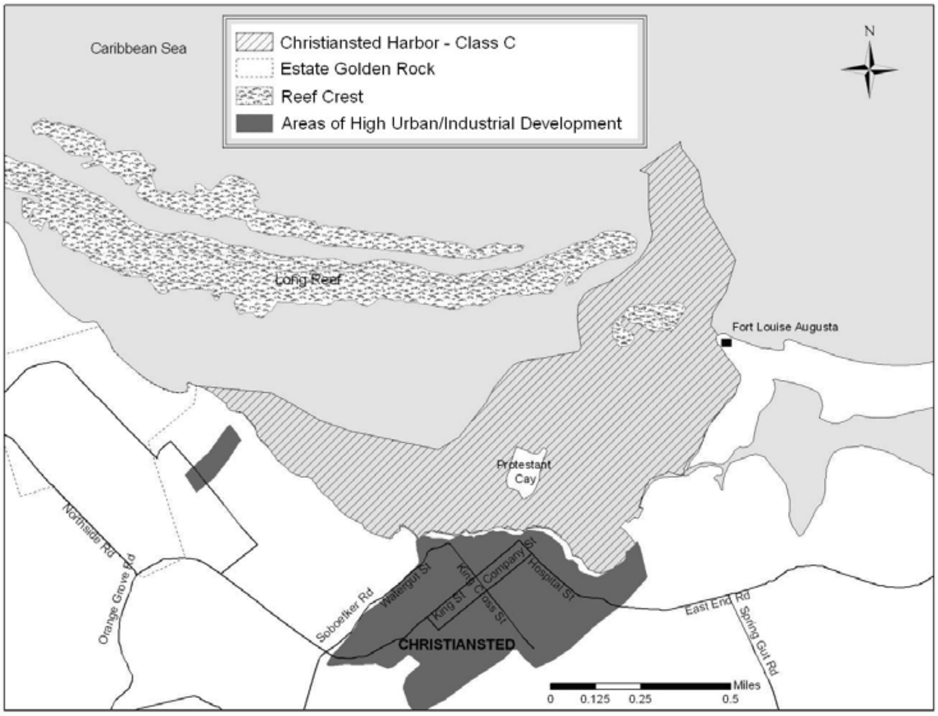

Figure 4. Class C waters in Christiansted, St. Croix

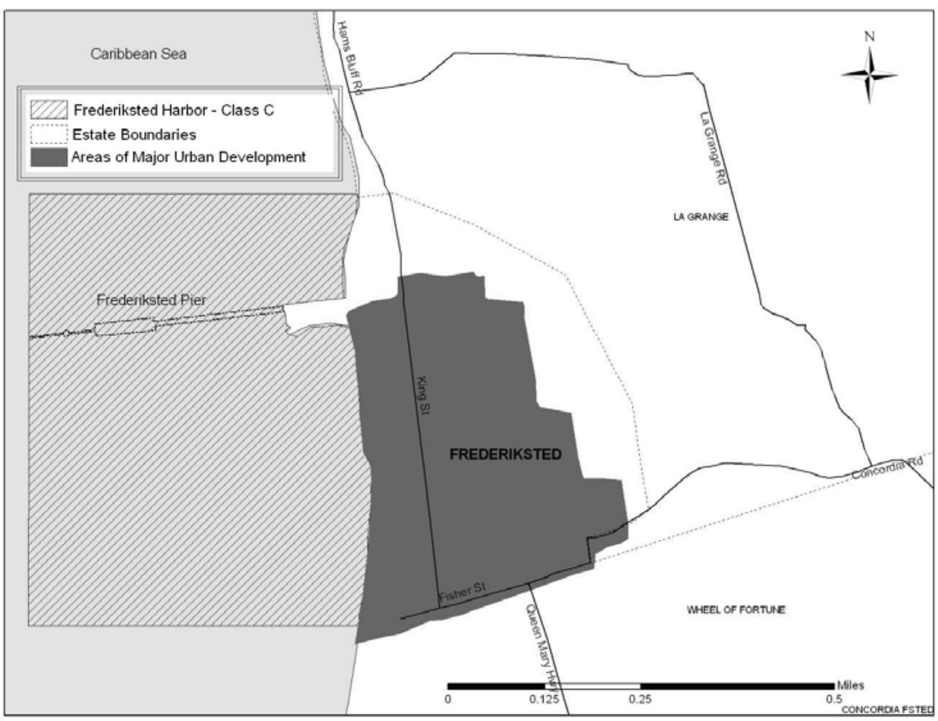

Figure 5. Class C waters off Frederiksted Harbor, St. Croix

Figure 6. Class C waters off southwest St. Croix

For St. John, Enighed Pond Bay (Figure 7) is the only Class C water around the island.

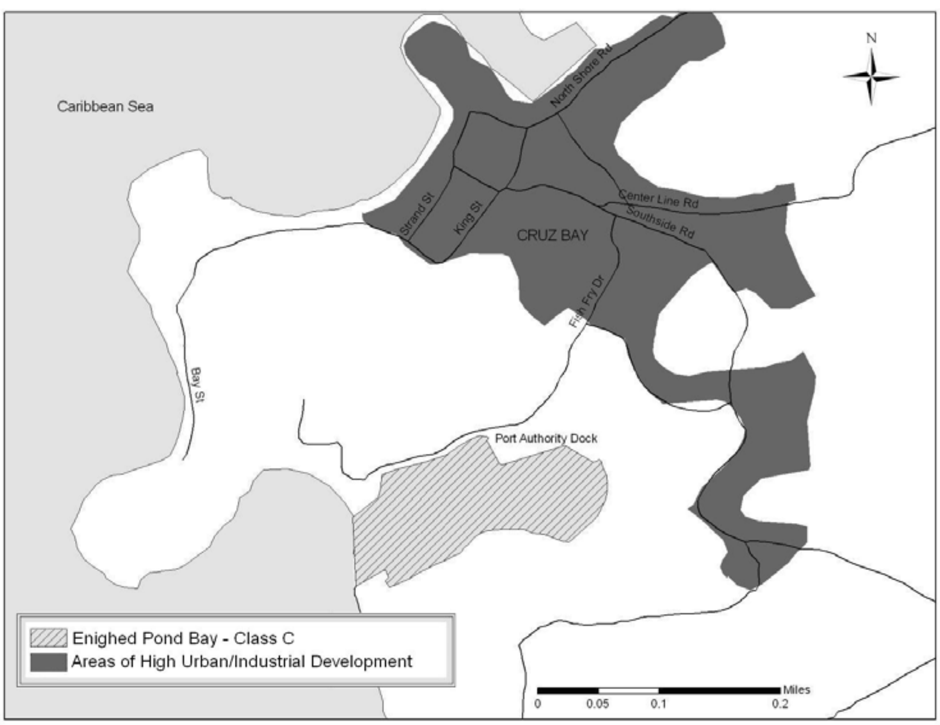

Figure 7. Class C waters in Enighed Pond, St. John

While this section was included in EPA's consultation request, the legal limits of Territorial waters will not result in effects to ESA-listed species or designated critical habitat. Therefore, this section of the VIWQS is not discussed further in this opinion.

3.4 §186-4 Classification of Water Designated Uses

3.4.1 Class of Inland Waters – Inland Fresh, Inland Brackish or Saline, Inland Groundwaters

These waters are designated for use in the maintenance and propagation of desirable species of wildlife and aquatic life including ESA-listed species, primary contact recreation, and as a potable water source, where applicable.

In terms of water quality criteria relevant to aquatic life, these waters must have a DO concentration of not less than 5.5 milligrams per liter (mg/l) except when due to natural forces. The VIDPNR had proposed a new turbidity standard applicable to inland waters but withdrew it until they collect sufficient water quality monitoring data specific to inland surface waters that can be used to establish an appropriate, fully protective standard.

Groundwater is designated for use as a potable water source and none of the criteria are relevant to aquatic life.

3.4.2 Class of Marine and Coastal Waters – Class A, Class B, Class C

Criteria for radioactivity, radium-226 and strontium-90, are included in the VIWQS for all coastal and marine waters. The basis for these criteria is not provided in the BE or other documentation submitted with the consultation request. They are identical to those used by Virginia in 1972 for areas where shellfish occur (USEPA 1972). It is not clear from historical documents whether these criteria were established to protect human health or the propagation of shellfish.

These waters are designated for use in the maintenance and propagation of desirable species of wildlife and aquatic life including ESA-listed species, primary contact recreation, and as a potable water source, where applicable.

Preservation of the unique characteristics of Class A waters is required to maintain their exceptional recreational, environmental or ecological significance. No new or increased discharges will be permitted in these waters.

Water quality criteria specific to Class A waters relevant to the protection of aquatic life include:

- Biocriteria - The biological condition shall be similar or equivalent to reference condition established for biological integrity within Class A waters
- DO **–** Cannot be less than 5.5 milligrams per liter (mg/l) except when due to natural forces
- pH **–** Natural conditions of pH must not be extended by more than +/- 0.1 pH unit and pH shall never be less than 7.0 or greater than 8.3

- Temperature – Cannot exceed 32°C at any time unless as a result of natural conditions and waste discharge cannot result in an increase in temperature greater than 1°C above natural conditions. In areas with coral reef ecosystems, temperature shall not exceed 25-29°C at any time and waste discharge cannot result in an increase in temperature greater than 1°C above natural conditions
- Phosphorus – Phosphorus as total phosphorus (TP) shall not exceed 50 micrograms per liter (µg/L) at any time
- Nitrogen – Nitrogen as total nitrogen (TN) shall not exceed 207 µg/L expressed as an annual geometric mean that shall not be exceeded in more than 10 percent of samples over a three-year period.
- Clarity – A secchi disc shall be visible at a minimum depth of one meter (m). In waters where the depth does not exceed one m, the bottom must be visible. In areas with coral reef ecosystems, a secchi disc shall be visible at a minimum depth of 15 m. In waters in coral reef ecosystems where the depth does not exceed 15 m, the bottom must be visible.
- Turbidity – A maximum of three nephelometric turbidity units (NTU) is permitted. In areas with coral reef ecosystems, a maximum of 1 NTU is permitted.

Water quality criteria for Class B waters relevant to the protection of aquatic life are the same as for Class A waters with the exception of the narrative criteria for biocriteria. For Class B waters, the biological condition shall reflect no more than a minimal departure from reference condition for biological integrity in terms of the structure of the biotic community and ecosystem function. Ecosystem functions must be maintained within the range of natural variability and native taxa must be maintained with some changes in biomass and/or abundance.

In addition, due to natural conditions in certain waterbodies, the turbidity criterion is not applicable to the following Class B waters:

St. Thomas waters:

- Mandahl Bay (marina)
- Vessup Bay
- Water Bay
- Benner Bay
- Mangrove Lagoon

St. Croix waters:

- Carlton Beach
- Good Hope Beach
- Salt River Lagoon (marina)
- Salt River Lagoon (Sugar Bay)
- Estate Anguilla Beach
- Buccaneer Beach

- Tamarind Reef Lagoon
- Green Cay Beach
- Enfield Green Beach

These waters were excluded from turbidity standards in the past and maps indicating the boundaries of these waters are not available. The VIDPNR has developed a work plan that includes the need to address this and define the actual area of these waters as well as determine whether or not their exclusion from turbidity requirements remains appropriate by the end of the next VIWQS triennial review scheduled for 2021. Any changes made as part of the next triennial WQSR review may require a separate ESA consultation and are not part of the current consultation.

Class C waters are designated for use in the maintenance and propagation of desirable species of wildlife and aquatic life including ESA-listed species, primary contact recreation, industrial water supplies, shipping, navigation, and as a potable water source, where applicable. For Class C waters, the biocriteria standard is also different from biocriteria adopted for class A and B waters. In Class C waters, evident changes in structure of the biotic community and minimal changes in ecosystem function are allowed. Evident changes in structure include loss of some rare native taxa and shifts in relative abundance of taxa (community structure) but sensitive-ubiquitous taxa should remain common and abundant and ecosystem functions should be fully maintained.

Water quality criteria specific to Class C waters relevant to the protection of aquatic life that vary from those for Class A and B waters include:

- DO – Cannot be less than 5.0 mg/l except when due to natural forces
- pH – Natural conditions of pH must not be extended by more than +/- 0.1 pH unit and pH shall never be less than 6.7 or greater than 8.5.

3.5 §186-5 General Water Quality Criteria

This section contains the narrative water quality criteria and numeric water quality criteria for toxic pollutants that apply to all Territorial waters. The narrative water quality criteria require all Territorial waters to meet generally accepted aesthetic qualifications and be capable of supporting diversified aquatic life (See Appendix A for details of narrative criteria). Narrative water quality criteria require that Territorial waters be free of substances attributable to municipal, industrial, or other discharges of wastes that include or affect: deposits; matter; turbidity; material; color; suspended, colloidal or settleable solids; oil and floating substances; taste and odor producing substances; substances and/or conditions; nuisance species; and downstream protection. The narrative criteria also include biocriteria that allow for the preservation, protection, and restoration of water resources in Class A, B, and C waters, wetlands, estuaries, mangroves, seagrass, coral reef, and other marine ecosystems.

This section also contains the numeric water quality criteria for toxic pollutants, which have already been adopted or are being proposed for adoption by the VIDPNR according to the most recent EPA national recommended Clean Water Act section 304(a) water quality criteria. The criteria were derived and adopted for the protection of freshwater and saltwater aquatic life from acute and chronic effects and the protection of human health (see tables in Appendix A Section 186-5 for the complete list of aquatic life criteria applicable to saltwater organisms for metals, pesticides, and other organic and inorganic substances).

3.6 §186-6 Thermal Policy

This section includes the criteria to protect Territorial waters and associated aquatic life, including listed species, from thermal pollution (see Appendix A Section 186-6 for details of thermal policy). The temperature criteria (Section 186-4 of the VIWQS) and associated thermal policy in terms of the temperature and variations in temperature allowed under the thermal policy are analyzed in this opinion.

3.7 §186-7 Mixing Zones

This section of the VIWQS allows for the establishment of mixing zones that apply to the discharge of treated wastewater to surface waters in USVI and establishes criteria for setting a mixing zone. Mixing zones must be limited in area or volume and allow for prompt mixing of the discharge with receiving waters. Mixing zones are not allowed as or to substitute for minimum treatment technology, cannot be used for pathogens or indicators of pathogens, cannot encroach on potable water intakes or areas where sessile organisms like shellfish are harvested, and cannot create nuisance conditions, accumulate pollutants in sediments or biota in toxic amounts, or disproportionately diminish use of surface waters. Mixing zones must ensure no lethality to organisms transiting through them, minimize impacts to aquatic life, and cannot interfere with biological communities or be established for discharges that would likely jeopardize the continued existence of any ESA-listed species or result in the destruction or adverse modification of any designated critical habitat. This section of the VIWQS merely establishes the criteria for creating mixing zones and does not include actual mixing zones that are meant to be protective of ESA-listed species and designated critical habitats. Therefore, we do not discuss mixing zones further in this opinion and instead focus on the potential for effects from water quality criteria that may be in VIDPNR-permitted discharges.

3.8 §186-10 Applicability of Standards

This section discusses the requirement that whatever provision under the VIWQS or other regulation or legal authority is most restrictive be applied in order to be most protective. This section of the VIWQS is not discussed further in this opinion as any effects to ESA-listed species and designated critical habitat associated with the application of this section of the VIWQS would be wholly beneficial.

3.9 §186-11 Natural Conditions

Natural waters may have characteristics outside the limits prescribed by the VIWQS. The criteria do not relate to violations of standards resulting from natural forces. The potential effects of natural forces such as storms and associated sediment transport that could lead to violations of standards in waters of the USVI are discussed in the stressors (Section 6) and environmental baseline (Section 8) of this opinion but the analyses in this opinion focus on criteria for permitted discharges and other applications of the VIWQS by the VIDPNR. This section is not discussed further in this opinion.

3.10 §186-13 Site-Specific Criteria

This section allows VIDPNR to modify criteria on a site-specific basis in response to local environmental conditions. Modifications must result in criteria that are still protective of aquatic life, be based on sound scientific rationale, and cannot result in jeopardy of any ESA-listed species or damage or adverse modification of any designation critical habitat. Less stringent aquatic life criteria may be developed when local water quality characteristics such as pH, hardness, temperature, or other physical characteristics alter the biological availability or toxicity of a pollutant or the sensitivity of species that occur at a particular site differs from species tested when developing the criteria. In some cases, modifications should be proposed to be more protective of ESA-listed species to ensure no jeopardy or adverse modification of designated critical habitat. Modifications may also require public review and will require EPA approval. Therefore, this section is not discussed further in this opinion as the criteria are meant to be protective of ESA-listed species and designated critical habitat and site-specific modifications to criteria requiring EPA approval would undergo ESA section 7 consultations.

3.11 §186-14 Water Quality Standards Variances

VIDPNR may allow a variance to criteria for a specific pollutant or pollutants in effluent when a designated use is not attainable in the short-term but might be attainable in the long-term (see Appendix A Section 186-14 for details of the requirements). A temporary modification to a designated use and associated water quality criteria might be allowed as long as the variance does not affect the underlying designated use of a waterbody, will not affect the water quality standard for the waterbody, and is not likely to jeopardize the continued existence of any ESA-listed species or result in the destruction or adverse modification of any designated critical habitat. A variance will not be allowed if the adoption of effluent limitations required under CWA sections 301(b) and 306 and the implementation of best management practices for nonpoint source control allow attainment of standards. Variances will not exceed 5 years or the terms of the Territorial Pollutant Discharge Elimination System (TPDES), whichever is less. Variances require EPA approval. Therefore, this section is not discussed further in this opinion as the requirements for variances are meant to be protective of ESA-listed species and designated critical habitat and granting of a variance requires EPA approval, requiring compliance with ESA section 7.

4 INTERRELATED AND INTERDEPENDENT ACTIONS

Interrelated actions are those that are part of a larger action and depend on that action for their justification. *Interdependent* actions are those that do not have independent utility apart from the action under consideration. NMFS has identified the issuance of TPDES permits (based on EPA-approved WQSR) and establishment of total maximum daily loads (TMDLs) for impaired waterbodies not meeting the applicable water quality standards as interdependent actions without EPA-approved WQSR. Similarly, VIDPNR's water quality monitoring program (Section 305(b) of the CWA) and funding from EPA to conduct and manage this program, as well as the listing of waters as impaired based on the results of monitoring (Section 303(d) of the CWA) are interrelated actions that also require EPA's approval of the VIWQS.

5 ACTION AREA

Action area means all areas affected directly, or indirectly, by the Federal action, and not just the immediate area involved in the action (50 C.F.R. §402.02). The action area for this consultation encompasses the Territorial waters of the USVI, which extend three nautical miles (nm) from the mean low water mark, and any surrounding waters affected by any permitted discharges that incorporate the criteria and create plumes that extend beyond the territorial waters. Water quality standards have been developed for inland waters (Class I) and all coastal and marine Territorial waters based on the division of these areas into Class A, B, or C (Figures 8 and 9).

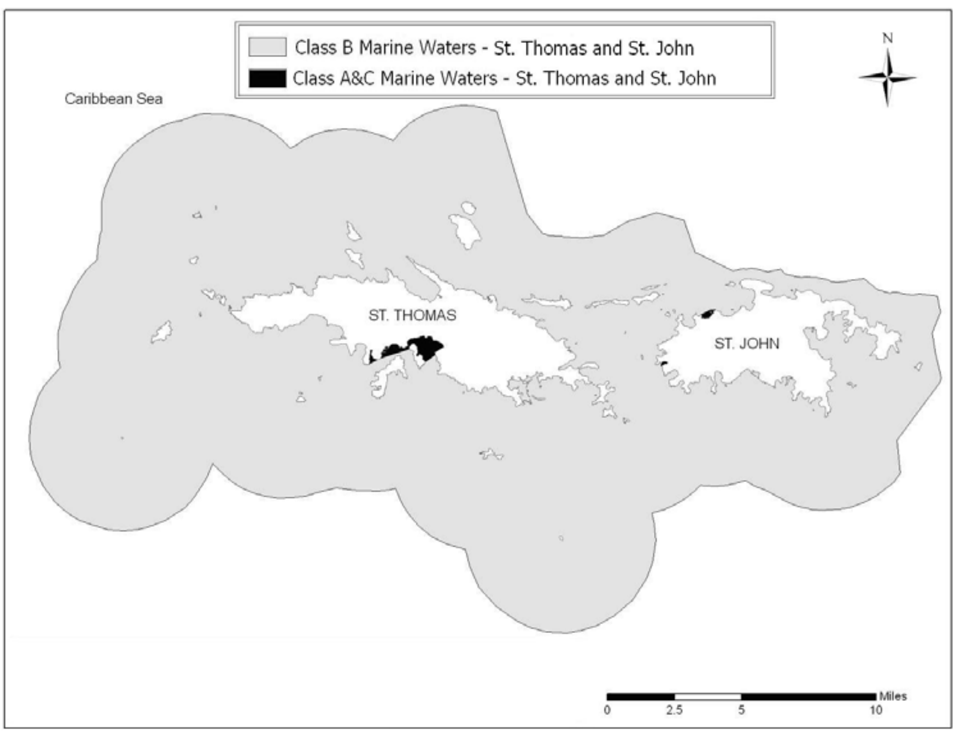

Figure 8. Classes of marine and coastal waters around St. Thomas and St. John from VIWQS with boundary of Territorial waters (3 nm) shown

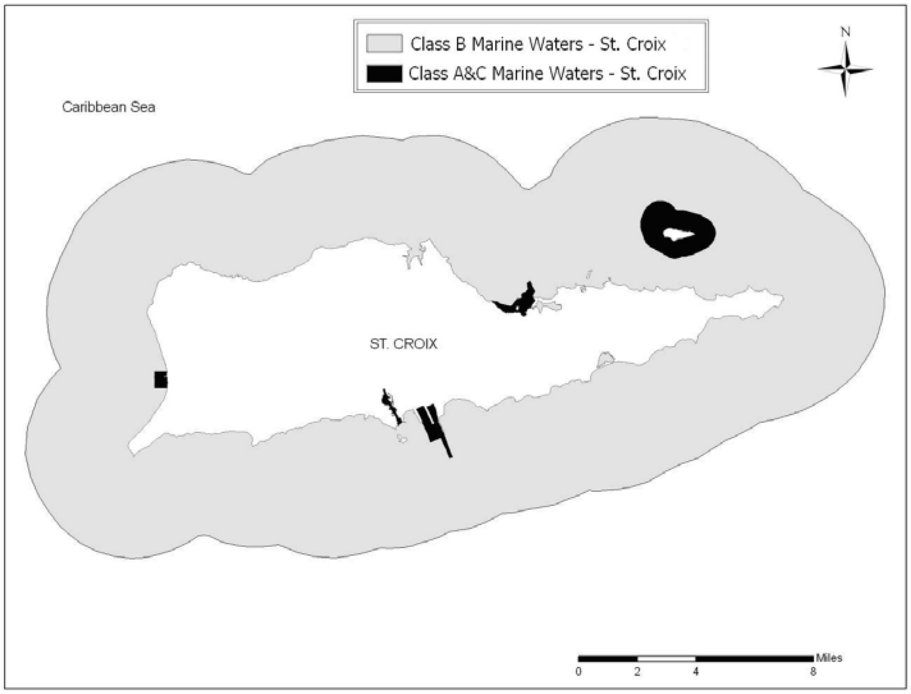

Figure 9. Classes of marine and coastal waters around St. Croix from VIWQS with boundary of Territorial waters (3 nm) shown

6 Status of Endangered Species Act Protected Resources

This section identifies the ESA-listed species and designated critical habitat that potentially occur within the action area (Table 1) that may be affected by the approval of the 2018 VIWQS by the EPA and the application of those standards in permitting by the VIDPNR. This section first identifies the species and designated critical habitats in the action area that may be affected but are not likely to be adversely affected by water quality conditions complying with the criteria in the 2018 VIWQS. The remaining species and designated critical habitats deemed likely to be adversely affected by water quality conditions resulting from VIDPNR's application of these water quality standards are carried forward through the remainder of this opinion.

The evaluation of adverse effects in this opinion begins by summarizing the biology and ecology of those species that are likely to be adversely affected and what is known about their life histories in the action area and the condition of designated critical habitat within the applicable critical habitat unit and in the action area. The status is determined by the level of risk that the ESA-listed species and designated critical habitat face, based on parameters considered in documents such as recovery plans, status reviews, and listing decisions.

This helps to inform the description of the species' current "reproduction, numbers, or distribution" as described in 50 CFR 402.02.

Table 1. Threatened and endangered species that may be affected by the EPA's proposed approval of the 2018 VIWQS

Species	**ESA Status**	Recovery Plan	Critical Habitat
Blue whale (*Balaenoptera musculus*)	E – 35 FR 18319, December 2,1970	07/1998	----
Fin whale (*Balaenoptera physalus*)	E – 35 FR 18319, December 2,1970	75 FR 47538	----
Sei whale (*Balaenoptera borealis*)	E – 35 FR 18319, December 2,1970	76 FR 43985	----
Sperm whale (*Physeter microcephalus*)	E – 35 FR 18319, December 2,1970	75 FR 81584	----
Nassau grouper (*Epinephelus striatus*)	T – 81 FR 42268, June 29, 2016	----	----
Giant manta ray (*Manta birostris*), Southwest Distinct Population Segment (DPS)	T – 83 FR 2916, January 22, 2018	----	----
Scalloped hammerhead shark (*Sphyrna lewini*), Central and Southwest Atlantic Distinct Population Segment (DPS)	T – 79 FR 38214, July 3, 2014	----	----
Oceanic whitetip shark (*Carcharinus longimanus*)	T – 83 FR 4153, January 30, 2018	----	----
Green sea turtle (*Chelonia mydas*), North Atlantic DPS	T – 81 FR 20057, April 6, 2016 (original listing 1978)	63 FR 28359	63 FR 46693
Green sea turtle (*Chelonia mydas*), South Atlantic DPS	T – 81 FR 20057, April 6, 2016	63 FR 28359	----
Hawksbill sea turtle (*Eretmochelys imbricata*)	E – 35 FR 8491, June 2, 1970	12/1993	Not in action area
Leatherback sea turtle (*Dermochelys coriacea*)	E – 35 FR 8491, June 2, 1970	63 FR 28359	44 FR 17710
Loggerhead sea turtle (*Caretta caretta*), Northwest Atlantic Ocean DPS	T – 76 FR 58868, September 22, 2011 (original listing 1978)	63 FR 28359	Not in action area
Elkhorn coral (*Acropora palmata*)	T – 71 FR 26852, May 9, 2006, and 79 FR 53852, September 10, 2014	80 FR 12146	73 FR 72210

Staghorn coral (*Acropora cervicornis*)	T – 71 FR 26852, May 9, 2006, and 79 FR 53852, September 10, 2014	80 FR 12146	73 FR 72210
Lobed star coral (*Orbicella annularis*)	T – 79 FR 53852, September 10, 2014	----	----
Boulder star coral (*Orbicella franksi*)	T – 79 FR 53852, September 10, 2014	----	----
Mountainous star coral (*Orbicella faveolata*)	T – 79 FR 53852, September 10, 2014	----	----
Pillar coral (*Dendrogyra cylindrus*)	T – 79 FR 53852, September 10, 2014	----	----
Rough cactus coral (*Mycetophyllia ferox*)	T – 79 FR 53852, September 10, 2014	----	----
T = threatened, E = endangered			

In its BE, the EPA also concluded that the proposed action may affect, but is not likely to adversely affect humpback whales, which were listed range-wide at the time. However, NMFS subsequently published a final rule on September 8, 2016 (81 FR 62260) identifying 14 DPSs for humpback whales. The West Indies DPS, which includes USVI, was found not to merit listing under the ESA. Therefore, humpback whales are not considered in this consultation.

Giant manta ray and oceanic whitetip shark were not included in the consultation initiation request or subsequent correspondence from EPA because these species were listed as threatened under the ESA in January 2018. Because the ESA requires that all listed species and their designated critical habitat that could be affected by a proposed action be considered, we included these species in this opinion as we have information indicating they are infrequently present in USVI waters.

6.1 Species and Designated Critical Habitat Not Likely to be Adversely Affected

NMFS uses two criteria to identify the ESA-listed or designated critical habitat that are not likely to be adversely affected by the proposed action, as well as the effects of activities that are interrelated to or interdependent with the Federal agency's proposed action. The first criterion is exposure, or some reasonable expectation of a co-occurrence, between one or more potential stressors associated with the proposed activities and ESA-listed species or designated critical habitat. If we conclude that an ESA-listed species or designated critical habitat is not likely to be exposed to the proposed activities, we must also conclude that the species or critical habitat is not likely to be adversely affected by those activities.

The second criterion is the probability of a response given exposure. ESA-listed species or designated critical habitat that co-occurs with a stressor of the action but is not likely to respond to the stressor is also not likely to be adversely affected by the proposed action. We applied these criteria to the ESA-listed species in Table 1 and we summarize our results below.

In the case of the 2018 VIWQS, ESA-listed species and designated critical habitat occur in waters affected by regulation of water quality by VIDPNR and will co-occur with water quality conditions that are subject to the criteria for temperature, water clarity pH, turbidity, nutrients, toxic pollutants, and enterococci bacteria.

The probability of an effect in a species or designated critical habitat is a function of exposure intensity and susceptibility of a species to the stressor's effects (i.e., probability of response). An action warrants a "may affect, not likely to be adversely affected" finding when its effects are wholly *beneficial, insignificant* or *discountable*. *Beneficial* effects have an immediate positive effect without any adverse effects to the species or habitat. Beneficial effects are usually discussed when the project has a clear link to the ESA-listed species or its specific habitat needs, and consultation is required because the species may be affected.

Insignificant effects relate to the size or severity of the impact and include those effects that are undetectable, not measurable, or so minor that they cannot be meaningfully evaluated. Insignificant is the appropriate effect conclusion when plausible effects are going to happen but will not rise to the level of constituting an adverse effect.

Discountable effects are those that are extremely unlikely to occur. For an effect to be discountable, there must be a plausible adverse effect (i.e., a credible effect that could result from the action and that would be an adverse effect if it did affect a listed species), but it is very unlikely to occur.

6.1.1 ESA-Listed Whales

We do not have recent survey data for ESA-listed whale species in USVI waters and we did not find recent stranding reports of ESA-listed whales in USVI waters. Blue, fin, sei, and sperm whales are oceanic and are predominantly found seaward of the insular shelf in the Caribbean. These species have broad ranges and their presence in the USVI is largely restricted to the winter migration period (approximately November to March), meaning any exposure to water quality conditions resulting from application of the criteria by the VIDPNR would be short-term. In addition, whale exposure to pollutants in water is expected to be undetectable because they breathe air and do not drink seawater. Therefore, we expect the effects of EPA's approval of the 2018 VIWQS and VIDPNR application of those criteria in regulating water quality on ESA-listed whales will be discountable. Because we expect the effects of the proposed action are not likely to adversely affect ESA-listed whales, we do not discuss these animals further in this opinion.

6.1.2 Giant Manta Ray

Giant manta rays are typically found offshore in the open ocean though these animals are sometimes found around nearshore reefs and estuarine waters, which are present in the action area. Giant manta rays have been observed infrequently in deeper waters off bays and over deep reefs around USVI (A. Dempsey, BioImpact, pers. comm. to L. Carrubba, NMFS, January 26, 2018 and February 26, 2018; R. Nemeth, University of the Virgin Islands, pers. comm. to L. Carrubba, NMFS, January 26, 2018). These sightings were recorded by the scientists during field work and include observations of giant mantas in the following locations: off Limetree Bay, Christiansted, and Cane Bay, St. Croix; off the south drop, St. John; and off Caret Bay/Vluck Point, Brewers Bay, and Great Bay, and on Grammanik Bank, St. Thomas. No nearshore cleaning sites or aggregations of these animals have been observed in USVI waters, only occasional sightings of one to three animals.

Giant manta rays feed in the water column on plankton. The rarity of giant manta rays in USVI waters and their preference for deeper, offshore areas means any exposure to water quality conditions resulting from VIDPNR applying the criteria would be short-term and rarely occur. Therefore, we believe the effects of EPA's approval of the 2018 VIWQS and VIDPNR application of those criteria in regulating water quality on giant manta rays will be discountable.

Giant manta rays could be affected if the VIWQS were set at levels that could cause reductions in the plankton on the animals feed. While there could be impacts to prey species, there are no known feeding aggregation sites in nearshore or offshore waters in USVI. Giant manta rays are highly mobile and forage in waters within and outside the USVI. Therefore, we expect that any impacts to those prey species in waters affected by the USVI criteria will have an insignificant effect on giant manta ray. Because we expect the effects of the proposed action are not likely to adversely affect giant manta rays, we do not discuss this species further in this opinion.

6.1.3 Oceanic Whitetip Shark

The oceanic whitetip shark is usually found offshore in the open ocean, along the continental shelf, or around oceanic islands in waters from the surface to at least 152 m in depth. Shark tagging data show movements by juveniles of this species in the Gulf of Mexico, along the east coast of Florida, Mid-Atlantic Bight, Cuba, Lesser Antilles, central Caribbean Sea, from east to west along the equatorial Atlantic, and off Brazil, Haiti, and Bahamas (Young et al. 2017). Fisheries data also indicate that, while catch of this species has declined, it has been part of fishery landings in the U.S. Caribbean (Young et al. 2017) meaning that the species is likely to be present in offshore waters of USVI.

The preference of oceanic whitetip sharks for deeper, offshore areas means any exposure to water quality conditions resulting from VIDPNR regulations applying the criteria would be short-term and rarely occur. Therefore, we believe the effects of EPA's approval of the 2018 VIWQS and VIDPNR application of those criteria in regulating water quality on oceanic whitetip sharks will be discountable.

While there could be impacts to prey species, oceanic whitetip sharks feed in very deep waters. Oceanic whitetip sharks are highly mobile and forage in waters within and outside the USVI, preferring open ocean waters. Therefore, we expect that any impacts to those prey species affected by the USVI criteria will have an insignificant effect on oceanic whitetip shark.

Because we expect the effects of the proposed action are not likely to adversely affect oceanic whitetip shark, we do not discuss this species further in this opinion.

6.1.4 Scalloped Hammerhead Shark (Central and Southwest Atlantic DPS)

NMFS Marine Recreational Informational Program data are not collected from the USVI. However, shark research conducted in St. Thomas and St. John in 2004 and 2005 resulted in the capture of a total of nine scalloped hammerhead sharks in Magen's Bay, St. Thomas over both years (DeAngelis 2006). The scalloped hammerhead sharks captured by DeAngelis (2006) were all neonates, indicating that the bay provides nursery habitat for the species. Commercial fisheries data for the U.S. Caribbean do not distinguish between hammerhead shark species but it is likely that adult sharks are present in deep offshore waters of the USVI based on catch information.

The finding that certain bays in St. Thomas and St. John serve as nursery habitat for the species means that younger life stages in these bays in particular could be exposed to water quality conditions resulting from application of the criteria by VIDPNR for longer periods of time than adult animals that are likely to be found in deeper waters. One of the bays that has been reported as impaired is Magen's Bay, St. Thomas, which is where scalloped hammerhead sharks were found in both years of the study. Impairment was due to DO levels, human fecal bacteria, and pH in 2010, 2012, and 2014, and turbidity in 2010, 2012, 2014, and 2016. There is no information to indicate that fecal coliform bacteria affect sharks or their prey. Changes in DO and pH are most likely to affect these animals due to the effects of these parameters on prey species. As discussed in Section 6.3.3.1, the DO criterion of 5.5 mg/L for waters in Magen's Bay is not expected to result in a redistribution of prey species.

There have been some studies that indicate acidification of the ocean can influence the sensory performance of fish. Sharks exposed to increased oceanic carbon dioxide levels showed decreased odor tracking behaviors and avoided food cues, as well as reducing attack behavior and attacking food less aggressively (Dixson et al. 2015). The pH impairment in Magen's Bay was not reported for 2016 and the pH criteria are not on the acid side of the pH spectrum, though lower allowed pH values combined with temperature and salinity could lead to more acidic waters in localized areas.

In terms of turbidity, sharks use odor to detect prey more than vision but high turbidity levels could still impair their ability to capture prey. Turbidity impairment seems to be a regular problem in Magen's Bay at the water quality monitoring stations, though the criterion of 3 NTUs in non-coral habitats is unlikely to be high enough to impede scalloped hammerhead sharks' ability to find prey. The water quality monitoring stations where impairments were recorded are adjacent to the beach in the interior of the bay where there is little habitat or prey likely to be used by scalloped hammerhead sharks (and we have no information indicating beach-goers have reported seeing sharks in the area of the beach). Given the size of Magen's Bay and the extent of coral and other habitats within the bay that provide foraging habitat for scalloped hammerhead sharks, particularly moving toward the entrance to the bay, these animals would be able to find other areas with adequate DO concentrations, lower turbidities, pH values closer to 8, and presence of prey. No other parameters were reported as impaired in bays where the presence of juvenile hammerhead sharks was recorded by DeAngelis (2006) and, as noted previously, the criteria included in the VIWQS do not appear to be outside the tolerance range of scalloped hammerhead sharks based on information available in the scientific literature. Therefore, we expect that impacts to those prey species affected by the USVI criteria will have an insignificant effect on scalloped hammerhead shark and that effects to the sharks themselves will also be insignificant.

Because we expect the effects of the proposed action are not likely to adversely affect scalloped hammerhead shark, we do not discuss this species further in this opinion.

6.1.5 Loggerhead Sea Turtles (Northwest Atlantic DPS)

Loggerhead sea turtles are not common in the U.S. Caribbean but there have been reports of limited nesting on the east coast of Puerto Rico and the island of Culebra, as well as on Buck Island, St. Croix (2 females reported nesting in the early 2000's but then no longer observed by National Park Service). There were infrequent stranding reports of this species (rarely equaling one per year) from the VIDPNR but there have been no reports of stranding of this species in waters of the U.S. Caribbean in the two years prior to completion of this consultation based on information in our records from other consultations. As stated in the listing rule that designated 9 DPSs for loggerhead sea turtles, the most important of the 5 recovery units for this DPS are the four units along the east coast from Georgia south and in the Gulf of Mexico from Florida to Texas due to the amount of nesting versus the Greater Caribbean unit (76 FR 58885, September 22, 2011).

Loggerhead sea turtles are omnivorous and opportunistic feeders with sub-adults and adults preying on benthic invertebrates and occasionally fish and plants (Parker et al. 2005; USFWS/NMFS 1998; Mckenzie et al. 1999; Gardner et al. 2003; Storelli et al. 2007; Pugh and Becker 2001). As for ESA-listed whales, giant manta ray, and oceanic whitetip sharks, loggerhead sea turtles could be affected if water quality conditions due to VIDPNR-authorized discharges result in declines in forage species or accumulation of toxic pollutants. Loggerhead sea turtles are rare in waters of the USVI with no known foraging areas for these animals reported in the Territory. Any animals that do enter USVI waters to forage are likely to be transiting through and are not resident so any potential for exposure to water quality conditions resulting from application of the criteria would be short-term. Therefore, we expect that impacts to those prey species affected by the USVI criteria will have an insignificant effect on loggerhead sea turtles in the Northwest Atlantic DPS and that effects to the sea turtles themselves will also be insignificant. Because we expect the effects of the proposed action are not likely to adversely affect loggerhead sea turtles, we do not discuss this species further in this opinion.

6.1.6 Leatherback Sea Turtles

Leatherback sea turtles are present in waters around the USVI during their nesting season, which peaks in May through July in the U.S. Caribbean. There are infrequent reports of strandings of this species from the VIDPNR, mainly due to vessel strikes and entanglement in fishing gear (around St. Croix only) during months when nesting occurs. At the primary nesting beach on St. Croix, the Sandy Point National Wildlife Refuge, nesting varied from a few hundred nests to a high of 1,008 in 2001, and the average annual growth rate was approximately 1.1 percent from 1986-2004 (TEWG 2007). From 2006-2010, Tiwari et al. (2013) report an annual growth rate of +7.5 percent in St. Croix and a three-generation abundance change of +1,058 percent. Limited nesting is reported on St. Thomas and St. John, likely due to the lack of extensive sandy beach areas in comparison with St. Croix.

Leatherback sea turtles appear to prefer the open ocean at all life stages (Heppell et al. 2003). Leatherbacks prey mainly on jellies (e.g., medusa, siphonophores, and salps). Leatherback sea turtles could be exposed to water quality conditions due to VIDPNR-authorized discharges as nesting adults or as hatchlings move through USVI waters when transiting to/from nesting beaches. These animals are not resident so any potential for exposure to water quality conditions resulting from application of the criteria would be short-term. Therefore, we expect that any impacts to leatherback sea turtles resulting from application of the USVI criteria will be insignificant. Because we expect the effects of the proposed action are not likely to adversely affect leatherback sea turtles, we do not discuss this species further in this opinion.

6.1.7 Leatherback Sea Turtle Critical Habitat

Critical habitat for the leatherback sea turtle includes waters adjacent to Sandy Point Beach, St. Croix. This area provides courting and breeding habitat and access to and from leatherback nesting habitat on Sandy Point Beach. The VIDPNR has proposed stricter criteria related to turbidity levels in nearshore waters and has retained a water clarity standard regardless of the designation of waters by class under the VIWQS. The proposed turbidity levels and the existing water clarity standards require that the waters adjacent to Sandy Point remain clear, meaning visibility and the corresponding ability of leatherbacks to encounter mates, if visual cues are part of this process, would not be affected. Therefore, we expect the effects of EPA's approval of the 2018 VIWQS and VIDPNR application of those criteria in regulating water quality will have no effect on leatherback sea turtle critical habitat because the ability of the habitat to function as a site for leatherback sea turtle courtship and mating will not be affected. We do not discuss leatherback sea turtle critical habitat further in this opinion.

6.2 Species and Critical Habitat Likely to be Adversely Affected

This opinion examines the status of each species and critical habitat that may be adversely affected by the proposed action. The species status section helps to inform the description of the species' current "reproduction, numbers, or distribution" as described in 50 C.F.R. 402.02. More detailed information on the status and trends of these ESA-listed species, 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 NMFS Web site: http://www.nmfs.noaa.gov/pr/species/esa/listed.htm.

6.2.1 Nassau Grouper

Species Description and Life History

The Nassau grouper, *Epinephelus striatus* (NMFS 2013), is a moderate-sized serranid fish with large eyes and a robust body. As with many serranids, the Nassau grouper is slow-growing and long-lived; estimates range up to a maximum of 29 years (Bush et al. 1996). Using length-frequency analysis, which tends to exclude younger animals, a theoretical maximum age at 95 percent asymptotic size is 16 years. Individuals of more than 12 years of age are not common in fisheries, with more heavily fished areas yielding much younger fish on average. Most studies indicate a rapid growth rate for juveniles, which has been estimated to be about 10 mm/month total length (TL) for small juveniles, and 8.4-11.7 mm/month TL for larger juveniles (Beets and Hixon 1994; Eggleston 1995). Maximum size is about 122 cm TL and maximum weight is about 25 kg (Humann and DeLoach 2002; Heemstra and Randall 1993; Froese 2010). Generation time (the interval between the birth of an individual and the subsequent birth of its first offspring) is estimated as 9-10 years (Sadovy and Eklund 1999). Male and female Nassau groupers reach sexual maturity at lengths between 40 and 45 centimeters standard length, about four to five years old. It is thought that sexual maturity is more determined by size, rather than age. Otolith studies indicate that the minimum age at maturity is between four and eight years; most groupers have spawned by age seven (Bush et al. 2006). Nassau groupers live to a maximum of 29 years.

Nassau groupers spawn once a year in large aggregations, in groups of a few dozen to thousands spawning at once. Nassau groupers move in groups towards the spawning aggregation sites parallel to the coast or along the shelf edge at depths between 20 and 33 m. Spawning runs occur in late fall through winter (i.e., a month or two before spawning is likely). Sea surface temperature is thought to be a key factor in the timing of spawning, with spawning occurring at waters temperatures between 25 and 26 ^{o}C. Spawning aggregation sites are located near significant geomorphological features, such as reef projections (as close as 50 m to shore) and close to a drop-off into deep water over a wide depth range (six to sixty m). Sites are usually several hundred meters in diameter, with soft corals, sponges, stony coral outcrops, and sandy depressions. Nassau groupers stay on the spawning site for up to three months, spawning at the full moon or between the new and full moons. Spawning occurs within twenty minutes of sunset over the course of several days. There have been about fifty known spawning sites in insular areas throughout the Caribbean; many of these aggregations no longer form. Current spawning locations are found in Mexico, Bahamas, Belize, Cayman Islands, the Dominican Republic, Cuba, Puerto Rico and the USVI.

Fertilized eggs are transported offshore by ocean currents. Thirty-five to forty days after hatching, larvae recruit from oceanic environment to demersal habitats (at a size of about 32 millimeters total length). Juveniles inhabit macroalgae, coral clumps, and seagrass beds, and are relatively solitary. As they grow, they occupy progressively deeper areas and offshore reefs, and can be found in schools of up to forty individuals. When not spawning, adults are most commonly found in waters less than one hundred meters deep. Nassau grouper diet changes with age. Juveniles eat plankton, pteropods, amphipods, and copepods. Adults are unspecialized piscivores, bottom-dwelling ambush suction predators (NMFS 2013).

Distribution

The Nassau grouper’s confirmed distribution currently includes “Bermuda and Florida (USA), throughout the Bahamas and Caribbean Sea” (e.g., Heemstra and Randall 1993). The occurrence of Nassau grouper from the Brazilian coast south of the equator as reported in Heemstra and Randall (1993) is “unsubstantiated” (Craig et al. 2011). The Nassau grouper has been documented in the Gulf of Mexico, at Arrecife Alacranes (north of Progreso) to the west off the Yucatan Peninsula, Mexico (Hildebrand et al. 1964). Nassau grouper is generally replaced ecologically in the eastern Gulf by red grouper (*E. morio*) in areas north of Key West or the Tortugas (Smith 1971). They are considered a rare or transient species off Texas in the northwestern Gulf of Mexico (Gunter and Knapp 1951; in Hoese and Moore 1998). The first confirmed sighting of Nassau grouper in the Flower Garden Banks National Marine Sanctuary, which is located in the northwest Gulf of Mexico approximately 180 kilometers (km) southeast of Galveston, Texas, was reported by (Foley et al. 2007). Many earlier reports of Nassau grouper up the Atlantic coast to North Carolina have not been confirmed. The Biological Report (Hill and Sadovy de Mitcheson 2013) provides a detailed description of the distribution, summarized in Figure 10.

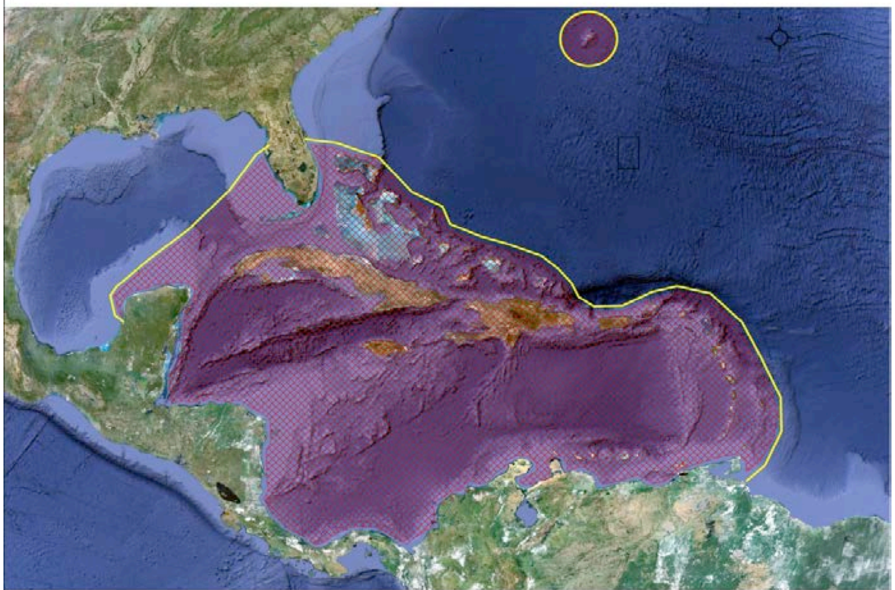

Figure 10. Range of Nassau grouper (*Epinephelus striatus*)

Population Dynamics

There is no range-wide abundance estimate available for Nassau grouper. The species is characterized as having patchy abundance due largely to differences in habitat availability or quality, and differences in fishing pressure in different locations (81 FR 42268). Although abundance has been reduced compared to historical levels, spawning still occurs and abundance is increasing in some locations, such as the Cayman Islands and Bermuda.

There is no population growth rate available for Nassau grouper. However, the available information from observations of spawning aggregations has shown steep declines (Aguilar-Perera 2006; Sala et al. 2001; Claro and Lindeman 2003). Some aggregation sites are comparatively robust and showing signs of increase (Whaylen et al. 2004; Vo et al. 2014).

Recent studies on Nassau grouper genetic variation has found strong genetic differentiation across the Caribbean subpopulations, likely due to barriers created by ocean currents and larval behavior (Jackson et al. 2014a).

Nassau grouper is distributed throughout the Caribbean, south to the northern coast of South America (Figure 10). Current Nassau grouper distribution is considered equivalent to its historical range, although abundance has been severely depleted.

Status

Historically, tens of thousands of Nassau grouper spawned at aggregation sites throughout the Caribbean. Since grouper species were reported collectively in landings data, it is not possible to know how many Nassau grouper were harvested, or estimate historic abundance. That these large spawning aggregations occurred in predictable locations at regular times made the species susceptible to over-fishing and was a cause of its decline. At some sites (e.g., Belize), spawning aggregations have decreased by over 80 percent in the last 25 years (Sala et al. 2001), or have disappeared entirely (e.g., Mexico; Aguilar-Perera 2006). Nassau groupers are also targeted for fishing throughout the year during non-spawning months. In some locations, spawning aggregations are increasing. Many Caribbean countries have banned or restricted Nassau grouper harvest, and it is believed that the areas of higher abundance are correlated with effective regulations (81 FR 42268). Since Nassau groupers are dependent upon coral reefs at various points in their life history, loss of coral reef habitat due to climate change. Increasing water temperatures may change the timing and location of spawning. Habitat degradation due to water pollution also poses a threat to the species. Nassau grouper populations have been reduced from historic abundance levels, and remain vulnerable to unregulated harvest, especially the spawning aggregations. NMFS determined that the species warrants listing as threatened.

Status of the Species in the Action Area

By 1970, Nassau grouper was common in the reef fish fishery of the USVI, where an aggregation in the 1970s contained an estimated 2,000-3,000 individuals (Olsen and Laplace 1979). During the 1980s, port sampling in the USVI showed that Nassau grouper accounted for 22 percent of grouper landings with 85 percent of the Nassau grouper catch coming from spawning aggregations (NMFS 2013). By 1981, “the Nassau grouper ha(d) practically disappeared from the local catches and the ones that d(id) appear (were) -small compared with previous years” (CFMC and NMFS 1985)and by 1986, the Nassau grouper was considered commercially extinct in the USVI/Puerto Rico region (Bohnsack et al. 1986). Fishing of this species was prohibited in federal waters by the Caribbean Fishery Management Council (CFMC) in 1990 and in Territorial waters in 2006 due to significant declines in landings of this species thought to be correlated with a decline in the overall population in the Caribbean due to overexploitation. There is evidence that spawning by Nassau grouper is occurring at what may be reconstituted or novel spawning sites in the USVI (Hill and Sadovy de Mitcheson 2013). In 2003, Nassau grouper were found aggregating in small numbers to spawn at Grammanik Bank off St. Thomas (Kadison et al. 2009). In 2005, the bank was closed seasonally from February – May by the CFMC to protect spawning aggregations of the yellowfin grouper. This has apparently benefited Nassau grouper as well because, since 2005, greater numbers, greater mean size, and a larger size range of Nassau grouper have been observed on Grammanik Bank spawning at the same time as the yellowfin grouper (Kadison et al. 2009).

Critical Habitat

No critical habitat has been designated for the Nassau grouper.

Recovery Goals

NMFS has not prepared a recovery plan for the Nassau grouper.

6.2.2 General Threats Faced by Green (North and South Atlantic DPS) and Hawksbill Sea Turtles

Sea turtles face numerous natural and man-made threats that shape their status and affect their ability to recover. Many of the threats are either the same or similar in nature for all listed sea turtle species, and those identified in this section are discussed in a general sense for all sea turtles. Threat information specific to a particular species is 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 (USFWS/NMFS 1991b;a); USFWS/NMFS (1993); (USFWS/NMFS 2008; NMFS 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 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. Marine debris is a problem due primarily to sea turtles ingesting debris and blocking the digestive tract, causing death or serious injury (Lutcavage et al. 1997; Laist et al. 1999). Schuyler et al. (2015) estimated that, globally, 52 percent of individual sea turtles have ingested marine debris. Gulko and Eckert (2003) estimated that between one-third and one-half of all sea turtles ingest plastic at some point in their lives; this figure is supported by data from Lazar and Gračan (2011), who found 35 percent of loggerheads had plastic in their gut. A Brazilian study found that 60 percent of stranded green sea turtles had ingested marine debris (Bugoni et al. 2001). Loggerhead sea turtles had a lesser frequency of marine debris ingestion. Plastic may be ingested out of curiosity or due to confusion with prey items. Marine debris consumption has been shown to depress growth rates in post-hatchling loggerhead sea turtles, increasing the time required to reach sexual maturity and increasing predation risk (McCauley and Bjorndal 1999). Sea turtles can also become entangled and die in marine debris, such as discarded nets and monofilament line (NRC 1990b; Lutcavage et al. 1997; Laist et al. 1999).

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).

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), ultimately affecting primary foraging areas of ESA-listed species including marine mammals, sea turtles, and fish. Marine species ranges are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2012). Though predicting the precise consequences of climate change on highly mobile marine species is difficult (Simmonds and Isaac 2007), recent research has indicated a range of consequences already occurring.

Hazen et al. (2012) examined top predator distribution and diversity in the Pacific Ocean in light of rising sea surface temperatures using a database of electronic tags and output from a global climate model. They predicted up to a 35 percent change in core habitat area for some key marine predators in the Pacific Ocean, with some species predicted to experience gains in available core habitat and some predicted to experience losses. Notably, leatherback turtles were predicted to gain core habitat area, whereas loggerhead turtles and blue whales were predicted to experience losses in available core habitat. McMahon and Hays (2006) predicted increased ocean temperatures will expand the distribution of leatherback turtles into more northern latitudes. The authors noted this is already occurring in the Atlantic Ocean. MacLeod (2009) estimated, based upon expected shifts in water temperature, 88 percent of cetaceans will be affected by climate change, with 47 percent predicted to experience unfavorable conditions (e.g., range contraction). Willis-Norton et al. (2015) acknowledged there will be both habitat loss and gain, but overall climate change could result in a 15 percent loss of core pelagic habitat for leatherback turtles in the eastern South Pacific Ocean.

Climate-related changes in important prey species populations are likely to affect predator populations. For ESA-listed species that undergo long migrations, if either prey availability or habitat suitability is disrupted by changing ocean temperature regimes, the timing of migration can change or negatively impact population sustainability (Simmonds and Eliott 2009).

Climate change may also result in significant impacts to the hatchling sex ratios of sea turtles may result (USFWS/NMFS 2007a;b). 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 to 35°C (Ackerman 1997). Increases in global temperature could skew future sex ratios toward higher numbers of females (NMFS and USFWS 2007). These impacts will be exacerbated by sea level rise. The loss of habitat because 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).

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 1990a). 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 over wash (USFWS/NMFS 2007a). Sea level rise from global climate change is also a potential problem for areas with low-lying beaches where sand depth is a limiting factor, as the sea may inundate nesting sites and decrease available nesting habitat (Baker et al. 2006; Daniels et al. 1993; Fish et al. 2005). The loss of habitat because 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, DO levels, nutrient distribution) could influence the distribution and abundance of lower trophic levels (e.g., phytoplankton, zooplankton, submerged aquatic vegetation, crustaceans, mollusk s, 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 (USFWS/NMFS 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.

6.2.3 Status of Green Sea Turtle (North and South Atlantic DPSs)

Species description

The green sea turtle (*Chelonia mydas*) is the largest of the hardshell marine turtles, growing to a weight of 350 lb (159 kg) and a straight carapace length of greater than 3.3 ft (1 m). It has a circumglobal distribution, occurring throughout nearshore tropical, subtropical and, to a lesser extent, temperate waters.

The species was listed under the ESA on July 28, 1978 (43 FR 32800) and separated into two listing designations: endangered for breeding populations in Florida and the Pacific coast of Mexico and threatened in all other areas throughout its range. On April 6, 2016, NMFS listed 11 DPSs of green sea turtles as threatened or endangered under the ESA (Figure 11; 81 FR 20057). Eight DPSs are listed as threatened: Central North Pacific, East Indian-West Pacific, East Pacific, North Atlantic, North Indian, South Atlantic, Southwest Indian, and Southwest Pacific. Three DPSs are listed as endangered: Central South Pacific, Central West Pacific, and Mediterranean.

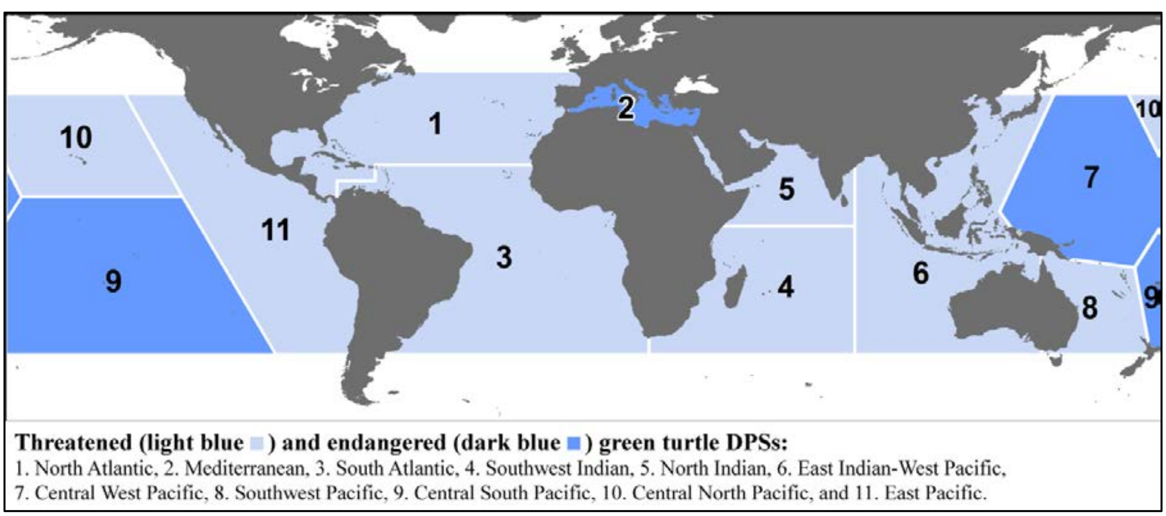

Figure 11. Map depicting DPS boundaries for green turtles

Life history

Age at first reproduction for females is 20 - 40 years. Green sea turtles lay an average of three nests per season with an average of 100 eggs per nest. The remigration interval (i.e., return to natal beaches) is 2 - 5 years. Nesting occurs primarily on beaches with intact dune structure, native vegetation and appropriate incubation temperatures during summer months. 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. Adult turtles exhibit site fidelity and migrate hundreds to thousands of kilometers from nesting beaches to foraging areas. Green sea turtles spend the majority of their lives in coastal foraging grounds, which include open coastlines and protected bays and lagoons. Adult green turtles feed primarily on seagrasses and algae, although they also eat jellyfish, sponges and other invertebrate prey.

Population dynamics

Abundance

Worldwide, nesting data at 464 sites indicate that 563,826 to 564,464 females nest each year (Seminoff et al. 2015).

<u>North Atlantic DPS</u>

Compared to other DPSs, the North Atlantic DPS exhibits the highest nester abundance, with approximately 167,424 females at 73 nesting sites; Figure 12), and available data indicate an increasing trend in nesting. The largest nesting site in the North Atlantic DPS is in Tortuguero, Costa Rica, which hosts 79 percent of nesting females for the DPS (Seminoff et al. 2015).

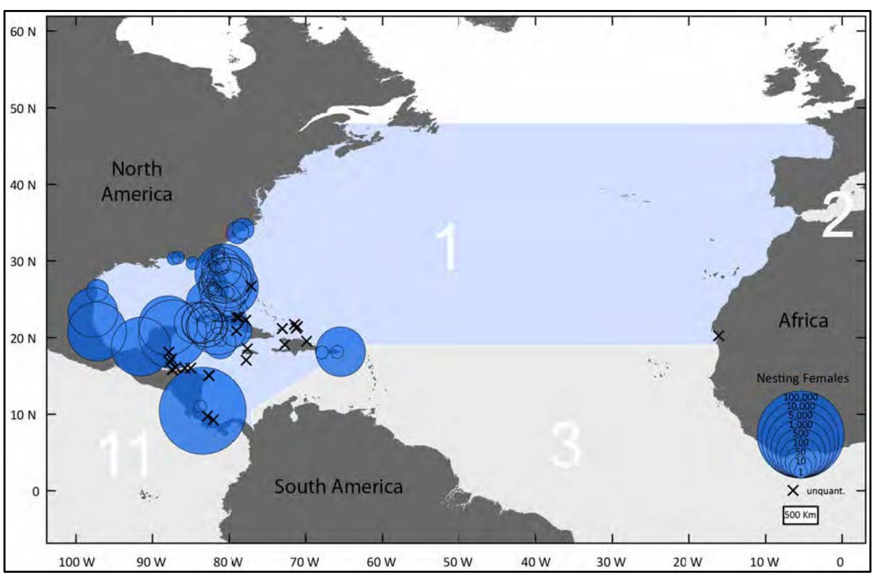

Figure 12. Geographic range of the North Atlantic DPS (1 on the map) and South Atlantic GPS (3 on the map), with location and abundance of nesting females (from Seminoff et al. 2015)

South Atlantic DPS

The South Atlantic DPS has 51 nesting sites, with an estimated nester abundance of 63,332. The largest nesting site is at Poilão, Guinea-Bissau, which hosts 46 percent of nesting females for the DPS (Seminoff et al. 2015).

Population Growth Rate

North Atlantic DPS

For the North Atlantic DPS, the available data indicate an increasing trend in nesting. There are no reliable estimates of population growth rate for the DPS as a whole, but estimates have been developed at a localized level. Modeling by Chaloupka et al. (2008) using data sets of 25 years or more show the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9 percent, and the Tortuguero, Costa Rica, population growing at 4.9 percent.

South Atlantic DPS

There are 51 nesting sites for the South Atlantic DPS, and many have insufficient data to determine population growth rates or trends. Of the nesting sites where data are available, such as Ascension Island, Suriname, Brazil, Venezuela, Equatorial Guinea, and Guinea-Bissau, there is evidence that population abundance is increasing.

Genetic Diversity

Globally, the green turtle is divided into eleven distinct population segments; available information on the genetic diversity for the North Atlantic and South Atlantic distinct population segments is presented below.

North Atlantic DPS

The North Atlantic DPS has a globally unique haplotype, which was a factor in defining the discreteness of the population for the DPS. Evidence from mitochondrial DNA studies indicates that there are at least four independent nesting subpopulations in Florida, Cuba, Mexico and Costa Rica (Seminoff et al. 2015). More recent genetic analysis indicates that designating a new western Gulf of Mexico management unit might be appropriate (Shamblin et al. 2015).

South Atlantic DPS

Individuals from nesting sites in Brazil, Ascension Island, and western Africa have a shared haplotype found in high frequencies. Green turtles from rookeries in the eastern Caribbean however, are dominated by a different haplotype.

Distribution

North Atlantic DPS

Green turtles from the North Atlantic DPS range from the boundary of South and Central America (7.5°N, 77°W) in the south, throughout the Caribbean, the Gulf of Mexico, and the U.S. Atlantic coast to New Brunswick, Canada (48°N, 77°W) in the north. The range of the DPS then extends due east along latitudes 48°N and 19°N to the western coasts of Europe and Africa (Figure 12).

South Atlantic DPS

The range of the South Atlantic DPS begins at the border of Panama and Colombia at 7.5°N, 77°W, heads due north to 14°N, 77°W, then east to 14°N, 65.1°W, then north to 19°N, 65.1°W, and along 19°N latitude to Mauritania in Africa. It extends along the coast of Africa to South Africa, with the southern border being 40°S latitude (Figure 12).

Status

We used information available in the 2007 5-Year Review (USFWS/NMFS 2007a) and 2015 Status Review (Seminoff et al. 2015) to summarize the status of the species, as follows.

Once abundant in tropical and subtropical waters, green sea turtles worldwide exist at a fraction of their historical abundance, as a result of over-exploitation. Globally, egg harvest, the harvest of females on nesting beaches and directed hunting of turtles in foraging areas remain the three greatest threats to their recovery. In addition, bycatch in drift net, long-line, set-net, pound-net and trawl fisheries kill thousands of green sea turtles annually. Increasing coastal development (including beach erosion and re-nourishment, construction and artificial lighting) threatens nesting success and hatchling survival. On a regional scale, the different DPSs experience these threats as well, to varying degrees. Differing levels of abundance combined with different intensities of threats and effectiveness of regional regulatory mechanisms make each DPS uniquely susceptible to future perturbations.

North Atlantic DPS

Historically, green turtles in the North Atlantic DPS were hunted for food, which was the principle cause of the population’s decline. Apparent increases in nester abundance for the North Atlantic DPS in recent years are encouraging but must be viewed cautiously, as the datasets represent a fraction of a green sea turtle generation, up to 50 years. While the threats of pollution, habitat loss through coastal development, beachfront lighting, and fisheries bycatch continue, the North Atlantic DPS appears to be somewhat resilient to future perturbations.

South Atlantic DPS

Though there is some evidence that the South Atlantic DPS is increasing, there is a considerable amount of uncertainty over the impacts of threats to the South Atlantic DPS. The DPS is threatened by habitat degradation at nesting beaches, and mortality from fisheries bycatch remains a primary concern.

Status Within the Action Area

Nesting by green sea turtles is reported in limited but increasing numbers on Buck Island, St. Croix, and on Sandy Point Beach, St. Croix, by the National Park Service (NPS) and the USFWS, respectively. Nesting of green sea turtles is also reported on other beaches of St. Croix by the VIDPNR and The Nature Conservancy, both of which monitor some beaches for sea turtle nesting activity. Green sea turtle nesting has been reported in the past on St. John by NPS as well, though NPS believes poaching has led to the elimination of green sea turtle nesting on St. John (https://www.nps.gov/viis/learn/nature/upload/Seaturtles1.pdf). There is no USVI-wide sea turtle nesting activity monitoring program so the only sources of reliable nesting data are from NPS and USFWS within the protected areas these agencies manage.

Juvenile, sub-adult, and adult green sea turtles are present year-round in USVI waters. The University of the Virgin Islands (UVI) has an in-water sea turtle monitoring program (NMFS Permit No. 20315) and reports that there is a large population of green sea turtles around St. Thomas where their monitoring program is focused. Based on observations by NMFS biologists during numerous site inspections around USVI, green sea turtles are common in the USVI around the main islands and offshore cays.

Critical Habitat

There is no designated critical habitat for the South Atlantic DPS. The North Atlantic DPS includes green sea turtle critical habitat designated on September 2, 1998, which includes waters surrounding Culebra Island, Puerto Rico, which is outside the action area of this consultation.

Recovery Goals

See the 1998 and 1991 recovery plans for the Pacific, East Pacific, and Atlantic populations of green turtles for complete down-listing/delisting criteria for recovery goals of the species. Broadly, recovery plan goals emphasize the need to protect and manage nesting and marine habitat, protect and manage populations on nesting beaches and in the marine environment, increase public education, and promote international cooperation on sea turtle conservation topics. For the Atlantic, which encompasses the North and South Atlantic DPSs, the recovery objectives are:

- The level of nesting in Florida has increased to an average of 5,000 nests per year for at least 6 years. Nesting data must be based on standardized surveys.
- At least 25 percent (105 km) of all available nesting beaches (420 km) is in public ownership and encompasses at least 50 percent of the nesting activity.
- A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.

- All priority one tasks have been successfully implemented.

6.2.4 Status of Hawksbill Sea Turtles

Species Description

The hawksbill turtle has a circumglobal distribution throughout tropical and, to a lesser extent, subtropical oceans (Figure 13). The hawksbill sea turtle has a sharp, curved, beak-like mouth and a “tortoiseshell” pattern on its carapace, with radiating streaks of brown, black, and amber. The species was first listed under the Endangered Species Conservation Act (35 FR 8491) and listed as endangered under the ESA since 1973. The status of the species is summarized below based on information available in the most recent 5-year reviews (USFWS/NMFS 2007b; NMFS and USFWS 2013).

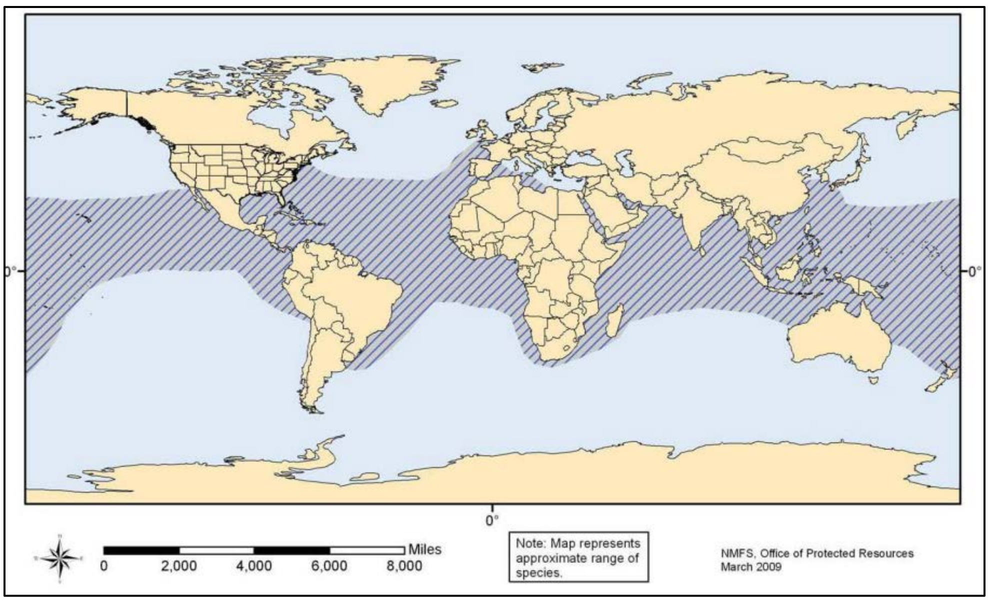

Figure 13. Map identifying the range of the endangered hawksbill sea turtle (http://www.nmfs.noaa.gov/pr/pdfs/rangemaps/hawksbill_turtle.pdf)

Life History

Hawksbill sea turtles reach sexual maturity at 20 – 40 years of age. Females return to their natal beaches every 2 – 5 years to nest (an average of three – 5 times per season). Clutch sizes are large (up to 250 eggs). Sex determination is temperature dependent, with warmer incubation producing more females. Hatchlings migrate to and remain in pelagic habitats until they reach approximately 22 – 25 cm in straight carapace length. As juveniles, they take up residency in coastal waters to forage and grow. As adults, hawksbills use their sharp beak-like mouths to feed on sponges and corals. Hawksbill sea turtles are highly migratory and use a wide range of habitats during their lifetimes (Musick and Limpus 1997; Plotkin 2003). Satellite tagged turtles have shown significant variation in movement and migration patterns. Distance traveled between nesting and foraging locations ranges from a few hundred to a few thousand km (Horrocks et al. 2001; Miller 1998).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section is broken down into: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the hawksbill sea turtle.

Abundance

Surveys at 88 nesting sites worldwide indicate that 22,004 – 29,035 females nest annually (USFWS/NMFS 2013). In general, hawksbills are doing better in the Atlantic and Indian Ocean than in the Pacific Ocean, where despite greater overall abundance, a greater proportion of the nesting sites are declining.

Population Growth Rate

From 1980 to 2003, the number of nests at three primary nesting beaches (Rancho Nuevo, Tepehuajes, and Playa Dos) increased 15 percent annually ; however, due to recent declines in nest counts, decreased survival at other life stages, and updated population modeling, this rate is not expected to continue (USFWS/NMFS 2013).

Genetic Diversity

Populations are distinguished generally by ocean basin and more specifically by nesting location. Our understanding of population structure is relatively poor. Genetic analysis of hawksbill sea turtles foraging off the Cape Verde Islands identified three closely-related haplotypes in a large majority of individuals sampled that did not match those of any known nesting population in the western Atlantic, where the vast majority of nesting has been documented (Monzón-Argüello et al. 2010). Hawksbills in the Caribbean seem to have dispersed into separate populations (rookeries) after a bottleneck roughly 100,000-300,000 years ago (Leroux et al. 2012).

Distribution

The hawksbill has a circumglobal distribution throughout tropical and, to a lesser extent, subtropical waters of the Atlantic, Indian, and Pacific Oceans. In their oceanic phase, juvenile hawksbills can be found in *Sargassum* mats; post-oceanic hawksbills may occupy a range of habitats that include coral reefs or other hard-bottom habitats, sea grass, algal beds, mangrove bays and creeks (Bjorndal and Bolten 2010; Musick and Limpus 1997).

Status

Long-term data on the hawksbill sea turtle indicate that 63 sites have declined over the past 20 to 100 years (historic trends are unknown for the remaining 25 sites). Recently, 28 sites (68 percent) have experienced nesting declines, 10 have experienced increases, three have remained stable, and 47 have unknown trends. The greatest threats to hawksbill sea turtles are overharvesting of turtles and eggs, degradation of nesting habitat, and fisheries interactions. Adult hawksbills are harvested for their meat and carapace, which is sold as tortoiseshell. Eggs are taken at high levels, especially in Southeast Asia where collection approaches 100 percent in some areas. In addition, lights on or adjacent to nesting beaches are often fatal to emerging hatchlings and alters the behavior of nesting adults. The species' resilience to additional perturbation is low.

Status Within the Action Area

In USVI, hawksbill nesting is reported on Buck Island, St. Croix, with 56-150 nests per year (Meylan 1988; Mortimer and Donnelly 2008a) and on Sandy Point Beach, St. Croix. Nesting also occurs to a lesser extent on other beaches around St. Croix, as well as on beaches of St. John, and St. Thomas.

Mortimer and Donnelly (2008b) 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 for 58 of the 83 sites and recent abundance trends (within the last 20 years) for 42 of the 83 sites. In terms of 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. Nine of the 10 sites showing increases in abundance were in the Caribbean including an increasing trend in nesting reported for Buck Island, St. Croix (Mackay 2006; Mortimer and Donnelly 2008b). The beaches of Buck Island are identified as an index site for hawksbill sea turtle recovery in the eastern Caribbean (NPS 2012).

In the Caribbean, hawksbills are known to feed almost exclusively on sponges (Meylan 1988; Van Dam and Diez 1997), although at times they have been seen foraging on other food items, notably corallimorphs and zooanthids (León and Diez 2000; Mayor 1998; Van Dam and Diez 1997). Coral reefs are reported as prime habitat for this species and area estimates of potential habitat for hawksbill turtles have been created using the distribution of coral reefs (Buitrago and Guada 2002; Prieto et al. 2001), also hawksbills, particularly juveniles, have been reported to use other habitats such as seagrass beds and mangrove-lined coastal embayments (Diez et al. 2003). Juveniles, sub-adults, and adult hawksbill sea turtles are reported as common around St. Thomas by UVI as part of their sea turtle monitoring program. Hawksbill sea turtles have been observed throughout USVI around the main islands and at offshore cays by NMFS biologists during site inspections.

Critical Habitat

NMFS designated critical habitat for hawksbill sea turtles on September 2, 1998 around Mona and Monito Islands, Puerto Rico, which is outside the action area for this consultation.

Recovery Goals

The 1992 and 1998 Recovery Plans for the U.S. Caribbean, Atlantic and Gulf of Mexico, and U.S. Pacific populations of hawksbill sea turtles, respectively, contain complete down listing/delisting criteria for each of their respective recovery goals. The following items were the top recovery actions identified to support in the Recovery Plans:

- Identify important nesting beaches
- Ensure long-term protection and management of important nesting beaches
- Protect and manage nesting habitat; prevent the degradation of nesting habitat caused by seawalls, revetments, sand bags, other erosion-control measures, jetties and breakwaters
- Identify important marine habitats; protect and manage populations in marine habitat
- Protect and manage marine habitat; prevent the degradation or destruction of important [marine] habitats caused by upland and coastal erosion
- Prevent the degradation of reef habitat caused by sewage and other pollutants
- Monitor nesting activity on important nesting beaches with standardized index surveys
- Evaluate nest success and implement appropriate nest-protection on important nesting beaches
- Ensure that law-enforcement activities prevent the illegal exploitation and harassment of sea turtles and increase law-enforcement efforts to reduce illegal exploitation
- Determine nesting beach origins for juveniles and subadult populations

6.2.5 General Threats Faced by ESA-Listed Corals

Corals face numerous natural and man-made threats that shape their status and affect their ability to recover. Because 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 Rules (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 that results in greater releases of CO_2 that is then absorbed by seawater. Reef-building corals produce skeletons made of the aragonite form of calcium carbonate. Ocean acidification reduces aragonite concentrations in seawater, making it more difficult for corals to build their skeletons. Ocean acidification has the potential to cause substantial reduction in coral calcification and reef cementation. Further, ocean acidification impacts adult growth rates and fecundity, fertilization, pelagic planula settlement, polyp development, and juvenile growth. Ocean acidification can lead to increased colony breakage, fragmentation, and mortality. Based on observations in areas with naturally low pH, the effects of increasing ocean acidification may also include reductions in coral size, cover, diversity, and structural complexity.

As CO_2 concentrations increase in the atmosphere, more CO_2 is absorbed by the oceans, causing lower pH and reduced availability of calcium carbonate. Because of the increase in CO_2 and other 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 (m^2) per day (Carpenter 1986), and thereby remove up to 90-100 percent 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 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 sublethal 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 two 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.

6.2.6 Status of ESA-Listed Corals

6.2.6.1 Elkhorn Coral (Acropora palmata)

Elkhorn coral occurs throughout coastal areas in the Caribbean, Gulf of Mexico, and southwestern Atlantic (Figure 14).

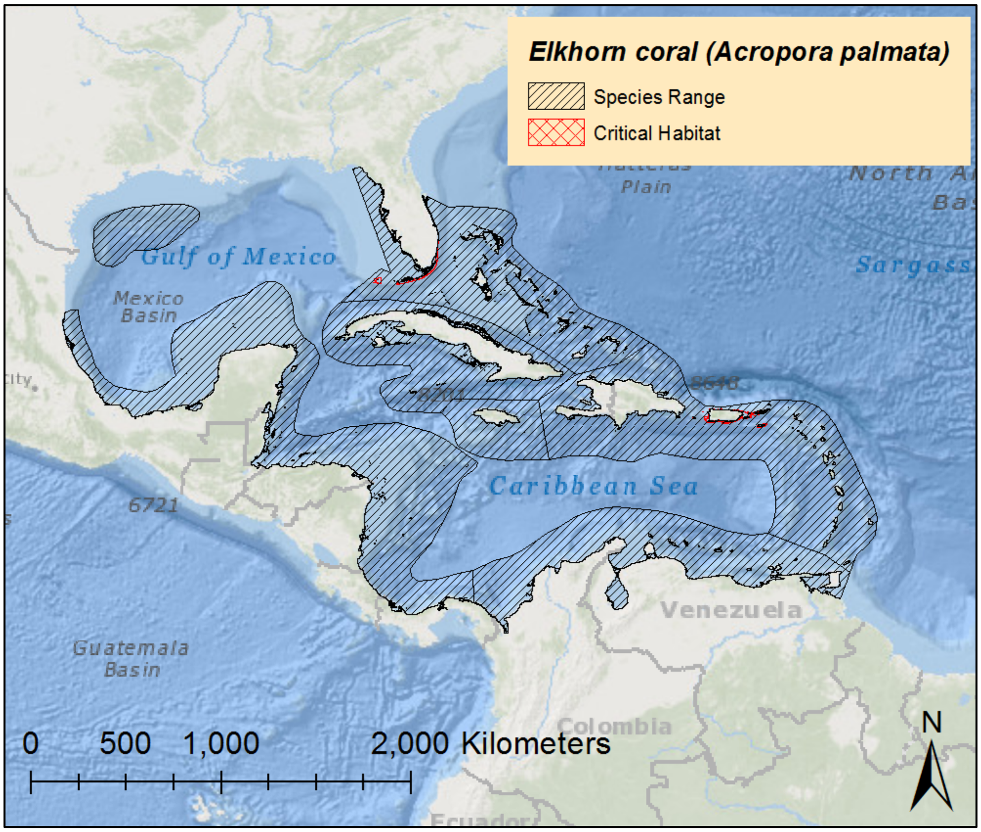

Figure 14. Map showing range of elkhorn coral

Elkhorn corals, as with all corals are composed of single polyp body forms, often present in numbers of hundreds to thousands creating dense clusters along the shallow ocean floor called colonies. Polyps are capable of catching and eating their own food, and have their own digestive, nervous, respiratory, and reproductive systems. In addition to being able to catch and eat their own food, elkhorn coral, along with most coral species contain zooxanthellae, a unicellular, symbiotic dinoflagellate, living within the endodermic tissues of individual polyps to provide photosynthetic support to the coral's energy budget and calcium carbonate secretion (NMFS 2005).

Elkhorn coral was listed as threatened under the ESA in 2006. In 2012, a proposal to change the listing to endangered was made, but in 2014 its threatened status was upheld. In 2008, critical habitat for elkhorn coral was designated in areas surrounding the Florida Keys, Puerto Rico, and portions of the USVI. Along with staghorn coral, elkhorn coral is the only other large, branching species of coral to produce and occupy vast complex environments within the Caribbean Sea's reef system. In all, there appears to be two distinct populations of elkhorn coral, a western Caribbean population and an eastern (Baums et al. 2005b) based on genetic analyses.

Life history

Elkhorn coral, like most stony corals, employ both sexual and asexual reproductive strategies to propagate. Sexual reproduction in corals includes gametogenesis, the process in which cells undergo meiosis to form gametes within the polyps. Since elkhorn coral is hermaphroditic, each polyp contains both sperm and egg cells that are released together in a "bundle", causing the coral gametes to develop externally from the parental colony. Elkhorn coral reproduces sexually after the full moon of July, August, and/or September, depending on location and timing of the full moon (ABRT 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 square inches (1,600 square cm [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. 2005b; 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 nine months are high based on settlement experiments (Szmant and Miller 2005).

Reproduction occurs primarily through asexual reproduction, generating multiple genetically identical colonies. Elkhorn coral can quickly monopolize large spaces of shallow ocean floor through fragment dissemination. A branch of elkhorn coral can be carried by waves and currents away from the mother colony to distances that range from 0.1 – 100 m, but fragments usually travel less than 30 m (NMFS 2005).

Because large colonies of elkhorn coral contain several thousand partially autonomous polyps, growth rates for the species are conveyed through the measurement of linear extensions of the organisms' skeletal branches. Depending on the size and location of the colony, physical growth rates for elkhorn corals range from approximately four to eleven centimeters per year. Branches are up to approximately 50 cm wide and range in thickness of about 4 - 5 cm. Individual colonies can grow to at least 2 m in height and 4 m in diameter (NMFS 2005). Total lifespan for the species is unknown (NMFS 2014).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section consists of abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the elkhorn coral.

Genetic samples from 11 locations throughout the Caribbean indicate that elkhorn coral populations in the eastern Caribbean (St. Vincent and the Grenadines, USVI, 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) (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 promotes dispersion of larval and gene flow between the eastern Caribbean and western Caribbean (Baums et al. 2006).

Colonial species present a special challenge in determining the appropriate unit to evaluate for abundance. However, the present population of Elkhorn coral is continuing at a very low abundance due to large declines in the past several decades (NMFS 2005). The western Caribbean is characterized by genetically depauperate 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. 2006).

Baums et al. (2006) concluded that the western Caribbean had higher rates of asexual recruitment and that the eastern Caribbean had higher rates of sexual recruitment. The research team claims that the postulated 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 for elkhorn coral. From the survey data, it can be inferred that genetic variability is more common in colonies within eastern populations as opposed to western. 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. 2005a). In contrast, sites within the eastern Caribbean displayed high variability. All 15 colonies sampled in Navassa had unique genotypes (Baums et al. 2006). Some sites have relatively high genotypic diversity such as in Los Roques, Venezuela (118 unique genotypes 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. 2006). In the Bahamas, about one third of the sampled colonies were unique genotypes, and in Panama between 24 and 65 percent of the sampled colonies had unique genotypes, depending on the site (Baums et al. 2006). A more-recent survey conducted along the coast of Puerto Rico found unique genotypes in 75 percent of the samples with high genetic diversity (Mège et al. 2014).

Elkhorn coral occurs in turbulent water on the back reef, fore reef, reef crest, and spur and groove zone in water ranging from one to thirty m in depth. Historically, elkhorn coral inhabited most waters of the Caribbean between one to five m depth. This included a diverse set of areas comprising of zones along Puerto Rico, Hispaniola, the Yucatan peninsula, the Bahamas, the southwestern Gulf of Mexico, the Florida Keys, the Southeastern Caribbean islands, and the northern coast of South America as seen in Figure 14 (Dustan and Halas 1987; Goreau 1959; Jaap 1984; Kornicker and Boyd 1962; Scatterday 1974; Storr 1964). While the present-day spatial distribution of elkhorn coral is similar to its historic spatial distribution, its presence within its range has become increasingly sparse due to declines in the latter half of the 20th century from a variety of abiotic and biotic threats.

Based on population estimates from both the Florida Keys and St. Croix, USVI, there are at least hundreds of thousands of elkhorn coral colonies. Absolute abundance is higher than estimates from these two locations given the presence of this species in many other locations throughout its range. The effective population size is smaller than indicated by abundance estimates due to the tendency for asexual reproduction. Across the Caribbean, percent cover appears to have remained relatively stable, albeit it at extremely low levels, since the population crash in the 1980s. Frequency of occurrence has decreased since the 1980s, indicating potential decreases in the extent of occurrence and effects on the species' range. However, the proportions of Caribbean sites where elkhorn coral is present and dominant have recently stabilized since the mid-2000s. There are locations such as the USVI where populations of elkhorn coral appear stable or possibly increasing in abundance and some such as the Florida Keys where population number appears to be decreasing.

Status

The decline in the total abundance of elkhorn coral has been attributed to a series of stressors consisting of disease, temperature-induced bleaching, excessive sedimentation, nitrification, pollution(i.e., oxybenzone from sunscreen), and large hurricanes/tropical storms (Brainard et al. 2011; Downs et al. 2016; Hernandez-Delgado et al. 2011; Mayor et al. 2006; Rogers 2012). It is believed that these effects act synergistically with one another thereby increasing the overall damage to already-stressed elkhorn coral colonies that have undergone disturbance by another threat. The current population trend appears to be steady, although there are places where populations continue to decrease and others where there appears to be modest or contained recovery (Miller et al. 2013). However, even if growth and recruitment end up surpassing mortality, this species requires prompt analysis and monitoring on a regional scale. Reasoning for this includes the current presence of areas with low genetic diversity and density within western Caribbean populations along with localized high rates of disease and bleaching (Miller et al. 2013).

Status of Species within the Action Area

In locations where historic quantitative data are available including USVI, there was a reduction of greater than 97 percent between the 1970s and early 2000s in elkhorn coral populations (ABRT 2005). Since the 2006 listing of elkhorn coral, continued population declines have occurred in some locations, including in the USVI, with certain populations of elkhorn coral decreasing up to an additional 50 percent or more (Lundgren and Hillis-Starr 2008; Muller et al. 2008; Williams et al. 2008; Colella et al. 2012; Rogers and Muller 2012). However, several studies describe elkhorn coral populations that are showing some signs of recovery or are stable including in the USVI (Mayor et al. 2006; Grober-Dunsmore et al. 2006; Rogers and Muller 2012).

In terms of density and abundance, maximum elkhorn coral density at ten sites in St. John was 0.18 colonies per m^2 (Muller et al. 2014). Mayor et al. (2006) surveyed 617 sites in Buck Island Reef National Monument, St. Croix, from May to June 2004 and extrapolated elkhorn coral density observed per habitat type to total available habitat. Within an area of 795 hectares, they estimated 97,232–134,371 (95 percent confidence limits) elkhorn coral colonies with any dimension of connected live tissue greater than one m. Mean densities (colonies ≥ 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.

At 8 of 11 sites in St. John, 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, USVI 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). However, the 2017 hurricanes are likely to have affected even these areas where elkhorn coral colony abundance was stable or increasing given the magnitude of the two hurricanes that severely impacted the Caribbean.

Critical Habitat

Critical habitat units for elkhorn and staghorn coral were designated in 2008 and include Florida (portions of Southeastern Florida and the Florida Keys), Puerto Rico, St. Thomas/St. John, and St. Croix. Elkhorn and staghorn coral critical habitat is described further in Section 7.2.7.

Recovery Goals

The 2015 Elkhorn Coral (*Acropora palmata*) and Staghorn Coral (*A. cervicornis*) Recovery Plan contains complete downlisting/delisting criteria for each of the two following recovery goals:

- Ensure population viability
    - Specific criteria include: 1) Preserving Abundance; 2) Maintaining Genotypic Diversity; and 3) Properly Observing and Recording Recruitment Rates
- Eliminate or sufficiently abate global, regional, and local threats

    Specific criteria include: 1) Developing quantitative recovery criterion through research to identify, treat, and reduce outbreaks of coral disease; 2) Controlling the Local and Global Impacts of Rising Ocean Temperature and Acidification; 3)

Reducing the Loss of Recruitment Habitat (if criterion 1, preserving abundance, is met then this objective is complete; 4) Reducing sources of nutrients, sediments, and contaminants; 5) Developing and adopting appropriate and effective regulatory mechanisms to abate threats; 6) Reducing impacts of natural and anthropogenic abrasion and breakage; and 7) Reducing impacts of predation.

6.2.6.2 Staghorn Coral (Acropora cervicornis)

Staghorn coral has the same range as elkhorn coral, occurring throughout coastal areas in the Caribbean, Gulf of Mexico, and southwestern Atlantic (Figure 14).

Staghorn coral is characterized by antler-like colonies with straight or slightly curved, cylindrical branches. The diameter of branches ranges form 0.25 – 5 cm (Lirman et al. 2010), and linear branch growth rates have been reported to range between three – 11.5 cm per year (ABRT 2005). The species can exist as isolated branches, individual colonies up to about 1.5 m diameter, and thickets comprised of multiple colonies that are difficult to distinguish from one another (ABRT 2005).

Staghorn coral naturally occurs on spur and groove, bank reef, patch reef, and transitional reef habitats, as well as on limestone ridges, terraces, and hard bottom habitats (Cairns 1982; Davis 1982; Gilmore and Hall 1976; Goldberg 1973; Jaap 1984; Miller et al. 2008; Wheaton and Jaap 1988). Historically it grew in thickets in water ranging from approximately 5 – 20 m in depth, though it has rarely been found to approximately 60 m (Davis 1982; Jaap 1984; Jaap et al. 1989; Schuhmacher and Zibrowius 1985; Wheaton and Jaap 1988). At the northern extent of its range, it grows in deeper water, 16-30 m (Goldberg 1973). Historically, staghorn coral was one of the primary constructors of mid-depth 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., 5 - 22 m; Miller et al. 2008). There is no evidence of range constriction, though loss of staghorn coral at the reef level has occurred (ABRT 2005).

Precht and Aronson (2004) suggest that coincident with climate warming, staghorn coral 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. 2014b).

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 60 cm and density ranged from one to ten colonies per fifteen m2 (García-Sais et al. 2013).

Life history

Relative to other corals, staghorn coral has a high growth rate that has 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 four to eleven centimeters per year (ABRT 2005). Annual linear extension has been found to be dependent on the size of the colony. New recruits and juveniles typically grow at slower rates. Stressed colonies and fragments may also exhibit slower growth.

Staghorn coral is a hermaphroditic broadcast spawning species. The spawning season occurs several nights after the full moon in July, August, or September depending on location and timing of the full moon and may be split over the course of more than one lunar cycle (Szmant 1986; Vargas-Angel et al. 2006). The estimated size at sexual maturity is approximately seventeen centimeters branch length, and large colonies produce proportionally more gametes than small colonies (Soong and Lang 1992). Basal and branch tip tissue is not fertile (Soong and Lang 1992). Sexual recruitment rates are low, and this species is generally not observed in coral settlement studies. Laboratory studies have found that certain species of crustose-coralline algae produce exudates that 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.

Population Dynamics

Staghorn coral historically was one of the dominant species on most Caribbean reefs, forming large, single-species thickets and giving rise to the 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, USVI, Belize), there was a reduction of approximately 92 to greater than 97 percent between the 1970s and early 2000s (ABRT 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 five to twenty 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.

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 (staghorn and elkhorn) decreasing up to an additional 50 percent or more (Colella et al. 2012; Lundgren and Hillis-Starr 2008; Muller et al. 2008; Rogers and Muller 2012; Williams et al. 2008). There are some small pockets of remnant robust populations such as in southeast Florida (Vargas-Angel et al. 2003), Honduras (Keck et al. 2005; Riegl et al. 2009), and Dominican Republic (Lirman et al. 2010). Additionally, Lidz and Zawada (2013) observed 400 colonies of staghorn coral along 44 miles (70.2 km) of transects near Pulaski Shoal in the Dry Tortugas where the species had not been seen since the cold-water die-off of the 1970s. Cover of staghorn coral increased on a Jamaican reef from 0.6 percent in 1995 to 10.5 percent in 2004 (Idjadi et al. 2006).

A report on the status and trends of Caribbean corals over the last century indicates that cover of staghorn coral has remained relatively stable (though much reduced) throughout the region since the large mortality events of the 1970s and 1980s. 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 percent of reefs), remained relatively stable at 30 percent through the 1990s, and decreased to approximately 20 percent of the reefs in 2000-2004 and approximately 10 percent in 2005-2011 (Jackson et al. 2014b).

Vollmer and Palumbi (2007) examined 22 populations of staghorn coral from nine regions in the Caribbean (Panama, Belize, Mexico, Florida, Bahamas, Turks and Caicos, Jamaica, Puerto Rico, and Curaçao) and concluded that populations greater than approximately 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 two km, suggesting that gene flow in staghorn coral may not occur at much smaller spatial scales (Garcia Reyes and Schizas 2010; Vollmer and Palumbi 2007). This fine-scale population structure was greater when considering genes of elkhorn coral were found in staghorn coral due to back-crossing of the hybrid *A. prolifera* with staghorn coral (Garcia Reyes and Schizas 2010; Vollmer and Palumbi 2007). Populations in Florida and Honduras are genetically distinct from each other and other populations in the USVI, Puerto Rico, Bahamas, and Navassa (Baums et al. 2010), indicating little to no larval connectivity overall. However, some potential connectivity between the USVI and Puerto Rico was detected and also between Navassa and the Bahamas (Baums et al. 2010).

Based on population estimates, there are at least tens of millions of colonies present in the Florida Keys and Dry Tortugas combined. Absolute abundance is higher than the estimate from these two locations given the presence of this species in many other locations throughout its range. The effective population size is smaller than indicated by abundance estimates due to the tendency for asexual reproduction. There is no evidence of range constriction or extirpation at the island level. However the species is absent at the reef level. 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, percent cover appears to have remained relatively stable since the population crash in the 1980s. 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 periods. 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. Percent benthic cover and proportion of reefs where staghorn coral is dominant have remained stable since the mid-1980s and since the listing of the species as threatened in 2006. We also conclude that population abundance is at least tens of millions of colonies, but likely to decrease in the future with increasing threats.

Status

The species has undergone substantial population decline and decreases in the extent of occurrence throughout its range due mostly to disease. Although localized mortality events have continued to occur, percent benthic cover and proportion of reefs where staghorn coral is dominant have remained stable over its range since the mid-1980s. 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 five to twenty 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. Its absolute population abundance has been estimated as at least tens of millions of colonies in the Florida Keys and Dry Tortugas combined and is higher than the estimate from these two locations due to the occurrence of the species in many other areas throughout its range. 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.

Status of Species in the Action Area

Staghorn corals have been reported on inshore colonized hard bottom, mid-shelf colonized hard bottom and coral reefs, and offshore shelf reefs around USVI, although numbers of colonies have largely not been quantified except in cases of proposed water resources development projects. The National Coral Reef Monitoring Program (NCRMP), which began in 2012 and is administered by NOAA's Coral Reef Conservation Program, has now completed 2 surveys in USVI. Based on the data, staghorn coral colonies have been observed at various locations around all three of the main islands (St. Thomas, St. John, and St. Croix) in 2013 and 2015 and in waters around Buck Island, St. Croix and Cane Bay, St. Croix in 2012 when the first NCRMP survey was conducted in these two locations only in St. Croix. The surveys are based on stratified random sampling in coral habitat types (colonized hard bottom, coral reef) in water depths less than 100 ft. This means that the same locations are not necessarily visited during each biannual sampling event as sampling locations are selected randomly so there are likely to be more colonies of these corals than found in the surveys. Staghorn coral colonies were observed more frequently around St. Thomas than St. John or St. Croix during the 2013 and 2015 NCRMP surveys but this does not necessarily indicate the species is more prevalent in St. Thomas in comparison to other areas in USVI.

Critical Habitat

As noted in Section 7.2.6.1, elkhorn and staghorn coral critical habitat is described further in Section 7.2.7.

Recovery Goals

The recovery goals for elkhorn and staghorn corals were described in the 2015 Elkhorn Coral (*Acropora palmata*) and Staghorn Coral (*A. cervicornis*) Recovery Plan and detailed in Section 7.2.6.1 (Elkhorn Coral). Two recovery goals were identified for Atlantic acroporid corals:

- Ensure population viability
- Eliminate or sufficiently abate global, regional, and local threats.

6.2.6.3 Pillar Coral (Dendrogyra cylindrus)

Pillar coral is present in the western Atlantic Ocean and throughout the greater Caribbean Sea, though absent from the southwest Gulf of Mexico (Tunnell 1988; Figure 15).

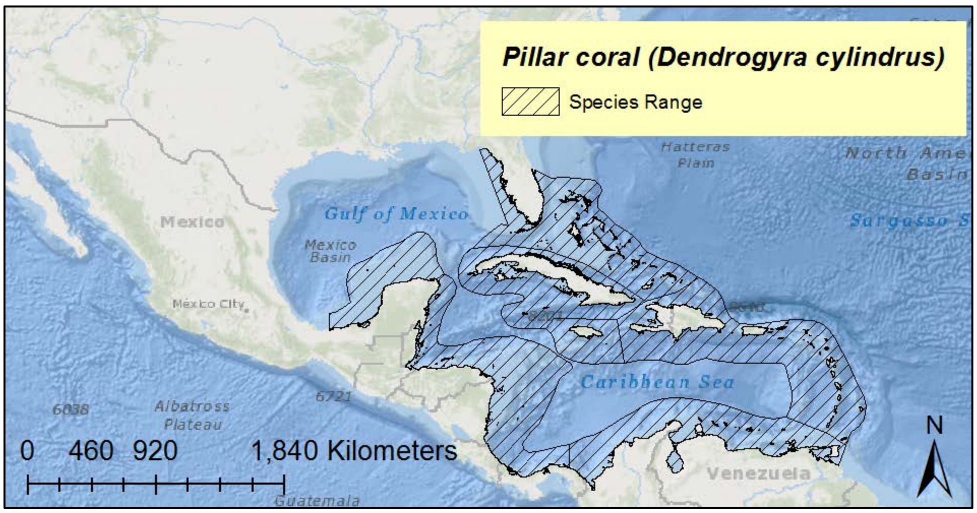

Figure 15. Range map for pillar coral

On September 10, 2014, NMFS listed pillar star coral as threatened. Pillar corals form tubular columns on top of encrusted foundations. Colonies are generally grey-brown in color and may reach approximately three m in height. Polyps' tentacles remain extended during the day, giving columns a furry appearance.

Brainard et al. (2011) identified a single known colony in Bermuda that is in poor condition. There is fossil evidence of the presence of the species off Panama less than 1,000 years ago, but it has been reported as absent today (FFWCC 2013). Pillar coral inhabits most reef environments in water depths ranging from approximately one to twenty-five m, but it is most common in water between approximately five to fifteen m deep (Acosta and Acevedo 2006; Cairns 1982; Goreau and Wells 1967).

Life history

Reported average growth rates for pillar coral have been documented to be approximately 1.8-2.0 centimeters per year in linear extension within the Florida Keys, compared to 0.8 centimeters per year as reported in Colombia and Curaçao. Partial mortality rates are size-specific with larger colonies having greater rates. Frequency of partial mortality can be high (e.g., 65 percent of 185 colonies surveyed in Colombia), while the amount of partial mortality per colony is generally low (average of three percent of tissue area affected per colony).

Pillar coral is a gonochoric broadcast spawning species with relatively low annual egg production for its size. The combination of gonochoric spawning with persistently low population densities is expected to yield low rates of successful fertilization and low larval supply. Sexual recruitment of this species is low, and reports indicate juvenile colonies are lacking in the Caribbean. Spawning has been observed to occur several nights after the full moon of August in the Florida Keys (Neely et al. 2013; Waddell and Clarke 2008) and in La Parguera, Puerto Rico (Szmant 1986). Pillar coral can also reproduce asexually by fragmentation following storms or other physical disturbance, but it is uncertain how much storm-generated fragmentation contributes to asexually produced offspring.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section consists of abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the pillar coral.

Pillar coral is uncommon but conspicuous with scattered, isolated colonies and is rarely found in aggregations. Benthic cover is generally less than one percent in monitoring studies. Mean density of pillar coral was approximately 0.5 colonies per ten m^2 in the Florida Keys between 2005 and 2007. In a study of pillar coral demographics at Providencia Island, Colombia, 283 pillar coral colonies were detected in a survey of 1.66 square kilometers (km^2) for an overall density of approximately 450 colonies per square mile (mi^2).

Information on pillar coral is most extensive for Florida. Pillar coral ranked as the least abundant to third least abundant coral species in stratified random surveys of the Florida Keys between 2005 and 2009 and was not encountered in surveys in 2012 (Miller et al. 2013). Pillar coral was seen only on the ridge complex and mid-channel reefs at densities of approximately 1 and 0.1 colonies per 10 m^2 (approximately 100 square feet [ft^2]), respectively, between 2005 and 2010 in surveys from West Palm Beach to the Dry Tortugas (Burman et al. 2012). In surveys conducted between 1999 and 2016 from Palm Beach to the Dry Tortugas, pillar coral was present at 2 percent of sites surveyed and ranged in density from 0 to 0.4 colonies per m^2 with an average density of 0.004 colonies per 10 m^2 (approximately 100 ft^2; NOAA NCRMP). In 2014, there were 714 known colonies of pillar coral along the Florida reef tract from southeast Florida to the Dry Tortugas. By 2017, many of these colonies had suffered tissue loss, and over half (57 percent) suffered complete mortality due to disease, most likely associated with multiple years of warmer than normal temperatures (Lewis et al. 2017; K. Neely and C. Lewis, Keys Marine Lab, unpublished data). The majority of these colonies were lost from the northern portion of the reef tract (Figure 16).

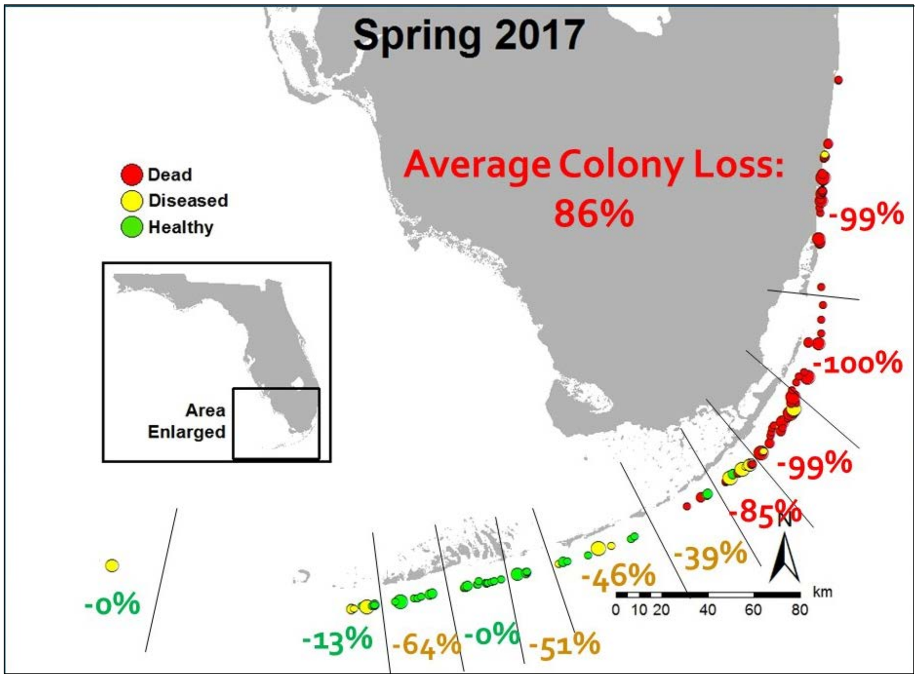

Figure 16. Condition of known pillar coral colonies in Florida between 2014 and 2017 (Figure courtesy of K. Neely and C. Lewis)

Density of pillar corals in other areas of the Caribbean is also low and on average less than 0.1 colonies per 10 m^2. The average number of pillar coral colonies in remote reefs off southwest Cuba was 0.013 ± 0.045 colonies per 10 m (approximately 32 ft) transect, and the species ranked sixth rarest out of 38 coral species (Alcolado et al. 2010). In a study of pillar coral demographics at Providencia Island, Colombia, a total of 283 pillar coral colonies were detected in a survey of 1.66 km^2 (0.6 mi^2) for an overall density of approximately 0.000017 colonies per 10 m^2 (approximately 100 ft2; Acosta and Acevedo 2006). In Puerto Rico, density of pillar coral ranged from 0 to 0.3 colonies per m^2 with an average density of 0.03 colonies per 10 m^2 (approximately 100 ft^2); it occurred at 4 percent of the sites surveyed between 2008 and 2016 (NOAA NCRMP).

Benthic cover is generally less than 1 percent in monitoring studies. Pillar coral's average cover was 0.002 percent on patch reefs and 0.303 percent in shallow offshore reefs in annual surveys of 37 sites in the Florida Keys between 1996 and 2003 (Somerfield et al. 2008). In surveys conducted in Florida between 1996 and 2016, cover of pillar coral ranged from 0 to 0.5 percent with an average of 0.0002 percent (NOAA NCRMP). In Puerto Rico, cover of pillar coral ranged between 0 and 4 percent with an average of 0.02 percent in surveys conducted between 2001 and 2016 (NOAA NCRMP). In Dominica, pillar coral comprised less than 0.9 percent cover and was present at 13.3 percent of 31 surveyed sites (Steiner 2003). Pillar coral was observed on 1 of 7 fringing reefs surveyed off Barbados, and cover was 2.7 ± 1.4 percent (Tomascik and Sander 1987).

Other than the declining population in Florida, there are two reports of population trends from the Caribbean. In monitored photo-stations in Roatan, Honduras, cover of pillar coral increased slightly from 1.35 percent in 1996 to 1.67 percent in 1999 and then declined to 0.44 percent in 2003 and to 0.43 percent in 2005 (Riegl et al. 2009).

Pillar coral is currently uncommon to rare throughout Florida and the Caribbean. Low abundance and infrequent encounter rate in monitoring programs result in small samples sizes. The low coral cover of this species renders monitoring data difficult to extrapolate to realize trends. The few studies that report pillar coral population trends indicate a general decline at some specific sites, though it is likely that the population remains stable at other sites. Low density and gonochoric broadcast spawning reproductive mode, coupled with no observed sexual recruitment, indicate that natural recovery potential from mortality is low.

Status

Pillar coral survival is susceptible to a number of threats, but there is little evidence of population declines thus far. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because pillar coral is limited to an area with high, localized human impacts and predicted increasing threats. Pillar coral inhabits most reef environments in water depths ranging from one to twenty-five m, but is naturally rare. Estimates of absolute abundance are at least tens of thousands of colonies in the Florida Keys, and absolute abundance is higher than estimates from this location due to the occurrence of the species in many other areas throughout its range. It is a gonochoric broadcast spawner with observed low sexual recruitment. Its low abundance, combined with its geographic location, exacerbates vulnerability to extinction. This is because increasingly severe conditions within the species' range are likely to affect a high proportion of its population at any given point in time. Also, low sexual recruitment is likely to inhibit recovery potential from mortality events, further exacerbating its vulnerability to extinction. We anticipate that pillar coral is likely to decrease in abundance in the future with increasing threats.

Status of Species within the Action Area

In surveys conducted in the USVI between 1992 and 2015, percent cover of pillar coral ranged from 0 to 6 percent with an average cover of 0.03 percent (NOAA NCRMP). In the USVI, 7 percent of 26 monitored colonies in permanent monitoring sites experienced total colony mortality between 2005 and 2007, though the very low cover of pillar coral (0.04 percent) remained relatively stable during this time period (Smith et al. 2013). Density of pillar coral ranged between 0 and 0.3 colonies per m^2 with an average density of 0.01 colonies per 10 m^2 (approximately 100 ft^2); it occurred in three percent of the randomly selected sites surveyed by NOAA between 2002 and 2015 (NOAA NCRMP). At permanent monitoring stations in the USVI, pillar coral was observed in low abundance at 10 of 33 sites and ranged in cover from less than 0.05-0.22 percent where present (Smith 2013).

Critical Habitat

No critical habitat has been designated for pillar coral.

Recovery Goals

No final recovery plans currently exist for pillar coral, however a recovery outline was published in 2014. The following short and long-term recovery goals are listed in the document:

Short-Term Goals:

- Increase understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling
- Through research, increase understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility
- Decrease locally manageable stress and mortality sources (e.g., acute sedimentation, nutrients, contaminants, over-fishing).
- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit *D. cylindrus*, *M. ferox*, and *Orbicella* spp.

Long-Term Goals:

- Cultivate and implement U.S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats.
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment.

6.2.6.4 Rough Cactus Coral (Mycetophyllia ferox)

Rough cactus coral occurs in the western Atlantic Ocean and throughout the wider Caribbean Sea (Figure 17).

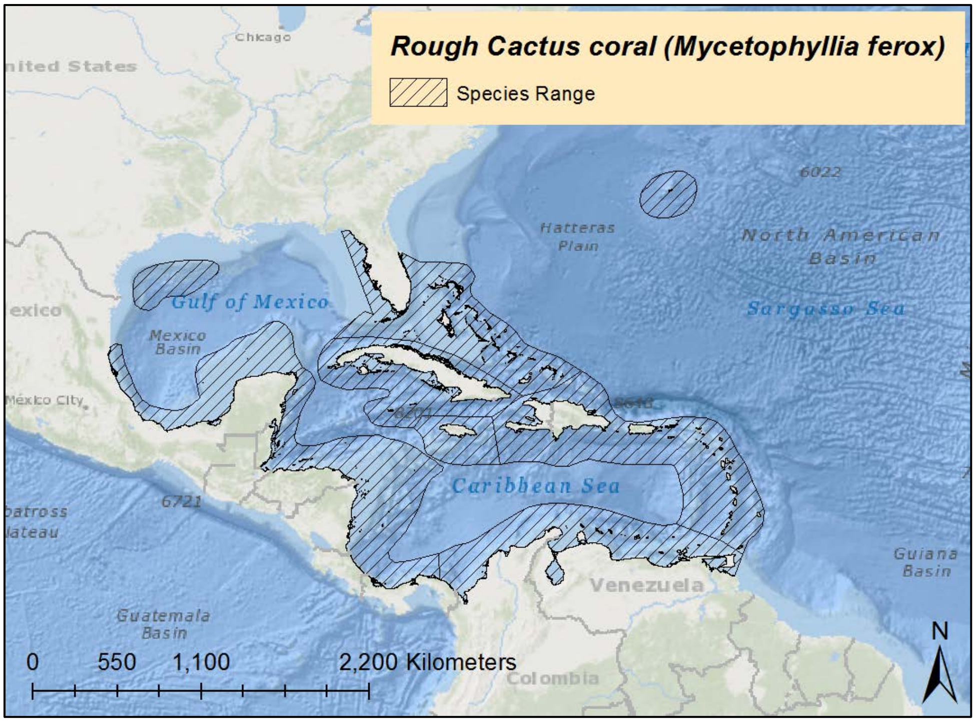

Figure 17. Range map for rough cactus coral

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

While rough cactus coral occurs in the western Atlantic Ocean and throughout the wider Caribbean Sea, it has not been reported in the Flower Garden Banks (Gulf of Mexico) or in Bermuda. It inhabits reef environments in water depths of five to ninety m, including shallow and mesophotic habitats (e.g/, > 30 m).

Life history

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

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section consists of abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the rough cactus coral.

Rough cactus coral is usually uncommon or rare according to published and unpublished records, indicating that it constitutes < 0.1 percent species contribution (percent of all colonies counted) and occurs at densities < 0.8 colonies per ten m^2 in Florida and at 0.8 colonies per 100 m transect in Puerto Rico sites sampled by the Atlantic and Gulf Rapid Reef Assessment (Veron and Stafford-Smith 2002, and AGRRA database as cited in; Brainard et al. 2011). Recent monitoring data (e.g., since 2000) from Florida (National Park Service permanent monitoring stations), La Parguera Puerto Rico, and St. Croix (USVI/NOAA Center for Coastal Monitoring and Assessment randomized monitoring stations) show *Mycetophyllia ferox* cover to be consistently low with occasional observations of cover by this species of up to two percent and no apparent temporal trend (Brainard et al. 2011).

Rough cactus coral may have been more abundant in the upper Florida Keys in the early mid-1970s (the methods are not well described for that study) than current observations based on data from Dustan (1977). Long-term Coral Reef Ecological Monitoring Program data from Florida containing species presence/absence information from fixed sites (stations) show a dramatic decline; for 97 stations in the main Florida Keys, occurrence had declined from 20 stations in 1996 to four stations in 2009; in Dry Tortugas occurrence had declined from eight out of twenty-one stations in 2004 to three stations in 2009 (Brainard et al. 2011).

According to the International Union for Conservation of Species (IUCN) Species Account and the CITES (Convention on International Trade in Endangered Species of Wild Flora and Fauna) species database, rough cactus coral occurs throughout the U.S. waters of the western Atlantic but has not been reported from Flower Garden Banks (Hickerson et al. 2008). The following areas include locations within federally protected waters where rough cactus coral has been observed and recorded (cited in Brainard et al. 2011): Dry Tortugas National Park; Virgin Island National Park/Monument; Florida Keys National Marine Sanctuary; Navassa Island National Wildlife Refuge; Biscayne National Park; Buck Island Reef National Monument.

On reefs where rough cactus coral is found, it generally occurs at abundances of less than one colony per approximately 10 m^2 and percent cover of less than 0.1 (Burman et al. 2012). Based on population estimates, there are at least hundreds of thousands of rough cactus coral colonies present in the Florida Keys and Dry Tortugas combined. Absolute abundance is higher than the estimate from these two locations given the presence of this species in many other locations throughout its range. Low encounter rate and percent cover coupled with the tendency to include *Mycetophyllia* spp. at the genus level make it difficult to discern population trends of rough cactus coral from monitoring data. However, reported losses of rough cactus coral from monitoring stations in the Florida Keys and Dry Tortugas (63-80 percent loss) indicate population decline in these locations. Based on declines in Florida, we conclude rough cactus coral has likely declined throughout its range and will continue to decline based on increasing threats. As a result, it is presumed that genetic diversity for the species is low.

Status

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

Status of Species within the Action Area

Benthic cover of rough cactus coral in the Red Hind Marine Conservation District off St. Thomas, which includes mesophotic coral reefs, was 0.003 ± 0.004 percent in 2007, accounting for 0.02 percent of coral cover, and ranking second to last out of 21 coral species (Nemeth et al. 2008). In the USVI between 2001 and 2012, rough cactus coral appeared in 12 of 33 survey sites and accounted for 0.01 percent of the colonized bottom and 0.07 percent of the coral cover, ranking as the 13th most common coral species (Smith 2013).

Critical Habitat

No critical habitat has been designated for rough cactus coral.

Recovery Goals

No final recovery plan currently exists for rough cactus coral, however a recovery outline was developed in 2014 to serve as interim guidance to direct recovery efforts, including recovery planning, until a final recovery plan is developed and approved for the five coral species listed in September 2014. The recovery goals are the same for all five species (see Section 7.2.6.3) with short and long-term goals.

6.2.6.5 Lobed Star, Mountainous Star, and Boulder Star Coral (Orbicella annularis, Orbicella faveolata, and Orbicella franksi)

Lobed, mountainous, and boulder star coral occur in the western Atlantic and greater Caribbean as well as the Flower Garden Banks. Lobed and mountainous star coral may be absent from Bermuda (Figure 18).

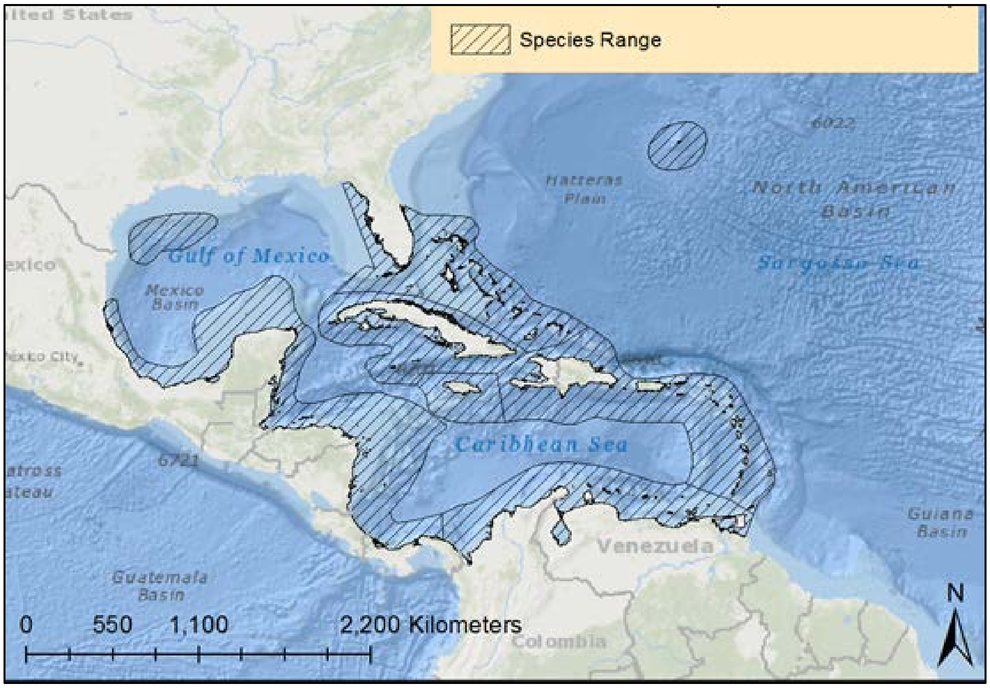

Figure 18. Range map for lobed, mountainous, and boulder star corals

On September 10, 2014, NMFS listed lobed star, mountainous star, and boulder star coral as threatened. Lobed star coral (*Orbicella annularis*), mountainous star coral (*Orbicella faveolata*), and boulder star coral (*Orbicella franksi*) are the three species in the *Orbicella annularis* star coral complex. These three species were formerly in the genus *Montastraea*; however, recent work has reclassified the three 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 Knowton 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 three 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 three 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 three separate species.

Lobed Star Coral

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

Lobed star coral is reported from most reef environments within the Caribbean (except for Bermuda) in depths of approximately 0.5-20 m. The star coral species complex is a common, often dominant component of Caribbean mesophotic (e.g., >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.

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.

Boulder Star Coral

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

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

Life history

The star coral species complex has growth rates ranging from 0.06-1.2 centimeters per year and averaging approximately one centimeter in linear growth per year. The reported growth rate of lobed star coral is 0.4 to 1.2 centimeters per year (Cruz-Piñón et al. 2003; Tomascik 1990). They grow slower in deep and murky waters.

All three species of the star coral complex are hermaphroditic broadcast spawners, with spawning concentrated on six to eight nights following the full moon in late August, September, or early October depending on location and timing of the full moon. All three 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 one to two hours earlier. Fertilization success measured in the field was generally below 15 percent for all three 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 two species of the *Orbicella* genus. In Puerto Rico, minimum size at reproduction for the star coral species complex was 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 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.

<u>Lobed Star Coral</u>

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>

Life history characteristics of mountainous star coral is considered intermediate between lobed star coral and boulder star coral especially regarding growth rates, tissue regeneration, and egg size. Spatial distribution may affect fecundity on the reef, with deeper colonies of mountainous star coral being less fecund due to greater polyp spacing. Reported growth rates of mountainous star coral range between 0.12 and 0.64 in (0.3 and 1.6 cm) per year (Cruz-Piñón et al. 2003; Tomascik 1990; Villinski 2003; Waddell 2005). Graham and van Woesik (2013) report that 44 percent of small colonies of mountainous star coral in Puerto Morelos, Mexico that resulted from partial colony mortality produced eggs at sizes smaller than those typically characterized as being mature. The number of eggs produced per unit area of smaller fragments was significantly less than in larger size classes. Szmant and Miller (2005) reported low post-settlement survivorship for mountainous star coral transplanted to the field with only 3-15 percent remaining alive after 30 days. Post-settlement survivorship was much lower than the 29 percent observed for elkhorn coral after 7 months (Szmant and Miller 2005).

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

Boulder Star Coral

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

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

Population Dynamics

Lobed Star Coral

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

In the Florida Keys, abundance of lobed star coral ranked 30 out of 47 coral species in 2005, 13 out of 43 in 2009, and 12 out of 40 in 2012. Extrapolated population estimates from stratified random samples were 5.6 million ± 2.7 million (SE) in 2005, 11.5 million ± 4.5 million (SE) in 2009, and 24.3 million ± 12.4 million (SE) in 2012. Size class distribution was somewhat variable between survey years, with a larger proportion of colonies in the smaller size classes in 2005 compared to 2009 and 2012 and a greater proportion of colonies in the greater than 36-in (90 cm) size class in 2012 compared to 2005 and 2009. Partial colony mortality was lowest at less than 4 in (10 cm; as low as approximately 5 percent) and up to approximately 70 percent in the larger size classes. In the Dry Tortugas, Florida, abundance of lobed star coral ranked 41 out of 43 in 2006 and 31 out of 40 in 2008. The extrapolated population estimate was 0.5 million ± 0.3 million (SE) colonies in 2008. Differences in population estimates between years may be attributed to sampling effort rather than population trends (Miller et al. 2013).

Colony density varies by habitat and location, and ranges from less than 0.1 to greater than one colony per approximately 100 ft^2 (10 m^2). In surveys of 1,176 sites in southeast Florida, the Dry Tortugas, and the Florida Keys between 2005 and 2010, density of lobed star coral ranged between 0.09 and 0.84 colonies per approximately 100 ft^2 (10 m^2) and was highest on mid-channel reefs followed by inshore reefs, offshore patch reefs, and fore-reefs(Burman et al. 2012). Along the east coast of Florida, density was highest in areas south of Miami at 0.94 colonies per 108 ft^2 (10 m^2) compared to 0.11 colonies per 108 ft^2 (10 m^2) in Palm Beach and Broward Counties (Burman et al. 2012). In surveys between 2005 and 2007 along the Florida reef tract from Martin County to the lower Florida Keys, density of lobed star coral was approximately 1.3 colonies per approximately 100 ft^2 ([10 m2] Wagner et al. 2010). Off southwest Cuba on remote reefs, lobed star coral density was 0.31 ± 0.46 (SE) per approximately 30 ft (10 m) transect on 38 reef-crest sites and 1.58 ± 1.29 colonies per approximately 30 ft (10 m) transect on 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).

Population trends are available from a number of studies. In a study of sites inside and outside a marine protected area (MPA) 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 percent in permanent monitoring stations in the Florida Keys (Waddell and Clarke 2008). Cover of lobed star coral declined 71 percent in permanent monitoring stations between 1996 and 1998 on a reef in the upper Florida Keys (Porter et al. 2001).

Mountainous Star Coral

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 three well-defined populations based on 5 genetic markers, but the populations were not stratified by geography, indicating they were shared among the three 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 three genotypes (Levitan et al. 2011) potentially indicating 30 percent clonality.

Extrapolated population estimates from stratified random samples in the Florida Keys were 39.7 ± 8 million (SE) colonies in 2005, 21.9 ± 7 million (SE) colonies in 2009, and 47.3 ± 14.5 million (SE) colonies in 2012. The greatest proportion of colonies tended to fall in the 4-8 in (10-20 cm) and 8-12 in (20-30 cm) size classes in all survey years, but there was a fairly large proportion of colonies in the greater than 36-in (90 cm) -size class. Partial mortality of the colonies was between 10 percent and 60 percent of the surface across all size classes. In the Dry Tortugas, Florida, mountainous star coral ranked seventh most abundant out of 43 coral species in 2006 and fifth most abundant out of 40 in 2008. Extrapolated population estimates were 36.1 ± 4.8 million (SE) colonies in 2006 and 30 ± 3.3 million (SE) colonies in 2008. The size classes with the largest proportion of colonies were 4-8 in (10-20 cm) and 8-12 in (20-30 cm), but there was a large proportion of colonies in the greater-than-36-in (90 cm) size class. Partial mortality of the colonies ranged between approximately 2 percent and 50 percent. Because these population abundance estimates are based on random surveys, differences between years may be attributed to sampling effort rather than population trends (Miller et al. 2013).

In a survey of 31 sites in Dominica between 1999 and 2002, mountainous star coral was present at 80 percent of the sites at 1-10 percent cover (Steiner 2003). In a 1995 survey of 16 reefs in the Florida Keys, mountainous star coral ranked as the coral species with the second highest percent cover (Murdoch and Aronson 1999). On 84 patch reefs (10 ft [3 m] to 16.5 ft [5 m] depth) spanning 149 mi (240 km) in the Florida Keys, mountainous star coral was the third most abundant coral species comprising 7 percent of the 17,568 colonies encountered. It was present at 95 percent of surveyed reefs between 2001 and 2003 (Lirman and Fong 2007). In surveys of 280 sites in the upper Florida Keys in 2011, mountainous star coral was present at 87 percent of sites visited (Miller et al. 2011). In 2003 on the East Flower Garden Bank, mountainous star coral comprised 10 percent of the 76.5 percent coral cover on reefs 105-132 ft (32-40 m), and partial mortality due to bleaching, disease, and predation were rare at monitoring stations (Precht et al. 2005).

Colony density ranges from approximately 0.1-1.8 colonies per 108 ft^2 (10 m^2) and varies by habitat and location. In surveys along the Florida reef tract from Martin County to the lower Florida Keys, density of mountainous star coral was approximately 1.6 colonies per 108 ft^2 (10 m2; Wagner et al. 2010). On remote reefs off southwest Cuba, density of mountainous star coral was 0.12 ± 0.20 (SE) colonies per 33 ft (10 m) transect on 38 reef-crest sites and 1.26 ± 1.06 (SE) colonies per 33 ft (10 m) transect on 30 reef-front sites (Alcolado et al. 2010). In surveys of 1,176 sites in southeast Florida, the Dry Tortugas, and the Florida Keys between 2005 and 2010, density of mountainous star coral ranged between 0.17 and 1.75 colonies per 108 ft^2 (10 m^2) and was highest on mid-channel reefs followed by offshore patch reefs and fore-reefs (Burman et al. 2012). Along the east coast of Florida, density was highest in areas south of Miami at 0.94 colonies per 108 ft^2 (10 m^2) compared to 0.11 colonies per 108 ft^2 (10 m^2) in Palm Beach and Broward Counties (Burman et al. 2012).

Boulder Star Coral

Boulder star coral is reported as common. In a 1995 survey of 16 reefs in the Florida Keys, boulder star coral had the highest percent cover of all species (Murdoch and Aronson 1999). In surveys throughout the Florida Keys, boulder star coral in 2005 ranked as the 26th most abundant out of 47 coral species, 32nd out of 43 in 2009, and 33rd out of 40 in 2012. Extrapolated population estimates from stratified random surveys were 8.0 ± 3.5 million (SE) colonies in 2005, 0.3 ± 0.2 million (SE) colonies in 2009, and 0.4 ± 0.4 million (SE) colonies in 2012. The authors note that differences in extrapolated abundance between years were more likely a function of sampling design rather than an indication of population trends. In 2005, the greatest proportions of colonies were in the smaller size classes of approximately 10-20 cm and approximately 20-30 cm. Partial colony mortality ranged from zero percent to approximately 73 percent and was generally higher in larger colonies (Miller et al. 2013).

In the Dry Tortugas, Florida, boulder star coral ranked fourth highest in abundance out of 43 coral species in 2006 and 8th out of 40 in 2008. Extrapolated population estimates were 79 ± 19 million (SE) colonies in 2006 and 18.2 ± 4.1 million (SE) colonies in 2008. Miller et al. (2013) notes the difference in estimates between years was more likely a function of sampling design rather than population decline. In the first year of the study (2006), the greatest proportion of colonies were in the size class approximately 20-30 cm with twice as many colonies as the next most numerous size class and a fair number of colonies in the largest size class of greater than 90 centimeters. Partial colony mortality ranged from approximately 10-55 percent. Two years later (2008), no size class was found to dominate, and proportion of colonies in the medium-to-large size classes (approximately 60-90 centimeters) appeared to be less than in 2006. The number of colonies in the largest size class of greater than 90 cm remained consistent. Partial colony mortality ranged from approximately 15-75 percent (Miller et al. 2013).

Abundance in Curaçao and Puerto Rico appears to be stable over an eight to ten year period. In Curaçao, abundance was stable between 1997 and 2005, with partial mortality similar or less in 2005 compared to 1998 (Bruckner and Bruckner 2006). Abundance was also stable between 1998-2008 at nine sites off Mona and Desecheo Islands, Puerto Rico. In 1998, four percent of all corals at six sites surveyed off Mona Island were boulder star coral colonies and approximately five percent in 2008; at Desecheo Island, about two percent of all coral colonies were boulder star coral in both 2000 and 2008 (Bruckner and Hill 2009).

Based on population estimates, there are at least tens of millions of colonies present in both the Dry Tortugas and USVI. Absolute abundance is higher than the estimate from these two locations given the presence of this species in many other locations throughout its range. The frequency and extent of partial mortality, especially in larger colonies of boulder star coral, appear to be high in some locations such as Florida and Cuba, though other locations like the Flower Garden Banks appear to have lower amounts of partial mortality. A decrease in boulder star coral percent cover by 38 percent and a shift to smaller colony size across five countries suggest that population decline has occurred in some areas; colony abundance appears to be stable in other areas. We anticipate that while population decline has occurred, boulder star coral is still common with the number of colonies at least in the tens of millions. Additionally, we conclude that the buffering capacity of boulder 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. We also anticipate that the population abundance is likely to decrease in the future with increasing threats.

The star coral species complex has growth rates ranging from 0.06-1.2 centimeters per year and averaging approximately one-centimeter linear growth per year. Boulder star coral is reported to be the slowest of the three species in the complex (Brainard et al. 2011). They grow slower in deep or murky waters.

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

Status

Lobed star coral

Lobed star coral was historically considered one of the most abundant species in the Caribbean (Weil and Knowton 1994). Percent cover has declined to between 37 percent and 90 percent over the past several decades at reefs at Jamaica, Belize, Florida Keys, The Bahamas, Bonaire, Cayman Islands, Curaçao, Puerto Rico, USVI, and St. Kitts and Nevis. Based on population estimates, there are at least tens of millions of lobed star coral colonies present in the Florida Keys and Dry Tortugas combined. Absolute abundance is higher than the estimate from these two locations given the presence of this species in many other locations throughout its range. Lobed star coral remains common in occurrence. Abundance has decreased in some areas to between 19 percent and 57 percent and shifts to smaller size classes have occurred in locations such as Jamaica, Colombia, The Bahamas, Bonaire, Cayman Islands, Puerto Rico, USVI, 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. We conclude that while substantial population decline has occurred in lobed star coral, it is still common throughout the Caribbean and remains one of the dominant species numbering at least in the tens of millions of colonies. We conclude that 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. We also conclude that the population abundance is likely to decrease in the future with increasing threats.

In the Florida Keys, abundance of lobed star coral ranked 30 out of 47 coral species in 2005, 13 out of 43 in 2009, and 12 out of 40 in 2012. Extrapolated population estimates from stratified random samples were 5.6 million ± 2.7 million (SE) in 2005, 11.5 million ± 4.5 million (SE) in 2009, and 24.3 million ± 12.4 million (SE) in 2012. Size class distribution was somewhat variable between survey years, with a larger proportion of colonies in the smaller size classes in 2005 compared to 2009 and 2012 and a greater proportion of colonies in the greater than 90 centimeters size class in 2012 compared to 2005 and 2009. Partial colony mortality was lowest at less than ten centimeters (as low as approximately five percent) and up to approximately 70 percent in the larger size classes. In the Dry Tortugas, Florida, abundance of lobed star coral ranked 41 out of 43 in 2006 and 31 out of 40 in 2008. The extrapolated population estimate was 0.5 million ± 0.3 million (SE) colonies in 2008. Differences in population estimates between years may be attributed to sampling effort rather than population trends (Miller et al. 2013).

As noted previously, in a study of sites inside and outside a MPA in Belize, lobed star coral cover declined significantly over a ten year period (1998/99 to 2008/09; Huntington et al. 2011). In a study of ten 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 percent in permanent monitoring stations in the Florida Keys (Waddell and Clarke 2008). Cover of lobed star coral declined 71 percent in permanent monitoring stations between 1996 and 1998 on a reef in the upper Florida Keys (Porter et al. 2001).

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 percent and 86 percent (thus, 14-82 percent 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.

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. Its absolute population abundance has been estimated as at least tens of millions of colonies in the Florida Keys and Dry Tortugas combined and is higher than the estimate from these two locations due to the occurrence of the species in many other areas throughout 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 lobed star coral is limited to an area 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. 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 also anticipate that the population abundance is likely to decrease in the future with increasing threats.

Mountainous Star Coral

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 percent and 48 percent at Mona and Desecheo Islands, respectively (Bruckner and Hill 2009). In 1998, 27 percent of all corals at 6 sites surveyed off Mona Island were mountainous star coral colonies, but this statistic decreased to approximately 11 percent in 2008 (Bruckner and Hill 2009). At Desecheo Island, 12 percent of all coral colonies were mountainous star coral in 2000, compared to 7 percent 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 percent. The total live area occupied by mountainous star coral declined by a mean of 65 percent, 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 percent 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 four 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).

Based on population estimates, there are at least tens of millions of colonies present in each of several locations including the Florida Keys, Dry Tortugas, and the USVI. Absolute abundance is higher than the estimate from these three locations given the presence of this species in many other locations throughout its range. Population decline has occurred over the past few decades with a 65 percent loss in mountainous star coral cover across five countries. Losses of mountainous star coral from Mona and Descheo Islands, Puerto Rico include a 36-48 percent reduction in abundance and a decrease of 42-59 percent 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. Partial colony mortality is lower in some areas such as the Flower Garden Banks. We conclude that mountainous star coral has declined but remains common and likely has at least tens of millions of colonies throughout its range. Additionally, as discussed in the genus section, we conclude 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.

Boulder Star Coral

Information on boulder star coral status and population 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.

Boulder star coral is reported as common. In a 1995 survey of 16 reefs in the Florida Keys, boulder star coral had the highest percent cover of all species (Murdoch and Aronson 1999). In surveys throughout the Florida Keys, boulder star coral in 2005 ranked 26th most abundant out of 47 coral species, 32nd out of 43 in 2009, and 33rd out of 40 in 2012. Extrapolated population estimates from stratified random surveys were 8.0 ± 3.5 million (SE) colonies in 2005, 0.3 ± 0.2 million (SE) colonies in 2009, and 0.4 ± 0.4 million (SE) colonies in 2012. The authors note that differences in extrapolated abundance between years were more likely a function of sampling design rather than an indication of population trends. In 2005, the greatest proportions of colonies were in the smaller size classes of approximately 4-8 in (10-20 cm) and approximately 8-12 in (20-30 cm). Partial colony mortality ranged from 0 percent to approximately 73 percent and was generally higher in larger colonies (Miller et al. 2013).

In the Dry Tortugas, Florida, boulder star coral ranked fourth highest in abundance out of 43 coral species in 2006 and 8th out of 40 in 2008. Extrapolated population estimates were 79 ± 19 million (SE) colonies in 2006 and 18.2 ± 4.1 million (SE) colonies in 2008. The authors note the difference in estimates between years was more likely a function of sampling design rather than population decline. In the first year of the study (2006), the greatest proportion of colonies were in the size class approximately 8-12 in (20-30 cm) with twice as many colonies as the next most numerous size class and a fair number of colonies in the largest size class of greater than three ft (90 cm). Partial colony mortality ranged from approximately 10-55 percent. Two years later (2008), no size class was found to dominate, and proportion of colonies in the medium-to-large size classes (approximately 24-36 in) appeared to be less than in 2006. The number of colonies in the largest size class of greater than three ft (90 cm) remained consistent. Partial colony mortality ranged from approximately 15-75 percent (Miller et al. 2013).

In 2003, on the east Flower Garden Bank, boulder star coral comprised 46 percent of the 76.5 percent coral cover on reefs approximately 105-131 ft (32-40 m) in depth. Partial coral mortality due to bleaching, disease and predation was rare in survey stations (Precht et al. 2005). In a survey of 31 sites in Dominica between 1999 and 2002, boulder star coral was present in 7 percent of the sites at less than 1 percent cover (Steiner 2003).

Reported density is variable by location and habitat and is reported to range from 0.02 to 1.05 colonies per approximately (~) 100 ft^2 (10 m^2). In surveys of 1,176 sites in southeast Florida, the Dry Tortugas, and the Florida Keys between 2005 and 2010, density of boulder star coral ranged between 0.04 and 0.47 colonies per ~100 ft^2 (10 m^2) and was highest on the offshore patch reef and fore-reef habitats (Burman et al. 2012). In south Florida, density was highest in areas south of Miami at 0.44 colonies per ~100 ft^2 (10 m^2) compared to 0.02 colonies per ~100 ft^2 (10 m^2) in Palm Beach and Broward Counties (Burman et al. 2012). Along the Florida reef tract from Martin County to the lower Florida Keys, density of boulder star coral was ~0.9 colonies per ~100 ft^2 (10 m2; Wagner et al. 2010). On remote reefs off southwest Cuba, colony density was 0.083 ± 0.17 (SD) per ~100 ft^2 (10 m^2) transect on 38 reef-crest sites and 1.05 ± 1.02 colonies per ~100 ft^2 (10 m^2) transect on 30 reef-front sites (Alcolado et al. 2010). The number of boulder star coral colonies in Cuba with partial colony mortality were far more frequent than those with no mortality across all size classes, except for 1 (i.e., less than ~20 in [50 cm]) that had similar frequency of colonies with and without partial mortality (Alcolado et al. 2010).

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

On the other hand, colony size has decreased over the past several decades. Bruckner conducted a survey of 185 sites (2010 and 2011) in 5 countries (The Bahamas, Bonaire, Cayman Islands, Puerto Rico, and St. Kitts and Nevis) and reported the size of boulder star coral and lobed star coral colonies as significantly smaller than mountainous star coral. The total mean partial mortality of boulder star coral was 25 percent. Overall, the total live area occupied by boulder star coral declined by a mean of 38 percent, and mean colony size declined from 210 in^2 to 131 in^2 (1356 cm^2 to 845 cm^2). At the same time, there was a 137 percent increase in small tissue remnants, along with a decline in the proportion of large (1,500 to 30,000 cm^2), completely alive colonies. Mortality was attributed primarily to outbreaks of white plague and yellow band disease, which emerged as corals began recovering from mass bleaching events. This was followed by increased predation and removal of live tissue by damselfish to cultivate algal lawns (Bruckner 2012).

Based on population estimates, there are at least tens of millions of colonies present in both the Dry Tortugas and USVI. Absolute abundance is higher than the estimate from these 2 locations given the presence of this species in many other locations throughout its range. The frequency and extent of partial mortality, especially in larger colonies of boulder star coral, appear to be high in some locations such as Florida and Cuba, though other locations like the Flower Garden Banks appear to have lower amounts of partial mortality. A decrease in boulder star coral percent cover by 38 percent and a shift to smaller colony size across 5 countries suggest that population decline has occurred in some areas; colony abundance appears to be stable in other areas. We anticipate that while population decline has occurred, boulder star coral is still common with the number of colonies at least in the tens of millions. Additionally, we conclude that the buffering capacity of boulder 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. We also anticipate that the population abundance is likely to decrease in the future with increasing threats.

Status of Species within the Action Area

Lobed Star Coral

Cover of lobed star coral at Yawzi Point, St. John, USVI declined from 41 percent in 1988 to approximately 12 percent by 2003 as a rapid decline began with the aftermath of Hurricane Hugo in 1989. The decline began 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 percent less than 7 in^2 [50 cm^2] in 1988 versus 70 percent in 2003) and lower proportion of large colonies (6 percent greater than 39 in^2 [250 cm^2] in 1988 versus three percent 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).

Star corals are the third most abundant coral by percent cover in permanent monitoring stations in the USVI. A decline of 60 percent 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).

Mountainous Star Coral

Mountainous star coral is the sixth most abundant species by percent cover in permanent monitoring stations in the USVI. The star coral species complex had the highest abundance at these stations and included all colonies where species identification was uncertain. Population estimates in the 19 mi^2 (49 km^2) of the Red Hind Marine Conservation District are at least 16 million colonies of mountainous star corals (Smith 2013).

Boulder Star Coral

In the USVI, boulder star coral is the second most abundant species by percent cover at permanent monitoring stations. However, because the species complex, which is the most abundant by cover, was included as a category prior to separating the three sibling species, it is likely that boulder star coral is the most abundant, when including mesophotic reefs. Population estimates of boulder star coral in the approximately 19-mi^2 (49 km^2) area of the Red Hind Marine Conservation District are at least 34 million colonies (Smith 2013).

Critical Habitat

No critical habitat has been designated for lobed star coral.

Recovery Goals

No final recovery plan currently exists for lobed star, mountainous star or boulder star coral, however a recovery outline was developed in 2014 to serve as interim guidance to direct recovery efforts, including recovery planning, until a final recovery plan is developed and approved for the five coral species listed in September 2014. The recovery goals are the same for all five species (see Section 7.2.6.3) with short and long-term goals.

6.2.7 Status of 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 mi^2 (3,442 km^2) of marine habitat; the Puerto Rico area, which comprises approximately 1,383 mi^2 (3,582 km^2) of marine habitat; the St. John/St. Thomas area, which comprises approximately 121 mi^2 (313 km^2) of marine habitat; and the St. Croix area, which comprises approximately 126 mi^2 (326 km^2) of marine habitat. The total area covered by the designation is thus approximately 2,959 mi^2 (7,664 km^2).

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

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

While algae, including crustose coralline algae and fleshy macroalgae, are natural components of healthy reef ecosystems, increased algal dominance since the 1980s has impeded coral recruitment. The overexploitation of grazers through fishing has also contributed to fleshy macroalgae persistence in reef and hard bottom areas formerly dominated by corals. Impacts to water quality associated with coastal development, in particular nutrient inputs, are also thought to enhance the growth of fleshy macroalgae by providing them with nutrient sources. Fleshy macroalgae are able to colonize dead coral skeleton and other hard substrate and some are able to overgrow living corals and crustose coralline algae. Because crustose coralline algae is thought to provide chemical cues to coral larvae indicating an area is appropriate for settlement, overgrowth by macroalgae may affect coral recruitment (Steneck 1986). Several studies show that coral recruitment tends to be greater when algal biomass is low (Rogers et al. 1984; Hughes and Jackson 1985; Connell et al. 1997; Edmunds et al. 2004; Birrell et al. 2005; Vermeij et al. 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 three sites in Puerto Rico, Torres (2001) found that low-density coral skeleton growth was correlated with increased re-suspended sediment rates and greater percentage composition of terrigenous sediment. In sites with higher carbonate percentages and corresponding low percentages of terrigenous sediments, growth rates were higher. This suggests that re-suspension of sediments and sediment production within the reef environment does not necessarily have a negative impact on coral growth while sediments from terrestrial sources increase the probability that coral growth will decrease, possibly because terrigenous sediments do not contain minerals that corals need to grow (Torres 2001).

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

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

7 Stressors Associated with the Proposed Action

Stressors are any physical, chemical, or biological entity that may induce an adverse response in either an ESA-listed species or their designated critical habitat. EPA's approval of the 2018 VIWQS would authorize the establishment of a new water quality criterion for total nitrogen, changes to other criteria such as water clarity, updated criteria for toxic pollutants , and enable the continued use of existing criteria including those for temperature, turbidity, pH, DO, enterococci bacteria, total phosphorus, radioactivity, and toxic pollutants.

The purpose of the criteria is to maintain or restore water quality conditions that support aquatic life. These criteria are evaluated to determine whether any have been set at levels that may directly or indirectly result in adverse effects to ESA-listed species and designated critical habitat. Some of the criteria specify the acceptable bounds for water quality parameters like pH and temperature or natural constituents like nitrogen and phosphorous while other criteria set maximum limits for toxic pollutants. For the purposes of this opinion, each of these are the stressors associated with this action.

Most of the criteria considered in this opinion are proximate stressors that can cause responses upon exposure. These include toxicants, radionuclides, DO, and sediment. Nitrogen and phosphorous are indirect stressors because, when in excess, they can stimulate eutrophication and generate stressors associated with excessive plant and algal growth: extremes in DO, reduced light penetration, smothering, and algal toxins. Indicators of eutrophy include diurnal extremes in DO, high biological oxygen demand, and low water clarity attributed to high chlorophyll a levels. Temperature and salinity are two important factors affecting eutrophy, complicating interpretation of nitrogen and phosphorous criteria. Evaluation of indirect stressors can include placing the criteria in the context of baseline conditions in waters that do not exhibit indicators of eutrophy.

7.1 Factors Influencing Stressor Effects

Information about how species are exposed to stressors, how those stressors cause adverse effects, and overall aquatic toxicity of the stressors relative to the criteria is useful in evaluating whether ESA-listed species or essential features of designated critical habitat are likely to respond to aquatic pollutant exposures at or below the criteria included in the 2018 VIWQSR.

7.1.1 Exposure

The primary exposure pathway for aquatic pollutants in freshwater and saltwater-gilled species (fish and invertebrates) is uptake via the gills as water continuously passes over the gill filaments to oxygenate blood and regulate ion balance. For saltwater fish, exposure to toxicants in water also occurs through ingestion because most marine fish "osmoregulate" by drinking water and excreting solute in order to maintain a lower concentration of solutes in their body fluids than saltwater. Most marine invertebrates have the same internal concentration of solutes as the water they live in and do not osmoregulate (Larsen et al. 2014). The exception is filter-feeding invertebrates, which ingest small quantities of seawater when feeding. For coral species, potential effects on zooxanthellae are attenuated the ability of coral to attract free-swimming dinoflagellates to serve as endosymbionts from the surrounding waters (Hagedorn et al. 2015; Takeuchi et al. 2017; Yamashita et al. 2014; Pasternak et al. 2004).

Sea turtles breathe air, but they also drink water and excrete solute to regulate their internal ion balance. Sea turtle exposures are less than those of marine fish because turtles do not drink continuously, whereas saltwater fish both drink and continuously pass water over their gills.

Allometric differences (e.g., body size, membrane area, organ size) are factors to be considered when evaluating toxicity data. A smaller individual generally succumbs to toxic effects more rapidly than a larger individual because it takes a longer time for exposures to reach critical concentrations within the tissues of the larger individual. Therefore, higher exposure concentrations would be needed to elicit the same response over a similar exposure period. For example, adult Nassau grouper are greater than 15 inches in length versus adult sheepshead minnows, a common toxicity test species, which are one to two inches in length. Nassau grouper larvae are initially pelagic. They settle in nearshore shallow waters in macroalgal and seagrass habitats after 1 to 2 months of floating with the ocean, beginning at just over one inch in length and moving into deeper and deeper water toward offshore reefs as they grow. It is this nearshore life stage that is most likely to be exposed to land-sourced pollutants.

The above information is helpful when addressing data gaps for species like sea turtles. There are no data for the actual effects of aquatic toxicants on survival, growth, or fitness that can be used to evaluate water quality criteria for these species. A majority of the marine fish and invertebrates inhabiting the same waters as sea turtles are allometrically disadvantaged because they are smaller and also have more intense exposures because they respire through gills. For these reasons, they are more likely to respond to toxicants in water. If NMFS makes a determination that EPA's approval of criteria for the toxicants evaluated in this opinion is not likely adversely affect these species groups, it is reasonable to expect EPA's approval of the criteria is not likely to adversely affect hawksbill and green sea turtles. The exception to this are those toxicants that accumulate in organisms and those that biomagnify through the food web. In poorly flushed marine systems, legacy inorganic and persistent organic toxicants can become incorporated into the biogeochemical cycle, with contaminants recycling between sediment and organism tissues through the trophic web, resulting in generational exposures. This is not expected to be a major factor for the waters surrounding the Caribbean Islands, as they are relatively open to ocean mixing and subject to tropical storms that seasonally flush and redistribute sediments, albeit also introducing additional land-sourced pollutants in stormwater runoff.

Aquatic pollutants may also result in indirect exposures to toxicants through the food web or result in indirect effects through altering the quantity or quality of prey. While there are data for toxic effects that may influence prey populations and tissue accumulation in prey or prey-like species under controlled laboratory conditions, information on dietary toxicity through food web exposures is extremely limited for marine environments. Information on prey items and foraging areas can suggest the potential for toxic exposures, but uncertainty in whether actual adverse effects will occur can be substantial.

Body burdens in sea turtles primarily result from diet. The presence of a contaminant in the tissues of an organism only confirms exposure and does not provide useful information about adverse effects on survival, development, growth, or fitness. The impact of this uncertainty on evaluating a location-specific criterion or activity is attenuated when the action area comprises a very small portion of a species foraging area and exposure to pollutants in seawater is expected to be minimal. Green and hawksbill sea turtles are likely to forage in waters affected by these criteria, but if adverse effects are not expected for forage species under the chronic criteria, adverse effects in sea turtles due to dietary exposures would not be expected unless the pollutant is persistent and potentially accumulates to toxic levels. Identifying toxic levels for sea turtle tissues would require extrapolations from other species and the attendant uncertainties, including differences in metabolism, life span, and diet.

7.1.2 Mechanisms of Effect

The probabilities and types of responses are influenced by the mechanism by which the stressor causes adverse effects. For example, mobile species are expected to be less affected by suspended sediments than stationary organisms like coral. For stressors that influence calcification such as the organochlorines and cadmium, we would be most interested in the response of marine calcifiers such as mollusks rather than worms, in the absence of toxicity data for ESA-listed coral species. For most toxic effects, the toxicant itself must be biologically available to interact in some way with the organism's tissues. Toxicant mechanisms include mimicry of natural biomolecules or substrates, interference with hormone signaling, or disruption of mitochondrial function subsequently generating superoxide and hydroxide (free radicals) that damage proteins, DNA, membranes and other biomolecules. When relying on mechanism of effect to determine if responses are likely, we must consider that toxicants can have multiple mechanisms and the biomolecules affected by toxicants may have multiple, unstudied functions. For example, we would not expect toxicants that interfere with neural transmitters to cause effects in plants because plants do not have nervous systems. Plants may be affected by a neurotoxicant, but the response is a result of some other, not neurologically based, mechanism for toxicity (Lin et al. 1972). Also, consider that if a species of concern does not have a receptor for a given substrate mimicked by the toxicant, that does not mean effects cannot occur. For example, corals do not have estrogen receptors, but there is evidence that estrogen is important in signaling during coral spawning events (Tarrant et al. 2003; Tarrant 2005; Tarrant 2003; Tarrant et al. 2004).

7.1.3 Applying Toxicity Data in this Opinion

We expect that ESA-listed species are extremely unlikely to respond to exposure under a CMC or CCC if that criterion is set at an exposure intensity that is orders of magnitude lower (i.e., by 100-fold or more) than the lowest reported acute lethal effect (e.g., LC50s or EC50s), or chronic exposure-response threshold (e.g., LOECs or ECXXs) for that substance. Interpreting criteria when the minimum exposures resulting in toxic response (i.e., LC50s, LOECs, MATCs) are not orders of magnitude greater than the criteria is somewhat more complicated. When the minimum reported LC50 or sublethal effects data is higher than, but not orders of magnitude greater than the criterion being evaluated, we need assess whether the magnitude of response at the criterion is not at some lower, but still unacceptable level from the standpoint of effects to ESA-listed species. As a generic example, an LC50 that is 20 percent higher than the CMC concentration for a species may indicate that exposures of that species under the proposed CMC concentration would result in 30 percent mortality. Exposure duration is also a key consideration when comparing toxicity data when evaluating water quality criteria for toxics. Some toxicants reach critical concentrations and act quickly, while others act over a longer time period. In some cases, the 96 hour LC50 exposure could be the same as a 6 hour LC50. Ideally, dose response data from which LC50 data are derived would be available to allow us to determine whether effects occur within one hour of exposure. Unfortunately, this is rarely the case.

Calculating an acute toxicity ratio in terms of relative percent mortality is one approach to assessing acute criteria against LC50 data. It was applied in the NMFS' Biological Opinion on EPA's approval of water quality criteria proposed by the state of Oregon (NMFS 2012b). This opinion for the VIWQS extends the approach to EC50 data for chronic exposure-response thresholds. This involves dividing a criterion by the LC50 or EC50 and multiplying that ratio by 0.5 to account for the 50% response represented by that LC50 or EC50 to estimate fractional response in the test population exposed to the toxicant at the proposed criterion, and therefore by inference, the potential response in field-exposed individuals.

We emphasize that the fractional responses arrived at using this approach is an estimate that can over or underestimate the response reflected by the actual exposure response relationship, if that data were available (e.g., Figure 19).

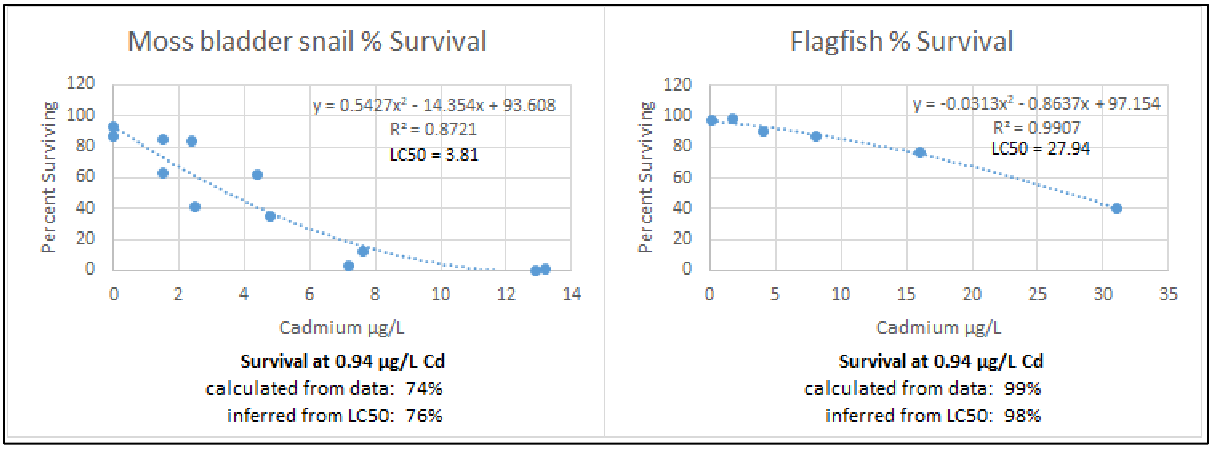

Figure 19. Example of inferring survival from an LC50: Survival at cadmium criterion of 0.94 µg/L calculated from dose-response relationship versus inferred from the LC50

The examples in Figure 19 show that while inferring survival from an LC50 does not perfectly represent dose-response estimates, it can provide a reasonable estimate. For a CMC to be protective of ESA-listed species, minimal to no fractional mortality should occur in surrogate species exposed to a toxicant at the CMC. Fractional mortality response estimates that are greater than 0 percent may also suggest insignificant effects for ESA-protected species, taking into consideration the response, abundance, and diversity within dataset, species represented, quality of the study, and the speed at which a toxicant may exert effects relative to the averaging time for the criterion (i.e., 1 hour for a CMC). For example, some chemicals, particularly pesticides, act rapidly and responses to exposures happen within a matter of a few hours in those individuals that will respond to the exposure, thereafter the exposure-response relationship plateaus. In such cases, the LC50 concentration after 6 hours exposure could be the same as the LC50 at 96 hours exposure. This issue is explored further in the following paragraphs.

Per the Organization for Economic Co-operation and Development guidelines (OECD 1992), Katz (1961) reported 24-, 48-, 72-, and 96-hour LC50s for three-spine stickelback exposed to various organochlorine and organophosphate pesticides. Figure 20 illustrates data from this study showing little to no effect of exposure duration on LC50s for heptachlor and malathion, and larger effects for other pesticides. The comparative percent difference between 24- and 96-hour LC50s for these other pesticides were: 36 percent for DDT, 30 percent for guthion, 24 percent for lindane, and 11 percent for chlordane (Katz 1961). Additionally, data for tilapia exposed to endosulfan show a difference of 36 percent between 24 and 96 hour LC50s (Kenneth and Seinen 2010).

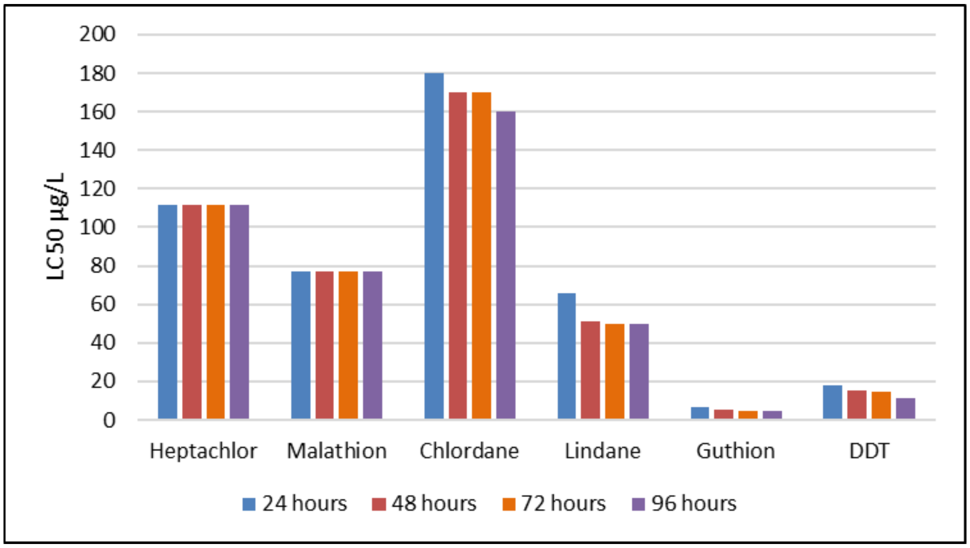

Figure 20. Effect of exposure duration on LC50 estimates for three-spine stickelback exposed to various pesticides (after Katz 1961)

Meanwhile, various studies for metals show percent differences between 24- and 96-hour LC50s ranging from ten to 74 percent (general examples in Figure 21, references are in footnote). Few studies report LC50s for the first hours of exposure, as mortality is usually not expected. However, a factorial study exploring the effects of exposure time and concentration on the toxicity of zinc and copper to rainbow trout reported no mortality in the first two hours of copper exposures ranging from 10 to 2,000 µg/L or in the first four hours of zinc exposures ranging from 5,000 to 13,000 µg/L (Gündoğdu 2008).

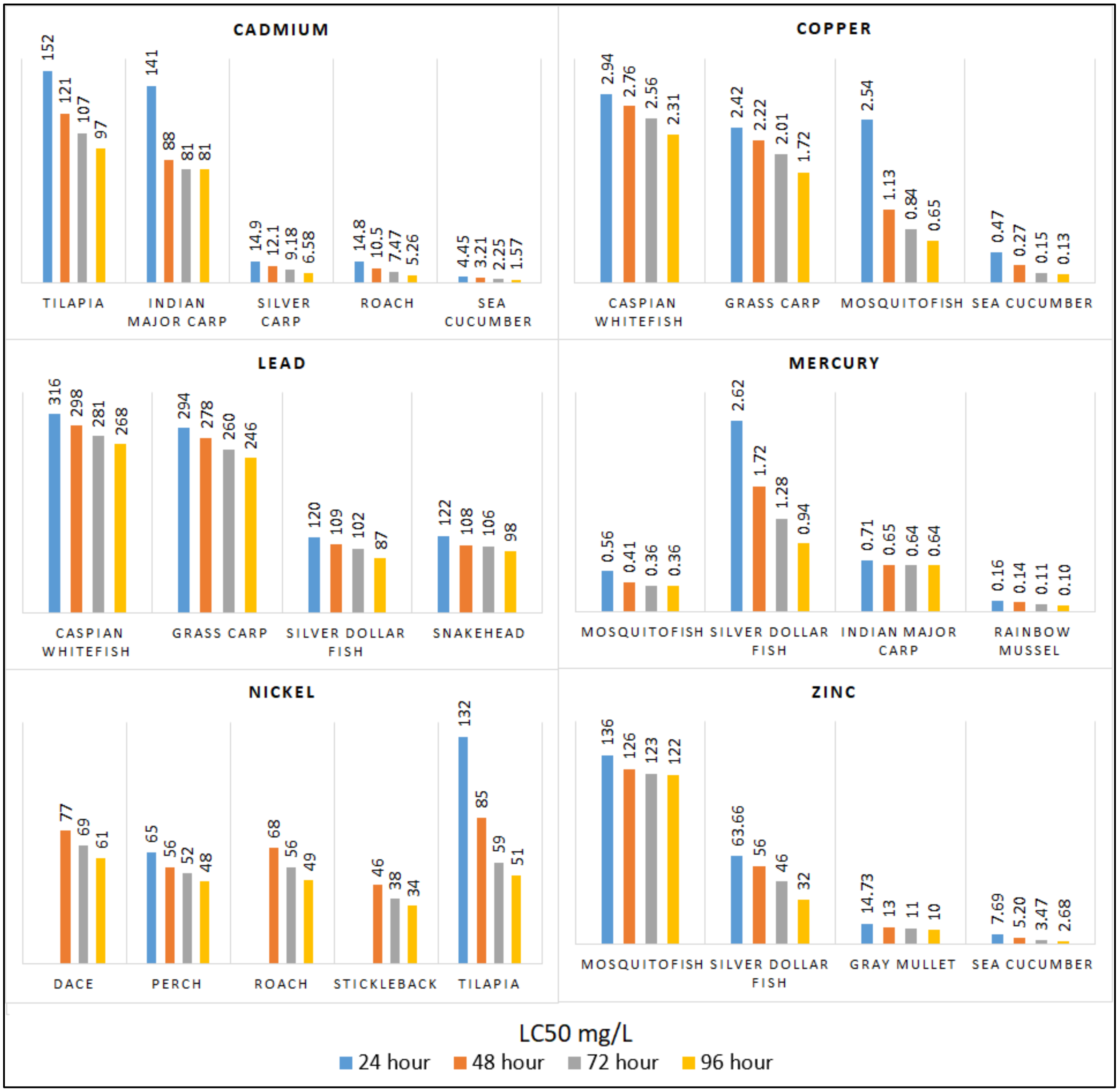

Figure 21. Influence of exposure duration on LC50s for fish exposed to various metals[2]

Evidence supports consideration of toxicity test exposure durations when evaluating proposed criteria in some, but not all cases. Factoring exposure durations into decisions on whether exclude criteria from further analysis will be applied on a case-by-case basis.

For evaluating proposed CCCs, in cases where the lowest reported NOEC is greater than the CCC and indicates a response magnitude that is considered biologically insignificant, we expect

[2] Various sources (Benjamin and Thatheyus 2012; Ebrahimpour et al. 2010; Erfanifar et al. 2016; Gharedaashi et al. 2013; Hedayati 2016; Lal et al. 2013; Li et al. 2016; Nekoubin et al. 2012; Pourkhabbaz et al. 2011; Rahimibashar and Alipoor 2012; Ramesh et al. 2019; Sadeghi and Imanpoor 2015; Svecevičius 2010; Valenti et al. 2005)

that responses in ESA-listed species exposed at or below CCC will be insignificant or are unlikely to occur. In cases where the magnitude of response at the NOEC is not available, the criterion averaging time of four-days is taken into consideration along with the speed at which a toxicant is expected or was observed to exert effects and exposure duration of the toxicity test. Where NOECs are not available, the magnitude of effect at the LOEC or MATC is taken into consideration similarly to the evaluation of the criterion using NOECs.

7.2 Identifying Criteria Not Likely to Adversely Affect Species Addressed in this Opinion

Below we provide a general evaluation of the proposed 2018 VIWQS criteria relative to the best available data to proactively identify, and exclude from further analysis, those criteria for substances that are unlikely to occur in waters affected by the proposed criteria or criteria that are unlikely to result in significant responses in the ESA-listed species evaluated in this opinion. EPA approval of such criteria is not likely to adversely affect ESA-listed species. The evaluation takes into consideration the diversity of species represented among the data, species' mean responses, allometric influences, the types of responses documented by researchers, the number and quality of the available toxicity studies, and the magnitude and types of effects reported. The probability of exposure in waters affected by the proposed criteria is also considered because this consultation specifically addresses EPA's approval of criteria used to regulate and assess water quality in the jurisdictional waters of the USVI.

> **Important:** *Discovery of toxicity data indicating ESA-listed species may respond to exposures at a CMC or CCC or monitoring that indicates a previously unexpected stressor is present or will be discharged where ESA-listed species occur could be considered "new information" and trigger reinitiation of consultation for EPA's approval of that VIDPNR-proposed criterion.*

7.2.1 Pesticides

Pesticides include herbicides, insecticides, fungicides, and antifoulants used on and within vessels and other marine structures. In addition to pesticides being used in the marine environment, they also enter the marine environment unintentionally through stormwater runoff, atmospheric deposition, and sewage discharge. For corals, specifically zooxanthellae, data for effects to algae, in particular dinoflagellates, are used to assess potential effects, although there are numerous data gaps in terms of exposures of algae for pesticides.

While the majority of pesticide use in the USVI is associated with domestic uses, agricultural uses could include livestock and cultivation of ornamental plants. The most recent census of agriculture for the islands indicated a 35 percent decline in farmed land between 2002 and 2007, with about 7 percent of total land in agricultural use in 2007 (USDA 2009), indicating that pesticide use may be limited in the USVI.

7.2.1.1 Cancelled and Restricted Use Organochlorine Pesticides

The criteria in the 2018 VIWQS include cancelled and restricted use pesticides. There should be no new discharge sources for cancelled pesticides, but these pesticides could enter the marine environment through transport of contaminated soil, as well as resuspension and redistribution of contaminated sediment during storm events, activities such as dredging, or illicit use of unused product.

While current agricultural pesticide use is expected to be limited and declining, soil disturbance during construction of residential and resort properties on previously agricultural land can redistribute legacy organochlorine pesticides that had both domestic and agricultural uses. Several to many half-lives (time for half of remaining residue to break down) have passed since the organochlorine pesticides in Table 2 ceased to be used or since their uses were restricted so the risks posed by these pesticides are expected to continue to decline over time.

Table 2. Half-lives of Organochlorine pesticides that are no longer in use, the number half-lives expended since end-of-use

Organochlorine Pesticide	Uses and year end-of-use	Half Life (years) [1]	Half Lives as of 2018
4,4'-DDT	Public health, pests, Canceled 1972	15	3.07
Aldrin	Crops, wood boring pests, Canceled 1987	up to 7	4.43
Alpha-Endosulfan	Crops, Canceled 2016	0.21	9.52
Beta-Endosulfan	Crops, Canceled 2016	0.41	4.88
Chlordane	Crops, landscaping, termites[2] Canceled 1975	10	4.30
Dieldrin	Crops, mosquitoes, wood boring pests, Canceled 1987	0.75	41.33
Endrin	Crops recreational areas, Canceled 1986	up to 12	2.67
gamma-BHC (Lindane)	Crops, lice, Canceled 2006	1.25	9.60
Heptachlor/ Heptachlor epoxide (metabolite)	Crops, termites, Canceled 1974	2	22
Methoxychlor	Crops, residential pests, Canceled 2016	0.33	6.08
Mirex	Wood boring pests, fire retardant additive, Canceled 1977	10	4.10
Pentachlorophenol	Wood preservative,[3] Restricted 1987	0.12	258.33
Toxaphene	Crops, livestock, Cancelled 1990	11	2.55
[1]Jayaraj et al. (2016). Organochlorine pesticides, their toxic effects on living organisms and their fate in the environment. Interdisciplinary Toxicology 9:90-100.			

[2]Currently EPA allows use on fire ants near transformers, but there are no products registered for this use in the US at the time of this writing. http://npic.orst.edu/factsheets/chlordanegen.pdf

[3]Restricted to use on telephone poles and their cross arms

The criteria for these pesticides still have regulatory relevance because these are persistent compounds that may still be present in marine habitats of the Caribbean (Whitall et al. 2014; Whitall et al. 2015; Menzies et al. 2013), particularly in sediments. However, the number of half-lives that have passed since they were used means these chemicals are not expected to be readily available in terms of potential exposure of marine species unless disturbance of contaminated sediment occurs. In cases such as dredging or construction activities or severe weather causing soil erosion and/or sediment resuspension, discharges from disturbed areas may contain concentrations of pesticides or pesticide metabolites approaching the criteria. While nonpoint sources of contaminated sediment are typically not monitored, the discharge of dredged or fill material into waters of the United States is regulated by the U.S. Army Corps of Engineers under section 404 of the Clean Water Act. No discharge of dredged or fill material may be permitted if: (1) a practicable alternative exists that is less damaging to the aquatic environment or (2) the nation's waters would be significantly degraded. Permit applicants must show that steps have been taken to avoid and minimize impacts and provide compensation for any unavoidable impacts. Activities conducted by the Corps, such as maintenance dredging of navigation channels, require sediment contaminant monitoring and management when polluted sediments are disturbed (paragraph 2.17 of USACE 2015).

Table 3 shows LC50 and chronic exposure-response thresholds for saltwater exposures of algae, fish, and invertebrates to organochlorine pesticides evaluated in this opinion. Nassau grouper or closely related species and ESA-listed corals or other stony corals are not represented among the species used in these toxicity tests. The hatched areas in identify species groups for which exposure-response data were not available. The ECOTOX did not include any data for exposures of saltwater species to mirex. There are no LC50 data for algae exposures to any of these pesticides or sublethal response data for algae exposed to chlordane, endosulfan, endrin, or methoxychlor. Finally, there are no sublethal response data for exposures of saltwater fish to chlordane, endosulfan, or toxaphene.

Table 3. ECOTOX data for concentrations* of cancelled organochlorine pesticides causing effects in marine species relative to the proposed criteria

Toxicant	Median Lethal Concentrations (LC50, mortality EC50) µg/L	Sublethal Effects/No Effects Concentrations (LOECs, NOECS, MATCS) µg/L	Saltwater CMC µg/L	Saltwater CCC µg/L
Aldrin		no CCC	1.3	no CCC
Algae				
Fish	2.03-126 (n=33)			
Invertebrates	410 (n=1)			
Chlordane			0.09	0.004
Algae				
Fish	6.4-180 (n=11)			
Invertebrates	0.4-220 (n=5)	1-330 (n=4)		
DDT			0.13	0.001
Algae		100 (n=7)		
Fish	0.26-115 (n=51)	2.4 (n=1)		
Invertebrates	0.07-43,010 (n=19)	0.4-408 (n=31)		
Dieldrin			0.71	0.0019
Algae		100 (n=4)		
Fish	0.5-40 (n=110)	10-124 (n=2)		
Invertebrates	0.3-49,180 (n=38)	0.49-640 (n=30)		
Endosulfan			0.034	0.0087
Algae				
Fish	0.09-3.45 (n=36)			
Invertebrates	0.04-9,230 (n=101)	60-269 (n=2)		
Endrin			0.037	0.0023
Algae				
Fish	0.05-5.6 (n=47)	0.12-345 (n=19)		
Invertebrates	0.4-10,000 (n=6)	0.08-790 (n=25)		
Heptachlor			0.053	0.0036
Algae		200,000 (n=1)		
Fish	0.8-240 (n=34)	1.9-5.7 (n=6)		
Invertebrates	0.03-3.4 (n=4)	0.33-10 (n=3)		
Lindane		no CCC	0.16	no CCC
Algae				
Fish	3.8-75 (n=33)			
Invertebrates	0.01-145,000 (n=19)			
Methoxychlor	no CMC		no CMC	0.03
Algae				
Fish		12-610 (n=6)		
Invertebrates		0.005-367 (4=22)		
Mirex	no CMC		no CMC	0.001
Algae				

Toxicant	Median Lethal Concentrations (LC50, mortality EC50) µg/L	Sublethal Effects/No Effects Concentrations (LOECs, NOECS, MATCS) µg/L	Saltwater CMC µg/L	Saltwater CCC µg/L
Fish				
Invertebrates				
Pentachlorophenol			13	7.9
Algae		2.7-5,500 (n=16)		
Fish	17-1,490,000 (n=51)	157-870 (n=22)		
Invertebrates	25-20,000 (n=135)	0.31-6,500 (n=44)		
Toxaphene			0.21	0.0002
Algae		14 (n=1)		
Fish	0.53-12 (n=11)			
Invertebrates	1.4-250 (n=5)	0.7-1,120 (n=8)		
* Hatched areas indicate data gaps				

In some cases, LC50 and chronic exposure-response thresholds include observations that are close to, or below, the exposure concentrations represented by their respective CMC and CCC. In many cases, estimates of fractional mortality using LC50 data suggest a mortality rate of greater than five percent under CMC exposure concentrations. Any exposures to cancelled pesticides historically used in the USVI would be the result of redistribution of legacy contamination through dredging or introduction of stormwater contaminated with legacy pesticides.

As mentioned previously, primarily domestic pesticide usage is expected in the USVI. For example, prior to cancellation in 1987, dieldrin was applied to crops and was also used to control mosquitoes and wood-boring pests. Dieldrin was likely used at some time in the USVI prior to cancellation. Prior to cancellation in 1974, heptachlor was applied to crops and used to control termites, so this pesticide was also likely used at some time in the USVI prior to cancellation. On the other hand, endosulfan was used only in agriculture to control pests in coffee, vegetables, and grain crops. It was not likely used much in the Virgin Islands.

Pentachlorophenol is a restricted-use wood preservative that was addressed in a recent Programmatic Environmental Impact Statement for the establishment of a Nationwide Public Safety Broadband Network for the Non-Contiguous United States (Goebel-Pereira 2017). The document acknowledged that most wood poles used for utility or telephone lines are treated with a pentachlorophenol to lessen wood rot and extend the life of the poles and that, once constructed, leachate from new treated poles could potentially enter surface water. Despite this, pentachlorophenol from new telephone poles was not expected to contaminate surface waters because of the chemical’s tendency to adhere to soils and its moderately rapid degradation rate in the environment. A review of current NPDES permits identified four wood treatment facilities in

Puerto Rico, but none in the USVI.[3] Finally, in response to severe damage caused by the 2017 hurricanes in the Caribbean, wood utility poles are being replaced with fiberglass composite poles that can withstand 200 mile per hour winds (USGAO 2019).

A survey for a broad array of contaminants in waters around St. John did not detect pentachlorophenol in coral tissues (Downs et al. 2011). A 1992 evaluation of aquatic use impairments in the Caribbean included assessments of pesticides in water and sediments. Aldrin, dieldrin, endrin, heptachlor and methoxychlor were detected in coastal waters off Puerto Rico at concentrations below 0.01 µg/L, but not in waters off the USVI (USEPA 1992). Sampling programs in 1974 indicated the presence of dieldrin in sediments at concentrations between 4 to 20 nanograms per gram (ng/g); however, dieldrin and heptachlor were not detected in samples of dredged material evaluated for the 1992 report (USEPA 1992).

An extensive study, that included sampling of coral, conch, fish, and sediment, monitored the St. Thomas East End Reserves for most of the organochlorine pesticides addressed by the criteria evaluated in this opinion. Exceptions were methoxychlor, pentachlorophenol, and toxaphene (Pait et al. 2016). The mean concentration of total DDT in the sediments was 0.047 ±0.025 ng/g. Concentrations of up to 0.609 ng/g were found in the Benner Bay and Mangrove Lagoon areas in samples collected in 2011. Most of the results for other organochlorines were below detection limits. The highest concentration of chlordane (alpha and gamma isomers) was detected at 0.85 ng/g. None of the sites had a chlordane concentration above the NOAA effects range median.[4] Monitoring of organochlorines in the tissues of fish, coral, and conch resulted in a few detections at concentrations considered to be very low Pait et al. (2016). A 2013 study did not detect heptachlor in samples of coral, fish, plankton, and detritus collected from coral reefs in Virgin Islands National Park and Virgin Islands Coral Reef National Monument (Bargar et al. 2013).

The above information provides evidence that cancelled and restricted organochlorine pesticides are not likely detectable in the waters and sediments affected by the 2018 VIWQS.

NMFS concludes that, because exposures to cancelled and restricted organochlorine pesticides in these waters are extremely unlikely, EPA approval of the VIDPNR-proposed CMC and CCC for organochlorine pesticides is considered discountable and therefore not likely to adversely affect Nassau grouper or ESA-listed coral. For this reason, these pesticides will not be discussed further in this opinion.

7.2.1.2 Organophosphate and Carbamate Pesticides

Organophosphate and carbamate pesticides were derived from nerve agents developed in World War II and have functionally replaced cancelled organochlorine pesticides. Their mode of action is the rapid inhibition of acetylcholinesterase causing uncontrollable firing of nerve impulses,

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[3] EPA Enforcement and Compliance History Online database accessed 4/9/2019

[4] The NOAA effects range median is a sediment concentration above which effects generally are expected to occur.(Long and Morgan 1991).

overstimulating and potentially shutting down the central nervous system (Amdur et al. 1993; Markey et al. 2007). Several of the organophosphate pesticides evaluated in this opinion are used only in agriculture, so the probability of exposure in waters affected by the proposed 2018 VIWQS is factored into the analysis. In general, insects are most sensitive to the effects of these pesticides and plants are the least sensitive (e.g., Russom et al. 2014b).

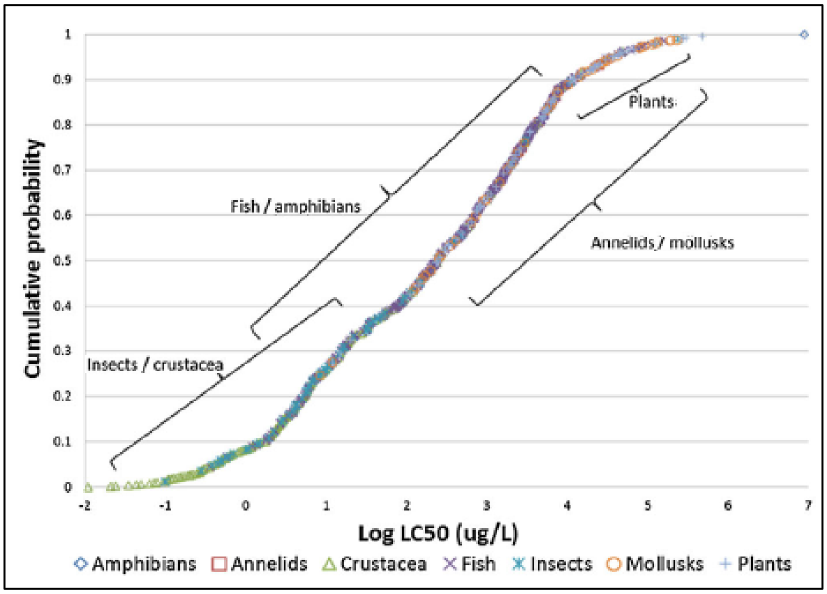

Figure 22. Relative sensitivity of species to organophosphates after' Russom et al. (2014a).

Table 4 shows LC50 and chronic exposure-response thresholds for saltwater exposures of algae, fish, and invertebrates to the organophosphates and the carbamate carbaryl evaluated in this opinion.

Table 4. ECOTOX data for organophosphate and carbamate pesticide exposure concentrations* causing effects in marine species relative to the proposed criteria for the protection of aquatic life

Toxicant	Median Lethal Concentrations (LC50, mortality EC50) µg/L	Sublethal Effects/No Effects Concentrations (LOECs, NOECS, MATCS) µg/L	Saltwater CMC µg/L	Saltwater CCC µg/L
Carbaryl			1.6	no CCC
Algae				
Fish	1,600-16,625 (n=27)			
Invertebrates	6.96-27,568 (n=50)			
Chlorpyrifos			0.011	0.0056
Algae		1-5,000 (n=12)		
Fish	0.4-1,000 (n=92)	0.9-750 (n=51)		
Invertebrates	0.01-22,500 (96)	0.002-2,0000 (n=131)		
Demeton			no CMC	0.1
Algae				
Fish				
Invertebrates				
Diazinon			0.82	0.82
Algae		1,000 (n=1)		
Fish	500-8,000 (n=22)	200-400 (n=4)		
Invertebrates	2.8-660,000 (n=48)	2.1-96,600 (n=32)		
Guthion			no CMC	0.01
Algae		1,000-15,000 (n=4)		
Fish		0.17-19.8 (n=11)		
Invertebrates		0.02-89,000 (n=22)		
Malathion			no CMC	0.1
Algae		17,880-159,380 (n=2)		
Fish		0.365-200 (n=10)		
Invertebrates		0.32-38,500 (n=21)		
* Hatched areas indicate data gaps				

While marine algae LC50s were not reported for organophosphates in ECOTOX, chronic exposure-response thresholds for algae were reported at concentrations that were many orders of magnitude greater than the organophosphate CCCs. This is consistent with the distribution of species sensitivities illustrated in Figure 22 from Russom et al. (2014a). This is not surprising because algae are not expected to be susceptible to acetylcholinesterase inhibition. Thus, coral zooxanthellae exposed to organophosphates or carbaryl at their respective criteria are not expected to respond, so the algal data for these pesticides will not be discussed further.

The following paragraphs place the fish and invertebrate toxicity data in Table 4 in context of the respective CMC and CCC to determine whether EPA's approval of proposed criteria for any of these pesticides requires further analysis in the effects section of this opinion. NMFS’ 2017 biological opinion on the reregistration of malathion, chlorpyrifos, and diazinon cannot be used to make inferences on organophosphate exposures of ESA-protected species in USVI waters because the exposure analysis in that opinion combined pesticide use information from Puerto Rico and the USVI (NMFS 2017).

Carbaryl

Fish LC50s for carbaryl ranged from 1,600 to 16,625 µg/L, representing eight species from six studies. These LC50s are orders of magnitude greater than the proposed CMC of 1.6 µg/L, suggesting that mortality would not occur in a population of fish exposed to carbaryl at the proposed CMC. These data provide evidence that mortality in Nassau grouper exposed to carbaryl at or below the proposed CMC of 1.6 µg/L is extremely unlikely.

Invertebrate LC50s for carbaryl ranged from 6.96 to 27,568 µg/L, which suggests that mortality could be around 11 percent in a population exposed to carbaryl at the proposed CMC of 1.6 µg/L. The dataset includes LC50s for 17 species from 14 studies. Among these data, species mean LC50s for nine species suggest some mortality, ranging from 0.6 to 11 percent, would occur due to exposures at or below the proposed CMC. The lowest LC50 among the data is for a 13-day exposure of adult pink shrimp (Bocquené and Galgani 1991). These data provide evidence that mortality in ESA-listed corals exposed to carbaryl at or below the proposed CMC of 1.6 µg/L may occur.

While sufficient data are not available to derive a CCC for saltwater exposures to carbaryl (USEPA 2012) and the ECOTOX did not include data for the effects of carbaryl on coral, a chronic exposure-response LOEC of 3.0 µg/L was reported for settlement impairment of *Acropora millepora* larva that were pre-exposed to carbaryl for 18 hours (Markey et al. 2007; Mayer 1987). The NOEC concentration in this test was 1 µg/L. As illustrated by Figure 20, acetylcholinesterase inhibitors are rapid-acting pesticides, suggesting effects under the brief exposure of the CMC one-hour averaging period may occur. At least 60 percent of larvae were impaired at the LOEC concentration. This data is particularly significant because the test species is from the same genus as two of the ESA-listed corals that occur in waters affected by the proposed criteria: elkhorn and staghorn coral.

Carbaryl is used to control outdoor biting pests, including ticks, fire ants, and fleas. These are domestic uses, so carbaryl is likely to be used in the USVI and occur in stormwater runoff.

NMFS concludes that the proposed CMC for carbaryl requires further evaluation in the effects section of this opinion because responses to carbaryl exposures are expected to be rapid and mortality and other responses may occur in ESA-listed corals exposed at or below the proposed carbaryl CMC of 1.6 µg/L.

Chlorpyrifos

Fish LC50s for chlorpyrifos ranged from 0.4 to 1000 µg/L, representing 14 species from eight studies. This suggests mortality rates of up to 1 percent may occur in a fish population exposed to chlorpyrifos at the proposed CMC of 0.011 µg/L. The lowest LC50s among these data were for four-hour exposures of early life stage tidewater silverside (Borthwick et al. 1985). Species mean LC50s suggest fractional mortality due to exposures at the CMC would be no higher than 0.5 percent in three out of the 14 species, with no mortality expected in the remaining species. The low fractional mortality only occurring in a minority of species tested provides evidence that the risk of mortality in Nassau grouper exposed to chlorpyrifos at or below the proposed CMC of 0.011 µg/L would be insignificant.

Invertebrate LC50s in ECOTOX for chlorpyrifos ranged from 0.01 to 22,500 µg/L, representing 28 species from 35 studies. This suggests that mortality could be around around 50 percent in a population exposed to chlorpyrifos at the proposed CMC of 0.011 µg/L. The lowest LC50 was a four-day exposure of the amphipod *Ampelisca abdita* (Scott and Redmond 1986), but this species' mean LC50 for chlorpyrifos exposures was 0.26 µg/L, indicating a fractional mortality of about two percent for exposures under the CMC. Overall, species mean LC50s suggest fractional mortality due to exposures at the CMC range from 1 to 12 percent in 12 out of the 28 species, with no mortality expected in the remaining species. Although not reported in ECOTOX, searches of the open literature provided information on exposures of a coral species. A four-day LC50 for the coral *Pocillopora damicornis* adult branches at 6 µg/L is notable (Tan Te 2019) as is a report of no mortality in the planulae of this species after four-day exposures to 100 µg/L chlorpyrifos (Acevedo 1991). These response thresholds are orders of magnitude greater than the proposed CMC, attenuate concerns posed by the ECOTOX invertebrate data and providing evidence that the risk of mortality in ESA-listed corals exposed to chlorpyrifos at or below the proposed CMC of 0.011 µg/L would be insignificant.

The chronic exposure-response thresholds for fish exposed to chlorpyrifos ranged from 0.9 to 750 µg/L, representing three species from four studies. These thresholds were orders of magnitude greater than the CCC of 0.0056 µg/L. The lowest observations, each at 0.9 µg/L, were for four-day NOECs for length, weight, and condition index of mummichog (Karen 1999). In addition, while difficult to associate with organism-level consequences, exposure-response thresholds for acetylcholinesterase inhibition are reported at exposure concentrations as low as 0.216 µg/L. This lowest acetylcholinesterase is a NOEC for three-day exposures of shiner perch (Troiano et al. 2013). The magnitude of difference between the range in response thresholds for chlorpyrifos and its proposed CCC provides evidence that Nassau grouper are extremely unlikely to respond to chlorpyrifos exposures at or below the proposed CCC of 0.0056 µg/L.

While the ECOTOX chronic exposure-response thresholds for invertebrate exposures to chlorpyrifos ranged from 0.001 to 20,000 µg/L, these data include 14 chronic exposure-response thresholds for development and fitness in the coral *A. millepora*, which are the most appropriate data for assessing the effects of chlorpyrifos to ESA-listed coral. The lowest among these thresholds are NOECs of 0.1 µg/L for metamorphosis effects and 30 µg/L for fertilization effects (Markey et al. 2007). Plots of the exposure-response data indicate that the response magnitudes at these NOECs are equivalent to controls. In addition, the lowest acetylcholinesterase inhibition NOEC reported for invertebrates was 98 µg/L for Pacific blue mussel exposed for 2 days (Kopecka-Pilarczyk 2010). The NOECs for a species from the same genus as two ESA-listed coral occurring at concentrations that are orders of magnitude higher than the CCC provide evidence that ESA-listed coral are extremely unlikely to respond to chlorpyrifos exposures at or below the proposed CCC of 0.0056 µg/L.

NMFS concludes that, because responses of Nassau grouper and ESA-listed coral to chlorpyrifos exposures at or below the both the proposed CMC and CCC is expected to be insignificant, EPA's approval of the proposed criteria is not likely to adversely affect these species. For this reason, the CMC and CCC for chlorpyrifos will not be discussed further in this opinion.

Diazinon

Fish LC50s for diazinon ranged from 500 to 8,000 µg/L, representing four species in 22 studies. These LC50s are orders of magnitude greater than the proposed CMC of 0.82 µg/L, suggesting that mortality would not be expected in a fish population exposed to diazinon under the CMC. The lowest LC50 was for three-day exposures of hirame flounder larvae (Menendez and Ishimatsu 1993). These data provide evidence that mortality in Nassau grouper exposed to diazinon at or below the proposed CMC of 0.82 µg/L is extremely unlikely.

Invertebrate LC50s for diazinon ranged from 2.8 to 100,000 µg/L, representing 17 species in 11 studies. This suggests that mortality could be around 15 percent in a population exposed to diazinon at the proposed CMC of 0.82 µg/L. The lowest LC50 among these data is for a four-day exposure of juvenile Daggerblade grass shrimp (Thursby and Berry 1988). These data provide evidence that mortality in ESA-listed corals exposed to diazinon at or below the proposed CMC of 0.82 µg/L may occur.

The lowest chronic exposure-response thresholds reported in ECOTOX for fish and invertebrate exposures to diazinon were NOECs above the proposed CCC of 0.82 µg/L, with thresholds at 200, and 2.1 µg/L, respectively. The only data for acetylcholinesterase inhibition in a saltwater species was an EC50 of 1,100 µg/L in Virginia oyster (Williams 1989). The fish NOEC was for hatching and survival over seven-day exposures of turbot embryos (Mhadhbi and Boumaiza 2012), and the invertebrate NOEC was for reduced progeny counts in juvenile opossum shrimp exposed for 22 days (Berry 1989). The dataset includes nine sources of data describing effects in one algae species, one fish species, and various life stages of six species of invertebrates providing evidence that responses of Nassau grouper and ESA-listed corals exposed to diazinon at or below the proposed CCC of 0.82 µg/L would be insignificant.

Since the LC50 data for invertebrates suggest mortality rates around 15 percent in invertebrates exposed to under the proposed CMC of 0.82 µg/L based on data for daggerblade grass shrimp. This species was not represented within the chronic exposure-response thresholds and the mysids that were represented were the most sensitive species invertebrate group. Taken together, chronic exposures under the proposed CCC would be expected result in effects in daggerblade grass shrimp, suggesting effects could occur in ESA-listed coral. We next consider the likelihood of exposures to diazinon in waters affected by the proposed criteria.

Diazinon is used in agriculture to control insects on fruit, vegetable, nut, and field crops. It is also used in ear tags for cattle. As described previously, agricultural activities on the USVI is limited and in decline. Diazinon would enter marine systems through storwater runoff in agricultural areas. Diazinon was not detected, with an analytical detection limit of 0.018 µg/L, among the pesticides monitored in stormwater at the St. Thomas East End Reserves (Pait et al. 2016). This stormwater monitoring deployed polar organic chemical integrative samplers at five bays to provide a representative snapshot of contaminants from low- and mid-density urban development. A 1992 evaluation of aquatic use impairments in the Caribbean detected diazinon in water and sediments at concentrations below 0.01 µg/L in Puerto Rico, but not in waters off the USVI (USEPA 1992). These reports, taken with the understanding that there are no domestic uses for diazinon, indicate that it is unlikely that diazinon would be detectable in waters and sediments of the USVI.

NMFS concludes that exposures to diazinon in waters affected by the 2018 VIWQS are expected to be extremely unlikely, and therefore discountable, so EPA approval of the proposed criteria for diazinon is not likely to adversely affect Nassau grouper and ESA-listed coral species. For this reason, diazinon is not discussed further in this opinion.

Demeton

Like diazinon, demeton is an agricultural pesticide that does not have domestic uses, so it is extremely unlikely that demeton will be detectable in stormwater runoff from the USVI. Only a CCC was proposed for this pesticide, but ECOTOX did not contain any chronic exposure-response thresholds for demeton and a search of Web of Science only produced information on analytical methodologies. The development of the demeton CCC described in the 1986 *Quality Criteria for Water "Gold Book"* (USEPA 1986) only provides two toxicity values for demeton: a two-day EC50 for pink shrimp of 63 µg/L and a one day LC50 of 550 µg/L. These values are several orders of magnitude greater than the CCC of 0.1 µg/L. Since fish are generally less sensitive to organophosphates than invertebrates (Figure 22), these data provide evidence that both Nassau grouper and ESA-listed corals are extremely unlikely to respond to demeton exposures at or below the proposed CCC of 0.1 µg/L.

NMFS concludes that exposures to demeton in waters affected by the 2018 VIWQS are expected to be extremely unlikely, and therefore discountable, so EPA approval of the proposed criteria for demeton is not likely to adversely affect Nassau grouper and ESA-listed coral species. For this reason, demeton is not discussed further in this opinion.

Guthion

Guthion is also an agricultural pesticide that does not have domestic uses. Only a CCC was proposed for guthion and all chronic exposure-response thresholds for fish exposures to guthion were above the CCC of 0.01 µg/L. Only two species are represented among the fish data, sheepshead minnow and mummichog. The lowest response thresholds among the data range from 0.17 to 0.34 µg/L and are for 28-day sheepshead minnow early life stage growth and survival tests (Morton et al. 1997). Response magnitudes at the NOEC were within three percent of the controls and growth was slightly higher than controls. Considering these data, it reasonable to expect that responses of Nassau grouper exposed to guthion under the proposed CCC of 0.01 µg/L would be insignificant.

The lowest invertebrate chronic exposure-response threshold reported for guthion was for 26-day exposures of opossum shrimp. The NOEC for the number of young per female of 0.02 µg/L resulted in a response magnitude within five percent of the controls (6.3±2.1 versus 6.4±2.25) and no difference in mean day at first brood (Morton et al. 1997). The dataset consisted of two species of crustaceans, two mollusk species, and one species of sea urchin, reporting effects on growth, fitness, development, and threshold mortality. Considering these data, it reasonable to expect that responses in ESA-listed corals exposed to guthion under the proposed CCC of 0.01 µg/L would be insignificant.

NMFS concludes that exposures to guthion in waters affected by the 2018 VIWQS are expected to be extremely unlikely, and therefore discountable, further, responses to exposures that may occur would be insignificant, so EPA approval of the proposed criteria for guthion is not likely to adversely affect Nassau grouper and ESA-listed coral species. For this reason, guthion is not discussed further in this opinion.

Malathion

A CMC for malathion was not proposed, so acute toxicity data for this pesticide will not be discussed in this opinion.

Malathion chronic exposure-response thresholds for only two fish species are represented in the dataset: sheepshead minnow and red drum. The lowest response thresholds among the fish thresholds range from 0.17 to 0.34 µg/L and are for 28-day sheepshead minnow early life stage growth and survival tests (Morton et al. 1997). Sheepshead minnow survival at the NOEC was within three percent of the controls and growth was slightly higher than controls at 0.17 µg/L, suggesting effects would not occur in exposures under the proposed CCC of 0.1 µg/L. Considering these data, it reasonable to expect that responses of Nassau grouper exposed to malathion under the proposed CCC of 0.1 µg/L would be insignificant.

Invertebrate chronic exposure-response thresholds for malathion ranged from 0.32 to 38,500 µg/L, representing seven species from six studies. The four-day threshold mortality NOEC for juvenile blue crab of 0.32 µg/L was 15 percent higher than mortality in controls (Wendel and Smee 2009). However, this NOEC is three-fold the proposed CCC of 0.1 µg/L, so a response magnitude far lower than the 15 percent would be exposed for exposures under the CCC. Considering these data, it reasonable to expect that responses of ESA-listed coral exposed to malathion under the proposed CCC of 0.1 µg/L would be insignificant.

NMFS concludes that, because responses of Nassau grouper and ESA-listed coral to malathion exposures at or below the proposed CCC is expected to be insignificant, EPA's approval of the proposed criteria is not likely to adversely affect these species. For this reason, the CMC and CCC for malathion will not be discussed further in this opinion.

7.2.2 Other Organics: Acrolein, Nonylphenol, Polychlorinated Biphenyls, Tributyltin

The pollutant 4-nonylphenol is used in the manufacture of the nonylphenol ethoxylate surfactants which degrade into 4-nonylphenol. Nonylphenol ethoxylate surfactants were once commonly used in household laundry detergents. EPA and the detergent manufacturers have cooperated to eliminate this use. In addition, nonylphenol ethoxylate use was voluntarily phased out in 2013 in liquid industrial laundry detergents and in 2014 industrial powder detergents. Though less toxic and persistent than 4-nonylphenol, nonylphenol ethoxylates are also highly toxic to aquatic organisms, and, in the environment, degrade into 4-nonylphenol, which is persistent and accumulates in sediment to concentration several orders of magnitude greater than concentrations in water (USEPA 2017).

PCBs are persistent biomagnifying chlorinated hydrocarbons consisting of carbon, hydrogen and chlorine atoms. PCBs were domestically manufactured for industrial and commercial applications from 1929 until manufacturing of these compounds was banned in 1979. The mechanism for PCB toxicity is induction of mixed function oxidases, interference with neurotransmitters, and generation of free radicals.

Tributyltin toxic effects include endocrine disruption in both fish and invertebrates (Zhang et al. 2009; Hagger et al. 2005; Huang et al. 2010; Min et al. 2018; Nakanishi 2008; Gooding et al. 2003). It now has restricted use applications, primarily for those that could lead to exposure of non-target aquatic organisms. Tributyltins were the main active ingredients in biocides and are used in the treatment and preservation of wood, antifouling of boats, antifungal action in textiles, and industrial water systems, but in many cases have been replaced by other chemicals.

Table 5 shows LC50 and chronic exposure-response thresholds for saltwater exposures of fish and invertebrates to the other organics evaluated in this opinion. The hatched areas indicate gaps in data for nonylphenol exposures of algae. PCB exposures of fish were all injection or dietary exposures and thus were not reported in the table.

Table 5. ECOTOX data for organic toxicant exposure concentrations* causing effects in marine species relative to the proposed criteria for the protection of aquatic life

Toxicant	Median Lethal Concentrations (LC50, mortality EC50) µg/L	Sublethal Effects/No Effects Concentrations (LOECs, NOECS, MATCS) µg/L	Saltwater CMC µg/L	Saltwater CCC µg/L
Nonylphenol			7	1.7
Algae		12-67 (n=9)		
Fish	17-6,104 (n=33)	20-420 (n=10)		
Invertebrates	2.7-100,000 (n=52)	0.01-1016 (n=114)		
PCBs			no CMC	0.03
Algae		0.1-1070 (n=16)		
Fish				
Invertebrates		0.098-3,000 (n=29)		
Tributyltin			0.42	0.0074
Algae	18-1,000 (n=2)	0.0003-360 (n=56)		
Fish	0.002-88 (n=47)	9.9-24 (n=3)		
Invertebrates	0.006-1,000,000 (n=130)	0.0001-100,000 (n=148)		
* Hatched areas indicate data gaps				

Nonylphenol

Fish LC50s for nonylphenol ranged from 17 to 6,104 µg/L, representing six species from six studies. These LC50s suggesting that mortality could be around 20 percent in fish populations exposed to nonylphenol at the CMC if 7 µg/L. The lowest LC50 was for a four-day exposures of winter flounder larvae (Lussier et al. 2000). However, the same study reports a two-day LC50 of 50 µg/L, indicating that exposure duration influences lethality of this toxicant. Data for other species confirm this, with LC50s for four-day exposures two or more times the LC50s reported for one-day exposures (Lussier et al. 2000). The CMC are applied as one-hour averages while the LC50s in this dataset are the result of exposures ranging from one to seven days. Mortality under the CMC, as applied, would not be expected to occur. These data provide evidence that the risk of mortality in Nassau grouper exposed to nonylphenol at or below the proposed CMC of 7 µg/L would be insignificant.

Invertebrate LC50s ranged from 2.7 to 100,000 µg/L, representing 16 species from 14 studies. Two out of 52 marine invertebrate LC50s were below the nonylphenol CMC of 7 µg/L. Both LC50s, 2.7 and 6.2 µg/L were reported by Shaukat et al. (2014) for brine shrimp exposed over two and three days. More than half of the remaining LC50s suggest greater than five percent fractional mortality would occur in invertebrates exposed under the proposed CMC. These data provide evidence that mortality in ESA-listed coral exposed to nonylphenol at or below the proposed CMC of 7 µg/L may occur.

Fish chronic exposure response thresholds for exposures to nonylphenol ranged from 20 to 420 µg/L, representing four species from three studies. These observations were above the proposed CMC of 1.7 µg/L, with the lowest NOEC of 20 µg/L for effects on growth in Atlantic salmon parr over 35 days (Arsenault et al. 2004). Considering these data, it reasonable to expect that responses of Nassau grouper exposed to nonylphenol at or below the proposed CCC of 1.7 µg/L would be insignificant.

Twenty-eight out of the 114 invertebrates chronic exposure-response thresholds for nonylphenol effects on fitness, development, mortality, and growth were below the CCC for this estrogen mimic. Affected organisms include seven species of crustaceans and molluscs exposed from one to 48 days, with eleven of these exposures shorter than the 4-day average intended to be applied to the CCC (see Table 6). Considering these data, it reasonable to expect that responses in ESA-listed coral exposed to nonylphenol at or below the proposed CCC of 1.7 µg/L may occur.

Table 6. Invertebrate NOEC observations below the CCC for nonylphenol

Species	Effect	NOEC µg/L	Source
Australian Barnacle	Development and Growth	0.6	(Billinghurst et al. 2001)
Harpacticoid Copepod	Development and Growth	0.473	(Marcial et al. 2003)
Mediterranean Mussel	Development and Growth	0.055	(Fabbri et al. 2014)
Opossum Shrimp	Mortality	0.529	(Ward and Boeri 1990)
	Development and Growth	0.65	(Hirano et al. 2009)
Pacific Oyster	Fitness	1	(Nice 2005)
	Development and Growth	0.28	(Nice et al. 2000)
Striped Barnacle	Population	0.31	(Billinghurst et al. 1998)
Taiwan Abalone	Development and Growth	1.205	(Liu et al. 2011)

NMFS concludes that the proposed CMC and CCC for nonylphenol requires further evaluation in the effects section of this opinion because mortality and other responses may occur in ESA-listed corals exposed at or below the proposed nonylphenol CMC and CCC.

PCBs

Algae chronic exposure-response thresholds for exposures to PCBs ranged from 0.1 to 1,070 µg/L (n=16), representing six species from five studies. The lowest observed PCB sublethal exposure-response threshold for algae was a NOEC of 0.1 µg/L for population abundance of *Coccolithus huxleyi* exposed for nearly six days during the exponential growth phase (log) (Fisher 1974). This lowest NOEC is above the proposed PCB CCC of 0.03 µg/L, providing evidence that responses in the zooxanthellae of ESA-listed coral exposed to PCBs at or below the proposed CCC of 0.03 µg/L would be insignificant.

Data for exposures of fish to PCBs in ambient salt water were not found in the ECOTOX and searches of the open literature returned data reporting delayed metamorphosis in Japanese flounder exposed for 38 days to Aroclor 1254 at concentrations as low as 0.1 µg/L and abnormal morphology at 1 µg/L (Dong et al. 2017). While the 0.01 µg/L NOEC in this study was below the proposed CCC of 0.03 µg/L, the 38-day exposure is considerably longer than the 4-day averaging period applied to this criterion, so we would not expect such effects over the averaging period. This provides evidence that responses in Nassau grouper exposed to PCBs at or below the proposed CCC of 0.03 µg/L would be insignificant.

Invertebrate chronic exposure-response thresholds for exposures to PCBs ranged from 0.098 to 3000 µg/L (n=29), representing five species from six studies. The lowest observed PCB sublethal exposure-response threshold for invertebrates was a NOEC of 0.098 µg/L for embryo development in the sea urchin, *Psammechinus miliaris*, exposed for 16 days (Anselmo et al. 2011). This lowest NOEC is above the proposed PCB CCC of 0.03 µg/L, providing evidence that responses in ESA-listed coral exposed to PCBs at or below the proposed CCC of 0.03 µg/L would be insignificant.

NMFS concludes that, because responses of Nassau grouper and ESA-listed coral (an zooxanthellae) to PCB exposures at or below the both the proposed CMC and CCC is expected to be insignificant, EPA's approval of the proposed criteria is not likely to adversely affect these species. For this reason, the CMC and CCC for PCBs will not be discussed further in this opinion.

Tributyltin

Algae LC50s for exposures to tributyltin ranged from 18 to 1,000 µg/L (n=2), representing two species from one study. This suggests that very little mortality (up to 1 percent) could occur in algae populations exposed under the tributyltin CMC of 0.42 µg/L. The lowest reported tributyltin LC50 for algae exposures was 18 µg/L for 5-day exposures of the diatom, *Skeletonema costatum* (Thain 1983). The response magnitude suggested by this LC50, taken with the longer duration than the one-hour averaging period, provides evidence that response of coral zooxanthellae would be insignificant.

Fish LC50s for exposures to tributyltin ranged from 0.002 to 88 µg/L, representing 11 species from 14 studies. Only one LC50 was below the CMC of 0.42 µg/L, this was for a four-day exposures of sheepshead minnow (E.G. and G. Bionomics 1981). The next lowest LC50 suggested a mortality rate as high 14 percent of a population exposed to tributyltin under the CMC. This LC50 was for four-day exposures of juvenile Chinook salmon (Short and Thrower 1987). However, this study also reported LC50s for 6 and 12 hour exposures at 20 and 54 µg/L, respectively, confirming that exposure duration influences lethality of this toxicant. Data for other species confirm this, with LC50s for four-day exposures two or more times the LC50s reported for one-day exposures (E.G. and G. Bionomics 1981; Buccafusco 1976; E.G. and G. Bionomics 1976). The CMC are applied as one-hour averages while the LC50s in this dataset are the result of exposures ranging from six hours to 22 days. Mortality under the CMC, as applied, would not be expected to occur. These data provide evidence that the risk of mortality in Nassau grouper exposed to tributyltin at or below the proposed CMC of 0.42 µg/L would be insignificant.

Invertebrate LC50s for exposures to tributyltin ranged from 0.006 to 1,000,000 µg/L (n=130), representing 41 species from 40 studies. This suggests that significant mortality could occur in invertebrate populations exposed under the tributyltin CMC of 0.42 µg/L. The lowest reported tributyltin LC50 for invertebrate exposures of 0.006 µg/L was for 4-day exposures of northern quahog larvae (Becerra-Huencho 1984). This same study reported a two day LC50 at 0.019 µg/L. Another study using this species reported LC50s for one and two-day exposures of northern quahog larvae 4.21 and 1.65 µg/L, respectively (Roberts 1987). Other LC50 observations that were below the CMC were for brine shrimp at 0.041 µg/L (Panagoula et al. 2002), and a calanoid copepod, at 0.015 µg/L (Kusk and Petersen 1997). However, species mean LC50s for these species were 140 µg/L for brine shrimp, 1.38 µg/L for Northern quahog, and 1.1 µg/L for the calanoid copepod. The brine shrimp average LC50 does not raise concern as it is orders of magnitude higher than the CMC. The species average LC50 for Northern quahog and calanoid copepod exceed the CMC by about three-fold, and the exposure durations were two to eight days long, far longer than the 1-hour averaging period applied to the CMC. The difference in exposure durations is notable since the one-day LC50 for was two and a half times the four-day LC50. Taking these factors into consideration, responses of ESA-listed corals to TBT exposures under the proposed CMC would be insignificant.

Algae chronic exposure-response thresholds for exposures to tributyltin ranged from 0.00031 to 360 µg/L (n=56), representing 21 species from 17 studies. The lowest reported tributyltin chronic exposure-response threshold for algae was an EC50 for population growth rate decline in *Platymonas* flagellate exposed for four days during exponential growth phase (Huang et al. 1996). Two other LOECs were below the CCC, at 0.001 µg/L, for threshold mortality over four-day exposures of brown algae zygotes and embryos (Burridge et al. 1995). Given the diversity of species represented and relative abundance of data indicating algae would not respond to tributyltin exposures under the proposed CCC, responses would not be expected in coral zooxanthellae exposed under the tributyltin CCC.

Fish chronic exposure-response thresholds for exposures to tributyltin ranged from 9.9 to 24 µg/L (n=3), representing two species from two studies. The lowest reported tributyltin chronic exposure-response threshold was a three-day EC50 for blastula development in dolphinfish (Adema-Hannes and Shenker 2008). The magnitude of difference between this EC50 and the CCC is such that effects are not expected at the criterion concentration, irrespective of the difference between the exposure duration and averaging period applied to the criterion. Considering these data, it reasonable to expect that responses in Nassau grouper exposed to tributyltin at or below the proposed CCC of 0.0074 µg/L would be insignificant.

Invertebrate chronic exposure-response thresholds for exposures to tributyltin ranged from 0.0001 to 1,000,000 µg/L (n=148), representing 28 species from 32 studies. The lowest reported tributyltin chronic exposure-response threshold is for exposure to a tributyltin hydride formulation. The responses were LOECs reported for purple sea urchin embryo development at two days, gastrula development at three days, and blastula development at three and a half days exposure (Roepke et al. 2005). Other chronic exposure-response thresholds reported at concentrations below the CCC include: a NOEC of 0.006 µg/L for growth of blue mussel larva over 33 days (Lapota et al. 1993), a calanoid copepod development EC50 of 0.003 µg/L over eight days (Kusk and Petersen 1997), and an Atlantic dogwinkle NOEC for penis growth, an indicator of imposex, at 0.00267 µg/L after exposures of between three months and one year (Davies and Bailey 1991). Only the purple sea urchin responses occurred over duration comparable to the CCC averaging period of four days. As a whole, the dataset included sublethal effects information for six crustaceans, nine mollusks, five other species of sea urchin, and one worm species. Given the abundance and diversity of invertebrate species in the dataset with response thresholds that do not indicate adverse effects would occur under the CCC, it is reasonable to expect that responses of ESA-listed corals to tributyltin exposures under the proposed CCC would be insignificant.

NMFS concludes that, because responses of Nassau grouper and ESA-listed coral to tributyltin exposures at or below the both the proposed CMC and CCC is expected to be insignificant, EPA's approval of the proposed criteria is not likely to adversely affect these species. For this reason, the CMC and CCC for tributyltin will not be discussed further in this opinion.

7.2.3 Inorganics: Chloride, Chlorine, Cyanide, Hydrogen Sulfide (VIWQS §186-4 and 5)

There are no aquatic life criteria proposed for chloride in saltwater in the 2018 VIWQS, so we do not discuss chloride further in this opinion.

Chlorine gas released into water first dissolves and then undergoes conversion into two forms of free chlorine: hypochlorous acid and the hypochlorite ion. If the water contains ammonia, the solution will also likely contain two forms of combined chlorine: monochloramine and dichloramine. All four of these forms of chlorine can be toxic to aquatic organisms. In saltwater, which contains bromide, addition of chlorine also produces hypobromous acid, hypobromous ion, and bromamines (USEPA 2010), which are referred to as chlorine-produced oxidants. Chlorine alone as Cl_2 is highly toxic and is often used as a disinfectant, including in wastewater treatment systems.

Cyanide and compounds of cyanide are commonly found near areas of high industrial activity and are used as pesticides. Under typical conditions in natural waters with a pH less than 9.2, most free cyanide converts to hydrogen cyanide, which is highly volatile. Thus, cyanide is not likely to bioaccumulate in aquatic life.

Hydrogen sulfide is largely generated by petroleum refineries and natural gas. Hydrogen sulfide also comes from coke ovens, paper mills, tanneries, and human sewage and animal waste.

Table 7 shows LC50 and chronic exposure-response thresholds for saltwater exposures of fish and invertebrates to the other organics evaluated in this opinion. The hatched areas indicate gaps.

Table 7. ECOTOX data for inorganic toxicant exposure concentrations* causing effects in marine species relative to the proposed criteria for the protection of aquatic life

Toxicant	Median Lethal Concentrations (LC50, mortality EC50) µg/L	Sublethal Effects/No Effects Concentrations (LOECs, NOECS, MATCS) µg/L	Saltwater CMC µg/L	Saltwater CCC µg/L
Chlorine			13	7.5
Algae				
Fish	90-230 (n=5)			
Invertebrates				
Cyanide			1	1
Algae		5.5-9,801,500 (n=28)		
Fish	30-372 (n=15)	29-56 (n=7)		
Invertebrates	1.33-860,000 (n-46)	5.6-1,000 (n=11)		
Hydrogen sulfide			no CMC	2
Algae				
Fish				
Invertebrates				

* Hatched areas indicate data gaps

7.2.4 Chlorine and Chlorine Produced Oxidants

Response thresholds reported by EPA for total residual chlorine for estuarine/marine organisms ranged from 7,500 to 46,600 µg/L based on NOECs and LOECs (for the Vessel General Permit program limits). These thresholds are far greater than the proposed CMC of 13 µg/L and CCC of 7.5 µg/L. Searches of the ECOTOX database did not provide LC50 data for chlorine and chlorine produced oxidants for invertebrates or algae or chronic exposure-response data for any marine species. A search of the Web of Science included several papers providing useful data for this assessment.

These include invertebrate LC50s ranged from 300 to 630 µg/L for American lobster larvae (Capuzzo et al. 1977; Wan et al. 2000) and the amphipod *Eohaustorius washingtonianus* (Wan et al. 2000), respectively, suggesting mortality rates at one to two percent for exposures at the CMC concentration of 13 µg/L.

These LC50s and chronic exposure-response thresholds are well above the CMC and CCC, and indicate that exposures of ESA-listed species, particularly corals, under the proposed CCC and the CMC are not likely to result in measurable responses so the chlorine CMC will not be discussed further in this opinion.

Reports of four-day EC01s for algal biomass of *Isochrysis galbana* and *Dunaliella salina* of 20 (10-30) and 190 (30-530) µg Chlorine (Cl) equivalents/L, respectively (Lopez-Galindo et al. 2010). NOECs reported for two diatom species *Achnanthes* spp and *Navicula pelliculosa* were reported at 20 and 40 µg Cl equivalents/L, respectively (Vannoni et al. 2018). Guo et al. (2017) reported photosynthesis decreases in the dinoflagellate *Prorocentrum minimum* at 1,000 µg Cl equivalents/L, with changes in transcription and expression of photosystem-regulated genes at 500 µg/L. This provides evidence that zooxanthellae would not be affected under the proposed CMC of 13 µg/L of the CCC of 7.5 µg/L for chlorine.

The only sublethal effects data found was for respiratory distress in American lobster at 50 µg/L (Capuzzo et al. 1976).

Fish LC50s for exposures to chlorine ranged from 90 to 230 µg/L (n=5), representing three species from three studies. This suggests that mortality rates around 7 percent could occur in fish populations exposed under the chlorine CMC of 13 µg/L. The lowest reported chlorine LC50 for fish exposures for 4-day exposures of spot (Bellanca and Bailey 1977). This same study reported a one-day LC50 of 140 µg/L, which is about ten-fold the CMC. Since the CMC are applied as one-hour averages, mortality under the CMC, as applied, would not be expected to occur. These data provide evidence that the risk of mortality in Nassau grouper exposed to chlorine at or below the proposed CMC of 13 µg/L would be insignificant.

While LC50 estimates for saltwater invertebrate exposures to chlorine were not found in ECOTOX or the open literature, an assessment modelling total residual oxidant in seawater used as cooling water for a power plant reports total residual oxidant LC50s for 11 marine invertebrate species ranging from 500 to 67,000 µg/L (Wang et al. 2008). The purpose of the work was to arrive at a benchmark for control of bio-fouling without irritating ambient marine organisms. These data suggest a maximum fractional mortality at the CMC of just over one percent for two of the species while no mortality is expected in the remaining species including the stony coral *Stylophora pistillata,* with an LC50 of 2,300 µg/L. These data provide evidence that the risk of mortality in ESA-listed coral exposed to chlorine at or below the proposed CMC of 13 µg/L would be insignificant.

Information on chronic exposure-response thresholds for chlorine in marine systems were not found in the ECOTOX or in searches of the open literature. Chlorine produced oxidants decay rapidly in seawater (Zeng et al. 2009), making it difficult to estimate effects over chrnic exposures in laboratory toxicity tests. The development of the chlorine CCC are described in the 1986 *Quality Criteria for Water "Gold Book"* (USEPA 1986), but toxicity data were not used in the derivation of marine criteria. Considering the expected rapid decay of chlorinated products in seawater, responses of Nassau grouper or ESA-listed coral to exposures at the proposed CCC of 7.5 µg/L is expected to be insignificant.

NMFS concludes that, because responses of Nassau grouper and ESA-listed coral to chlorine exposures at or below the both the proposed CMC and CCC are expected to be insignificant, EPA's approval of the proposed criteria is not likely to adversely affect these species. For this reason, the CMC and CCC for chlorine will not be discussed further in this opinion.

Cyanide

Fish LC50s for exposures to cyanide ranged from 30 to 372 µg/L (n=15), representing nine species from eight studies. This suggests that mortality rates around 2 percent could occur in fish populations exposed under the cyanide CMC of 1 µg/L. The lowest reported cyanide LC50 for fish exposures was 30 µg/L for 4-day exposures of cobia (Dung et al. 2005). This study included a one-day LC50 estimate of 105 µg/L, about 60 percent higher than LC50 estimates for four-day exposures. Since the CMC are applied as one-hour averages, mortality under the CMC, as applied, would not be expected to occur. These data provide evidence that the risk of mortality in Nassau grouper exposed to cyanide at or below the proposed CMC of 1 µg/L would be insignificant.

Invertebrate LC50s for exposures to cyanide ranged from 4.2 to 60,800 µg/L (n=27), representing ten species from seven studies. This suggests that mortality rates around 12 percent could occur in invertebrate populations exposed under the cyanide CMC of 1 µg/L. The lowest reported cyanide LC50 for invertebrate exposures was 4.2 µg/L for 4-day exposures of rock crab (Johns and Gentile 1981). While this study did not provide a one-day LC50 estimate, Carr et al. (1986) reported LC50 estimates from one-, two-, three-, and four-day exposures of the archiannelid worm, *Dinophilus gyrociliatus* indicating a greater than 60 percent difference between one and four day LC50 estimates, suggesting that mortality in invertebrates, if any, would be minimal or not occur for exposures under the CMC of 1 µg/L . These data provide evidence that the risk of mortality in ESA-listed coral exposed to cyanide at or below the proposed CMC of 1 µg/L would be insignificant.

Algae chronic exposure-response thresholds for exposures to cyanide ranged from 5.5 to 9801500 µg/L (n=28), representing three species from six studies. The lowest reported cyanide chronic exposure-response threshold for algae was a 14-day MATC of 5.5 µg/L (Total) for red algae (*Champia parvula*) reproduction (Steele and Thursby 1983). This lowest threshold is above the CCC concentration of 1 µg/L, so effects are not expected in algae species exposed under the proposed CCC for cyanide.

Fish chronic exposure-response thresholds for exposures to cyanide ranged from 29 to 56 µg/L (n=7), representing two species from two studies. The lowest reported cyanide chronic exposure-response threshold for fish was a 28-day NOEC of 29 µg/L (Active ingredient) for sheepshead minnow (*Cyprinodon variegatus*) embryo growth (Schimmel 1981). This lowest threshold is above the CCC concentration of 1 µg/L, so effects are not expected in fish species exposed under the proposed criterion, providing evidence that responses of Nassau grouper exposed to cyanide at or below the proposed CCC would be insignificant.

Invertebrate chronic exposure-response thresholds for exposures to cyanide ranged from 5.6 to 1000 µg/L (n=11), representing four species from four studies. The lowest reported cyanide chronic exposure-response threshold for invertebrates was a two-day NOEC of 5.6 µg/L (Total) for prickly scallop (*Chlamys asperrimus*) egg development (Pablo et al. 1997). This lowest threshold is above the CCC concentration of 1 µg/L, so effects are not expected in invertebrate species exposed under the proposed criterion. The data provide evidence that responses of ESA-listed coral exposed to cyanide at or below the proposed CCC would be insignificant.

NMFS concludes that, because responses of Nassau grouper and ESA-listed coral to cyanide exposures at or below the both the proposed CMC and CCC are expected to be insignificant, EPA's approval of the proposed criteria is not likely to adversely affect these species. For this reason, the CMC and CCC for cyanide will not be discussed further in this opinion.

7.2.5 Metals and Metalloids

Some metals, including copper, nickel, and zinc, are essential to organism function, others such as mercury have only adverse effects. Essential metals, when not in excess amounts, are structural and functional components of enzymes and vitamins. For example, metals and metalloids that are micronutrients (e.g., selenium, zinc) can be beneficial to coral health and growth when they occur at naturally low levels. However, at increased concentrations, they can be toxic. While toxic metals and essential metals at excess levels have metal-specific pathways for causing adverse effects, the one pathway to toxic effects that metals have in common is the generation of free radicals that indiscriminately oxidize proteins, nucleic acids, and cell membranes. This damage stimulates defensive measures and repair or replacement of cells. Accumulating damage depletes energy reserves and damages tissues, impairing organ function. Examples of other pathways to adverse effects include copper's interference with fish olfaction, affecting predator detection and cadmium's mimicry of calcium, affecting bone loss and calcium-dependent neuromuscular functions. The pathway to adverse effects does not require actual uptake into body tissues as adverse effects may occur due to metal impairment of gill membranes.

7.2.5.1 Probability of Exposure to Metals and Metalloids

Due to the geologic formation (Alminas et al. 1994) of the Virgin Islands and the tourism-based economy of the USVI, for most of the metals, NMFS does not expect Nassau grouper or ESA-listed coral in waters affected by the USVIWQS would ever be exposed to elevated concentrations of anthropogenic origin. Because this consultation specifically addresses EPA approval of criteria used to regulate and assess water quality in the jurisdictional waters of the USVI, we consider the probability that ESA-listed species would be exposed to criteria contaminants above natural background in these waters when making our determination regarding the potential for adverse effects in Nassau grouper or ESA-listed coral. NMFS considered the existing land and sea uses and examined environmental monitoring reports and permitting data from EPA's Enforcement Compliance History Online (ECHO) database to evaluate the likelihood of exposures.

A 1992 evaluation of aquatic use impairments in the Caribbean reported that arsenic, cadmium, chromium, copper, lead, zinc, aluminum, and mercury were detected at low levels suspended sediment samples analyzed between 1972 and 1979. In 1986, an analysis of water and sediment samples identified cadmium, copper, lead mercury, nickel selenium, and zinc at nine station locations in the USVI. Copper and mercury were detected most frequently. Copper was associated with boatyards and yacht harbors and mercury was attributed to atmospheric deposition from global sources. This study did not find contamination that was significant enough to impair aquatic life or human health. Because of these findings, similar monitoring project proposals were not funded by the USVI (USEPA 1992).

One hundred and eighty-five chemical contaminants were analyzed in sediments, and a series of sediment toxicity bioassays were conducted along with a characterization of the benthic infaunal community, as part of the Sediment Quality Triad, to assess the presence and effects of chemical contaminant stressors within the St. Thomas East End Reserves. Zinc, copper, lead, and mercury were detected above a NOAA sediment quality effects-range low guideline at one or more sites, indicating effects were possible. Copper was above detected the NOAA effects-range medium guidelines at one site, indicating effects were likely. Coral (*Porites astreoides*) and conch (*Lobatus gigas*) were among the species evaluated for contaminants in tissues. Contaminants found in coral were similar to the concentration ranges reported in corals from other reef areas in the U.S. Caribbean while conch had lower contaminant body burdens relative to published data from south Florida and some other areas of the Caribbean. The body burdens in conch were lower than available FDA action levels for molluscan shellfish consumption.

Whitall et al. (2015) related sediment metal concentrations in Coral and Fish Bays on St. John to crustal erosion to determine if the metals were from a natural source (i.e., erosion). Arsenic, chromium, copper, lead, and zinc were well correlated with aluminum concentrations, an indicator of crustal origin. However, while correlated with aluminum, copper exceeded the sediment quality guidelines at three locations and, overall, copper concentrations were disproportionately high, suggesting anthropogenic contributions.

The ECHO database lists 90 permits for discharging effluent to waters of the USVI. These are analogous to the NPDES permits and are called TPDES. Only four permits require monitoring for metals; chromium under an administratively continued permit for a closed petroleum manufacturing facility, copper from a rum distillery and two power suppliers, and zinc from the rum distillery. The power plants and retired petroleum refinery discharge to the coast. Oil contamination of groundwater beneath the retired petroleum refinery is being addressed by EPA under authority of the Resource Conservation and Recovery Act. The effluent limit exceedances report from ECHO dates back to 2007 and indicates that chromium was not among the pollutants for any reported exceedances. Discharge monitoring reports from the two power plants did not indicate exceedances of limits for copper.

The following section evaluates the implications of exposures to metals at the proposed criteria.

7.2.5.2 Probability of Responses to Metals and Metalloids Under the Proposed Criteria

Metal data in ECOTOX are typically expressed as total and dissolved concentrations. Total concentrations reflect metal present in an unfiltered sample while dissolved concentrations are the metals in a sample that has been filtered at 0.45 µm. Because the dissolved metals are most readily biologically available, the criteria were derived based on dissolved metal concentrations. Accordingly, in preparing the metals data for this evaluation, the conversion factors provided in Section 186-5 of the proposed 2018 VIWQS were applied to any data expressed as total concentrations. Concentration types expressed as active ingredient or formulation are excluded because these were exposures to complex metal compounds, such as antifouling agents, for which the metal makes up only a fraction of the mass. Concentrations expressed in terms of only the labile fraction were also excluded because this fraction is not readily relatable to the dissolved fraction the proposed criteria reflect. When selecting data, the influence of salinity on the uptake and toxicity of metals was considered. Higher salinity attenuates metal effects in some species, particularly marine species, and increases toxicity for others, particularly estuarine species (Hall and Anderson 1995; McLusky et al. 1986; Zhou et al. 2017; Bryant et al. 1985; Deruytter et al. 2015). Salinity over 97 percent of the ocean ranges between 33 and 37 ppt (Stumm and Morgan 1996). Guan et al. (2015) reported global, annually averaged tolerance limits for coral reefs of 28.7-40.4 Practical Salinity Units. Some of the saltwater toxicity tests reported in ECOTOX were at lower, estuarine salinities or above this tolerance range, so these data were excluded from our analysis for ESA-listed coral. Nassau grouper, however, are euryhaline and could occur in estuarine waters with salinities as low as 0.5 ppt, so tests conducted with estuarine salinities were retained in the saltwater toxicity data for fish.

Aluminum, chromium III and iron are included among the proposed aquatic life criteria for freshwater species, but no acute or chronic criteria are proposed for saltwater organisms, so these criteria will not be discussed further in this opinion.

Table 8 shows LC50 and chronic exposure-response thresholds for metals and metalloids evaluated in this opinion. Algae are includes in the to represent coral zooxanthellae, fish to represent Nassau grouper, and invertebrates to represent ESA-listed coral. The hatched areas in the table indicate data gaps. This table also includes information regarding the expected environmental behavior and potential for biomagnification of these metals and metalloids, as well as summarizing the toxicity data information on toxicity from the ECOTOX database.

Table 8. Environmental behavior and ECOTOX data for metal and metalloid toxicant exposure concentrations* causing effects in marine species relative to the proposed criteria for the protection of aquatic life

Metal Species group	Environmental Behavior	Median Lethal Concentrations (LC50, mortality EC50) µg/L	Sublethal Effects/No Effects Concentrations (LOECs, NOECS, MATCS) µg/L	Saltwater Acute CMC µg/L	Saltwater Chronic CCC µg/L
Arsenic	Dependent on chemical form: Accumulates in organisms			69	36
Algae			1.24-1,879,226 (n=5)		
Fish		10,300-29,247 (n=8)	2,650-654,591 (n=22)		
Invertebrate		508-17,200 (n=16)	0.1-150,000 (n=33)		
Cadmium	Partitions to sediment, low levels in water: Accumulates at all levels of food web in plants and animals			40	8.8
Algae		298.2-3,181 (n=4)	0.2-72,886 (n=54)		
Fish		149.1-56,640 (n=190)	13.78-911,081 (n=30)		
Invertebrate		7.95-611,310 (n=341)	0.06-454,258 (n=336)		
Chromium (VI)	Partitions to sediment, low levels in water: Low accumulation likelihood			1100	50
Algae			228.39-15,094 (n=8)		
Fish		14171-198,600 (n=25)	2482.5-55,906 (n=24)		
Invertebrate		56.49177-179,125 (n=121)	<0.001-125,118 (n=63)		
Copper	Partitions to sediment, low levels in water: Low accumulation likelihood in fish, higher likelihood in mollusks			4.8	3.1
Algae		8.22-100 (n=3)	0.2-20,750 (n=439)		
Fish		9.88-117196 (n=224)	8.3-7,055 (n=227)		
Invertebrate		0.83-2388136 (n=710)	0.07-830,000 (n=1650)		
Lead	Dependent on water chemistry and organic material: Accumulates at all levels of food web in plants and animals, but doesn't biomagnify			210	8.1
Algae		16,167-19,020 (n=2)	4.4-47,550 (n=30)		
Fish		4.76-206,367 (n=34)	706-31,003 (n=5)		
Invertebrate		19.69-5,347,853 (n=99)	16.17-19,020 (n=78)		
Mercury	Dependent on chemical form, methylmercury			1.8	0.94
Algae		3.4-4.68 (n=2)	0.21-374 (n=18)		

Metal Species group	Environmental Behavior	Median Lethal Concentrations (LC50, mortality EC50) µg/L	Sublethal Effects/No Effects Concentrations (LOECs, NOECS, MATCS) µg/L	Saltwater Acute CMC µg/L	Saltwater Chronic CCC µg/L
Fish Invertebrate	highly biologically available: Accumulates and magnifies at all levels of the food web	30.6-19,550 (n=42) 0.85-10,872 (n=93)	1.24-94 (n=20) 0.85-35,445 (n=69)		
Nickel Algae Fish Invertebrate	Partitions to suspended particles in water: Low accumulation likelihood	 7878-396,000 (n=18) 10.59-161,355 (n=59)	 5.84-22,275 (n=5) 7437.87-11,150 (n=3) 0.99-1,196,910 (n=37)	74	8.2
Selenium Algae Fish Invertebrate	Dependent on chemical form, found in sediment and in water: Accumulates in aquatic food web	 289.4-85668 (n=27) 123.8-261,906 (n=24)	 149.7-174,604 (n=4) 49.9-9,786 (n=25) 199.6-9,980 (n=11)	290	71
Silver Algae Fish Invertebrate	Found in sediment and some in aqueous phase: Limited accumulation in algae, mollusks, and other aquatic organisms	 4-54,400 (n=78) 0.19-3,400 (n=20)	 17.97-21,758 (n=58) 95.55-9,460 (n=13) 0.76-1,655,000 (n=621)	1.9	no CCC
Zinc Algae Fish Invertebrate	Partitions to suspended particles in water, some in aqueous phase: Low accumulation likelihood	 25.54-170,280 (n=71) 7.38-1,655,000 (n=367)	 17.974-21758 (n=58) 95.546-9,460 (n=13) 0.76-165,5000 (n=621)	90	81

* Hatched areas indicate data gaps.

Arsenic

While arsenic occurs in two oxidation states: arsenic-V and the more toxic arsenic-III, the proposed criteria were derived from data for arsenic-III, but are applied to total arsenic. The ECOTOX search did not return any LC50 data for algae nor did a search of a literature database for additional data. Arsenic in marine algae has been studied, but studies did not assess the effects of arsenic on algae (Sanders and Riedel 1993; Neff 1997; Diop et al. 2016). Impacts to marine algae are not expected because those papers that were found discussed arsenic uptake and transformation by algae, arsenic tolerance in algae, or evaluated marine algae's role in accumulation of arsenic in marine food webs. Exposure of coral zooxanthellae to arsenic at the proposed CMC is therefore not expected to result in adverse effects.

Fish LC50s for exposures to arsenic ranged from 10,300 to 29,247 µg/L (n=8), representing seven species from six studies. This suggests that mortality would not be expected to occur in fish populations exposed under the arsenic CMC concentration of 69 µg/L. The lowest LC50 among these was for 4-day exposures of striped bass (Dwyer et al. 1992). These data are several orders of magnitude higher than the proposed CMC, suggesting that mortality would not be expected to occur in fish populations exposed under the arsenic CMC and supporting a determination that mortality in Nassau grouper exposed under the CMC would be extremely unlikely to occur.

Invertebrate LC50s for exposures to arsenic ranged from 508 to 17,200 µg/L (n=16), representing 11 species from seven studies. This suggests that mortality rates of about seven percent could occur in invertebrate populations exposed under the arsenic CMC of 69 µg/L. The lowest LC50 reported was 508 µg/L for four-day exposures of the calanoid copepod *Acartia clausi* (Lussier et al. 1985). Exposure duration dependent toxicity of arsenic was demonstrated by exposures for four, eight, and 16 day providing LC50s of 96,000 µg/L, 70,000 µg/L, and 47,000 µg/L, respectively (Madsen 1992). Considering these data, mortality from exposures under the one-hour averaging period of the arsenic CMC is not expected in invertebrates, supporting a determination that the risk of mortality in ESA-listed coral exposed under the proposed CMC would be insignificant.

Algae chronic exposure-response thresholds for exposures to arsenic ranged from 1.24 to 1,879,226 µg/L (n=5), representing three species from two studies. This suggests that sublethal responses or threshold mortality could occur in algae populations exposed under the arsenic CCC of 36 µg/L. The two lowest thresholds were a NOEC and an EC50 for growth inhibition in the green algae *Chlorella salina* (Karadjova et al. 2008). These were the only reported thresholds that were below the CCC and the water used for the arsenic exposures in this study used seawater of the Bulgarian Black Sea coast. These waters are known to be polluted (Dineva 2011) and the Black Sea as a whole is the subject of a multinational effort to restore water quality (Resteanu et al. 2015). Because the exposures in the Karadjova et al. (2008) are most likely mixtures of arsenic with other unknown pollutants, these data are excluded from consideration.

As discussed previously, there is an abundance of data on arsenic tolerance and the role of algae in arsenic biogeochemistry, but not toxicity to algae. For this reason, we expect responses of coral zooxanthellae exposed to arsenic at the proposed CCC would not result in adverse effects.

Fish chronic exposure-response thresholds for exposures to arsenic ranged from 2,650 to 65,4591 µg/L (n=14), representing three species from six studies. This suggests that sublethal responses or threshold mortality would not be expected to occur in fish populations exposed under the arsenic CCC of 36 µg/L. The lowest reported threshold was a ten day NOEC for threshold mortality in pink salmon (Holland 1960). Because this lowest threshold is orders of magnitude above the CCC concentration of 36 µg/L, responses are not expected in fish species exposed under the proposed criterion, supporting a determination that responses of Nassau grouper under the proposed criterion would be insignificant.

Invertebrate chronic exposure-response thresholds for exposures to arsenic ranged from 0.1 to 150000 µg/L (n=33), representing four species from 14 studies. This suggests that sublethal responses or threshold mortality could occur in invertebrate populations exposed under the arsenic CCC of 36 µg/L. The lowest reported threshold was a LOEC for the harpacticoid copepod, *T. japonicus* time-to-develop from nauplii to the copepodid stage after four days exposure to arsenic-III and arsenic-V (Lee et al. 2008). However, this this contrasts with the NOECs from the same study for fecundity, sex ratio, threshold mortality, and number of offspring at 100 µg/L, the highest concentration tested over two generations of exposure. *T. japonicus* undergo 11 molts prior to metamorphosis into copepodids (Lee et al. 2008), whereas the ESA-listed corals considered in this opinion do not molt prior to metamorphosis into polyps. This 0.1 µg/L LOEC data appears to reflect disturbances in molting processes that do not occur in coral species. The data also include an 11 µg/L LOEC for development of purple sea urchin embryos (Garman et al. 1997) and NOECs of 631 µg/L for threshold mortality and number of young produced in post-larval opossum shrimp (Lussier et al. 1985). The significance of the purple sea urchin 11 µg/LOEC to ESA-listed corals in the Virgin Islands is attenuated by other data indicating that effects are not expected to occur at the CCC concentration and information about arsenic biogeochemistry in the Caribbean. Arsenic is naturally elevated in Caribbean waters due to the volcanic origin of the highly erodable soils the Islands (Criaud and Fouillac 1989; Bundschuh et al. 2012) and marine species are known to accumulate, metabolize, and adapt to naturally elevated arsenic (Price et al. 2013; Di Carlo et al. 2017; Khokiattiwong et al. 2009; Price et al. 2009; Ruiz-Chancho et al. 2013). Taken together, this information provides evidence that responses of ESA-listed coral exposed under the proposed CCC would be insignificant.

NMFS concludes that, because responses of Nassau grouper and ESA-listed coral to arsenic exposures at or below both the proposed CMC and CCC are expected to be insignificant, EPA's approval of the proposed criteria is not likely to adversely affect these species. For this reason, the CMC and CCC for arsenic will not be discussed further in this opinion.

Cadmium

The EPA's cadmium CMC and CCC guidelines for marine and estuarine waters were changed to 33 and 7.9 µg/L, respectively in 2016. These changes are due to inclusion of new toxicity studies, including sensitive genera. The proposed VIWQS CMC and CCC predate the current guidelines at, 40 and 8.8 µg/L, respectively.

Algae LC50s for exposures to cadmium were reported at 298 to 3,181 µg/L (n=4) for two and four-day exposures of the dinoflagellate, *Gonyaulax polyedra* and green algae, *Tetraselmis gracilis* (Okamoto et al. 1996; Okamoto et al. 1999). The lowest LC50 was reported for the dinoflagellate. This suggests that mortality rates around seven percent could occur for exposures at the cadmium CMC concentration of 40 µg/L. A seven percent mortality rate in coral endosymbionts may not substantially affect coral, which are able to attract free-swimming dinoflagellates from the surrounding waters (Hagedorn et al. 2015; Takeuchi et al. 2017; Yamashita et al. 2014; Pasternak et al. 2004).

Fish LC50s for exposures to cadmium ranged from 149 to 14,581,980 µg/L (n=190), representing 28 species from 31 studies. This suggests that mortality rates around 13 percent could occur in fish populations exposed under the cadmium CMC concentration of 40 µg/L. The lowest reported cadmium LC50 for fish exposures was for 11-day exposures of Atlantic silverside (Voyer et al. 1979) however, this was for exposures at low salinity, 10 ppt. Voyer et al. (1979) also reported a 11 day cadmium LC50 for this species at 566 µg/L for exposures at 20 ppt salinity. Nassau grouper are actually euryhaline and can be found in estuarine waters with low salinities, so an exposure to water pollutants at 10 ppt is plausible. Data for sea bass provide an example of the exposure duration-dependent toxicity of cadmium, showing a steady decline in the LC50 concentration with longer exposure durations until the LC50 approaches 3,500 µg/L at five days exposure (Figure 23; Gelli et al. 2004). This suggests that mortality would not be expected in fish exposed to cadmium at the CMC concentration of 40 µg/L over the one-hour averaging period applied to acute criteria. This supports a determination that the risk of mortality in Nassau grouper exposed under the proposed cadmium CMC would be insignificant.

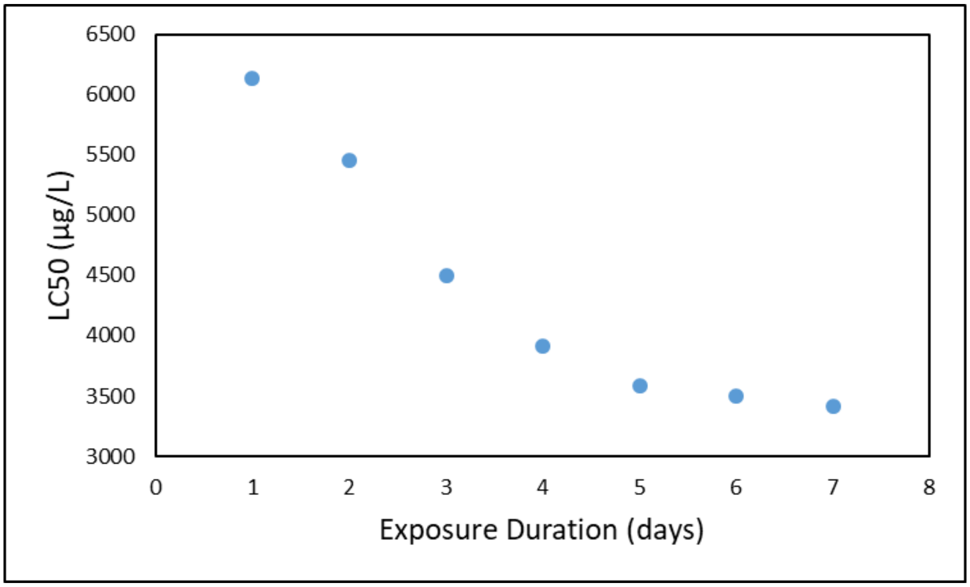

Figure 23. Relationship between exposure duration and LC50 for sea bass exposed to cadmium (after Gelli et al. 2004)

Invertebrate LC50s for exposures to cadmium ranged from 8 to 611,310 µg/L (n=341), representing 110 species from 94 studies. This suggests that significant mortality could occur in invertebrate populations exposed under the cadmium CMC of 40 µg/L. The lowest LC50 was for 4-day exposures of the mysid, *Acanthomysis costata* (Asato and Reish 1988). A total 88 of the reported LC50s suggest a fractional mortality of five percent or greater in 38 species. In addition, the exposure durations in these studies range from one hour to 45 days. Considering this, mortality in ESA-listed coral exposed to cadmium under the CCC may occur.

Algae chronic exposure-response thresholds for exposures to cadmium ranged from 0.2 to 72,886 µg/L (n=54), representing 21 species from 23 studies. This suggests that sublethal responses or threshold mortality could occur in algae populations exposed under the cadmium CCC of 8.8 µg/L. Three of these studies report thresholds for four species below the CCC concentration (Wang and Wang 2009; Mitchelmore et al. 2007; Visviki and Rachlin 1994). Wang and Wang (2009) reports a four-day LOEC of 0.2 µg/L for growth effects in the dinoflagellate *Prorocentrum minimum* (Wang and Wang 2009). Meanwhile, threshold mortality observations for a different species of dinoflagellate, *Gonyaulax polyedra*, were reported as a NOEC of 49.7 µg/L and LOEC of 99.4 µg/L (Okamoto et al. 1999). Considering these data on potential effects of cadmium on dinoflagellates, and understanding that coral are able to attract free-swimming dinoflagellates to serve as zooxanthellae, effects on coral endosymbiont species are not expected to result in responses in ESA-listed coral.

Fish chronic exposure-response thresholds for exposures to cadmium ranged from 13.78 to 911,081 µg/L (n=30), representing nine species from 12 studies. The lowest reported threshold was an 80-day NOEC for egg growth in hirame flounder (Cao et al. 2010). These data suggest that sublethal effects or threshold mortality will not occur in fish exposed to cadmium under the

CMC of 8.8 µg/L, providing evidence that responses in Nassau grouper exposed to under this CMC would be insignificant.

Invertebrate chronic exposure-response thresholds for exposures to cadmium ranged from 0.06 to 454258 µg/L (n=336), representing 70 species from 71 studies. This suggests that sublethal responses or threshold mortality could occur in invertebrate populations exposed under the cadmium CCC of 8.8 µg/L. The lowest reported threshold was a 50-day NOEC for egg growth in the common cuttlefish (Lacoue-Labarthe et al. 2010). While 42 thresholds were below the CCC, most of these were reported below the CCC (32 out of) were from the same study reporting one- to ten day LOECs or NOECs for settling success of sponge larvae over exposures of 4.97 µg/L (Ringwood 1992; Arizza et al. 2009; Martin-Diaz et al. 2005; Carr et al. 1985; Voyer and McGovern 1991). This is a large and diverse dataset, with multiple observations from different studies for many of the species represented. Averaging thresholds for fitness, growth, and threshold mortality LOECs and NOECs within species did not suggest adverse effects would occur in marine invertebrates under the CCC.

Most importantly, this dataset includes a five-hour LOEC and NOEC of 4,970 and 1,988 µg/L, respectively, for fitness in *Acropora tenuis* (Reichelt-Brushett and Harrison 2005), a Pacific coral that is the same genus as two of the ESA-listed coral species in the Caribbean. These data provide evidence that responses of ESA-listed corals to cadmium exposures under the CCC of 8.8 µg/L would not be significant.

Although effects on dinoflagellates that serve as coral endosymbionts may occur, NMFS concludes that responses of Nassau grouper and ESA-listed coral to cadmium exposures at or below both the proposed CMC and CCC are expected to be insignificant, therefore EPA's approval of the proposed criteria is not likely to adversely affect these species. For this reason, the CMC and CCC for cadmium will not be discussed further in this opinion.

Chromium (VI)

Data for the effects of chromium (VI) on marine algae were not available in ECOTOX and searches of a literature database only produced papers on chromium (VI) biosorption by marine algae, biochemical responses, or uptake and distribution in marine organisms. These biosorption data (e.g. Farzadkia et al. 2012) suggest that adverse effects would not occur in coral zooxanthellae exposed to chromium (VI) under the proposed CMC or CCC.

Fish LC50s for exposures to chromium (VI) ranged from 14,171 to 198,600 µg/L (n=25), representing 11 species from 11 studies. This suggests that mortality rates of around 4 percent could occur in some fish populations exposed under the chromium (VI) CMC of 1,100 µg/L. The lowest LC50 was reported for four-day exposures of Atlantic silverside larva (Cardin 1985). All but one LC50 suggest mortality rates of between one and three percent for exposures of one to seven days. However, the increasing LC50 concentrations for exposures of minnows over one (198,600 µg/L), four (90,363 µg/L), and seven days (43,693 ug/L; Jop et al. 1986) provide evidence that, based on exposure duration, mortality of fish would not be expected to result from exposures to chromium (VI) at the CMC of 1,100 μg/L over the one-hour averaging period. Considering data for other metals and the relatively low fractional mortality expected for four-day exposures at the CMC concentration, it is reasonable to expect that exposures of fish averaging 1,100 μg/L chromium (VI) over one hour would not likely result in mortality. This provides evidence that mortality of Nassau grouper exposed to chromium (VI) under the CMC would be insignificant.

Invertebrate LC50s for exposures to chromium (VI), mostly as potassium dichromate, ranged from 56 to 179,125 μg/L (n=121), representing 40 species from 29 studies. Eight of the reported LC50s were below the CMC concentration of 1,100 μg/L and the majority of the other LC50s indicate mortality rates of greater than five percent and up to 42 percent. Exposure durations in this dataset ranged from six hours to 59 days. The degree of mortality suggested at the CMC concentration is such that, even accounting for the one-hour averaging period applied to the criterion relative to the exposure periods used in these tests, mortality would likely occur in invertebrate populations exposed under the proposed CMC criterion. This provides evidence that mortality would be expected in ESA-listed coral exposed to chromium (VI) under the CMC of 1,100 μg/L.

Algae chronic exposure-response thresholds for exposures to chromium (VI) ranged from 228.39 to 15094 μg/L (n=8), representing three species from two studies. While this dataset does not include threshold data for *Symbontium* sp. or dinoflagellates, the data provided suggests that sublethal responses or threshold mortality would not be expected to occur in algae populations exposed under the chromium (VI) CCC of 50 μg/L.

Fish chronic exposure-response thresholds for chromium (VI) ranged from 2,482 to 55,906 μg/L (n=24), representing two species, sheepshead minnow and silver salmon, from three studies (Jop et al. 1995; Holland 1960; McCulloch and Rue 1989). These values include NOECs for growth and mortality that are well above the CCC of 50 μg/L, indicating that effects are not expected in fish exposed to chromium (VI) under the CCC. These data provide evidence that responses of Nassau grouper exposed to chromium (VI) under the CMC would be insignificant.

Invertebrate chronic exposure-response thresholds for exposures to chromium (VI) ranged from <0.001 to 125,118 μg/L (n=63), representing 23 species from 26 studies. This suggests that sublethal responses or threshold mortality could occur in invertebrate populations exposed under the chromium (VI) CCC of 50 μg/L.

Because this consultation specifically addresses EPA approval of criteria used to regulate and assess water quality in the jurisdictional waters of the USVI, we are also able to consider the probability that ESA-listed species would be exposed to chromium (VI) in these waters when making our determination. The majority of the toxicity data are for potassium dichromate, a potent oxidizer with industrial applications. Potential industrial sources of chromium (VI) in water include not only discharges from local activities, but atmospheric transport. Approximately one-third of the atmospheric releases of chromium are believed to be in the hexavalent form, chromium (VI) (Johnson et al. 2006). The estimated atmospheric half-life for chromium (VI) reduction to chromium (III) was reported in the range of 16 hours to about five days (Kimbrough et al. 1999). In natural waters soluble chromium (VI) is eventually be reduced to chromium (III) by organic matter or other reducing agents in water (Richard and Bourg 1991). While chromium was recently monitored for and detected in USVI sediments, oxidation state was not discussed. A single facility on St. Croix was required to monitor chromium in its discharges. The facility is no longer operational, its permit is listed as administratively continued, and the civil enforcement case report indicates remediation and brownfield conversion is provided for (USEPA 2016). NMFS considers this to indicate that at this time there are no potential sources of chromium (VI) to USVI waters, so exposures of aquatic life to chromium (VI) is expected to be extremely unlikely. NMFS concludes that, because it is extremely unlikely that Nassau grouper and ESA-listed coral would be exposed to chromium (VI) in USVI waters, EPA's approval of the proposed criteria is not likely to adversely affect these species. For this reason, the CMC and CCC for chromium(VI) will not be discussed further in this opinion.

Copper

Algae LC50s for exposures to copper ranged from 8.2 to 100 µg/L (n=3), representing two species from two studies. This suggests that mortality rates of around 29 percent could occur in some algae populations exposed under the copper CMC of 4.8 µg/L. This lowest reported LC50 was for five-day exposures of green algae, *Enteromorpha* sp. (Fletcher 1989). However, the data include two and four day LC50s for the dinoflagellate, *Gonyaulax polyedra*, of 100 and 66 µg/L, respectively (Okamoto et al. 1999). Because coral *Symbiodinium* endosymbionts are dinoflagellates, these data, suggesting mortality rates of less than five percent at the CMC concentration, are more suitable for evaluating the CMC. Based on these data, implications for ESA-listed coral zooxanthellae are expected to be insignificant.

Fish LC50s for exposures to copper ranged from 10 to 117,196 µg/L (n=224), representing 49 species from 49 studies. This suggests that mortality rates of around 24 percent could occur in some fish populations exposed under the copper CMC of 4.8 µg/L. The lowest reported LC50 was for four-day exposures of summer flounder embryos (Cardin 1985). This contrasts with another study reporting a five day LC50 for embryos of the same species at 704 µg/L (CH2M Hill 1999), suggesting no mortality would occur at the CMC exposure concentration. The dataset includes exposure-duration response data similar to those illustrated in Figures 20 and 22, so it is appropriate to consider LC50 data in context of exposure duration when evaluating the CMC at it is applied as a one hour average. This is important considering the factorial study exploring the effects of exposure time and concentration on the toxicity of copper to reported no mortality in the first two hours of copper exposures ranging from 10 to 2,000 µg/L (Gündoğdu 2008). A majority (n=116) of the remaining LC50s do not suggest mortality would occur for exposures to copper at the CMC concentration for exposures ranging from 12 hours to five days. The bulk of the remaining LC50s (n=107) suggesting a mortality rate of less than 5 percent could occur at this concentration for exposures ranging from two hours to 14 days. The two-hour LC50 was 339 µg/L, suggesting a fractional mortality of only 0.7% in northern anchovy embryos (W. Jr. Rice et al. 1980), if exposed at the CMC concentration. The remaining four LC50s suggest mortality rates of seven to 18 percent for exposures lasting two to seven days to copper at the CMC concentration. Considering breadth and diversity of the LC50 data for fish and the dominance of data suggesting no or very low fractional mortality for days-long exposures at the CMC concentration of 4.8 µg/L, it is reasonable to expect that exposures averaging 4.8 µg/L copper over one hour would not likely result in mortality. This provides evidence that mortality of Nassau grouper exposed to copper under the CMC would be insignificant.

Invertebrate LC50s for acute exposures to copper ranged from 0.83 to 2388136 µg/L (n=710), representing 167 species from 166 studies. This suggests that significant mortality could occur in invertebrate populations exposed under the copper CMC of 4.8 µg/L. The lowest reported LC50s were for 4-day exposures of queen conch veliger and Karuma shrimp mysis and nauplii (Bambang et al. 1995; Garr 2012). However, the study of queen conch veliger evaluated copper effects on both fed and unfed larvae, with mortality in fed larvae (a more likely scenario in the wild) too low to estimate an LC50 (Garr 2012). More importantly, among these data are 34 LC50s for six different species of coral: *Pocillopora damicornis*, *Goniastrea aspera*, *Galaxea fascicularis*, *Acropora tumida*, and *Platygyra acuta*. The LC50s for these species range from 27 to 461 µg/L (Kwok and Ang 2013; Bao et al. 2011; Esquivel 1986; Sabdono 2009; Reichelt-Brushett and Harrison 2004), suggesting mortality rates of up to nine percent for exposures at the CMC of 4.8 µg/L (Figure 24).

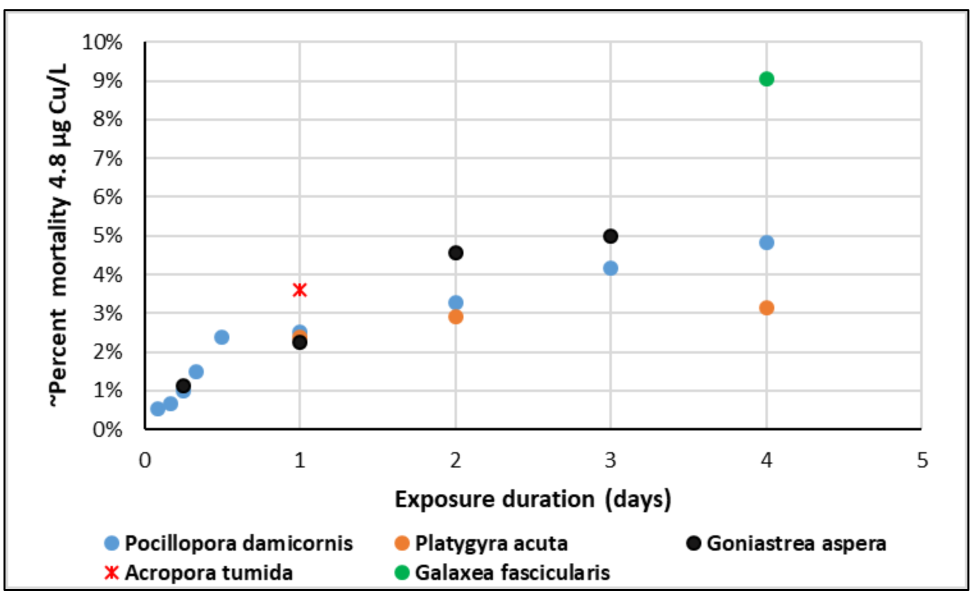

Figure 24. Estimated mortality over time in coral species exposed to copper at the CMC concentration of 4.8 µg/L

The data for *Pocillopora damicornis* (blue dots) and *Goniastrea aspera* (black dots) in Figure 24 suggests mortality rates below one percent for exposures at the CMC for durations of less than six hours. Considering these data, it is reasonable to expect that mortality would not likely occur in corals exposed to an average copper concentration of 4.8 µg/L over one hour. This provides evidence that the risk of mortality in ESA-listed coral exposed to copper under the CMC would be insignificant.

Algae chronic exposure-response thresholds for copper ranged from 0.2 to 20,750 µg/L (n=439), representing 69 species from 115 studies. The lowest reported threshold was a three-day LOEC of 0.2 µg/L for population growth in the diatom, *Bellerochea polymorpha,* during the exponential growth phase (Levy et al. 2007). This suggest effects would occur in algae exposed under the copper CCC of 3.1 µg/L. The dataset includes 41 mortality and population response thresholds for ten dinoflagellate species ranging from 3.3 to 78 µg/L, including a MATC of 10.68 µg/L for a 23-day exposure for *Symbiodinium microadriaticum* (Goh and Chou 1997). Bielmyer et al. (2010) reported NOECs for population effects from 3.3 to 16.8 µg/L for 35-day exposures of *Symbiodinium* sp. The dataset also includes a NOEC for population effects in *Symbiodinium muscatinei* exposed for 42 days to 83 µg/L copper (Mitchelmore et al. 2003). Considering these data, adverse effects are not expected in coral zooxanthellae exposed to copper at the CCC.

Fish chronic exposure-response thresholds for exposures to copper ranged from 8.3 to 7055 µg/L (n=227), representing 25 species from 43 studies. This suggests that sublethal responses or threshold mortality would not be expected to occur in fish populations exposed under the copper CCC of 3.1 µg/L. However, these data include a three day EC50 for blastula development in the dolphinfish (Adema-Hannes and Shenker 2008) and four day EC50 for behavioral aberrations in juvenile common goby that, while above the CCC of 3.1 µg/L, suggest a fractional response of (Ringwood 1992) around 12 percent, suggesting adverse effects may occur in Nassau grouper exposed to copper under the CCC.

Invertebrate chronic exposure-response thresholds for copper ranged from 0.07 to 830,000 µg/L (n=1650), representing 176 species from 276 studies. The lowest reported threshold was a 32-hour EC10 for development in Pacific oyster (Worboys et al. 2002). A total of 88 observations were below the CCC. Most importantly, the data include 84 observations for 12 species of coral, including two ESA-listed species, *Acropora formosa, Acropora longicyathus, Acropora surculosa, Acropora tenuis, Goniastrea aspera, Goniastrea retiformis, Montipora verrucosa,* Mountainous star coral (threatened), *Platygyra daedalea, Pocillopora damicornis*, Rice Coral, and Staghorn Coral (threatened). Thresholds for coral ranged from 1.66 to 415 µg/L, with the lowest a six-hour NOEC reported for *Goniastrea aspera* was below the 3.1 µg/L CCC, at 1.66 µg/L (Reichelt-Brushett and Harrison 1999). The fertilization rate at this concentration was actually slightly higher than that of controls (93±4.03 versus 90±3.23). A 35-day LOEC for growth in *Pocillopora damicornis* reported at 3.35 µg/L indicated a 30 percent decline in growth (Bielmyer et al. 2010). The one-day EC50 of for embryo development alterations in mountainous star coral, a threatened species suggest a response magnitude of 7.5 percent. The confidence interval reported with this EC50 indicates the EC50 may actually be as low as 7.4 µg/L, resulting in a response of 21% for exposures at the copper CCC. These data are highly relevant to the effects of the copper CCC on ESA-listed corals evaluated in this opinion. Considering the number of observations indicating effects may occur in other invertebrates at or below the copper CCC and the response thresholds reported for coral, adverse effects may occur in ESA-listed coral exposed to copper under the CCC.

NMFS determines that, because responses of Nassau grouper and ESA-listed coral to acute exposures to copper at or below the proposed CMC are expected to be insignificant, EPA's approval of the proposed CMC for copper is not likely to adversely affect these species. For this reason, the copper CMC for acute exposures will not be discussed further in this opinion.

However, the proposed CCC for copper requires further evaluation in Section 9 of this opinion because mortality and other responses may occur in Nassau grouper and ESA-listed coral exposed at or below the proposed copper CCC during chronic exposures.

Lead

Algae LC50s for exposures to lead were reported at 16,167 and 19,020 µg/L for four and two-day exposures of the dinoflagellate, *Gonyaulax polyedra* (Okamoto et al. 1999). This suggests that very little mortality (up to 0.6 percent) could occur in algae populations exposed under the lead CMC of 210 µg/L. Adverse effects in ESA-listed coral due to effects of this CMC on zooxanthellae are therefore not expected.

Fish LC50s for exposures to lead ranged from 4.75 to 206,367 µg/L (n=34), representing 12 species from 12 studies. This suggests that significant mortality could occur in fish populations exposed under the lead CMC of 210 µg/L. The lowest reported lead LC50 was for four-day exposures of pre-larval Japanese anchovy (Cherkashin et al. 2004). However, this appears to be an outlier observation because the next highest LC50 is 1,455 µg/L, suggesting a mortality rate of seven percent in a variety of marine sculpin, the cabezon, for four-day exposures at the CMC concentration of 210 µg/L (Dinnel et al. 1983). Most importantly, the data include LC50s for exposures of grouper at 40,418 µg/L, 21,398 µg/L, 19,069 µg/L, and 16,167 µg/L for one-, two-, three-, and four-day exposures, respectively (Siammai and Chiayvareesajja 1988). These data, for a species of the same genus as Nassau grouper *Epinephelus* sp., provide evidence that mortality of Nassau grouper exposed to lead at or below the CMC of 210 µg/L over the one-hour averaging period for this criterion would be insignificant.

Invertebrate LC50s for exposures to lead ranged from 19.7 to 5,347,853 µg/L (n=99), representing 43 species from 34 studies. This suggests that significant mortality could occur in invertebrate populations exposed under the lead CMC of 210 µg/L. The lowest reported LC50 was for three-day exposures of San Francisco brine shrimp, *Artemia franciscana* (MacRae and Pandey 1991). However, the dataset includes five LC50s for the stony coral, *Goniastrea aspera*, at 13,380 µg/L, 11,127 µg/L, 9,549 µg/L, 9405 µg/L, and 9405 µg/L for six-hour, one day, 18-hour, two day, and three-day exposures, respectively (Reichelt-Brushett and Harrison 2004). These data for a species of stony coral provide evidence that mortality of ESA-listed coral exposed to lead at or below the CMC of 210 µg/L over the one-hour averaging period for this criterion would be insignificant.

Algae chronic exposure-response thresholds for lead ranged from 4.4 to 47,550 µg/L (n=30), representing 13 species from 15 studies. Five observations were below the CCC of 8.1 µg/L as EC50s ranging from 4.4 to 5.7 µg/L for growth in exposed diatom *Skeletonema costatum* populations during the exponential growth phase from one to 11 days (Rivkin 1979). As EC50s below the CCC of 8.1 µg/L, these data suggest that effects would occur in algae at the criterion concentration. However, this dataset also includes chronic exposure-response mortality thresholds for dinoflagellates indicating a four day NOEC of 238 µg/L (Okamoto et al. 1999). Based on these data, adverse effects in ESA-listed coral due to effects of this CMC on zooxanthellae are therefore not expected.

Fish chronic exposure-response thresholds for exposures to lead ranged from 706 to 31,003 µg/L (n=5), representing four species from five studies. This suggests that sublethal responses or threshold mortality would not be expected to occur in fish populations exposed under the lead CCC of 8.1 µg/L, providing evidence that responses of Nassau grouper exposed to lead at the CCC would be insignificant.

Invertebrate chronic exposure-response thresholds for exposures to lead ranged from 16 to 19020 µg/L (n=78), representing 27 species from 24 studies. This suggests that sublethal responses or threshold mortality would not be expected to occur in invertebrate populations exposed under the lead CCC of 8.1 µg/L. The lowest reported threshold was a NOEC for opossum shrimp post-larva reproduction at nearly twice the CCC (Lussier et al. 1985). In addition, NOECs for mortality in exposed larvae and fertilization success in gametes of the stony coral *Goniastrea aspera* were 9,510 and 5,188 µg/L, respectively (Reichelt-Brushett and Harrison 2004;2005). These data provide evidence that responses of ESA listed coral exposed to lead at the CCC would be insignificant.

NMFS concludes that, because responses of Nassau grouper and ESA-listed coral to lead exposures at or below both the proposed CMC and CCC are expected to be insignificant, EPA's approval of the proposed criteria is not likely to adversely affect these species. For this reason, the CMC and CCC for lead will not be discussed further in this opinion.

Mercury

Algae LC50s for exposures to mercury were reported at 3.4 and 4.7 µg/L for four- and two-day exposures of the dinoflagellate *Gonyaulax polyedra*, respectively (Okamoto et al. 1999), suggesting up to 26 percent mortality for algae exposed at the CMC concentration of 1.8 µg/L. While an exposure duration effect on LC50 is implied by these two observations, the estimated mortality of 26 percent at the CMC is too high to make inferences on whether exposure for one-hour at the CMC concentration would adversely affect coral zooxanthellae.

Fish LC50s for exposures to mercury ranged from 30.6 to 19,550 µg/L (n=42), representing 12 species from 12 studies. This suggests that mortality rates around three percent could occur in fish populations exposed under the mercury CMC of 1.8 µg/L. The lowest LC50 was for four-day exposures of spot (D'Asaro 1985). Data for the common goby illustrate the exposure duration dependence of mercury lethality with a one day LC50 concentration about 3.5 times the four day LC50 concentration (Vieira et al. 2009), suggesting that one-hour exposures at the CMC concentration of 1.8 µg/L would not result in substantial mortality of fish populations. This provides evidence that the risk of mortality in Nassau grouper exposed to mercury at or below the CMC of 1.8 µg/L over the one-hour averaging period for this criterion would be insignificant.

Invertebrate LC50s for exposures to mercury ranged from 0.85 to 10,871 µg/L (n=93), representing 56 species from 39 studies. This suggests that significant mortality could occur in invertebrate populations exposed under the mercury CMC of 1.8 µg/L. The lowest reported LC50 was for two-day exposures of European oyster larva (Connor 1972). The LC50s for 18 species signal mortality rate of greater than five percent at the CMC. However, the data include LC50s for the foraminiferan, *Pararotalia spinigeraa* (Bresler and Yanko 1995). Foraminifera are single celled shelled protists. Those with endosymbionts are considered suitable surrogates for corals in pollution studies because they have similar water-quality requirements as stony corals (Hallock et al. 2003). The one day LC50s reported for this species ranged from 16 to 4,798 µg/L based on the influence of the dissolved organic carbon from decaying seaweed in the test water. The lowest LC50, suggesting about 5.6 percent mortality, was reported for exposures in clear, natural seawater (i.e., low dissolved organic carbon). These data provide evidence that the mortality may occur in ESA-listed corals exposed to mercury at or below the CMC of 1.8 µg/L.

Algae chronic exposure-response thresholds for mercury ranged from 0.21 to 374 µg/L (n=18), representing seven species from nine studies. These include ten observations below the CCC concentration of 0.94 µg/L and six EC50s suggesting threshold responses of up to 22 percent. The lowest reported threshold was a four day NOEC for threshold mortality in the dinoflagellate *Gonyaulax polyedra* (Okamoto et al. 1999), suggesting that algae, including zooxanthellae, may be affected by exposures at the CCC.

Fish chronic exposure-response thresholds for exposures to mercury ranged from 3.12 to 94 µg/L (n=19), representing two species from five studies. This suggests that sublethal responses or threshold mortality would not be expected to occur in fish populations exposed under the mercury CCC of 0.94 µg/L. However, the lowest threshold was a four day EC50 for in the juvenile common goby Considering the duration of exposures for these lowest thresholds that are four fold the CCC concentrations, responses of fish to exposures at the CCC of 0.94 µg/L are not expected, providing evidence that responses in Nassau grouper exposed at the CCC would be insignificant.

Invertebrate chronic exposure-response thresholds for exposures to mercury ranged from 0.85 to 35445 µg/L (n=69), representing 19 species from 12 studies. This suggests that sublethal responses or threshold mortality could occur in invertebrate populations exposed under the mercury CCC of 0.94 µg/L. The lowest reported threshold was an day1 hour LOEC for behavioral abnormalities in two-spot California octopus (Anderson et al. 1989). In addition, four EC50s suggested responses of up to ten percent for developmental effects in Pacific oyster, the sea urchin, *Paracentrotus lividus*, and the tubeworm, *Hydroides elegans*, exposed to mercury at the CCC concentration of 0.94 µg/L for two days or less (Glickstein 1978; His et al. 1999; Gopalakrishnan et al. 2008) and up to 30 percent mortality in blue crab embryos exposed at that concentration for five to six days (Howe et al. 2014). These data provide evidence that adverse effects may occur in ESA-listed coral exposed to mercury under the CCC.

Because this consultation specifically addresses EPA approval of criteria used to regulate and assess water quality in the jurisdictional waters of the USVI, we are also able to consider the probability that ESA-listed species would be exposed to mercury in these waters when making our determination.

Potential sources of mercury in water include not only discharges from local activities, global atmospheric transport of mercury from volcanic activity and coal combustion can result in mercury deposition to areas far from the source (Costa et al. 2012; Fitzgerald et al. 2018). Considering that existing monitoring data has not identified environmentally significant concentrations of mercury and there are no apparent point sources in the USVI (see section 1.1.1.1), NMFS expects that it is extremely unlikely that Nassau grouper and ESA-listed coral would be exposed to mercury from a regulated source in USVI waters. The EPA's approval of proposed criteria for a substance that is not likely to occur above global background levels contributed in waters affected by the VIWQS is therefore not likely to adversely affect these species. For this reason, the CMC and CCC for mercury will not be discussed further in this opinion.

Nickel

The ECOTOX did not include LC50s for exposures of marine algae to nickel and a search of the open literature was unfruitful. The effects of acute exposures under the nickel CMC toxicity on zooxanthellae of ESA-listed coral cannot be estimated.

Fish LC50s for exposures to nickel ranged from 7,878 to 396,000 µg/L (n=18), representing seven species from six studies. The lowest reported LC50 was for four-day exposures of Atlantic silverside (Cardin 1985). This suggests that mortality would not be expected in fish populations exposed under the nickel CMC of 74 µg/L, providing evidence that the risk of mortality in Nassau grouper exposed to nickel at and below the CMC would be insignificant.

Invertebrate LC50s for exposures to nickel ranged from 10.59 to 161,355 µg/L (n=59), representing 24 species from 22 studies. The lowest reported LC50 was for one-day exposures of for San Francisco brine shrimp nauplii (Asadpour et al. 2013). Two other studies reported LC50s for opossum shrimp at 150 µg/L, signaling 25 percent mortality at the CMC (Lussier and Walker

1985; Gentile and Cardin 1982). Overall, a total of 13 of the LC50s suggested mortality rates of five percent or more for exposures at the proposed nickel CMC of 74 µg/L. This suggests that significant mortality could occur in invertebrate populations exposed under the proposed nickel CMC, providing evidence that mortality in ESA-listed coral could occur as a result of exposures to nickel at or below the proposed CMC.

Algae chronic exposure-response thresholds in ECOTOX for nickel ranged from 5.8 to 22,275 µg/L (n=5), representing four species from four studies. DeForest and Schlekat (2013) published a compendium of nickel EC10s, not reported in ECOTOX, for four algae species ranging from 96.7 to 17,891 µg/L (Golder 2007; Parametrix 2007a;e;c). The lowest reported threshold was a seven day NOEC for population effects in the chrysophyte *Aureococcus anophagefferens* (Wei et al. 2013). The remaining data include EC50s suggesting population growth responses of zero to 2.8 percent for the green algae, *Chlorella vulgaris*, the diatom *Ditylum brightwellii*, and blue-green algae (*Cyanobium* sp.) exposed to nickel for three- to five days at the CCC concentration of 8.2 µg/L (Canterford and Canterford 1980; Latala and Surosz 1997; Alquezar and Anastasi 2013). This suggests that coral zooxanthellae may be affected by chronic exposures to nickel under the CCC, but effects to ESA-listed coral would be tempered by their ability to attract free-swimming dinoflagellates to serve as endosymbionts.

Fish chronic exposure-response thresholds for nickel were reported in ECOTOX for inland silverside included NOECs for threshold mortality and growth of 7,438 µg/L and a threshold mortality LOEC of 11,150 µg/L (Lussier and Walker 1985). DeForest and Schlekat (2013) published a compendium of nickel EC10s for sheepshead minnow and topsmelt silverside at 3,599 and 20,760 µg/L, respectively (Golder 2007; Hunt et al. 2002). These data suggest that fish would not respond to exposures to nickel at the CCC concentration of 8.2 µg/L. These data provide evidence that responses in Nassau grouper exposed to nickel under the proposed CCC would be insignificant.

Data on the chronic toxicity of nickel to invertebrates should be considered in context of recent evidence that nickel exposures at low concentrations may actually stimulate growth in coral. Exposure to nickel enriched water at 3.5 µg/L over four weeks resulted in a 1.49-fold and 1.64-fold increase in growth in *Dendrophyllia arbuscula* and *Pocillopora damicornis*, respectively (Biscéré et al. 2018). Toxicity thresholds in ECOTOX for invertebrates under chronic exposures to nickel ranged from 0.99 to 1,196,910 µg/L (n=37), representing 11 species from 16 studies. This suggests that sublethal responses or threshold mortality could occur in invertebrate populations exposed under the nickel CCC of 8.2 µg/L. DeForest and Schlekat (2013) published a compendium of nickel EC10s, some not reported in ECOTOX, for various life stages of ten invertebrate species ranging from 2.9 to 431 µg/L (Bielmyer et al. 2005; Gentile and Cardin 1982; Parametrix 2007b;g;f;d; Hunt et al. 2002; Novelli et al. 2003; Pagano 2007). The lowest reported threshold was a NOEC for mortality in embryos of the snail *Marisa cornuarietis* exposed to nickel for up to 12 days (Sawasdee and Kohler 2009). This same study reported the threshold mortality LOEC and six day development NOEC at 9.9 µg/L, an exposure concentration just above the proposed nickel CCC of 8.2 µg/L. The remaining thresholds were an order of magnitude greater than the proposed nickel CCC concentration and included a NOEC of 1,980 µg/L for gamete survival in the stony coral species *Goniastrea aspera* (Reichelt-Brushett and Harrison 2005). More recent data assess nickel effects on to gamete fertilization, *Acropora aspera* was the most sensitive species to nickel, with a NOEC below 280 µg /L), followed by *A. digitifera* with an EC10 of 2000 µg /L and *Platygyra daedalea*, with an (EC10 greater than 4,610 µg/L (Gissi et al. 2017). The single study reporting a 12 day NOEC concentration below the proposed CCC of 8.2 µg/L contrasts with the majority of response thresholds, includingthose specifically for coral species, indicating invertebrates would not respond to exposures at and below this concentration. Based on these data responses of ESA-listed coral to exposures under the nickel CCC would be insignificant.

Because this consultation specifically addresses EPA approval of criteria used to regulate and assess water quality in the jurisdictional waters of the USVI, we are also able to consider the probability that ESA-listed species would be exposed to nickel in these waters when making our determination regarding the potential for adverse effects identified for ESA-listed coral under the CMC. None of the NPDES permits tracked in EPA's Enforcement Compliance History Online database require monitoring of nickel in discharges, so NMFS expects that nickel is not released from regulated point sources in appreciable amounts. A 1986 analysis of water and sediment samples evaluated for heavy metals for nine station locations in the USVI detected nickel, but did not find contamination that was significant enough to impair aquatic life or human health. Because of these findings, similar monitoring project proposals were not funded by the USVI (USEPA 1992). These reports indicate that nickel concentrations in waters around the USVI are not expected to ever reach acute exposures at the CMC concentration of 74 µg/L.

NMFS concludes that EPA's approval of the proposed criteria for nickel is not likely to adversely affect Nassau grouper or ESA-listed coral because:

1) It is extremely unlikely that ESA-listed coral would experience acute exposures to nickel in USVI waters;
2) Responses of ESA-listed coral to chronic exposures low levels of nickel in USVI waters at the proposed CCC would be insignificant, if not beneficial, and
3) Responses of Nassau grouper nickel exposures at or below both the proposed CMC and CCC are expected to be insignificant

For these reasons, reason, the CMC and CCC for nickel will not be discussed further in this opinion.

Selenium

Selenium is a trace essential mineral incorporated into some antioxidant proteins, so it is expected to be present in biological tissues. In excess amounts, it is toxic. The ECOTOX database did not include acute toxicity values for algae exposures to selenium. The 1987 criteria document reported a four-day EC50 for *Skeletonema costatum* of 7,930 µg/L. In addition, growth of *Chlorella* sp., *Platymonas subcordiformis,* and *Fucus spiralis* actually increased at selenium(IV) concentrations from 10 to 10,000 ~g/L. Based on this information, EPA concluded that these saltwater plants would not be adversely affected by selenium concentrations that do not affect saltwater animals (USEPA 1987a).

Fish LC50s for exposures to selenium ranged from 598 to 85,668 µg/L (n=25), representing seven species from five studies. The lowest LC50 reported was a four-day exposure of haddock larva (Cardin 1985). This suggests that mortality rates of around 24 percent could occur in fish populations exposed under the proposed selenium CMC of 290 µg/L. Meanwhile other species average LC50s suggested mortality rates at the CMC concentration ranging from zero to four percent for exposures lasting one to four days. However, the haddock data is taken as evidence that mortality could occur in Nassau grouper as a result of exposures to selenium at or below the proposed CMC.

Invertebrate LC50s for exposures to selenium ranged from 123.85 to 261,906 µg/L (n=24), representing ten species from eight studies. The lowest LC50 was for three-day exposures of Virginia oyster embryo (Smith 1979). This suggests that significant mortality could occur in invertebrate populations exposed under the selenium CMC concentration of 290 µg/L. Five other species LC50s indicated mortality rates of five percent or greater at this concentration, These data provide evidence that mortality could occur in ESA-listed coral as a result of exposures to selenium at or below the proposed CMC.

Algae chronic exposure-response thresholds for exposures to selenium ranged from 149.7 to 174,604 µg/L (n=4), representing three species from a single study. This suggests that sublethal responses or threshold mortality would not be expected to occur in algae populations exposed under the selenium CCC of 71 µg/L, providing evidence that the zooxanthellae of ESA-listed coral would not be affected by exposures under the CCC.

Fish chronic exposure-response thresholds for exposures to selenium ranged from 49.9 to 9,786 µg/L (n=25), representing three species from 16 studies. The lowest threshold was reported in juvenile red sea bream: a 28-day LOEC for growth and a 14 day NOEC for weight gain (Kim and Kang 2014). This suggests that sublethal responses or threshold mortality could occur in fish populations exposed under the selenium CCC of 71 µg/L, providing evidence that Nassau grouper could be adversely affected as a result of chronic exposures to nickel at or below the proposed CCC.

Invertebrate chronic exposure-response thresholds for exposures to selenium ranged from 199.6 to 9,980 µg/L (n=11), representing six species from four studies. The minimum threshold was a one-hour EC50 for filtration rate of brown mussel (Watling 1981). This suggests that sublethal responses or threshold mortality would not be expected to occur in invertebrate populations exposed under the selenium CCC of 71 µg/L, providing evidence that responses of ESA-listed exposed under the proposed CCC would be insignificant

Considering that existing monitoring data has not identified environmentally significant concentrations of selenium and there are no apparent point sources in the USVI (see section 1.1.1.1), NMFS expects that it is extremely unlikely that Nassau grouper and ESA-listed coral would be exposed to anthropogenic selenium in USVI waters. The EPA's approval of proposed criteria for a substance that is not likely to occur above natural background levels in waters affected by the VIWQS is therefore not likely to adversely affect these species. For this reason, the CMC and CCC for selenium will not be discussed further in this opinion.

Silver

The ECOTOX database did not include acute toxicity values for algae exposures to silver. The criteria document (USEPA 1987b) reported a four-day EC50 for the diatom, *Skeletonema costatum*, was 130 µg/L based on cell counts and 170 µg/L based on chlorophyll-a (USEPA 1978). The EC65 for decline in chlorophyll-a was 52 µg/L for the diatom *Thalassiosira pseudonana* and was 13 µg/L for the dinoflagellate *Gymnodinium splendens* (Wilson and Freeberg 1980), suggesting rate of chlorophyll-a decline of about ten percent could occur in algae exposed to silver under the CMC of 1.9 µg/L. This suggests some adverse effects could occur in coral zooxanthellae exposed under the CMC.

Fish LC50s for exposures to silver ranged from 4 to 54,400 µg/L (n=78), representing 13 species from 11 studies. The lowest reported LC50 was for four-day exposures of summer flounder larva (Lussier and Cardin 1985) and suggests that mortality rates of around 24 percent could occur in some fish populations exposed under the silver CMC concentration of 1.9 µg/L. Mortality rates ranging from one to 14 percent were indicated in 17 of the LC50s reported. While the data set provided evidence that exposure over the one-hour averaging period for the CMC would not result in mortality for some species, this inference cannot be extended the summer flounder data, which is taken as evidence that mortality could occur in Nassau grouper as a result of exposures to silver at or below the proposed CMC.

Invertebrate LC50s for exposures to silver ranged from 17 to 3,400 µg/L (n=20), representing eight species from 11 studies. The lowest LC50 was for a ten-day exposure of amphipod, *Ampelisca abdita* (Berry et al. 1999). This suggests that a mortality rate of 5.5 percent could occur in invertebrate populations exposed under the silver CMC of 1.9 µg/L. Two and four day average LC50s of 496 and 92 µg/L, respectively, for the calanoid copepod, *Acartia tonsa*, provides evidence that silver LC50s are influenced by exposure duration (Pedroso et al. 2007; Lussier and Cardin 1985). Taking this into consideration, mortality would not be expected to occur in invertebrate populations exposed to silver at the CMC over the one-hour averaging time, providing evidence that mortality in ESA-listed coral exposed under the CMC would be insignificant.

The proposed VIWQS did not propose a CCC for silver, so the implications of chronic exposures of ESA-listed species to silver are not evaluated in this opinion.

Because this consultation specifically addresses EPA approval of criteria used to regulate and assess water quality in the jurisdictional waters of the USVI, we are also able to consider the probability that ESA-listed species would be exposed to silver in these waters when making our determination.

Considering that existing monitoring data has not identified environmentally significant concentrations of silver and there are no apparent point sources in the USVI (see section 1.1.1.1), NMFS expects that it is extremely unlikely that Nassau grouper and ESA-listed coral would be exposed to anthropogenic silver in USVI waters. The EPA's approval of proposed criteria for a substance that is not likely to occur above natural background levels in waters affected by the VIWQS is therefore not likely to adversely affect these species. For this reason, the CMC and CCC for silver will not be discussed further in this opinion.

Zinc

Zinc is an essential trace mineral that is toxic in excess amounts. The ECOTOX database and EPA criteria documents did not include acute toxicity values for saltwater algae exposures to zinc.

Fish LC50s for exposures to zinc ranged from 25.54 to 170,280 µg/L (n=71), representing 28 species from 24 studies. The lowest LC50 was for a four-day exposure of sand flounder pre-larva (Cherkashin et al. 2004). This suggests that significant mortality could occur in fish populations exposed under the zinc CMC of 90 µg/L. The LC50s for thirteen other species suggested mortality ranging from one to 32 percent mortality could occur in fish under the proposed zinc CMC. The sand flounder data is taken as evidence that mortality could occur in Nassau grouper as a result of exposures to zinc at or below the proposed CMC.

Invertebrate LC50s for exposures to zinc ranged from 7.38 to 1,655,000 µg/L (n=367), representing 109 species from 91 studies. While lowest LC50 was for a 24-day exposure of the mysid *Acanthomysis costata* (Hunt et al. 1996), LC50s for 51 two-and four-day exposure were also below the proposed zinc CMC of 90 µg/L. This suggests significant mortality could occur in invertebrate populations exposed to zinc under the CMC.

Algae chronic exposure-response thresholds for exposures to zinc ranged from 17.97 to 21,758 µg/L (n=58), representing 17 species from 22 studies. This suggests that sublethal responses or threshold mortality could occur in algae populations exposed under the zinc CCC of 81 µg/L. The lowest reported threshold was a two day EC50 for population changes in the diatom, *Bellerochea polymorpha* (as cited in Walsh et al. 1988). Because this EC50 is less than the CCC, effects could occur in zooxanthellae exposed under the CCC.

Fish chronic exposure-response thresholds for exposures to zinc ranged from 95.55 to 9,460 µg/L (n=13), representing seven species from nine studies. This suggests that sublethal responses or threshold mortality would not be expected to occur in fish populations exposed under the zinc CCC of 81 µg/L.

Invertebrate chronic exposure-response thresholds for exposures to zinc ranged from 0.76 to 1,655,000 µg/L (n=621), representing 82 species from 111 studies. This suggests that sublethal responses or threshold mortality could occur in invertebrate populations exposed under the zinc CCC of 81 µg/L.

Considering that existing monitoring data has not identified environmentally significant concentrations of zinc and there are no apparent point sources in the USVI (see section 1.1.1.1), NMFS expects that it is extremely unlikely that Nassau grouper and ESA-listed coral would be exposed to anthropogenic zinc in USVI waters. The EPA's approval of proposed criteria for a substance that is not likely to occur above natural background levels in waters affected by the VIWQS is therefore not likely to adversely affect these species. For this reason, the CMC and CCC for zinc will not be discussed further in this opinion.

7.2.6 Ionizing Radiation: Gross Beta, Radium-226, And Strontium-90 (VIWQS §186-4)

Ionizing radiation results from the radioactive decay of unstable isotopes such as radium-226 and strontium-90 and subsequent release of electromagnetic radiation or subatomic particles that have sufficient energy to ionize the tissues or materials the radiation passes through. Ionization damages tissues by generating free radicals and breaking chemical bonds in the proteins, membranes, and other biomolecules as radiation passes through tissue. Free radicals are reactive molecules that damage tissues by oxidizing proteins, membranes, and nucleic acids. Symptoms of acute exposures resulting from accidents or extreme events are characterized by cell death due to massive radiation damage. Health effects of lower level exposures are classified as stochastic because, while the initiation of effects is probabilistic and proportional to the exposure, the actual severity of effects is independent of the radiation dose received because decay products interact with multiple targets resulting in a cascade of impacts and free radical generation (Hinton 1998).

A 2014 risk analysis for the effects of ionizing radiation on fish and wildlife reported that it is the unregulated, inadvertent releases that are of greatest concern (NYSDEC 2014). Among aquatic organisms, fish are the most sensitive to the effects of radiation, with developing fish embryos particularly vulnerable (UNSCEAR 1996). However, this comparison did not include coral and because radium-226 and strontium-90 are calcium analogs[5], corals may suffer development effects due to exposure to ionizing radiation.

The VIWQS include criteria for radiological activity expressed in units of picocuries per liter that are identical to those used by Virginia in 1972 for areas where shellfish occur (USEPA 1972) as noted in Section 3. The basis behind the criteria for gross beta particles, radium-226, and strontium-90 are not provided in the BE or other documentation submitted with the consultation request. It is not clear whether the radium-226 and strontium-90 criteria were originally established to protect human health or the propagation of shellfish (and thus protection of aquatic life). The gross beta criterion of 1,000 picocuries per liter, in the absence of strontium-

[5] Radium and strontium mimic calcium and can be incorporated into bony tissue (Norris and Kisielesk 1948).

90 and alpha emitters, is a drinking water standard for the protection of human health that was published in the Federal Register in 1962 (27 FR 44). The tolerance for effects in biota is higher than that for human health.

Responses of aquatic species to radiation exposures are reported in terms of total absorbed dose or dose per unit time (Grays, Gy or Gy/hour). The surface area of the body intercepting emitted radiation influences absorbed dose. Radiological activity may also be expressed as Becquerel (Bq), which corresponds to one disintegration per second. One Bq is equivalent to 27 picocuries per liter. Modeling by the International Atomic Energy Agency (IAEA 1992) provides internal and external absorbed dose estimates for fish (Table 9). Using these estimates, the expected internal and external absorbed doses for radium-226 under the proposed criteria would be 3.39×10^{-1} and 8.54×10^{-7} micrograys per hour (µGy/hr), respectively and the strontium-90 expected internal and external absorbed doses would be 1.15×10^{-3} and 1.15×10^{-4} µGy/hour, respectively.

Table 9. Expected absorbed radiation estimates under the proposed criteria for radium-226 and strontium-90

	Absorbed Dose (µGy/hour) per Bq/L		VIWQS Criteria in Bq/L	Expected Absorbed Dose at proposed criteria (µGy/hour)	
Stressor	**Internal**	**External**		**Internal**	**External**
Radium-226	3.1	7.8×10^{-6}	0.11	3.39×10^{-1}	8.54×10^{-7}
Strontium-90	3.1×10^{-3}	3.1×10^{-4}	0.37	1.15×10^{-3}	1.15×10^{-4}

Adverse effects are reported in saltwater species at doses that are many orders of magnitude greater than expected absorbed doses for fish under the VIWQS criteria. Doses of three to 25 µGy/hour did not result in adverse effects in plaice, a saltwater fish. Invertebrates are much less sensitive to ionizing radiation than fish. Doses ranging from 2,100 to 73,,000 µGy/hour did not result in adverse effects in blue crabs, scallops, or polychaetes (UNSCEAR 2008;1996; NYSDEC 2014; IAEA 1992).

Predicted no effect dose rate values for generic ecosystems exposed to radioactive substances also suggest the VIWQS criteria (3 picocuries per liter for radium-226 and 10 picocuries per liter for strontium-90) are orders of magnitude below levels at which effects could occur (Agüero et al. 2008). The absorbed hazard dose rate at which 95 percent of species are expected to be unaffected is estimated to be between 23.8 and 336 µGy/hour (Agüero et al. 2008). While species' sensitivity distributions describe the potential for effects on ecosystem structure and do not ensure protection of vulnerable species (see discussion in Section 2.1), the lower confidence interval at the HDR_5 (hazardous dose rate), 23.8 µGy/hour, is about 70 times that of the highest expected absorbed dose rate under the VIWQS criteria for radiological activity.

Gross beta absorbed dose rates under the VIWQS criteria cannot be calculated using radionuclide-specific dose factors because there are many beta emitters that could contribute to the absorbed dose. However, when measuring gross beta, instrumentation is calibrated using cesium-137 and cesium-137 (IAEA 1992). Using cesium-137 as a surrogate estimate of gross beta exposure under the VIWQS criterion (1,000 picocuries per liter in the absence of strontium-90 and alpha emitters), internal and external absorbed doses would be 5.1 and 2.1×10^{-3} µGy/hour, respectively. These exposures are also far lower than the absorbed dose where adverse effects have been reported in saltwater fish.

Given the potential internal and external absorbed doses of ionizing radiation based on the allowed criteria under the VIWQS in comparison to the absorbed dose rates where adverse effects have been reported in saltwater fish, blue crabs, scallops, or polychaetes, we believe the criteria for radiological activity will have no effect on ESA-listed species and do not consider radiological activity further in this consultation.

7.2.7 Water Quality Parameters: Dissolved Oxygen, Temperature, pH, Alkalinity Secchi Depth, Turbidity

7.2.7.1 Dissolved Oxygen

Inputs from terrestrial areas that contribute to decreases in DO, including inputs of sewage, can lead to hypoxia. In a laboratory experiment with larval seven-band grouper, varying temperature regimes and disinfection with ozonated seawater had no effect on the frequency of abnormality in developing larvae but exposure to low DO concentrations had a significant effect on the incidence of abnormalities during both the early and mid-somite stages (Uji et al. 2015). Exposure to hypoxia at environmentally realistic levels (14 and 22 percent saturation versus 97 percent saturation in natural seawater) during somitogenesis led to abnormal muscle development in the trunk of seven-band grouper larvae (Uji et al. 2015). Because somites give rise to skeletal muscle, vertebrae, and cartilage, abnormal development due to hypoxia can affect survival. DO levels also influence the distribution and availability of prey species. Prey species for species such as Nassau grouper are expected to move out of areas with low DO but the DO criteria of 5.0 mg/L in the VIWQS is not likely to result in any prey redistribution.

A study of hypoxia in a semi-enclosed coastal bay in Panama found that hypoxic conditions were in the bottom layer of the stratified water column, leading to mortality of sponges, corals, crustaceans, gastropods, and echinoderms, thick mats of bacteria, and an exclusion of consumers that would normally eat dead organisms below a certain depth (Altieri et al. 2017). Oxygen levels below 0.5 mg/L were observed at bottom sites while all areas with depths of less than 4 m had DO concentrations above 4.8 mg/L (Altieri et al. 2017). The most hypoxic waters were those with greater depths near land where terrestrial inputs were expected to be greater and included untreated sewage and where exchange with the open ocean was poor. Altieri et al. (2017) noted that, although the event caused mass bleaching and mortality of corals and other organisms, laboratory experiments showed that not all coral species were as sensitive to hypoxia. Hypoxia was also found to play an important role in coral tissue loss during coral-algae interaction processes (Haas et al. 2014). Haas et al. (2014) found that algae were significantly more tolerant to low oxygen concentrations (2 to 4 mg/L) than corals and that corals could only tolerate reduced oxygen concentrations only until a certain combination of exposure time and concentration occurred. The DO criteria of 5.5 mg/L in Class A and B waters and 5 mg/L in Class C waters is higher than the concentrations in the studies discussed above at which hypoxia and other effects to fish, prey species, invertebrates, corals, and algae were reported. Therefore, we expect the proposed DO criteria will not have adverse effects to ESA-listed species and do not consider DO further in this opinion.

7.2.7.2 Temperature

Water temperature influences hatchling rates of fish eggs, larval sizes at hatch, time for yolk sac absorption, energy reserve storage efficiency, and larval growth and survival. In a laboratory study of leopard grouper, the highest hatch rate occurred for eggs incubated in temperatures between 24 and 30°C, which is similar to that observed in some island areas in Mexico where leopard grouper broodstock have been found (Gracia-López et al. 2004). Gracia-López et al. (2004) also documented a tendency toward lower hatch rates at temperatures different from the 26.3°C optimum and an inverse relationship between temperature and hatchling period duration with embryonic development being faster as temperature increased but larval size being greater at lower temperatures. Higher temperatures were also found to have faster growth rates, a shorter yolk absorption period but lowered energy designated for tissue growth, and no apparent change in survival rate (Gracia-López et al. 2004). Juvenile Nassau grouper demonstrated higher growth rates with increasing temperatures (up to 31°C) associated with increases in feeding rates (Ellis et al. 1997). Temperature was found to influence survival in an experiment simulating responses to fishing on the part of adult coral grouper. Fish that reacted strongly to the fishing simulation by vigorously swimming away were found to have lower survival rates, particularly when temperatures were 30°C or more (Clark et al. 2017). Fish were also found to suffer mortality faster at a higher temperature (33oC; Clark et al. 2017). Similarly, Messmer et al. (2017) found that larger individuals of leopard coral grouper were more thermally sensitive than smaller conspecifics apparently due to a restricted capacity on the part of large fish to increase mass-specific maximum metabolic rate at higher temperatures (33°C compared with 28.5°C). Thus, increasing temperatures may lead to declines in adult body size, though increasing temperatures may also lead to faster larval and juvenile development. For fish that spawn in deep water areas such as groupers, eggs typically hatch offshore and then larvae move inshore. The distance from the shore to areas where spawning occurs, and the volume of water enables mixing and reduces temperatures of thermal plumes originating from shoreline and nearshore discharges. In addition, deeper waters tend to be cooler except during events such as prolonged periods of elevated sea surface temperatures.

For sea turtles, temperature regime increases generally lead to female-biased nests (Hill et al. 2015). Acevedo-Whitehouse and Duffus (2009) proposed that the rapidity of environmental changes, such as those resulting from global warming, could harm immunocompetence and reproductive parameters in wildlife to the detriment of population viability and persistence. An example of this is the altered sex ratios observed in sea turtle populations worldwide (Mazaris et al. 2008; Reina et al. 2009; Robinson et al. 2009; Fuentes et al. 2010). This does not appear to have yet affected population viabilities through reduced reproductive success, although nesting and emergence dates of days to weeks in some locations have changed over the past several decades (Poloczanska et al. 2009). Altered ranges can also result in the spread of novel diseases to new areas via shifts in host ranges (Simmonds and Eliott 2009; Schumann et al. 2013).

In terms of in-water temperatures, wide-ranging species that inhabit waters of varying temperatures such as sea turtles, marine mammals, and pelagic fish tolerate a wide range of temperatures. However, recent cold stunning events such as in Florida during winter when drops in water temperatures have led to stranding of various sea turtle species indicate that sea turtles have a temperature tolerance range. Water temperatures also influence the distribution and availability of foraging habitat and prey for wide-ranging and resident species.

Reef corals tend to thrive in areas with mean temperatures within a narrow range (typically 25-30^{o}C). Short-term exposures (days) to temperature increases of a few degrees (i.e., 3-4^{o}C increase above climatological mean maximum summer temperature) or long-term exposures (several weeks) to minor temperature increases (i.e., 1-2^{o}C above mean maximum summer temperature) can cause significant thermal stress in the form of bleaching and mortality in most coral species (Berkelmans and Willis 1999; Jokiel and Coles 1990; Fitt and Warner 1995). Such temperature thresholds are variable in time (e.g., season) and geographic location and may be nonlinear. Even in areas where corals have adapted to extremely high summer (and low winter) temperatures such as in the Arabian Gulf and Northwestern Hawaiian Islands, corals bleach when their normal maximum and minimum temperature tolerances are exceeded (Riegl 2002; Hoeke et al. 2006; Brainard et al. 2011). High temperatures are a significant cause of coral bleaching. Coral can withstand mild to moderate bleaching but severe, repeated, or prolonged bleaching can lead to colony mortality. On the Florida Reef Tract, the prevalence of black band disease, which is considered a major contributor to the decline of reef-building coral species in Florida and elsewhere, was found to increase significantly when water temperatures exceeded 29^{o}C (Lewis et al. 2017). Black band disease in 2014 and 2015 appeared immediately following bleaching events driven by high sea surface temperatures (Lewis et al. 2017). A similar pattern of bleaching due to high temperatures followed by disease outbreaks was documented in USVI, particularly following the 2005 Caribbean coral bleaching event (Miller et al. 2009).

The temperature criteria for areas containing coral reef ecosystems and non-coral areas in the 2018 VIWQS (Section 186-4) and the thermal policy (Section 186-6) are within the tolerance ranges reported for ESA-listed species. Therefore, we expect the exposure of ESA-listed species and their habitat to the proposed temperature criteria and thermal policy will not result in adverse effects to ESA-listed species and their habitat and we do not consider the proposed temperature criteria and thermal policy further in this opinion.

7.2.7.3 pH

The pH of water affects the normal physiological functions of aquatic organisms, including the exchange of ions with the water and respiration. Coastal water pH varies with salinity. Beyond a certain limit, changes in pH do not continue to evoke a response in marine organisms as much as changes in salinity. Seawater pH decreases with decreasing salinity, meaning that the effects of acidification can be more significant locally where there are riverine inputs, for example.

In response to acid rain problems occurring in the eastern United States, the physiological effects of acid stress on fish and other aquatic life have been well documented. A number of researchers have proposed that the toxic action of hydrogen ions on fish under acidic conditions involves production of mucus on the gill epithelium, which interferes with the exchange of respiratory gasses and ions across the gill; precipitation of proteins within the epithelial cells; and/or acidosis of the blood (also affecting oxygen uptake; Ellis 1937; Westfall 1945; AFS 1979; Boyd 1990). Hence, respiratory distress and osmotic imbalance are the primary physiological symptoms of acid stress in fish. In addition, primary productivity of freshwater aquatic ecosystems is reduced considerably below pH of 5.0, which in turn, reduces the food supply for higher organisms. Thus, fish that remain present would likely experience reduced numbers or growth rates (Alabaster and Lloyd 1980).

The physiological effects on aquatic life induced by high pH (>9) have been studied less than those at low pH. This is likely because high pH waters are less common (Doudoroff and Katz 1950; Alabaster and Lloyd 1980). Several researchers concluded that the toxic mode of action of hydroxyl ions (i.e., high pH values) is hypertrophy of mucus cells at the base of the gill filaments and destruction of gill and skin epithelium, with effects on the eye lens and cornea (Alabaster and Lloyd 1980; Boyd 1990).

Increases in ocean acidification can be more drastic in local areas where there are influxes of freshwater, leading to reduced calcification of corals due to decreases in pH but also in salinity. As pH declines, the linear extension rate and skeletal density of corals decrease (Hoegh-Guldberg et al. 2007), leading not only to impacts to individual colonies but to the value of coral areas as habitat for other species. Decreasing pH also leads to bleaching as corals experience stress and increasing acidification can override any acclimatization to thermal stress (Anthony et al. 2008). Studies in both the Caribbean (Crook et al. 2012) and the Pacific (Fabricius et al. 2011) have found that very few hard corals can survive and grow below a pH of 7.7 and aragonite saturation of 2.9. Corals from the genus Porites were still found in these conditions and *Siderastrea radians* (Caribbean) but at low abundance and typically only when nutrient concentrations were high (Crook et al. 2012). Major reef-building corals did not tolerate these conditions regardless of nutrient input. The size of coral colonies, even in tolerant species, declined as pH became more acidic, likely because growth slowed under more acidic conditions. Similar to corals, research has shown that the percent cover of several sponge species declines significantly with increases in acidification and that the community composition of sponges shifts as these bioerode under acidic conditions and other species become more common

(Goodwin et al. 2014).The 2018 VIWQS require that pH be maintained between 7 and 8.3, meaning waters tending toward neutral (7) or slightly basic (8.3). Because these pH values may result in impacts to prey species, habitat, and ESA-listed species such as corals, the proposed pH criteria are discussed further in this opinion.

7.2.7.4 Secchi Depth

The zone in which photosynthesis can occur in coastal and marine waters is restricted to a relatively thin layer from the water surface to the depth in which 1 percent of the photosynthetically active radiation (PAR) entering the water remains (the euphotic zone). A coarse way to evaluate underwater light penetration is to use a secchi disc rather than an instrument to measure PAR. Secchi depth is a visual measure of water transparency. Secchi depth varies with changes in the optical properties of water because light is attenuated by suspended solids, colored dissolved organic matter, and chlorophyll and photosynthetic pigments of phytoplankton (Luhtala and Tolvanen 2013). Reduced water clarity can lead to reduced coral biodiversity and increased macroalgal cover, community shifts toward heterotrophic filter feeders, and a proliferation of filter-feeding bioeroding organisms. Prolonged shading also limits the depth distribution of corals and seagrass (Fabricius et al. 2014). River discharges to shallow nearshore waters strongly affect water clarity. Fabricius et al. (2014) found an annual mean photic depth reduction of 20 percent across the shelf of the Great Barrier Reef in wet years, which is a significant loss of light for photosynthetic organisms such as corals and seagrass. The effects of light on growth and survival of early life stages of marine fish vary significantly among species with Nassau grouper, striped bass, and Atlantic cod demonstrating increased feeding, survival, and growth of larvae under increased light intensities (Copeland and Watanabe 2006).

The numeric criteria for secchi depth in the 2018 VIWQS require that the secchi disc be visible at a minimum depth of 1 m or the bottom be visible in water depths less than 1 m. In areas containing coral reef ecosystems, the criteria for secchi depth is that it be visible at 15 m or the bottom be visible if depth is less than 15 m. We expect these criteria to allow adequate light penetration to maintain the existing depth distributions of benthic habitat such as coral and seagrass in the USVI and to not inhibit development, growth and other functions of various life stages of fish and other marine organisms. Therefore, we do not expect the secchi depth criteria to result in adverse effects to ESA-listed species or their habitat and we do not consider secchi depth further in this opinion.

7.2.7.5 Turbidity

Erosion is a natural occurrence in aquatic systems where the flow or movement of water scours loose sediment from stream banks and shorelines. However, stream bank erosion can be exacerbated by anthropogenic sources which disturb soils or alter hydrology (e.g., logging, construction, paving) and is commonly correlated with urbanization and high percentages of impervious surfaces in watersheds. Impervious surfaces in a watershed increase the natural flow and volume of water during rain events causing increased scouring and sediment transport potential. Stormwater erosion directly contributes to spikes in total suspended solids, turbidity, and nutrient concentrations in the water column, as well as causing indirect water chemistry changes.

During high flow events, eroded sediment is transported as suspended solids until it reaches low flow areas where it settles out of solution and sinks to the bottom. Excessive sediment loads introduced into receiving waterbodies can pose significant environmental health risks; sediment can cause smothering and disruption of aquatic habitats, reduce light penetration and transport many other potentially harmful pollutants (e.g. hydrocarbons, heavy metals, phosphorus) (Duncan et al. 1999). Direct effects of suspended materials on invertebrates and fish are complex, ranging from behavioral to physiological to toxicological. Suspended sediments have been documented to have a negative effect on the survival of fish and benthic organisms. In a frequently cited review paper prepared by Newcombe and Jensen (1996), sublethal effects (e.g. increased respiration rate) were observed in eggs and larvae of fish when exposed to total suspended solids (TSS) concentrations as low as 55 mg/L for one hour. Increased turbidity associated with suspended sediments can reduce primary productivity of algae as well as growth and reproduction of submerged vegetation (Jha and Swietlik 2003). In addition, once in the system, resuspension and deposition can "recycle" sediments so that they exert water column and benthic effects repeatedly over time and in multiple locations.

Studies have found that survival of coral recruits in warmer waters (water temperatures of 30^{o}C) is better if fine sediments are less than 6.55 NTU (Fourney and Figueiredo 2017); coral disease increases by orders of magnitude in reefs exposed to sediment plumes (Pollock et al. 2014); undisturbed reefs consistently have turbidity values less than 1 NTU (Fourney and Figueiredo 2017); and high sediment concentrations affect reproduction (Jones et al. 2015).

Observed impacts to ESA-listed species from elevated sediment and turbidity levels fall into several broad categories such as avoidance or behavioral responses, feeding and hunting, breeding and egg survival, habitat loss, juvenile survival and physical damage. The potential cumulative effect of these impacts includes reduced disease and parasite resistance, reduced growth, and degraded health of individual organisms. Population reductions can take place both through direct mortality in the short term and reduced reproductive success in the long term. Undisturbed reef areas often have turbidity levels of less than 1 NTUs and the data gaps and scale of existing benthic maps in USVI mean that areas containing ESA-listed corals and habitat for other listed species may be present in areas where the 3 NTU "non-coral" turbidity value may be applied, resulting in impacts to habitat or the species themselves. Therefore, the potential effects of the "non-coral" (3 NTU) turbidity criterion on ESA-listed species and their habitat are discussed further in this opinion.

7.2.8 Fecal Bacteria (VIWQS §186-4)

EPA recommended the use of the fecal indicator bacteria *Escherichia coli* and enterococci to determine the level of fecal contamination in waters and establish the 2012 Recreational Water Quality Criteria protecting waters designated for primary contact recreational use (EPA 2015).

The VIWQS include fecal indicator bacteria as a standard for potable water and enterococci as an ambient standard including in marine waters. As discussed previously in this opinion, EPA expressed its position that the ESA section 7 consultation request did not include criteria adopted to protect human health. However, we consider the potential effects of the enterococci standard in this opinion because of the ESA requirements to analyze all effects of the action on our species and existing scientific studies indicate fecal contamination from human sewage discharge to marine waters negatively affects corals.

It has been suggested that enterococci bacteria may not be reliable indicators in tropical and subtropical environments, including coastal waters, because of the ability of these bacteria to replicate in water and sediment and be held in sediment reservoirs (Bonkosky et al. 2009; Lee et al. 2006; Phillips et al. 2011). In addition, other animals also contribute to fecal contamination of coastal waters, meaning that high concentrations of these bacteria may not be due to human sewage contamination. In some areas with large seabird colonies, for example, high concentrations of fecal indicator bacteria are due to the rookeries, as indicated by tests that use microbial source tracking (MST) methods. MST is being used to identify the source of fecal contamination with molecular markers that can amplify DNA sequences from potential microbial fecal indicators (Bonkosky et al. 2009).

Human sewage has been identified as the likely source for the strain of *Serratia marcescens* which caused outbreaks of white pox disease in threatened elkhorn coral in the Caribbean (Sutherland et al. 2010). Diseases such as white pox are listed as major contributors to the decline of elkhorn coral. Discharges represent more direct and undiluted exposure “pulses” and expose elkhorn to *Serratia marcescens* and cause an outbreak of whitepox disease.

Marine mammals can serve as hosts and become infected by bacterial pathogens associated with sewage (Thompson et al. 2005; Venn-Watson et al. 2010; Grillo 2001). At this time, the impact of sewage-associated bacteria found in marine mammals is uncertain because the role of infectious disease has not been traditionally investigated in cetacean population health assessments. When pathogens are detected in tissues, it is often unknown if the pathogen was associated with a chronic or acute infection, underlying or immediate cause of death, or simply an incidental finding (Venn-Watson et al. 2010). In general, small, isolated cetacean populations with limited gene pools and populations with nutritional challenges are expected to be at higher risk of infectious disease host susceptibility and disease transmission (Venn-Watson et al. 2010).

While marine turtles are known to harbor human pathogens such as *Pseudomonas*, *Mycobacterium*, *Salmonella*, *Vibrio*, and *Chlamydia*, these pathogens are typically opportunistic organisms which are found throughout the environment and in a wide range of vertebrate species (Glazebrook and Campbell 1990). While discharges containing sewage pathogens can enhance the risk of exposures resulting in infection, a causal link has not been established between exposure to human sewage pathogens and adverse effects in marine turtles. While several ubiquitous pathogens are found in both humans and fish, those associated with sewage are not known to adversely affect marine and anadromous fish.

At this time, NMFS does not have a good understanding of the potential impact of human sewage indicator pathogens on the population health of marine mammals, sea turtles, and fish. Understanding is complicated by our ability to discriminate between primary infections and opportunistic infections in wounded or weakened animals. However, based on available information, we expect the ambient standard for human fecal bacteria may result in adverse effects to ESA-listed corals. Therefore, in Section 9 we consider the effects of exposure to the human fecal bacteria criteria only on ESA-listed corals. The effects of exposure to fecal bacteria criteria on other ESA-listed species are not considered further in this opinion.

7.3 Indirect Stressors (VIWQS §186-4)

Nitrogen (N) and phosphorous (P) are indirect stressors because they stimulate eutrophication and stressors associated with excessive plant and algal growth: extremes in DO, reduced light penetration, smothering, and algal toxins when present in high concentrations. Under natural conditions, essential nutrients contribute to the proper structure and function of ecosystems. However, in excessive quantities, nutrients can have adverse effects on ecosystems and rank as one of the top causes of water resource impairment (Bricker et al. 2008; USEPA 2014).

Eutrophication alters the composition and species diversity of aquatic communities through intensifying competition by those species, native or invasive, that are better adapted to eutrophic environments (Nordin 1985; Welch et al. 1988; Carpenter et al. 1998; Smith 1998; Smith et al. 1999). Eutrophication can have cascading effects that change ecosystem structure at numerous trophic levels. Nuisance levels of algae and other aquatic vegetation (macrophytes) can develop rapidly in freshwater and marine habitats in response to nutrient enrichment when other factors (e.g., light, temperature) are not limiting. The relationship between nuisance algal growth and nutrient enrichment is well-documented (Welch et al. 1992; Dodds et al. 1997; Chetelat et al. 1999; Vannieuwenhuyse and Jones 1996). Increases in nutrient loading to nearshore waters that increase algal and phytoplankton growth, affect coral habitat function related to coral settlement and growth.

In addition to outcompeting native aquatic plants and sessile organisms such as corals for space and light, the proliferation of nuisance algae can lead to harmful algal blooms (e.g., brown tides, toxic *Pfiesteria piscida* outbreaks, some types of red tides) that contain potent toxins from microalgae. In marine systems, algal toxins have caused massive fish kills, along with deaths of whales, sea lions, dolphins, manatees, sea turtles, birds, and wild and cultured fish and invertebrates (Landsberg 2002; Shumway et al. 2003). Eutrophication is believed to be a likely contributor to the increased occurrence of harmful algal blooms (Heisler et al. 2008). Eutrophication has also been linked to increases in bacteria biomass (Carr et al. 2005). Bacteria have been associated with mortality of fish, turtles, and alligators (Shotts et al. 1972). Harmful algal blooms are not common in the USVI but there may be a relationship between pulses of nutrients and blooms of other algal species and cyanobacteria in coral and seagrass habitats in summer months. Van Houtan et al. (2014) found that green sea turtles were more likely to develop fibropapilloma (FP) tumors due to the concentrations of arginine nitrogen the animals were consuming in macroalgae from eutrophied waters. In areas with chronic high nutrient inputs in the Florida Keys, reef coral die-off occurred and was attributed to microbial diseases responding to high nutrient inputs (Lapointe et al. 2004).

The accumulation of algal biomass through excessive productivity can reduce available habitat, and the decay of this organic matter may lead to reductions in DO in the water column, which in turn can cause problems such as fish kills and the release of toxic substances or phosphates that were previously bound to oxidized sediments (Chorus and Bartram 1999). Because hypoxia, defined as concentrations below 0.2 ml/L (Kamykowski and Zentara 1990), often occurs in estuaries or nearshore areas where the water is poorly mixed, nursery habitat for fish and shellfish is often affected. Without nursery grounds, the young animals cannot find the food or habitat they need to reach adulthood. This causes years of weak recruitment to adult populations and can result in an overall reduction or destabilization of important stocks. High biomass blooms can also directly inhibit growth of beneficial vegetation by blocking sunlight penetration into the water column (Onuf 1996). Macroalgal blooms reduce sunlight penetration and can overgrow or displace seagrasses and corals as well as foul beaches (Valiela et al. 1997). Bloom-inflicted mortalities can degrade habitat quality indirectly through altered food webs or hypoxic events caused by the decay of dead animals (Lopez et al. 2008). Tew et al. (2013) found a 16:1 ratio of N:P to be ideal in allowing for the development of adequate phytoplankton to then nourish the zooplankton community that served as prey for larval and early stage juvenile grouper.

In addition, direct toxic effects can occur in waters with elevated nitrogen in the form of ammonia, with the more toxic form, ammonium, more prevalent at high pH. In addition to the potential for ammonia toxicity under high nutrient loadings, toxins may be produced by some algae that thrive in eutrophic conditions. Accumulation and increased turnover of algal and plant biomass (i.e., death, decay, nutrient release) generates suspended solids in the form of organic particulates and phytoplankton, contributing to turbidity from natural and human-caused erosion and sediment resuspension. Increased turbidity affects light penetration into the water column and the ability of aquatic plants to photosynthesize and survive and the effectiveness of sight-dependent behaviors such as foraging, reproductive displays, and predator evasion. Decreased light penetration affects the degree of coverage of the substrate by plants and benthic organisms that are reliant on plants. Suspended organic matter can eventually accumulate on and smother plants, animals, and benthic habitats. Increases in plant and microbial biomass or productivity may result in negative ecological effects by:

- altering food resources: the amount and type of food resources or their palatability (e.g., changes in algal cell size affects filter-feeding animals);
- increased microbial infection of invertebrates or fish;
- altering habitat: light penetration, diurnal DO cycle, changes in benthic interstitial space, availability of macrophytes as habitat;

- stimulating generation of toxins: some algae that thrive in eutrophic conditions can be toxic to fish and invertebrates;
- increasing mortality through favoring nitrogen pathways increasing the formation of toxic unionized ammonia; and
- changes in community structure, even without overall increases in primary producers, due to alterations to nutrient availability ratios.

The proposed criterion for TN is in keeping with values from other geographies such as Florida and Hawai'i and was set based on studies from USVI. Therefore, we do not expect the TN criterion to result in adverse impacts to ESA-listed species or their habitats. However, because of the existing criterion for TP and the relationship between N and P, there could be adverse effects to ESA-listed species and their habitat associated with the existing TP criteria and the ratio of N:P that will result from the proposed criteria. For this reason, TP and the N:P ratio that will result from the proposed TN criterion are discussed further in this opinion.

7.4 Summary: Stressors Considered Further in this Opinion

In summary, the following stressors are considered further in this opinion because, as discussed above, the criteria for these pesticides, metals, nutrients, and other stressors, may result in adverse effects to ESA-listed species exposed to the stressors:

- CMC for carbaryl
- CMC and CCC for nonylphenol
- CCC for copper
- pH
- "Non-coral" turbidity (3 NTU)
- Enterococci
- TP
- N:P ratio resulting from the proposed TN criterion

The potential effects of these stressors on ESA-listed corals, green and hawksbill sea turtles, Nassau grouper, and elkhorn and staghorn coral critical habitat are discussed in Section 9.

8 ENVIRONMENTAL BASELINE

The "environmental baseline" includes the past and present impacts of all Federal, state, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 C.F.R. §402.02).

The environmental baseline for this Opinion includes several activities that affect the survival and recovery of green, leatherback, and hawksbill sea turtles; Nassau grouper; scalloped hammerhead sharks; elkhorn, staghorn, rough cactus, pillar, lobed star, mountainous star, and boulder star corals; and the ability of designated elkhorn and staghorn coral critical habitat in the action area to support its intended conservation function. We describe these activities below.

8.1 Fisheries

ESA-Listed Sea Turtles and Fish

There are federally managed fisheries that operate in federal waters from three nautical miles from shore for USVI out to the limits of the Exclusive Economic Zone (EEZ). Threatened and endangered sea turtles are adversely affected by fishing gears used throughout the insular shelf in the action area. Nassau grouper and scalloped hammerhead sharks may also be adversely affected by fishing gear and these animals have also been fisheries targets. There are also commercial and recreational fisheries in Commonwealth and Territorial waters that are regulated by the VIDPNR. 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 (VIDPNR unpublished data, Sea Turtle Stranding Network). Entanglement in nets, trap lines, and fishing line accounted for 27 percent of reported sea turtle strandings around St. Croix for the period from 1982-2010 with 43 percent of the turtles entangled in line being greens and 48 percent hawksbills (VIDPNR stranding data). Fewer data were available from St. Thomas and St. John, but they reflect similar trends with 40 percent of strandings caused by entanglement in fishing gear in St. Thomas (of which 88 percent were greens and 12 percent were hawksbills) and 22 percent in St. John (of which 100 percent were greens; VIDPNR stranding data). The USVI TCRMP found derelict fishing gear in the some of the reefs surveyed as part of the program and indications of fishing pressure at some permanent monitoring sites (Smith et al. 2011). Abandoned or lost fishing gear can also affect the quality of refuge and foraging habitat for green and hawksbill sea turtles and Nassau grouper as abandoned gear can lead to abrasion and breakage in hard bottom and coral reef habitats and have shading impacts on seagrass and macroalgae if the gear is large enough such as traps and nets.

Nassau grouper were an important component of the fishery and were targeted in federal and Territorial fisheries until fishing was prohibited (in federal waters in 1990 and in Territorial waters in 2006). Fishing in Territorial waters occasionally targeted juveniles in nearshore areas in addition to adults. As the fishery became more diminished, younger life stages were targeted, leading to the prohibition of fishing for this species year-round in federal and Territorial waters.

For all fisheries for which there is a Fishery Management Plan (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 likely to adversely affect, but not likely to jeopardize the continued existence, of sea turtle species. Formal Section 7 consultations have been conducted on the following fisheries occurring in the action area and found fisheries actions to be likely to adversely affect threatened and endangered sea turtles: Caribbean Reef Fish and Caribbean Spiny Lobster FMPs under the jurisdiction of the CFMC. Anticipated levels of take associated with these actions reflect the impact on sea turtles and other listed species of each activity anticipated from the date of the ITS forward in time in the waters of the EEZ 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 three years, 51 lethal takes of hawksbill sea turtles with no more than three non-lethal takes over three years, and 48 lethal takes of leatherback sea turtles over three years. No take of loggerhead sea turtles under this FMP is anticipated due to the scarcity of this species in the U.S. Caribbean. Anticipated levels of take under the Spiny Lobster FMP are 12 lethal takes of green and hawksbill sea turtles over three years and 9 lethal takes of leatherback sea turtles over three years. Informal Section 7 consultations were also completed for the Caribbean Coral and Queen Conch FMPs. NMFS concluded that the implementation of the Coral and Queen Conch FMPs is not likely to adversely affect ESA-listed sea turtles or Nassau grouper. A section 7 consultation is ongoing for the Caribbean Reef Fish FMP to include consideration of the possible effects on Nassau grouper because this species could be captured as bycatch.

The Southeast Region also has established anticipated levels of take for HMS fisheries. Anticipated levels of take under the Coastal Migratory Pelagics FMP are 2 lethal takes for leatherbacks and hawksbills and 14 lethal takes of greens over three years; under the Dolphin-Wahoo FMP, 12 leatherback takes with no more than 1 lethal and up to three green or hawksbill takes with no more than 1 lethal over 1 year; under the HMS-Pelagic Longline FMP, 1,764 leatherback takes with no more than 252 lethal and 105 green and/or hawksbill takes with no more than 18 lethal over three years; and under the HMS-Shark Fisheries FMP, 18 leatherback takes with no more than 9 lethal, 57 green takes with no more than 33 lethal, and 18 hawksbill takes with no more than 9 lethal.

ESA-Listed Corals and Elkhorn and Staghorn Coral Critical Habitat

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 TCRMP. 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. 2011). Fishing pressure measured by the number of registered commercial fisherman versus shelf areas with less than 64 m depths is approximate 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. 2011). 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 proposed Amalago Bay project over the course of the TCRMP leading to impacts to the shelf edge reef (Smith et al. 2011).

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 and proposed corals and their habitat are 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 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 elkhorn and staghorn 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 has also completed Opinions for the Reef Fish and Spiny Lobster FMPs as part of Section 7 consultations to consider the potential impacts of the fisheries on elkhorn and staghorn corals and their designated critical habitat. These opinions determined that these fisheries would adversely affect but not jeopardize elkhorn and staghorn corals and would adversely affect but not destroy or adversely modify their critical habitat. The adverse effects include: breakage and abrasion from trap use, nets, vessel anchoring, and fishing line and increased algal cover due to the loss of herbivorous fish. NMFS believes that the effects from these federal fisheries to star coral species would be similar to those described for elkhorn and staghorn corals. As noted above, many of the intense fishing activities are also on going in shallow waters that are within the jurisdiction of the Territory. These activities are mainly net and spearfishing. Net fishing in particular can have detrimental effects on corals and coral habitat as noted above.

8.2 Vessel Operations and Traffic

ESA-Listed Sea Turtles and Fish

Potential sources of adverse effects to hawksbill, green, and leatherback sea turtles from federal vessel operations in the action area include operations of the U.S. Coast Guard (USCG) and NOAA. NMFS and the USCG completed a programmatic consultation for the USCG's Aids-to-Navigation (ATON) 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 concluded that ATON maintenance activities were not likely to adversely affect sea turtles. NMFS is currently working on a national programmatic consultation that will determine the magnitude of the adverse impacts resulting from all ATON maintenance nationwide, including those in the U.S. Caribbean. Various NOAA programs are also working with OPR to ensure their vessel and research operations are in compliance with the ESA.

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 sea turtles.

Adverse effects to Nassau grouper from federal vessel operations could occur if vessels operate in spawning aggregation sites during spawning if the vessels have seawater intakes. Larvae of Nassau grouper could be affected by entrainment or impingement depending on the size, flow rate, and location of the seawater intake. However, there are few federal vessels of a size large enough to require constant seawater intake to cool the engines (such as NOAA vessels and U.S. Coast Guard cutters) and none of these vessels are stationed in the USVI or have frequent transit routes through USVI waters.

Commercial and recreational vessel traffic can have adverse effects on sea turtles via propeller and boat-strike injuries. NMFS and the USCG completed an informal section 7 consultation for the Caribbean Marine Event Program in 2009 for 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. A programmatic consultation was completed with the USCG for their Caribbean Marine Event Program (FPR-2017-9233) that included all activities that may be covered by the USCG under the program. The programmatic consultation concluded that the continuation of the Marine Events Program in the U.S. Caribbean was not likely to adversely affect ESA-listed species and their habitat, including green and hawksbill sea turtles and Nassau grouper.

Stranding data reported 77 sea turtles (loggerhead, leatherback, green, and hawksbill) around St. Croix from 2001-2010 (VIDPNR, unpublished data, Sea Turtle Stranding Network). Of these, 4 green, 2 leatherback, and 1 unknown species of sea turtle could be confirmed to have been impacted by boats (VIDPNR unpublished data, Sea Turtle Stranding Network). Thus, approximately 9 percent 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 (VIDPNR unpublished data, Sea Turtle Stranding Network). Similarly, 22 percent of the reported strandings around St. John and 25 percent 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 (VIDPNR unpublished data, Sea Turtle Stranding network). 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.

Vessel operation and the associated proliferation of docks and other boating facilities have resulted in the loss or degradation of refuge and foraging habitat, for green and hawksbill sea turtles and Nassau grouper due to impacts to seagrass and coral habitats from propeller scarring, propeller wash, accidental groundings, and in-water construction. Coastal runoff, marina and dock construction, dredging, industrial operations, increased underwater noise, and boat traffic can degrade marine habitats used by sea turtles and Nassau grouper. 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 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 in the action area year-round as can Nassau grouper.

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 and it is likely there are numerous groundings that go unreported despite causing damage to green and hawksbill sea turtle and Nassau grouper habitats. As part of the Section 7 process for dock, port, and marine construction activities under the jurisdiction of the U.S. Army Corps of Engineers (USACE), NMFS also considers the impacts of the vessel traffic from the operation of these facilities and any measures to avoid and minimize adverse impacts to green and hawksbill sea turtles and Nassau grouper.

ESA-Listed Corals and Elkhorn and Staghorn Coral Critical Habitat

Federal vessel operations in the action area that have the potential to adversely affect ESA-listed corals and their habitat include operations of the USCG and NOAA. As noted above, NMFS and the USCG completed a programmatic consultation on August 5, 2013, for the USCG's ATONS program in portions of Florida, Puerto Rico, and the USVI (SER-2011-3196, *Maintenance of Existing Fixed and Floating Aids to Navigation [ATON] within Sectors Miami and Key West, Florida, and Sector San Juan, Puerto Rico*). NMFS determined the ATON program would adversely affect elkhorn and staghorn corals and elkhorn and staghorn critical habitat but would not result in jeopardy for the two coral species or result in the destruction or adverse modification of critical habitat. NMFS determined that the maintenance of ATONs would result in the taking of 11.5 ft^2 of staghorn and 69 ft^2 of elkhorn corals due to pile driving or spudding. NMFS also determined that there would be impacts to 1,726 ft^2 of dead coral skeleton containing the essential feature of elkhorn and staghorn coral critical habitat. These impacts were for all of the U.S. Caribbean and a portion of Florida so impacts to elkhorn and staghorn corals and their critical habitat in USVI associated with the USCG's ATON program would be minimal. Potential impacts to the 5 Atlantic/Caribbean coral species that were listed in 2014 were not considered in the 2013 ATON consultation. NMFS is currently working on a national programmatic consultation that will determine the magnitude of the adverse impacts resulting from all ATON maintenance nationwide, including those in the U.S. Caribbean, on all species and critical habitat occurring within the action area of the consultation. Various NOAA programs are also working with OPR to ensure their vessel and research operations are in compliance with the ESA. 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 elkhorn and staghorn coral critical habitat.

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 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. 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. St. Thomas has several areas where vessels congregate around the island in addition to port facilities in Redhook and Charlotte Amalie. Similarly, numerous bays around St. John that are outside of the National Park and Coral Monument are experiencing increases in the number of vessels mooring in the bays, including Coral Bay where there is a marina and another is proposed, and there are port facilities in Enighed Pond and Cruz Bay. 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.

8.3 Research Activities

Sea turtles are the focus of research activities authorized by Section 10 permits under the ESA. Regulations developed under the ESA allow for the issuance of permits allowing take of certain ESA-listed species for the purposes of scientific research under Section 10(a) (1) (a) of the ESA. Authorized activities 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. At this time, the University of the Virgin Islands UVI holds a NMFS research permit for take of sea turtles (NMFS Permit No. 20315).

Section 10 permits are not required for research on Nassau grouper, and rough cactus coral, pillar coral, lobed star, mountainous star, and boulder coral because they are listed as threatened species, which do not automatically receive the protections of ESA Section 9 upon listing. ESA Section 4(d) gives NMFS the discretion to issue a rule to extend any or all of the ESA Section 9 protections to threatened species. No ESA Section 4(d) rule has been promulgated to extend ESA

Section 9 protections to these species at this time. 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.

There has been and continues to be, research on Nassau grouper spawning aggregations in waters of the USVI, particularly Grammanik Bank, but most of the research is done using visual censuses rather than capturing animals. When fish are captured, they are captured live and returned to the water to limit any mortality from fishery independent research activities.

There are numerous research activities associated with ESA-listed corals and their habitat that were ongoing in USVI waters prior to the 2017 hurricanes and that have restarted in many cases. These include NCRMP monitoring activities and monitoring and restoration activities funded through NOAA's Coral Reef Conservation Program (CRCP), as well as activities carried out by NOAA RC and partners and by researchers from UVI and other universities. There were coral farms, used to propagate ESA-listed corals, in particular elkhorn and staghorn coral, for outplanting on coral habitats in order to increase the live cover of ESA-listed corals on habitats where they were historically present in the USVI.

8.4 Coastal and Marine Development

Federal agencies such as the USACE are responsible for permitting of coastal and marine development activities including the construction of docks, boardwalks along the shoreline, and dredging, all of which are activities that have been permitted within the last 5 years in the action area by the USACE. We have conducted consultations with the USACE for those projects that had the potential to affect ESA resources under our purview. EPA is also responsible for permitting, including under the NPDES program.

Sources of pollutants along the coast of USVI include stormwater runoff from coastal development, 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. As noted previously, water quality monitoring studies by the VIDPNR 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 best management practices on construction sites, and failure of on-site disposal methods (Rothenberger et al. 2008). These factors result in increased sedimentation and nutrient transport, bacterial contamination, and trash and other debris entering surface and nearshore waters from developed areas. DEP reports declines in water quality around USVI based on monitoring data. The 2012 impaired waters list included 98 sites and the 2016 list includes 89 sites throughout USVI, indicating that water quality continues to be an issue throughout USVI. The 2016 impaired waters list includes sites around St. Croix, St. Thomas, and St. John (https://www.epa.gov/sites/production/files/2017-02/.../2016_usvi_303d_list.pdf).

Coastal runoff, marina and dock construction, dredging, industrial operations, increased underwater noise, and boat traffic can degrade marine habitats used by sea turtles and Nassau grouper, as well as ESA-listed corals. The development of marinas and docks can negatively affect nearshore habitats, including designated critical habitat for elkhorn and staghorn coral. An increase in the number of docks built thereby increases boat and vessel traffic, as discussed in Section 8.2. Fueling and pump-out facilities that serve marinas and vessels in other areas such as mooring fields can sometimes discharge oil, gas, and sewage into sensitive coastal habitats. Although these contaminant concentrations do not likely affect 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 the USVI year-round. Similarly, different life stages of Nassau grouper may be in offshore and nearshore waters. Therefore, hawksbill and green sea turtles and Nassau grouper, as well as ESA-listed corals analyzed in this Opinion may be exposed to and accumulate terrestrial contaminants that are released into the marine environment during their life cycles.

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 percent, while it was only 39 percent 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) .

A 4-year monitoring study of the reef complex in Caret Bay before, during, and after a terrestrial construction project showed a significant difference among transects and depths. Sedimentation rates between 10 – 14 mg/cm^2 per day in some reef sites led to a 38 percent increase in the number of coral colonies experiencing pigment loss compared to reef sites exposed to rates between 4 – 8 mg/cm^2 per day (Nemeth and Sladek Nowlis 2001). Bleaching of corals was strongly correlated to sedimentation rate, indicating that corals bleached in response to sediment stress.

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 by 1-2 orders of magnitude (Rogers et al. 2008). Nearshore waters adjacent to highly developed watersheds typically average over 10 mg per cm^2 per day, in contrast to nearshore waters adjacent to less developed watersheds, which average less than 4 mg per cm^2 per day, and offshore reefs that are not associated with a land mass that average less than 0.5 mg per cm^2 per day (Rogers et al. 2008; Smith et al. 2008a). During a severe rain event, sediment load can increase to > 30 mg per cm^2 per day (Rogers et al. 2008). Over the rainy season, sediment flux rates from developed watersheds were up to 360 mg per cm^2 per day (Gray et al. 2008). Developed watersheds around St. John were also found to increase the input of terrestrially derived sediments by 15 times, in comparison to undeveloped watersheds, and mean organic matter flux rates by up to 10 times. Influxes of sediment USVI waters are common following rain events, possibly due to ineffective sediment control measures during terrestrial construction or unpaved roads, as was demonstrated in St. John (Ramos-Scharron and Macdonald 2007; Ramos-Scharrón and MacDonald 2007). These changes in sedimentation affect the quality of the elkhorn and staghorn 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 (Fabricius 2005; Babcock and Smith 2000). 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. 2008c). 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 Sladek Nowlis 2001). Reef sites exposed to average sedimentation rates between 10 and 14 mg per cm^2 per day showed a 38 percent increase in the number of coral colonies experiencing pigment loss compared to reef sites exposed to sedimentation rates between 4 and 8 mg per cm^2 per day (Nemeth and Sladek Nowlis 2001). The findings of Nemeth and Sladek 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. 1984; Rogers et al. 2008). 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 percent of the time in the (Nemeth and Sladek Nowlis 2001) study associated with a development project. Bleaching of corals was strongly correlated to sedimentation rate, indicating that bleaching can be a response to sediment stress.

8.5 Water Quality

According to the 2017 discharge monitoring reports, permitted discharge volumes in the USVI ranged from negligible amounts to 98 million gallons of water per day discharged by Virgin Islands Water & Power Authority into Christiansted Harbor, St. Croix. The next highest discharge would be from Limetree Bay Terminals (petroleum services), St. Croix, which discharged an average of 8.7 million gallons of wastewater per day in 2017.

Table 10 lists the locations of impaired waters in the USVI and the reason for the impairment as well as the year impaired from 2010 to 2018 (but note that the 2018 summary is in a draft that was published for public comment). Nassau grouper and green and hawksbill sea turtles could be present in any of these areas and ESA-listed corals and their designated critical habitat could also be present in the majority of these waters with the exception of those that are entirely or partially estuarine such as Salt River Lagoon in St. Croix.

Table 10. Impaired waters from the USVI Integrated Water Quality Monitoring & Assessment Reports (https://dpnr.vi.gov/environmental-protection/water-quality-management-program/)

Island	General Location Of Monitoring Stations	Class	Impairment	Years Impaired
St. Croix	Frederiksted Harbor	C	DO	2010, 2012, 2014
			Turbidity	2010, 2012, 2014
			Enterococci	2018
	Prosperity, nearshore	B	Turbidity	2010, 2014, 2016
			Enterococci	2018
	Sprat Hall Beach	B	DO	2010, 2012, 2014
			Phosphorus	2010, 2012, 2014

			Turbidity	2010, 2012, 2014
			Enterococci	2016
	Cane Bay	B	Turbidity	2010, 2012, 2014, 2016
			Phosphorus	2016
			DO	2018
			Enterococci	2018
	Baron Bluff, subwatershed	B	DO	2010, 2012, 2014
			Enterococci	2010, 2012, 2014
			Turbidity	2010, 2012, 2014
	Salt River Lagoon, Marina	B	Enterococci	2010, 2012, 2014
			Turbidity	2010, 2012, 2014
	Salt River Bay	B	Enterococci	2010, 2012, 2014, 2016
			Turbidity	2010, 2012, 2014, 2016
			DO	2016
	St. Croix-by-the-Sea	B	pH	2010, 2012, 2014
			Turbidity	2010, 2012, 2014, 2016
	Long Reef Backreef, West	C	Enterococci	Prior to 2010
	Long Point Bay	B	DO	2016
			Turbidity	2016, 2018
	Christiansted Harbor	B	pH	2016
			Turbidity	2010, 2012, 2014, 2016
	Christiansted Harbor, East	C	DO	2010, 2012, 2014
			Enterococci	2010, 2012, 2014

			Turbidity	2010, 2012, 2014
			pH	2016
	Beauregard Bay	B	Enterococci	2018
			Secchi Depth	2010, 2012, 2014
			Turbidity	2010, 2012, 2014
			pH	2016
			Phosphorus	2018
	Buccaneer Beach	B	DO	2010, 2014
			Enterococci	2018
	Punnett Bay*	B	Turbidity	2010, 2012, 2014, 2016
			Enterococci	2018
	Green Cay Beach*	B	Enterococci	2010, 2012, 2014
			Turbidity	2010, 2012, 2014, 2016
	Southgate Subwatershed, Offshore*	B	DO	2010, 2014, 2016
			Enterococci	2010, 2014, 2016, 2018
			Turbidity	2010, 2014, 2016
	Teague Bay	B	DO	2010, 2012, 2014
			pH	2010, 2012, 2014
			Turbidity	2010, 2012, 2014
	Teague Bay Backreef	B	Enterococci	2018
			pH	2010, 2012, 2014
			Turbidity	2010, 2012, 2014, 2016

	Grapetree Bay	B	DO	Prior to 2010, 2014
	Turner Hole Backreef	B	Enterococci	2010, 2012, 2014, 2016
			Turbidity	2010, 2012, 2014, 2016
	Buck Island Backreef	A	Temperature	2018
	Bugby Hole Backreef	B	Enterococci	2010, 2012, 2014
			Phosphorus	2010, 2012, 2014
			Turbidity	2010, 2012, 2014, 2016
			pH	2016
	Canegarden Bay	B	Phosphorus	2010, 2014
			Turbidity	2010, 2014
			DO	2016
	Hess Oil Virgin Islands Harbor	C	DO	2010, 2014
			Enterococci	2010, 2014
			Phosphorus	2010, 2014
			Temperature	2010, 2014
			Turbidity	2010, 2014
	Limetree Bay	B	DO	2016
	Martin-Marietta Alumina Harbor	C	DO	Prior to 2010, 2016
	Manning Bay/Estate Anguilla Beach	B	Phosphorus	2010, 2014
			Turbidity	2010, 2014
			DO	2016
	HOVENSA West	B	Enterococci	Prior to 2010
			DO	2016
	HOVENSA Subwatershed, Offshore	B	Temperature	2018
	Diamond Subwatershed, Offshore	B	DO	2010, 2014, 2016
			Enterococci	2010, 2014
			Phosphorus	2010, 2014

			Secchi Depth	2010, 2014
			Toxicity	2010, 2014
			Turbidity	2010, 2014
	Carlton Beach	B	DO	Prior to 2010, 2014, 2016
			Turbidity	Prior to 2010, 2014
	Good Hope Beach	B	Enterococci	2010, 2012, 2014
			DO	2016
	Sandy Point, Nearshore West	B	DO	2010, 2012, 2014
			Enterococci	2010, 2012, 2014
			Turbidity	2010, 2012, 2014
St. John	Caneel Bay	B	DO	Prior to 2010
			Turbidity	Prior to 2010
	Hawksnest Bay	B	DO	2010, 2012
			Enterococci	2018
			Turbidity	2018
	Trunk Bay	A	DO	Prior to 2010
			Enterococci	2018
	Cinnamon Bay	B	DO	Prior to 2010
			Enterococci	2018
	Maho Bay/Francis Bay	B	DO	2010
			Turbidity	2010
	Coral Harbor*	B	Enterococci	2010, 2012, 2014
			pH	2010, 2012, 2014
			Turbidity	2010, 2012, 2014, 2016
			Phosphorus	2016
			DO	2016

	Coral Bay	B	Enterococci	2018
	Round Bay*	B	Enterococci	2012
	Great Lameshur Bay**	B	pH	2014
			Turbidity	2014
			Phosphorus	2016
	Genti Bay, Nearshore**	B	Turbidity	2014
	Fish Bay**	B	pH	2010
			Turbidity	2010, 2016
			Phosphorus	2016
			DO	2016
	Rendezvous Bay**	B	Turbidity	2010, 2012, 2016
	Chocolate Hole	B	DO	2010, 2012
	Great Cruz Bay	B	DO	2010, 2012, 2014
			pH	2010, 2012, 2014
			Turbidity	2010, 2012, 2014, 2016
			Enterococci	2018
	Cruz Bay	B	DO	2012, 2014
			Enterococci	2012, 2014
			pH	2012, 2014
			Secchi Depth	2012, 2014
			Turbidity	2012, 2014, 2016
			Phosphorus	2016
	Great Cruz Bay Watershed, Offshore	B	Turbidity	Prior to 2010
			Enterococci	2018
	Southwest St. John HUC 14, Offshore	B	Turbidity	2014
St. Thomas	Botany Bay	B	Enterococci	2010
			Turbidity	2016
	Stumpy Bay	B	Turbidity	Prior to 2010, 2016

	Santa Maria Bay	B	DO	2010
			Turbidity	2010, 2016
	Caret Bay	B	DO	Prior to 2010
			Turbidity	Prior to 2010, 2016
	Dorothea	B	DO	2010, 2012
			Turbidity	2010, 2012, 2016
	Hull Bay	B	DO	2010, 2012, 2014
			pH	2010, 2012, 2014
			Turbidity	2010, 2012, 2014, 2016
			Enterococci	2018
	Magen's Bay	B	DO	2010, 2012, 2014
			Enterococci	2010, 2012, 2014
			pH	2010, 2012, 2014
			Turbidity	2010, 2012, 2014, 2016
	Mandahl Bay (Marina)	B	DO	2010, 2012, 2014, 2016
			Enterococci	2010, 2012, 2014
			Phosphorus	2018
			pH	2010, 2012, 2014
			Turbidity	2016
	Sunsi Bay	B	DO	2010, 2012
	Spring Bay	B	DO	Prior to 2010
	Mandahl Bay Subwatershed, Offshore	B	DO	2010, 2012
			Turbidity	2010, 2012, 2016
			Enterococci	2016

	Water Bay	B	DO	2010, 2012, 2014
			pH	2010, 2012, 2014
			Enterococci	2018
	Smith Bay	B	DO	2010, 2012, 2014
			Turbidity	2010, 2012, 2014
			Enterococci	2018
	St. John Bay	B	DO	2010, 2012, 2014
			Turbidity	2010, 2012, 2014
			Enterococci	2018
	Red Bay	B	DO	Prior to 2010, 2016
			Turbidity	Prior to 2010, 2016
			Enterococci	2018
	Vessup Bay	B	Enterococci	2010, 2012, 2014
			Temperature	2010, 2012, 2014
			Turbidity	2016
	Red Hook Bay	B	Turbidity	2010, 2012, 2016
			Enterococci	2018
	Great Bay	B	DO	2010, 2012, 2014
			Turbidity	2010, 2012, 2014, 2016
			Enterococci	2018
	Cowpet Bay	B	DO	Prior to 2010
			Turbidity	2016
			Enterococci	2018

	Nazareth Bay	B	Turbidity	2010, 2012, 2014, 2016
			Enterococci	2018
	Benner Bay Lagoon Marina**	B	Enterococci	2010, 2012, 2014
			Turbidity	2016
			DO	2016
	Mangrove Lagoon**	B	Enterococci	2010, 2012, 2014
			Temperature	2010, 2012, 2014
			Turbidity	2016
	Frenchman Bay Subwatershed East**	B	Turbidity	2010, 2012, 2016
			Phosphorus	2016
			Enterococci	2018
	Frenchman Bay**	B	DO	2010, 2012, 2014
			Turbidity	2010, 2012, 2014, 2016
			Enterococci	2016
	Limetree Bay**	B	DO	2010, 2012
			Turbidity	2010, 2012, 2016
			Enterococci	2018
	Morningstar Bay**	B	Enterococci	2010, 2012
			Turbidity	2010, 2012, 2016
	Pacquereau Bay**	B	Turbidity	2016
	St. Thomas Harbor, Inner	C	Turbidity	2010, 2012, 2014
			Enterococci	2016
	Gregerie Channel	B	Turbidity	2016
			Enterococci	2018
	Sprat Bay	B	Turbidity	2016
			Enterococci	2018

	Hassel Island at Haulover Cut to Regis Point	C	Turbidity	2010, 2014
	Druif Bay	B	Turbidity	Prior to 2010, 2016
			Enterococci	2018
	Flamingo Bay	B	Turbidity	2010, 2016
			Enterococci	2018
	Lindbergh Bay	B	DO	2010, 2012, 2014
			Turbidity	2010, 2012, 2014, 2016
			Enterococci	2018
	Cyril E. King Airport Subwatershed, Offshore	B	DO	Prior to 2010
			Turbidity	2016
	Perseverance Bay, Offshore	B	Turbidity	2010, 2012, 2016
			Enterococci	2018
	Krum Bay	C	Turbidity	2016, 2018
	Brewers Bay	B	DO	2010, 2012, 2014
			Turbidity	2010, 2012, 2014, 2016
			Enterococci	2018
	Fortuna Bay	B	DO	2010
			Enterococci	2010
* = high priority, ** = medium priority, rest are considered low priority for the development of assessment unit-specific TMDL				

Of the 34 assessment units in St. Croix, 16 in St. John, and 39 in St. Thomas found to have impairments due to non-compliance with VIWQS (Table 10), the majority support habitat for green and hawksbill sea turtles, Nassau grouper, and ESA-listed corals based on NOAA benthic maps (Kendall et al. 2001)for USVI and information from site inspections conducted by NMFS biologists in many of these locations. Assessment units that may not support habitat for these species include Salt River Lagoon Marina, Tamarind Reef Lagoon (Southgate Lagoon), parts of Limetree Bay that include the HOVENSA piers, Martin-Marietta Alumina Harbor, HOVENSA West, and Diamond Subwatershed Offshore if in area of former rum distillery discharge (that has now moved), St. Croix; and Mandahl Bay (Marina), Benner Bay Lagoon Marina, Mangrove

Lagoon, and portions of St. Thomas Inner Harbor, St. Thomas. Water quality impairment is likely to result in exposure of green and hawksbill sea turtles, Nassau grouper, ESA-listed corals, and elkhorn and staghorn coral critical habitat. Based on the impaired waters information in Table 10, ESA-listed species are more likely to be exposed to low DO, enterococci, and high turbidity throughout the USVI with 21, 10, and 22 waters impaired due to DO levels; 25, 12, and 35 waters impaired due to turbidity levels; and 22, 9, and 29 waters impaired due to enterococci levels in St. Croix, St. John, and St. Thomas, respectively. In some assessment units in St. Croix, there is likely to be exposure to high phosphorus concentrations (8 assessment units), low or high pH (7 units), poor water clarity as secchi depth (2 units), toxicity (1 unit), and temperature (3 units; Table 10). In St. John, there is likely to be exposure to high phosphorus concentrations (4 units), low or high pH (5units), and poor water clarity as secchi depth (1 unit; Table 10). In St. Thomas, there is likely to be exposure to high phosphorus concentrations (2 units), low or high pH (4 units), and temperature (2 units; Table 10).

We also searched the EPA's STOrage and RETrieval (STORET) data warehouse for water quality data from the USVI to determine the ranges in water quality parameters in order to assess the potential exposure of ESA-listed species and their habitat to constituents at levels or concentrations that could lead to impacts to these resources. However, there appear to be data quality issues because some of the numbers reported for certain parameters seem unlikely, such as DO values of 0 mg/L recorded in several locations in St. Croix, including Christiansted in an area open to the Caribbean Sea with a depth of 3 m. STORET includes the water quality monitoring data from the VIDPNR that encompasses the areas in Table 10, as well as water quality data from NPS and others, such as TetraTech, Inc. that was contracted by VIDPNR to perform some of the water quality sampling for the Territory in the past. Sites with DO impairment had reported values of 0 to 5.49 mg/L (the standard is 5.5 mg/L). Sites with impairment due to coliform bacteria had reported values ranging from 31 CFU/100 mL to 3760 CFU/100 mL (standard is 30 CFU/100 mL as 30-day mean and 110 CFU/100 mL 10 percent of the time over the same 30 days).

Water clarity measured as secchi depth ranged from 0 in 2 meters of water in St. Thomas and St. Croix in 2002 and 2003, respectively, to the bottom being visible in 11 meters of water (standard is 15 m visibility in coral areas, depth permitting, 1 m everywhere else). Total suspended solids in water column samples from St. John, St. Thomas, and St. Croix ranged from 0 mg/L to 120 mg/L. Many of the high values for total suspended sediment recorded for St. Thomas, St. John, and St. Croix had a note in the file that samples were collected following periods of prolonged heavy rainfall, particularly in December 2013. There were some negative turbidity values in the STORET data, which we do not consider valid and therefore do not report here. The rest of the recorded turbidity values range from zero, which may simply mean the probe used to measure turbidity was not sensitive enough to record actual turbidity less than 1, to 366 NTU (the standard is 1 NTU in coral areas and three NTU everywhere else). The highest reported turbidity values appear to have been collected in water bottles, which can also affect the results. There are also notes for some of the high turbidity values reporting that probe calibration difficulties were encountered. If we compare the reported STORET values with data compiled from water quality sampling during in-water construction projects based on a review of our project files from 2011 to 2016, we find that turbidity values typically remain below 1 NTU but do occasionally exceed this value during in-water construction, particularly following rain events over 0.5 inches, but turbidity in these instances is usually under 2.5 NTU. An exception occurred in November 2015 during dredging and construction in the area of the Crown Bay cruise ship pier, St. Thomas, when turbidity reached 4.5 NTU on the bottom and 3.4 NTU at the surface in the immediate area of the dredging (BioImpact Inc. 2015).

Total nitrogen values in the STORET data range from 52 µg/L to 274 µg/L (the proposed new standard for Total N is 207 µg/L) with the majority of actual measured values falling below 210 µg/L. Total P values in the STORET data range from 0.9 µg/L to 580 µg/L with an outlier of 1500 µg/L in a sample from St. Croix in 2005, but over 90 percent of the Total P values in STORET were under 50 µg/L (the current standard). Measured pH values in the STORET data for USVI range from 0 to 11.21 with two values that are either significant outliers or use a different measurement unit that was not reported in STORET or are entry errors; 303 (St. Thomas in August 2008) and 728 (St. Croix in July 2015). Temperature values ranged from 20.2 to 39.8°C with low temperature outliers of 0°C reported in 2000, 2001, and 2002 in St. Croix and 2001 in St. Thomas and 2.75°C reported in St. Thomas in 2008, and high temperature outliers of 51.89 and 57.02 reported in 2001 and 2003, respectively, in St. Thomas (the standard is 25 to 29°C in coral areas and 32°C everywhere else). Salinity values in STORET range from 0.18 to 40.09 ppt with some outliners from 65.43 off Hassel Island, St. Thomas in 2005 to 80.16 at the HOVENSA pier in St. Croix in 2005. Given that the sampling locations are all along the coast of St. Thomas, St. Croix, and St. John, the extreme high and low salinity values are unlikely and not consistent with values measured during water quality sampling for in-water construction projects based on a review of our files. Therefore, the utility of the STORET data may be low if the quality of the data is poor, but STORET is the only source of long-term water quality monitoring data available for the Territory at this time.

8.6 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. Flooding from tropical storms and hurricanes also causes significant sedimentation of nearshore areas resulting in impacts to benthic habitats used by green and hawksbill sea turtles and Nassau grouper. In-water habitat for green and hawksbill sea turtles and Nassau grouper is temporarily lost or temporarily or permanently degraded (depending on the magnitude of the storm).

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 containing the essential feature of elkhorn and staghorn coral critical habitat and areas that also provide habitat to green and hawksbill sea turtles and Nassau grouper, 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 25). 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. This means that many of the habitat areas using by ESA-listed corals, as well as Nassau grouper and green and hawksbill sea turtles likely remain in a condition that can support these species in terms of providing habitat for growth, foraging, and refuge, depending on the species.

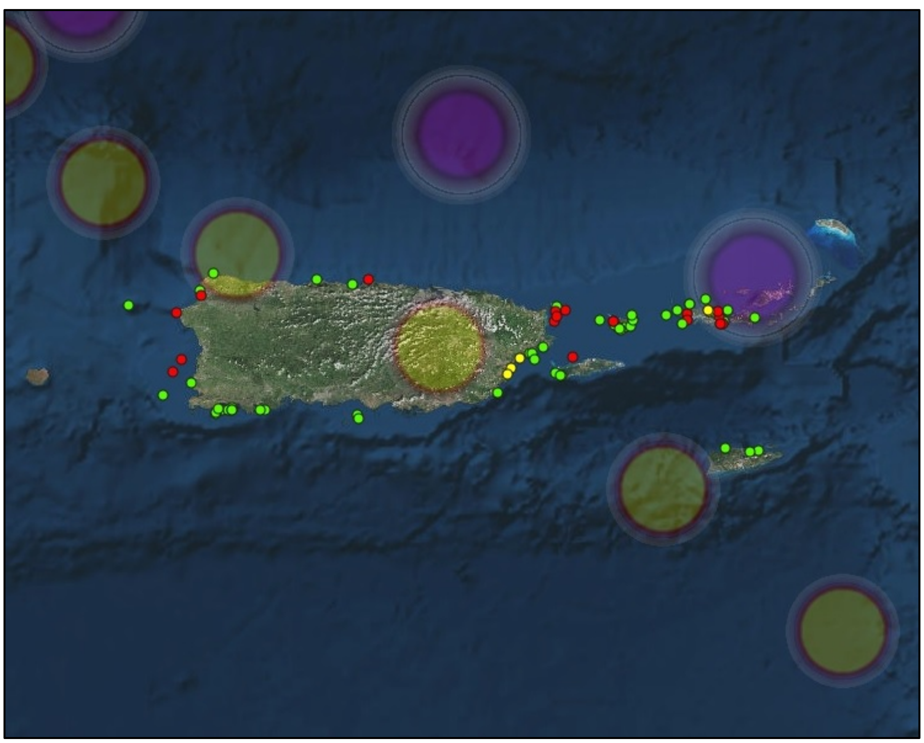

Figure 25. Map showing tracks of Hurricanes Irma (large purple dots) and Maria (large yellow dots) in area where Puerto Rico and USVI are located and results of coral surveys conducted to date. Small green dots indicate areas where coral surveys indicated no triage was needed, red dots indicate areas where triage was needed, and yellow dots indicate areas where the need for triage is still under evaluation (NOAA Restoration Center, https://noaa.maps.arcgis.com/apps/MapJournal/index.html?appid=4f7e03fe4c3748849426d15e124 91d22)

8.7 Synthesis of Baseline Impacts

In summary, several factors adversely affect green and hawksbill sea turtles, Nassau grouper, ESA-listed corals, and elkhorn and staghorn coral critical habitat 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 impact on sea turtles based on stranding data, although there are also records of vessel strikes associated with the operation of recreational vessels. Over the past 5 years, the impacts to sea turtles 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, although these fisheries do not occur in the action area. Poaching is another factor that is likely to continue affecting sea turtles in the action area. Other environmental impacts, including the effects of vessel operation, scientific research, coastal and marine development and associated pollution, and natural phenomena had and are expected to have adverse effects on sea turtles in the action area.

As for sea turtles, poaching may also affect Nassau grouper. Nassau grouper were actively fished in federal waters until 1990 and in USVI waters until 2006; leading to significant declines in the population but fishery regulations prohibiting the catch of this species has effectively eliminated this impact. The greatest adverse impact on Nassau grouper may be the degradation of nearshore habitats that serve as nursery areas for younger life stages of the species due to coastal and marine development and associated declines in water quality now that fishing of this species is prohibited.

Coastal development and the associated transport of land-based sources of pollutants to marine waters has the greatest adverse impact to ESA-listed corals and elkhorn and staghorn coral critical habitat. Other impacts, including vessel operation leading to marine debris, accidental groundings and spills, and fishing activities leading to abandoned or lost gear that can damage coral colonies and their habitat are also expected to have adverse effects on ESA-listed corals and elkhorn and staghorn coral critical habitat in the action area. Natural phenomena such as hurricanes can also have significant adverse effects.

Based on the information discussed in this section, the environmental baseline for sea turtles, Nassau grouper, ESA-listed corals, and elkhorn and staghorn coral critical habitat in the action area is not pristine and has been degraded, particularly by coastal and marine development associated with the construction of residential, commercial, and tourist facilities and point and non-point source discharges to the Caribbean Sea and Atlantic Ocean.

9 Effects of the Action

Section 7 regulations define "effects of the action" as the direct and indirect effects of an action on the species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action, that will be added to the environmental baseline (50 C.F.R. §402.02). Indirect effects are those that are caused by the proposed action and are later in time, but are reasonably certain to occur (50 C.F.R. §402.02). This effects analyses section is organized following the stressor-exposure-response- risk assessment framework. This means we identify stressors associated with the proposed action, in this case, the pollutants for which criteria were proposed, evaluate the response of ESA-listed species to exposure at the proposed criteria, and assess the risk to individuals of each ESA-listed species and the populations to which they belong.

The jeopardy analysis relies upon the regulatory definition of "to jeopardize the continued existence of a listed species," which is "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 C.F.R. §402.02). Therefore, the jeopardy analysis considers both survival and recovery of the species.

The destruction and adverse modification analysis considers whether the action produces "a direct or indirect alteration that appreciably diminished the value of critical habitat for the conservation of a listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features" (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. We intend the phrase "significantly delay" in development of essential features to encompass a delay that interrupts the likely natural trajectory of the development of physical and biological features in the designated critical habitat to support the species' recovery. In the preamble to the proposed rule (and repeated in the final rule) issuing a new regulatory definition of "destruction or adverse modification," the meaning of "appreciably diminish" was clarified by explaining that the relevant question is whether the reduction in the value of critical habitat for the conservation of a listed species has some relevance because we can recognize or grasp its quality, significance, magnitude, or worth in a way that negatively affects the value of the critical habitat as a whole for the conservation of a listed species (79 FR 27060, May 12, 2014; 81 FR 7214, February 11, 2016).

As described in Section 7 *Stressors Associated with the Proposed Action*, NMFS elected to use the existing data as the best available for determining the potential extent of effects to ESA-listed green and hawksbill sea turtles, Nassau grouper, elkhorn, staghorn, rough cactus, pillar, lobed star, mountainous star, and boulder star corals, and elkhorn and staghorn coral critical habitat from the pH (Section 7.2.7.3), "non-coral" turbidity Section 7.2.7.5), enterococci (Section 7.2.8), and TP and the N:P ratio resulting from the proposed TN criterion (Section 7.3). NMFS also elected to use existing data as the best available for determining the potential extent of effects to ESA-listed corals from the CMC for carbaryl (Section 7.2.1.2), the CMC and CCC for nonylphenol (Section 7.2.2), and the CCC for copper (Section 7.2.5).

9.1 Discountable and Insignificant Effects

We have determined that green and hawksbill sea turtles, Nassau grouper, ESA-listed corals, and elkhorn and staghorn coral critical habitat may be adversely affected by the proposed action. However, some of the effects of the proposed action to these species and designated critical habitat will be discountable or insignificant and therefore not likely to result in adverse effects. Discountable and insignificant effects to green and hawksbill sea turtles, Nassau grouper, and elkhorn and staghorn coral critical habitat that were not discussed in detail in Section 7 are discussed here and not discussed further in this opinion.

9.1.1 Green and Hawksbill Sea Turtles

pH: For sea turtles, pH affects the distribution of prey species and/or foraging habitat depending on the species. Acidification of coastal waters can affect herbivorous green sea turtles if the distribution of seagrass changes in response to changing pH, though some predictions indicate seagrass growth could be negatively affected by increased temperatures and changes in salinity (Hawkes et al. 2009). Hawksbill sea turtles, that become more specialized in terms of prey as they age, may also be affected by declines in prey abundance. Research has shown that the percent cover of several sponge species declines significantly with increases in acidification and that the community composition of sponges shifts with bioeroding sponges and other species that likely do not serve as prey for hawksbills becoming more common (Goodwin et al. 2014). There have been a number of impairments due to pH in St. Croix, St. John, and St. Thomas, including in areas where sea turtle nesting is reported and foraging habitat for greens and hawksbills is present, such as Teague Bay in St. Croix, Coral Bay in St. John, and Magen's Bay in St. Thomas. These impairments have resulted in more acidic and more basic pH values, but the continued presence of green and hawksbill sea turtles in many of the areas where pH impairment was recorded during quarterly sampling indicates the impairments did not have significant impacts on the availability of prey species or habitat used by these animals. The VIWQS require that pH be maintained between 7 and 8.3, meaning waters tending toward neutral (7) or slightly basic (8.3). While these pH values may result in impacts to prey species or foraging habitat for green and hawksbill sea turtles in localized areas, large areas of refuge and foraging habitat will remain available. Therefore, we believe the pH criteria is not likely to adversely affect green and hawksbill sea turtles because its application by VIDPNR will result in insignificant effects. We do not discuss the effects of pH on green and hawksbill sea turtles further in this opinion.

Total Nitrogen and Total Phosphorus: There are few studies on the response of sea turtles to nutrients. However, a study by Van Houtan et al. (2014) found that green sea turtles in eutrophic waters that consumed large quantities of macroalgae were more likely to develop fibropapilloma (FP) tumors due to the concentrations of arginine nitrogen the animals were consuming. The incidence of FP in green sea turtles in USVI is low and the TN criterion of 207 µg/L for total nitrogen, proposed based on measured values of total nitrogen in water quality sampling around the Territory is lower than nitrogen in various forms reported in the Van Houtan et al. (2014) study.

We were not able to find studies on phosphorus and sea turtles but the most likely effects would be related to impacts of high phosphorus concentrations on habitat resulting in the loss or degradation of foraging habitat and associated changes in prey availability. As noted in Section 7.3 and discussed further in Section 9.3, the total phosphorus criterion in the VIWQS of 50 µg/L is high compared to levels that allow corals to survive, grow, and have demonstrated reproductive success so there could be some impacts to habitat used by green and hawksbill sea turtles. However, there are no studies indicating that this phosphorus concentration results in detrimental effects to prey species such as sponges for hawksbill sea turtles or seagrass for green sea turtles.

Therefore, we believe the application of the total nitrogen and total phosphorus criteria to water quality conditions by the VIDPNR is not likely to result in adverse effects to green and hawksbill sea turtles because it will be insignificant. We do not discuss the effects of nitrogen and phosphorus on green and hawksbill sea turtles further in this opinion.

9.1.2 Nassau Grouper

Total Nitrogen and Total Phosphorus: Ratios of N:P were found to be important in allowing for the development of adequate phytoplankton to then nourish the zooplankton community that served as prey for larval grouper in a tank experiment (Tew et al. 2013). Depending on when nutrient inputs occur and the balance between nitrogen and phosphorus, a ratio of 16:1 N to P found to be ideal for development of the planktonic food web (Tew et al. 2013). High densities of zooplankton are necessary to provide adequate prey for larval and early stage juvenile grouper. In the case of the proposed total nitrogen criterion of 207 µg/L and the existing standard of 50 µg/L for phosphorus, these concentrations are lower than the amounts of fertilizer used in the Tew et al. (2013) study. The N:P ratio allowed in the 2018 VIWQS based on the proposed TN criterion and the existing TP criterion will be 4:1, which is much lower than that shown by Tew et al. (2013). The TN criterion was proposed based on observed concentrations from water quality monitoring around the USVI and is in keeping with the criterion included in WQS from other states and territories. In addition, a higher ratio such as that found be Tew et al. (2013) could lead to excess growth of algae and eutrophication in real world conditions where nitrogen can be a limiting factor in the growth of plankton. As discussed further in Section 9.3, the TP criterion, which contributes to the difference in N:P ratios between the proposed criteria in the 2018 VIWQS and the Tew et al. (2013) study, is likely high compared to levels that allow corals to survive, grow and have demonstrated reproductive success. This means there could be impacts to habitat used by Nassau grouper. However, there are no studies indicating that this phosphorus concentration results in detrimental effects to prey species for Nassau grouper. Given the extensive habitat available to Nassau grouper around the USVI and the likelihood that nutrient inputs at ratios of 4:1 would occur due to nutrient pulses in specific locations, Nassau grouper would continue to find suitable refuge and foraging habitat and prey. Therefore, we believe the exposure of Nassau grouper to the proposed total nitrogen criterion and the existing total phosphorus criterion are not likely to result in adverse effects because it will be insignificant. We do not discuss the effects of nutrients on Nassau grouper further in this opinion.

9.1.3 Elkhorn and Staghorn Coral Critical Habitat

Enterococci Bacteria: In terms of elkhorn and staghorn coral critical habitat, we do not believe the criteria for enterococci bacteria will result in a decrease in function of the essential feature and find the effects of the criteria are not likely to adversely affect elkhorn and staghorn coral critical habitat. However, because the presence of human fecal bacteria is an indicator of sewage contamination, which includes nutrients, there could be an overgrowth of coral habitat by macroalgae, leading to a decline in function of the essential feature. This effect is discussed in relation to TN and TP in Section 9.3.3. We do believe there may be an effect on ESA-listed coral colonies from the criteria for enterococci bacteria, which is discussed in Section 9.3.3 and sections that follow. We do not discuss the effect of the enterococci bacteria criteria on elkhorn and staghorn coral critical habitat further in this opinion.

Pesticides: Carbaryl and Nonylphenol: As discussed in Sections 7.2.1.2 and 7.2.2, we believe the acute criterion for carbaryl and the acute and chronic criterion for nonylphenol will result in exposure concentrations that may adversely affect ESA-listed corals. In terms of elkhorn and staghorn coral critical habitat, we do not expect the application of the criteria for carbaryl and nonylphenol to result in a decrease in function of the essential feature. We do believe there may be an effect on ESA-listed coral colonies from the proposed criteria for carbaryl and nonylphenol, which is discussed in Section 9.3.3. We do not discuss the effect of the carbaryl and nonylphenol criteria application on elkhorn and staghorn coral critical habitat further in this opinion.

Metals: Copper: As discussed in Section 7.2.5, the chronic toxicity criterion for copper was found to be too high to be protective to marine invertebrates, which are expected to include coral. The effects of this metal on ESA-listed corals are discussed further in Section 9.3.3. In terms of elkhorn and staghorn coral critical habitat, the presence of copper in areas containing the essential feature in concentrations permitted under the proposed acute and chronic criteria is not likely to increase the amount of sediment on hard substrate where corals can grow or increase the amount of macroalgae on hard substrate. Therefore, we expect the effect of application of the criteria for copper by the VIDPNR on elkhorn and staghorn coral critical habitat will be insignificant and we do not discuss it further in this opinion.

9.2 Exposure Analysis

When evaluating a numeric water quality criterion, the exposure estimate in the analysis is straightforward: the exposure is the concentration proposed to be the criterion. The exposure analysis also identifies, as possible, the number, age or life stage, and gender of the individuals likely to be exposed to the actions' effects and the population(s) or subpopulation(s) those individuals represent. The criteria are applied to discharges from construction activities on land and in the water that generate non-point source discharges, which includes in-water sediment plumes from activities such as dredging. The criteria are also applied to point and non-point discharges from commercial and industrial operations that generate process water and/or stormwater. Single-family home construction and associated lot clearing is one of the few activities that may not require local permits that would include required compliance with the VIWQS depending on the size of the project. The criteria will also be applied to waters under USVI jurisdiction for the assessment of impairment status and as recovery goals for waters found to be impaired. Therefore, any ESA-listed species and their habitat present in USVI waters or waters affected by water quality conditions due to VIDPNR authorized discharges will be exposed. The following sections describe, to the extent possible given available information, the number, age or life stage, and gender of the individuals occurring in waters affected by VIDPNR's use of the water quality criteria in regulating water quality in USVI waters found to merit further consideration due to the potential for adverse effects to ESA-listed species, and the population(s) or subpopulation(s) those individuals represent.

9.2.1 Green (North and South Atlantic DPS) and Hawksbill Sea Turtles

When we evaluated the stressors of the action (Section 7), we concluded that green and hawksbill sea turtles would likely respond to exposures to the following criteria: pH, "non-coral" turbidity standard of 3 NTUs, and TP standard. Further analysis of the pH and TP criteria (Section 9.1.1) led us to conclude that these criteria are not likely to adversely affect green and hawksbill sea turtles. Therefore, only the potential adverse effects of the "non-coral" turbidity criterion on sea turtles are discussed further in this and following sections.

Juvenile and adult green sea turtles are common in USVI waters, particularly in bays containing seagrass and coral habitats. Green sea turtle nesting has been reported on some beaches in St. Thomas and in greater numbers on beaches around St. Croix and on Buck Island, St. Croix, but no nesting of this species is reported on St. John. Juvenile and adult hawksbill sea turtles are common in nearshore and offshore waters of USVI in areas containing coral habitats and rocky ledges. Hawksbill nesting is reported on all of the main islands, as well as on some of the offshore cays, though mainly in small numbers with the exception of Sandy Point Beach, St. Croix.

The National Park Service conducted hawksbill sea turtle foraging assessments in Buck Island Reef National Monument from 1994-2004 and observed juvenile and adult green sea turtles and adult, subadult, and juvenile hawksbills with juvenile and subadult hawksbills recaptured year-round (Hart et al. 2013). The U.S. Geological Survey (USGS) began in-water sea turtle surveys in March 2012 and observed 505 green sea turtles, 69 hawksbills, and 2 animals that could not be identified to species from 2012 to 2014 (Hart et al. 2014). Of the sea turtles captured by USGS for measurements and health assessments as part of the in-water surveys, 77 percent were greens and 33 percent were hawksbills and all were considered juveniles and subadults based on carapace size (Hart et al. 2014). Thus, the results of the surveys around Buck Island, St. Croix, indicate that young green and hawksbill sea turtles are year-round residents.

Sea turtle surveys have also been conducted for some development projects in St. Thomas and St. Croix, including the Water and Power Authority projects to convert the power plants on these two islands to use liquefied propane gas as fuel. Sea turtle surveys conducted as part of these projects reported a number of green and hawksbill sea turtle sightings, with green sea turtles being seen more often than hawksbills in survey locations off St. Thomas (from Krum Bay toward offshore mooring off southwest corner of St. Thomas) and Water Island, and St. Croix (in area of Christiansted). Sea turtle surveys by UVI, focused in the Brewer's Bay area of St. Thomas, also report large numbers of green sea turtles and some hawksbills present year-round. However, because there are no comprehensive nesting and in-water sea turtle survey programs, it is not possible to estimate the population of juvenile and subadult resident turtles or juvenile, subadult and adult migratory turtles in waters of USVI that could be affected by water quality constituents. Resident turtles are more likely to be affected because these individuals are exposed to contaminants on a longer-term basis in comparison to transitory individuals and some contaminants could bioaccumulate. Green sea turtles could also be affected by declines in cover or condition of seagrass beds where these turtles forage. Similarly, hawksbill sea turtles could be affected by declines in cover or condition of prey species, particularly sponges but also seagrass and other organisms that younger individuals consume because these life stages are less specialized consumers.

Therefore, exposure of green and hawksbill sea turtles to water quality conditions meeting the proposed criteria would occur in nearshore and offshore areas throughout the USVI though more commonly around St. Thomas and St. Croix. These exposures are likely to affect resident juveniles, and transient adults and juveniles, as well as hatchlings of both species around islands where nesting has been reported. We do not have an estimate of the resident population of hawksbill and green sea turtles in USVI, though NMFS estimated a total population of 219 adult green sea turtles and 441 adult hawksbill sea turtles around St. Croix based on nesting data from 2009 for a consultation. There are also resident juveniles of each species in USVI.

9.2.2 Nassau Grouper

When we evaluated the stressors of the action (Section 6), we concluded that Nassau grouper would likely respond to exposures to the following criteria: pH, “non-coral” turbidity criterion of 3 NTU, and TP. Further analysis of the TP criterion (Section 9.1.2) led us to conclude that this criterion is not likely to adversely affect Nassau grouper. Therefore, only the potential adverse effects of pH and “non-coral” turbidity on Nassau grouper are discussed further in this and following sections.

Based on studies by NOAA's National Ocean Service in certain bays around USVI, Nassau grouper are rare, though adult groupers of various species were more common inside the no-take National Monuments than in other areas (Pittman et al. 2014). No Nassau grouper were sighted from 2001 to 2010 in 1,149 visual surveys (Pittman et al. 2014). A 1979 survey found adult Nassau grouper were present in low abundance around Buck Island (Gladfelter et al. 1980) meaning the species may never have been common within this reserve. In 2011, as part of the

TCRMP, two Nassau grouper were sighted during surveys off St. Croix, including at Lang Bank (Smith et al. 2011). In contrast, adult grouper density was low (maximum of one grouper observed across survey areas) but relatively constant in Virgin Islands National Park, St. John from 2002 to 2011, although Nassau grouper were not sighted in 417 underwater visual surveys in the Park (Pittman et al. 2014). Similarly, a survey of Coral and Fish Bays, St. John, indicated that groupers were more common in Coral Bay than in Fish Bay but species observed were rarely Nassau grouper (Menza et al. 2012). Surveys of the St. Thomas East End Reserve did not find Nassau grouper despite the presence of smaller fish of other grouper species using the mangrove/seagrass/coral habitat complex within the reserve (Pait et al. 2016), which is a habitat utilized by juvenile Nassau grouper based on surveys from other Caribbean islands and the Florida Keys (NMFS 2013). The 2011 TCRMP did observe Nassau grouper off Black Point, College Shoal, Ginsburgs Fringe, Grammanik Bank, and Flat Cay, St. Thomas, and the species is reported in traps at Buck Island, St. Thomas (Smith et al. 2011). Grammanik Bank is also one of the sites were Nassau grouper spawning has been reported beginning in 2005 (Kadison et al. 2009).

Therefore, Nassau grouper exposure to water quality conditions meeting the proposed criteria could occur as juveniles in nearshore habitats, including mangrove roots, seagrass beds, coral reefs, and colonized hard bottom, and as adults in deeper waters associated with coral habitats. Juveniles are more likely to be exposed to pulses of land-based contaminants in stormwater runoff to nearshore waters than adults are but all life stages could be affected by declines in habitat quality due to the impacts of contaminants such as sediment on corals in particular. Prey species of juvenile and adult Nassau grouper could also be affected by the presence of contaminants, resulting in impacts to Nassau grouper associated with any declines in the abundance or quality of prey due to contaminants.

9.2.3 ESA-Listed Corals and Elkhorn and Staghorn Coral Critical Habitat

When we evaluated the stressors of the action (Section 7), we concluded that ESA-listed corals and elkhorn and staghorn coral critical habitat would likely respond to exposures to the following criteria: CMC for carbaryl; CMC and CCC for nonylphenol; CCC for copper; pH; enterococci; “non-coral” turbidity; and the existing TP standard, including in combination with the proposed TN standard due to the resulting ratio of N:P. Further analysis of the potential effects of these stressors on elkhorn and staghorn coral critical habitat (Section 9.1.3) led us to conclude that enterococci, carbaryl, nonylphenol, and copper are not likely to adversely affect this habitat. Therefore, only the potential adverse effects of “non-coral” turbidity on elkhorn and staghorn coral critical habitat, as well as the potential adverse effects of carbaryl, nonylphenol, enterococci, copper, pH, “non-coral” turbidity, and TP on all ESA-listed coral species are discussed further in this and following sections.

The TCRMP has 14 permanent monitoring sites around St. Croix, three around St. John and 16 around St. Thomas covering nearshore and offshore reefs in depths from 6 to 63 m, some of which have been monitored since 2001. Based on the results of this monitoring, many of the sites are dominated by corals from the star coral species complex (Smith et al. 2011). Flat Cay, St. Thomas, also supports populations of elkhorn and staghorn coral as does Buck Island, St. Croix. Rough cactus coral was reported at Little St. James (off St. Thomas). Surveys from development projects with in-water components conducted in 2002 to 2016 also reported the presence of elkhorn, staghorn, pillar, and species from the star coral complex at sites around USVI (NOAA NCRMP). Rough cactus coral was rarely reported. Detailed surveys have not been conducted following the 2017 hurricanes so the status of ESA-listed coral populations in the TCRMP monitoring stations and other locations around USVI is not known at this time. Because of the lack of comprehensive surveys of coral habitats around USVI, it is not possible to estimate the number of ESA-listed corals of each species present around each of the main islands and offshore cays.

ESA-listed corals, particularly colonies in nearshore waters, may be exposed to water quality conditions meeting the proposed criteria as larvae, recruits, juveniles, and adults throughout the USVI. Once settlement occurs, individuals cannot move away from any contaminant plumes and can be affected by exposure. Corals may absorb some contaminants into their tissues, or be smothered by sediment or algae from blooms associated with eutrophication due to excess nutrients. Recruitment may also be affected if settlement habitat quality degrades due to influxes of sediment and nutrients that promote algal growth in areas containing coral colonies.

The St. Thomas/St. John elkhorn and staghorn coral critical habitat unit comprises approximately 121 mi^2 (313 km^2) of marine habitat and the St. Croix area comprises approximately 126 mi^2 (326 km^2). Of the area within the St. Thomas/St. John unit, approximately 26 mi2 are likely to contain the essential element of ESA-designated elkhorn and staghorn coral critical habitat based on the amount of coral, rock reef, colonized hard bottom, and other coralline communities mapped by NOAA's National Ocean Service Biogeography Program in 2000 (Kendall et al. 2001). Of the area within the St. Croix unit, approximately 90 mi^2 are likely to contain the essential feature of ESA-designated elkhorn and staghorn coral critical habitat based on the number of coral habitats mapped. The other areas within the St. Thomas/St. John and St. Croix units are dominated by sand and unconsolidated bottom, seagrass beds with varying densities of coverage, and uncolonized hard bottoms (Kendall et al. 2001).

The essential feature of elkhorn and staghorn coral critical habitat is present largely in nearshore areas around the USVI and could be exposed to water quality conditions meeting the criteria through stormwater runoff. Contaminants such as sediment could degrade the essential feature or result in its no longer functioning as hard substrate free of sediment that is needed to allow for the settlement and growth of elkhorn and staghorn corals. Similarly, high nutrient concentrations can stimulate algal growth, leading to a dominance of algae on coral habitat, also leading to degradation or loss of the essential feature as corals cannot settle and grow in areas with high algal cover.

9.3 Response Analysis

Given the exposure estimated above, in this section we describe the range of responses that may be demonstrated by Nassau grouper, hawksbill, and green (North and South Atlantic DPS) sea turtles, and ESA-listed corals, as well as elkhorn and staghorn coral critical habitat that may result from the stressors associated with the criteria identified in Section 7 and not determined to have discountable or insignificant effects. These criteria include "non-coral" turbidity criterion of 3 NTUs for green and hawksbill sea turtles, Nassau grouper, ESA-listed corals and elkhorn and staghorn coral critical habitat; pH criteria for Nassau grouper, ESA-listed corals and elkhorn and staghorn coral critical habitat; enterococci, carbaryl (CMC), nonylphenol (CMC and CCC), and copper (CCC) for ESA-listed corals; and TP criterion for ESA-listed corals and elkhorn and staghorn coral critical habitat. For the purposes of consultation, our assessment tries to detect potential lethal, sub-lethal (or physiological), or behavioral responses that might reduce the fitness of individuals. Our response analysis weighs evidence of adverse consequences that potentially occur against evidence for attenuating conditions that suggest adverse consequences may be offset or may not occur in USVI waters. For example, considering the likelihood of adverse responses due to exposure under the numeric criteria for cancelled and agricultural pesticides against the probability of exposure to cancelled and agricultural pesticide.

There are other toxic pollutants, such as those related to personal care products and pharmaceutical products that have been shown to affect the health of various freshwater and marine organisms, including fish and corals for which criteria are not proposed in the 2018 VIWQS. Many of these pollutants act as endocrine disruptors and affect processes such as reproduction, immune response, and growth. Endocrine disruptors are commonly identified in sewage effluent and even advanced treatment systems do not remove these compounds.

9.3.1 Green and Hawksbill Sea Turtle

Turbidity: There is little information regarding the effects of sediment on sea turtles and the responses of sea turtles to turbid waters. Given the life history traits of sea turtles, the most likely responses to sediment are behavioral changes to avoid sediment plumes due to the associated impacts on vision and ability to find prey and avoid predators. Avoidance of turbid waters by hawksbill sea turtles was suggested by a study systematically sampling Golfo Dulce, Costa Rica (Chacon-Chaverri et al. 2015). Capture success declined during spring rainy season when the waters sampled became more turbid. High turbidity also affects habitat for green and hawksbill sea turtles, potentially leading to the degradation or loss of some areas of refuge and foraging habitat if sediment inputs are chronic. In addition, hawksbill sea turtles may have to forage over larger areas to find the sponge species that are the preferred prey, particularly of later life stages, which could have fitness consequences on these animals. The 2018 VIQWS require turbidity levels of 1 NTU in areas containing corals, which are likely to also be the areas where preferred sponge species that serve as prey for hawksbill sea turtles are present. The 3 NTU criterion for non-coral areas is high in comparison to values in water quality monitoring data from in-water construction projects, including dredging, based on a review of our files and is likely to result in degradation of foraging habitat used by green and hawksbill sea turtles. Water quality sampling for in-water projects indicate that baseline turbidity levels are below 1 NTU in most nearshore areas around USVI. Because the VIWQS do not define how the coral versus non-coral areas are determined for application of the turbidity criteria, application of the higher standard may occur in areas containing important refuge and foraging habitat for green and hawksbill sea turtles. The EPA noted that the 2000 NOAA benthic maps (Kendall et al. 2001) are used to determine marine habitat types in their response to this draft opinion. However, because the scale of these maps is one-acre pixels, the maps are now almost 20 years old, and some areas were classified as "unknown" due to cloud cover or other interference that did not allow for mapping using aerial photographs, there are likely to be errors in designation of coral versus non-coral habitats. The chronic effects of elevated turbidity are likely to result in impacts to juvenile green and hawksbill sea turtles in particular as these animals will have to forage over larger areas, expending more energy, and may have difficulty finding prey in areas where elevated turbidity has caused permanent changes to coral and seagrass habitats. Thus, application of the 3 NTU turbidity criterion may result in adverse effects, particularly for juvenile green and hawksbill sea turtles.

9.3.2 Nassau Grouper

pH: As discussed in Section 7.2.7.3, seawater pH decreases with decreasing salinity. Salinity affects egg hatchling rate of fish, larval survival, yolk consumption efficiency, and growth, as well as egg buoyancy, which can affect egg survival. In a laboratory study of leopard grouper, the highest hatch rate was calculated to occur at a salinity of 31.7 ppt with no hatching at extreme high and low temperatures and abnormalities in larvae hatched at 8 and 56 ppt (Gracia-López et al. 2004). Gracia-López et al. (2004) also documented a longer notochord at hatch at 32 and 40 ppt, but larvae grew better in terms of growing longer in low salinities and survival rates of larvae maintained at different salinities were consistently high. Eggs were buoyant with salinity above 34 ppt but started to sink below this concentration and reached the bottom of the test tube at 30 ppt (Gracia-López et al. 2004). Similarly, growth of juvenile orange-spotted grouper exposed to varying pH was suppressed in more acidified seawater (pH 7.4 to 7.6) in comparison to control waters with a pH from 8.1 to 8.3 (Shao et al. 2016). Juvenile goliath grouper collected from a mangrove estuary in Colombia were exposed to different salinities to determine the tolerance of the fish to freshwater. Fish that were transferred directly to freshwater died within the first 24 to 48 hours but fish in waters that were transitioned gradually from salinities of 27 to 30 ppt to freshwater salinity of less than 1 ppt over 96 hours did not suffer mortality (Chapman et al. 2014). The smallest individuals in the group that was placed in freshwater with no transition in salinities survived less time, indicating there is a size-related salinity tolerance in this species as has been found in other species (Chapman et al. 2014).

These tolerances may be affected by acidification. Many fish species may not be able to compensate for acid-base changes in ocean chemistry leading to behavioral changes, depressed metabolism, reduced function of blood oxygen transport, changes in gene expression that can affect growth, increased otolith deposition that may affect hearing sensitivity and orientation

(Oxenford and Monnereau 2017). The VIWQS require that pH be maintained between 7 and 8.3, meaning that pH values toward the lower end of the standard could result in suppression of growth in juvenile Nassau grouper, particularly given that juveniles utilize shallow, nearshore habitats where pH may be more variable. In addition, the tolerance of juvenile Nassau grouper for changes in other water quality parameters, such as salinity and temperature, is likely to be negatively affected in waters with lower pH values that are still in compliance with the pH criteria required by the 2018 VIWQS.

Foraging habitat for Nassau grouper may also be affected by changes in pH, particularly coral areas preferred by adults, but also seagrass and mangrove habitats used by juveniles as nursery areas. Local factors such as freshwater discharges to marine environments can influence the degree to which pH changes in a particular area. Prey species such as other fish are likely to be affected by changes in pH such that shifts in abundance or distribution of these species will occur. Because fish species that are prey for Nassau grouper may have varying sensitivities to more acidic waters, some may suffer reduced reproductive success in terms of hatching rates of eggs and/or growth and survival of larvae that could reduce the availability of some prey species. These effects are likely to be more common in shallow nearshore waters juvenile Nassau grouper utilize as nursery habitat. Therefore, the proposed pH criteria could result in adverse effects to Nassau grouper associated with developmental effects to juvenile grouper as well as effects to prey in terms of distribution and availability.

Turbidity: Laboratory experiments with the green grouper demonstrated that exposure to various concentrations of suspended solids (0, 50, 100, 200, 1,000, and 2,000 mg/L) for 6 weeks had varying effects based on the amount of suspended solids to which the fish were exposed (Au et al. 2004). Only those fish exposed to 2,000 mg/L had mortality rates greater than 50 percent. None of the concentrations led to a reduction in metabolic capacity to the extent that growth was affected in fish that survived and fish appeared to become increasingly tolerant to sediment over time. On the other hand, there were sublethal effects to fish that survived in the form of damage to gill structure, apparently due to abrasion by sediment particles, with these effects being more pronounced with exposure to higher concentrations of suspended solids (Au et al. 2004). Similarly, Wong et al. (2013) found that juvenile orange-spotted grouper exposed to suspended sediment concentrations of 8, 32, and 128 mg wet sediments/L demonstrated hypoxic-like symptoms due to sediment clogging gills. Lowe et al. (2015) documented effects to gills of juvenile snapper exposed to turbidities ranging from 10 to 160 NTU for a month. While there is no direct conversion from NTU to mg/L because the amount of suspended sediments versus turbidity depends on particle size and type, the results indicate that exposure of Nassau grouper to sediment concentrations under the VIWQS turbidity criteria are not expected to have fitness consequences for the fish. Given that experiments have used different species of grouper, it is likely that exposure to high levels of sediment would have similar effects on Nassau grouper. The 3 NTU turbidity criterion that applies to areas that serve as nursery habitat for juvenile Nassau grouper is lower than sediment levels determined to have chronic effects on fish in the form of impacts to gill structure due to abrasion and clogging by sediment particles.

Chronic turbidity leads to impacts to marine habitats such as seagrass, mangrove roots, and corals, used by different life stages of Nassau grouper. Sediment may smother seagrass and sediment in the water column in high concentrations affects the amount of light reaching seagrass, reducing the ability of these plants to photosynthesize. Sediment smothering of mangrove root communities can cause a shift in the species colonizing submerged mangrove roots, leading to potential losses of smaller fish species that serve as prey for juvenile Nassau grouper. Similarly, sediment may smother corals and affect the ability of zooxanthellae to photosynthesize, leading to shifts in coral communities especially if sediment exposure is chronic. Changes in these habitats may also change the species utilizing them, which could reduce the amount of prey available to Nassau grouper. Long-term monitoring of some reefs on the southern Great Barrier Reef demonstrates that, as coral cover declined, there was a decrease in prey biomass and a shift in dominant prey species. Nassau grouper may also avoid areas with sediment plumes because turbidity impairs visual acuity thus affecting activities requiring vision, including foraging, and likely leading to behavioral changes. The foraging success of juvenile snapper was reduced due to short-term exposure to turbidity (less than 10 to 160 NTU; Lowe et al. 2015). Nassau groupers forage through ambush, which relies on sight (NMFS 2013). The turbidity criteria in coral and non-coral areas do not exceed turbidities under which foraging success of juveniles of other saltwater fish species were shown to have reduced foraging success, or effects to gills as described above. However, because there are no studies specific to Nassau grouper in tropical waters, waters of the USVI are characterized by turbidity rarely over 1 NTU (based on monitoring data from in-water construction projects), and there is no standard for suspended solids, the three NTU criterion may not be protective of Nassau grouper due to chronic impacts to refuge and foraging habitat and the ability of the animals to find prey. Therefore, there may be adverse effects to Nassau grouper as a result of the 3 NTU "non-coral" turbidity criterion.

9.3.3 ESA-Listed Corals and Elkhorn and Staghorn Coral Critical Habitat

pH: Increases in ocean acidification can be more drastic in local areas where there are influxes of freshwater, leading to reduced calcification of corals due to decreases in pH but also decreases in salinity. As pH declines, the linear extension rate and skeletal density of corals decrease (Hoegh-Guldberg et al. 2007), leading not only to impacts to individual colonies but to the value of coral areas as habitat for other species. Decreasing pH also leads to bleaching as corals experience stress and increasing acidification can override any acclimatization to thermal stress (Anthony et al. 2008). Studies in both the Caribbean (Crook et al. 2012) and the Pacific (Fabricius 2005) have found that very few hard corals can survive and grow below a pH of 7.7 and aragonite saturation of 2.9. Corals from the genus *Porites* were still found in these conditions and *Siderastrea radians* (Caribbean) but at low abundance and typically only when nutrient concentrations were high (Crook et al. 2012). Major reef-building corals did not tolerate these conditions regardless of nutrient input. The size of coral colonies, even in tolerant species, declined as pH became more acidic, likely because growth slowed under more acidic conditions. Because the 2018 VIWQS allow a pH range of 7 to 8.3, the criteria, particularly the lower end of the range, are not protective of ESA-listed corals, which are unlikely to survive and grow in waters with a pH of 7.7 or less. In terms of elkhorn and staghorn coral critical habitat, the pH standard may lead to erosion of the structure of the habitat at more acidic pH values, reducing the function of the essential feature to provide substrate for growth and settlement of elkhorn and staghorn corals.

Turbidity: 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 (Fabricius 2005; Babcock and Smith 2000). However, the type of sediment is also important in terms of the effects to corals. Fourney and Figueiredo (2017) found that high concentrations of coarse sediment was not detrimental to coral recruits but fine sediment from dredging a boat basin considerably increased turbidity and coral recruit mortality. Fourney and Figueiredo (2017) also found that coral recruits survived warm conditions (water temperatures of 30°C) better if fine sediments were kept at less than 6.55 NTU. On reefs without disturbance, turbidity was always less than 1 NTU and sediment was coarse, meaning grain sizes were large and resuspended sediment settled quickly rather than remaining in the water column (Fourney and Figueiredo 2017). Coral health studies conducted in an area of the Great Barrier Reef exposed to a dredging-associated sediment plume gradient found that reefs exposed to the sediment plume longest has two-fold higher levels of coral disease relative to reefs with little to no exposure to the sediment plume (Pollock et al. 2014). While the non-coral criterion of 3 NTU is lower than the turbidity found in the Fourney and Figueiredo (2017) study to affect corals during sediment pulses from a dredging project, this turbidity level is likely to result in chronic effects to ESA-listed corals and elkhorn and staghorn coral critical habitat. As found by Fourney and Figueiredo (2017), undisturbed reefs always had turbidity less than 1 NTU, which is supported by water quality sampling conducted for numerous in-water construction projects around the USVI based on a review of our records where control areas outside the in-water disturbance footprint consistently had turbidity values less than 1 NTU.

Developed watersheds around St. John were found to increase the input of terrestrially derived sediments to nearshore waters 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. High sediment concentrations also affect light penetration, which may impact mass spawning if enough sediment is present in the water column to affect the amount of light reaching corals for longer times. Lack of synchronicity in the release of gamete bundles would affect fertilization and the possibility of hybridization if temporal changes in mass spawning of some species occur (Jones et al. 2015). Sediment may also bind to egg-sperm bundles and reduce their ascent rate, also affecting rates of fertilization (Jones et al. 2015).

While the 2018 VIWQS require that a maximum turbidity of 1 NTU be maintained in coral areas, a turbidity of 3 NTU is permitted in non-coral areas. It is not entirely clear what the distinction is between these two areas given that water quality monitoring data from projects around USVI in various habitat types indicate that even during dredging projects, turbidity values rarely exceed 1 NTU outside the in-water construction footprint (as long as in-water sediment control measures associated with construction projects are adequate). These data also indicate that turbidities up to 4 NTU are very rare and usually associated with rain events including in pre-construction monitoring prior to any disturbance of marine sediments.

This contrasts with studies from terrestrial development projects, which will largely be part of the application of the 2018 VIWQS that have found significant transport of sediment to nearshore waters and associated increases in turbidity to 10 NTU or more (Nemeth and Sladek Nowlis 2001; Rogers et al. 2008). Given the number of impairments around the USVI for turbidity, sediment transport to nearshore waters appears to be a significant stressor. While the standards contain definitions of coral ecosystem components (§186-2), there are no maps or other tools that indicate how these will be determined when assessing which NTU level is appropriate. As discussed in Section 9.3.1, the use of NOAA benthic maps from 2000, while a good starting point, has limitations that are likely to result in the mischaracterization of some areas as non-coral. Therefore, there may be fitness consequences to ESA-listed corals in terms of effects on settlement and growth, due to the 3 NTU turbidity criterion and the application of it by the VIDPNR. The function of the essential feature of elkhorn and staghorn coral critical could also be affected as sediment cover impedes coral recruitment to habitat and the growth and survival of recruits.

Total Phosphorus and N:P Ratio: 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). Similarly, in a study in the Florida Keys, Lapointe et al. (2004) found that nutrient concentrations, particularly of ammonia, decreased proportionally with distance from shore. In areas with chronic higher nutrient inputs, largely due to sewage and agricultural discharges, algal blooms, eutrophication, and reef coral die-off were the result in the Keys with much of the coral reef die-off attributed to microbial diseases that respond to high nutrient inputs in a manner similar to algae (Lapointe et al. 2004). D’Angelo and Wiedenmann (2014) found there could be negative consequences to the marine environment if the concentration of nitrogen is high enough to promote phytoplankton growth, leading to decreases in light penetration.

Ward and Harrison (2000) concluded that it is essential to maintain water quality on coral reefs within ecologically appropriate limits to ensure successful reproduction of corals because, while corals may continue to grow and survive in areas with elevated nutrient concentrations, elevated nutrient concentrations can lead to corals producing smaller and fewer eggs and in some cases less testes material. Wiedenmann et al. (2013) found that nutrient imbalance seemed to have the most severe impacts on corals rather than over-enrichment. Wiedenmann et al. (2013) and Rosset et al. (2017) also found starvation of phosphate leading to a lowered threshold of light and temperature-induced bleaching. Wiedenmann et al. (2013) and Rosset et al. (2017) concluded that a replete balance of nutrients was approximately 6.5 µM dissolved inorganic nitrogen (nitrate and nitrite) and approximately 0.3 µM phosphate, which are equivalent to 813 µg/L inorganic nitrogen and 28.5 µg/L phosphate. Similarly, Harrison and Ward (2001) found fertilization rates of *Acropora longicyathus* (a Pacific species) were significantly reduced at all nutrient levels selected for the study (1, 10, and 100 µM ammonium, phosphate, and combined ammonium and phosphate) with phosphate having more deleterious effects. For ammonium, the micromolar concentrations used in the Harrison and Ward (2001) study are equivalent to 18 µg/L, 180 µg/L, and 1800 µg/L. For phosphate, the micromolar concentrations used in the Harrison and Ward (2001) study are equivalent to 95 µg/L, 950 µg/L, and 9500 µg/L.

Koop et al. (2001) experimented with nutrient concentrations on patch reefs that are part of the Great Barrier Reef in Australia, adding ammonium, phosphate, and both to some of the reefs and maintaining others as controls with no additional nutrient additions. For the most part, biological processes were not affected during an initial low-loading phase of the experiment over the course of a year with the exception of reproduction, which was affected under all nutrient treatments, similar to the Harrison and Ward (2001) study. In the second year of the experiment when high nutrient doses (36.2 µM [653 µg/L] ammonium, 5.1 µM [484 µg/L] phosphate) were added to reefs, Koop et al. (2001) documented mortality in some coral species, stunted growth with high nitrogen additions, and greater calcification and linear extension but lower skeletal density with high phosphorus additions, among other changes. However, it is important to note that these high loading doses are unlikely to occur under regularly occurring conditions in coral areas. Some of these studies of nutrient effects to corals are similar to those done for fish, such as the Tew et al. (2013) study of nutrients and a grouper species that found the preferred ratio of N:P was 16:1 to maintain the food web and insure successful growth of juvenile grouper. Overall, all studies found that phosphorus concentrations should be a small fraction of nitrogen concentrations. Based on these studies, it is unlikely that the proposed TN criterion will result in measurable effects to ESA-listed corals or elkhorn and staghorn critical habitat alone, but the TN criterion could contribute to macroalgal growth on hard substrate in combination with TP, as discussed further below.

A dissolved inorganic nitrogen concentration of approximately 1 µM (126 µg/L) and a soluble reactive phosphorus concentration of approximately 0.1 µM (9.5 µg/L) were noted as threshold concentrations for macroalgal overgrowth of seagrass and coral reefs around seabird mangrove rookeries on the Belize Barrier Reef (Lapointe et al. 1993). These concentrations were exceeded on several reefs in Discovery Bay, Jamaica, where macroalgal blooms were also observed (Lapointe 1997). In the Florida Keys, background nitrate concentrations were typically 0.1 to 0.2 µM (6.2 to 12.4 µg/L) and background orthophosphate concentrations were typically 0.01 to 0.02 µM (9.5 to 19 µg/L) with nutrient concentrations increasing by orders of magnitude during influxes of cold waters from internal tidal bores (Leichter et al. 2003).

When we compare the proposed TN criterion of 207 µg/L with that of other areas such as Florida and Hawai'i, the concentration proposed for application under the 2018 VIWQS is within the same range as those included in these states for areas containing corals. Therefore, it is likely the proposed TN standard is within the range of nitrogen concentrations that are not likely to result in adverse biological changes to ESA-listed corals and may actually lead to an increase in growth of zooxanthellae.

The 2018 VIWQS includes a new standard for TN of 207 µg/L and maintains the existing standard for TP of 50 µg/L. If we use the ratio of N:P based on values reported in other studies, we find that those in studies in coral areas range from 28:1 to 10:1 while the ratio under the proposed TN and TP criteria will be 4:1. Therefore, based on the results of studies cited above, there could be some impacts to the function of the essential feature of elkhorn and staghorn coral critical habitat if the application of the proposed TN criterion in combination with the existing TP standard results in a TN:TP ratio that promotes magroalgal growth. The current monitoring program in USVI does not include measures of phytoplankton biomass in relation to nutrient concentrations that could serve as an indicator of excess nutrients.

The TP value included in the VIWQS is orders of magnitude higher than that included in the Hawai'i WQS (16 µg/L for open ocean waters receiving less than three million gallons per day of freshwater discharge per shoreline mile). As discussed previously, it will also result in an imbalance in the N to P ratio in comparison to ratios reported from studies in coral areas and for some fish species, particularly a grouper. In terms of the existing TP standard, we believe that, while phosphorus limitation has been found to limit biological processes in corals (Wiedenmann et al. 2013; Rosset et al. 2017), the current TP value is higher than some concentrations of phosphate alone that were found to have negative effects on corals (Harrison and Ward 2001), including ESA-listed species. The TP concentration may also result in increases in growth of macroalgae, leading to impacts to the essential feature of elkhorn and staghorn coral critical habitat, particularly in combination with the TN criterion. Therefore, there may be fitness consequences to ESA-listed corals due to the existing total phosphorus criterion related to impacts to skeletal integrity. The function of the essential feature of elkhorn and staghorn coral critical could also be affected as phosphorus loading leads to increases in growth of macroalgae, which would reduce the quality of the habitat and reduce the potential for growth and survival of elkhorn and staghorn coral recruits.

Enterococci Bacteria: Fecal bacteria are one component of sewage discharges that affect corals but sewage also contains nutrients, sediments, and other compounds that can affect coral health. These other constituents are discussed in the various subsections contained in Sections 9.2 and 9.3 of this document related to other water quality criteria such as nitrogen, phosphorus, turbidity, and some metals. Human fecal contamination was detected using MST methods (see Section 7.2.8) over a large geographic area in southwestern Puerto Rico with higher levels of contamination in fringing coastal and mid-shelf reefs, likely due to their proximity to non-point sources along the coast (Bonkosky et al. 2009). Futch et al. (2011) found a similar pattern in the

Florida Keys with fecal indicator bacteria showing a declining trend in concentration and frequency of detection from nearshore to offshore. On the other hand, Futch et al. (2011) found human enteric viruses at similar frequencies regardless of sampling location likely due to differences in susceptibility to biotic and abiotic degradation.

Coral mucus was found to accumulate enteric bacteria and viruses even when these were not detected in the water column or were present at levels below 2.5 CFU/100 ml, meaning some coral heads could serve as indicators of fecal contamination (Lipp et al. 2002). Bacterial indicators including fecal coliform bacteria, enterococci, or *Clostridium perfringens* were detected in approximately 67 percent of the coral surface microlayer at levels between 5 and 1,000 CFU/100 ml (Lipp et al. 2002). Enterovirus nucleic acid sequences, which indicate human-specific waste is present, were detected in approximately 93 percent of the surface microlayer samples but only once in the water column based on cell cultures (Lipp et al. 2002). Similarly, Martins et al. (2007) found genetic material from one or more human enteric viruses in 40 of 100 coral and water samples collected over three years. Three quarters of the coral samples came from lobed star corals, which was the most common species in the study area (Lipp et al. 2007). As in other studies, Lipp et al. (2007) found human enteric viruses more often in coral mucus than in the immediately overlying water column. The difference between water column concentrations and concentrations of bacteria from human waste led Wetz et al. (2004) to state that the inability of fecal indicators to predict the presence of enteric viruses in marine waters, particularly in the tropics, was confirmed.

A study in St. Croix examined the relationship between the prevalence of coral disease and the relative exposure to sewage effluent in coral reef sites on the west shore of St. Croix (Butler Bay and Frederiksted; (Butler Bay and Frederiksted; Kaczmarsky et al. 2005). Kaczmarsky et al. (2005) found the Frederiksted site, which was adjacent to a sewage bypass outfall, had a significantly higher disease (black band disease and white plague type II) prevalence in hard corals examined during the study, as well as lower coral diversity. Kaczmarsky et al. (2005)) also speculated that the increasing trend in sewage discharges over the 15 years prior to the study contributed to the decline in massive coral species in the Frederiksted area, with the death of large colonies, including of elkhorn coral, coinciding with the period of increasing frequency of sewage bypasses.

Patterson et al. (2002) first identified a common fecal enterobacterium, *Serratia marcescens*, as the causal agent of white pox. *Escherichia coli* was used as a control in an experiment to test for this causal agent of white pox and the elkhorn fragments exposed to this control in the study appeared to remain healthy over the 26-day study (Patterson Sutherland et al. 2011). *Serratia marcescens* strain PDR60 was confirmed as the source of the disease after being isolated from diseased elkhorn coral, wastewater, a species of predatory snail that preys preferentially on elkhorn coral and is thought to spread the disease, and a non-host coral (Patterson Sutherland et al. 2011). The presence of the pathogen in a non-host coral species suggests that other coral species may serve as reservoirs for disease-causing pathogens even if the species themselves are not affected by the pathogen (Patterson Sutherland et al. 2011). Patterson Sutherland et al. (2011) also found evidence that some genotypes of elkhorn coral are more resistant to the disease and that disease incidence or severity may increase with increasing temperature. On the other hand, higher temperatures, biological processes, and exposure to ultraviolet radiation may also lead to faster viral decay in the marine environment, which can reduce infectivity (Wetz et al. 2004).

The existing enterococci bacteria criteria is 30 CFU/100 ml 30-day geometric mean and no more than 10 percent of the samples collected in the same 30 days exceeding 110 CFU/100 ml. While the studies cited above do not provide suggested levels of bacteria to be protective of corals, they do indicate that levels measured in the water column are likely much lower than concentrations accumulating on coral heads. In addition, scientific literature indicates that enterococci bacteria is not a good indicator in tropical waters, appears to have little direct effect on corals, but can be an indicator of sewage contamination and the presence of other bacteria and viruses contained in human waste that lead to disease in corals, including ESA-listed corals. STORET data indicate that the existing criteria are frequently exceeded around St. Croix and the information in Table 10 indicates that the criteria are exceeded in sites around USVI. Therefore, there may be fitness consequences to ESA-listed corals from the existing enterococci bacteria criteria due to the likelihood that the presence of the allowed concentration of enterococci bacteria would mean high concentrations of this and other human fecal bacteria on ESA-listed corals, potentially leading to increased disease prevalence.

Carbaryl and Nonylphenol: Pesticides can affect non-target marine organisms like corals and their zooxanthellae.

As discussed in Section 7.2.1.2, the aquatic life criterion for acute exposure to carbaryl is 1.6 µg/L but data in ECOTOX indicate this concentration will affect reproduction, development, and survival. The CMC is higher than exposure concentrations that were observed to cause mortality and have sublethal developmental effects in *Acropora millepora* (Markey et al. 2007), a Pacific coral of the same genus as elkhorn and staghorn corals. The study by Markey et al. (2007) indicates that the proposed acute exposure concentration will lead to significant reduction in fertilization, settlement, and metamorphosis in a Pacific acroporid coral species. Therefore, there may be fitness consequences to ESA-listed corals from application of the acute criterion for carbaryl.

Two LC50s for brine shrimp out of 52 marine invertebrate reported LC50s were below the nonylphenol CMC of 7 µg/L (Shaukat et al. 2014). Four other LC50s were less than four-fold the CMC, including 10.5 µg/L for brine shrimp (Section 7.2.2). The few marine invertebrate species for which data are available are not closely related taxonomically to corals but data did indicate effects to maturation and growth. Growth is an important factor for coral species because they must reach a minimum size before they are able to sexually reproduce.

Coral endocrine systems include estradiol-17-beta (Atkinson and Atkinson 1992; Tarrant et al. 2004) and are expected to be affected by endocrine disrupters like nonylphenol. Our understanding of the role of estrogens in coral species is growing (Tarrant 2005). 17β-estradiol is found within coral tissues and is released during mass spawning events (Atkinson and Atkinson 1992; Tarrant et al. 1999). As a pollutant, ambient estradiol is biologically active in corals, with treatment resulting in 29 percent fewer egg-sperm bundles in rice coral and 13 to 20 percent reduced growth rates in finger coral fragments (Tarrant et al. 2004). Further evidence for steroidal modulation of coral reproduction is the apparent lunar periodicity of estradiol levels and clearance hormones measured in coral tissues before and after reproductive events (Rougee et al. 2015). Yet cnidarians contain no known orthologs of vertebrate estrogen receptors (Tarrant 2003). While these data provide strong evidence that an estrogen mimic such as nonylphenol would have reproductive effects on coral, mechanism (s) for such effects have not been established.

Recent work from the National Center for Coastal Science in Charleston, South Carolina evaluated the effects of nonylphenol on fertilization success in mountainous star coral using the Organization for Economic Co-operation and Development testing guidelines. The exposure concentrations of 0, 0.05, 1, 100, and 300 mg/L did not closely bracket the VIDPNR's criterion of 1.7 µg/L for chronic exposure to nonylphenol. The 1 mg/L exposure did not differ significantly from the control while exposures at 100 mg/L reduced fertilization rates by nearly half (C. Woodley, NOAA National Center for Coastal Science, pers. comm. to P. Shaw-Allen, NMFS OPR, November 2, 2017). These data suggest a NOEC of 1 µg/L, which is below the criterion. The proximity of the NOEC to the criterion of 1.7 µg/L, the underlying variability of the data, and the absence of exposures between 1 and 100 µg/L make it difficult to definitively determine whether or not coral fertilization rates would be affected at the criterion. Based on experience with testing various coral species at the Center for Coastal Science, elkhorn coral are more sensitive to toxicants than mountainous star coral, so the data for mountainous star coral likely underestimate the potential for toxic effects to more sensitive species of corals, such as elkhorn and staghorn corals.

While there are no published nonylphenol toxicity data for species that are taxonomically closely related to coral, the available toxicity data indicate that adverse effects to growth and development occur in other marine invertebrates exposed to nonylphenol at concentrations below the chronic and acute criterion. In addition, unpublished toxicity data indicate the potential for exposure at the criterion to affect fertilization success because responses in a coral species known to be robust to toxicant exposures (mountainous star coral) likely under estimated effects in more sensitive species. Taken together, this information leads NMFS to conclude that the chronic and acute exposure criteria for nonylphenol are likely to adversely affect the fitness of ESA-listed coral species.

Copper:

Information from studies of other metals included in the VIWQS, such as nickel, on Indo-Pacific corals indicate that levels at which effects to coral growth and fitness were observed are orders of magnitude higher than the concentrations permitted by the standards. While the response of ESA-listed corals in the Caribbean could vary, other studies such as those using copper did not find differences in responses between Caribbean and Pacific corals to vary by orders of magnitude. Other demonstrated sublethal effects of heavy metals include induction of heat shock proteins (Venn et al. 2009) and oxidative enzymes (Yost et al. 2010). The most significant sublethal effect might be disruption of coral reproductive processes. Concentrations at which reproductive effects occur vary with both metal type and coral species, but copper, zinc, nickel, lead, and cadmium have been shown to inhibit coral fertilization (Heyward 1988; Reichelt-Brushett and Harrison 1999;2000;2005) and nickel has been shown to cause mortality of larval and inhibit settlement (Goh 1991).

As discussed in Section 7.2.5, the chronic criterion for copper is likely to cause adverse effects to ESA-listed corals based on toxicity data from studies using corals and other marine invertebrates.

In a study testing the effects of exposure to copper in staghorn, mountainous star, and cauliflower coral (an Indo-Pacific species), after exposure to copper concentrations from 2 – 20 µg/L for 5 weeks Bielmyer et al. (2010) found significantly different sensitivities between species. Copper was shown to accumulate in the zooxanthellae and tissue of staghorn and mountainous star coral. Staghorn and cauliflower corals showed effects with exposure to copper concentrations as low as 4 µg/L in terms of impacts to zooxanthellae photosynthesis and skeletal growth (Bielmyer et al. 2010). Mountainous star and staghorn corals also demonstrated significant decreases in carbonic anhydrase (enzymes that play a major role in the carbon supply for calcium carbonate precipitation and in carbon-concentrating mechanisms for symbiont photosynthesis (Bertucci et al. 2013)).For mountainous star coral this happened at a concentration of 10 µg/L while for staghorn coral it occurred at the lower concentration of 4 µg/L. The aquatic life criteria adopted in the VIWQS for copper are 4.8 µg/L (acute) and 3.1 µg/L (chronic) in saltwater, meaning the acute criteria is higher than the concentration documented to cause effects in staghorn coral. Because elkhorn coral, which is in the closely related to staghorn (to the point where the species are known to interbreed), is also sensitive to contaminants as discussed for nonylphenol above, it is likely that elkhorn coral would have the same response to exposure to the criteria as staghorn. Table 11 contains information regarding the effects of copper on ESA-listed coral species from various studies. On the other hand, data from ECOTOX for Pacific acroporid corals reported mortality at an LC 50 of 80 µg/L in larvae after 1 day of observations (Bao et al. 2011) and an EC50 of 35 µg/L in larvae after 2 days of observations with decreased settling (Reichelt-Brushett and Harrison 1999). EC50s ranging from 11.4 to 45.2 µg/L in gametes after several hours of observations with decreases in survival and fertilization (Reichelt-Brushett and Harrison 2005; Victor and Richmond 2005).

Table 11. Effects of copper on ESA-listed coral species from various studies indicating the exposure and duration of exposure leading to effects

Toxicant	Species	Exposure	Duration	Effect	Citation
Cu	*Orbicella faveolata*	4 µg/L	5 weeks	No effect	(Bielmyer et al. 2010)
		10 µg/L		Increased copper concentration in soft tissue; physiological impairment	
		20 µg/L			
	Acropora cervicornis	4 µg/L	5 weeks	impaired photosynthesis	
		10 µg/L		impaired photosynthesis; bleaching; physiological impairment	
		20 µg/L		impaired photosynthesis; bleaching; decreased growth; physiological impairment	
Cu	*Orbicella faveolata*	LOEC: 12.7 µg/L	24 hours	reduced larval swimming behavior	(Rumbold and Snedaker 1997)
		EC50: 24.9 µg/L		reduced larval swimming behavior	
Cu	*Orbicella franksi*	30 µg/L	48 hours	DNA damage; expression of cellular and oxidative stress genes; expression of genes associated with biomineralization	(Schwarz et al. 2013)
LOEC = lowest observed effect concentration; EC50 = dilution at which 50 percent of the organisms displayed the effect or median effective concentration					

As discussed in Section 6.3.1.4, five observations for three species of coral, including one ESA-listed species, showed copper sublethal responses. These responses included effects on growth. Observations for other invertebrates also indicated sublethal effects at or below the chronic copper criterion. There appears to be a significant difference in the sensitivity of Atlantic versus Pacific acroporid corals to copper. Based on the results of these studies compared to the copper criteria, there may be fitness consequences to ESA-listed corals, particularly elkhorn and staghorn coral, associated with chronic exposure to copper in discharges, although there is considerable uncertainty due to the variability in the data available.

9.4 Risk Analysis

The results of our exposure and response analyses concluded the following:

- For green and hawksbill sea turtles:
    - The three NTU turbidity standard is likely to result in impacts to juvenile green and hawksbill sea turtles because of habitat impacts associated with turbidity resulting in these animals having to forage over larger areas, which has energetic costs, and the potential difficulty in finding prey in areas chronically affected by turbidity resulting in damage to or loss of coral and seagrass habitats.
- For Nassau grouper:
    - The pH values toward the lower end of the range allowed under the criteria (i.e., toward 7) are likely to result in suppression of growth in juvenile Nassau grouper in shallow nearshore areas these animals use as nursery habitat and lower pH is likely to impact foraging habitat such as coral reefs and prey species.
    - The three NTU turbidity standard is likely to have chronic effects in the form of impacts to juvenile habitat.
- For ESA-listed corals:
    - pH values of 7.7 or less, which would comply with the standard, are likely to affect coral growth, lead to stress responses such as bleaching, which will make corals more vulnerable to disease and other stressors, and decrease survival and reproductive success.
    - The 3 NTU standard for non-coral areas could affect ESA-listed coral colonies that may be present in habitats such as seagrass beds on patch reefs because it is not clear how coral habitats areas will be delineated when applying the standards.
    - The TP standard is likely to cause greater calcification and linear extension but lower skeletal density, leading to a weakening of the structure of coral colonies, as well as reducing fertilization rates.
    - The criteria for carbaryl (CMC) and nonylphenol (CMC and CCC), and the lack of standards for pesticides included in antifouling paints used on vessels are likely to affect coral settlement and metamorphosis.
    - The copper criterion (CCC) is likely to affect photosynthesis by zooxanthellae, as well as skeletal growth.
- For elkhorn and staghorn coral critical habitat:
    - Acidic pH values are likely to lead to erosion of dead coral skeletons and other calcium carbonate-based hard substrate that are part of the structure of reefs, impacting the function of the essential feature of critical habitat.

- The TP standard is likely to lead to overgrowth of hard substrate by macroalgae, affecting the essential feature of critical habitat particularly through impacts on the settlement and growth of elkhorn and staghorn coral.

In this section, we assess the consequences of the responses to the individuals that have been exposed, the populations those individuals represent, and the species those populations comprise. Whereas the Response Analysis identified the potential responses of ESA-listed species to the proposed action, this section summarizes our analysis of the expected risk to individuals, populations, and species given the expected exposure to those stressors and the expected responses to those stressors.

We measure risks to individuals of endangered or threatened species using changes in the individuals' fitness, which may be indicated by changes the individual's growth, survival, annual reproductive success, and lifetime reproductive success. When we do not expect ESA-listed animals exposed to an action's effects to experience reductions in fitness, we would not expect the action to have adverse consequences on the viability of the populations those individuals represent or the species those populations comprise.

9.4.1 Green and Hawksbill Sea Turtles

This risk analysis is related to the potential chronic effects of elevated turbidity, particularly in juvenile refuge and foraging habitat, on green and hawksbill sea turtles. 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 (NPS 2004;2005) based on sizes of turtles captured that were all less than those of nesting hawksbills. Macdonald et al. (2013) 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 (Macdonald et al. 2013). Overlap in home ranges suggests the areas provide sufficient resources to be shared by neighboring green sea turtles. Wershoven and Wershoven (1992) 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.

The application of the turbidity criteria is not expected to directly affect juvenile green and hawksbill sea turtles but we do expect indirect effects due to chronic impacts of turbidity on refuge and foraging habitat and prey species availability. Because of the strong tendency toward site fidelity for foraging and refuge habitat, the loss or degradation of habitat within a juvenile 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. Chronic effects in the form of habitat degradation and lack of prey species could lead to reduced recruitment of juveniles to certain areas, particularly given the criterion for non-coral areas and the exceptions to the standards that may generate chronic turbidity in these areas, as well as the frequent non-compliance with the turbidity criteria throughout USVI. The effects on juvenile green and hawksbill sea turtles are discussed further in Section 11.

9.4.2 Nassau Grouper

This risk analysis is related to the potential effects of the application of the pH criteria and potential chronic effects of the application of the turbidity criteria on Nassau grouper and their foraging habitats and prey species. Nassau grouper were once common in the USVI but overfishing, particularly targeting of spawning aggregations of the species, led to a dramatic decline in the population of Nassau grouper. Population declines led to fishery closures, first at specific spawning aggregation sites and then throughout federal waters and eventually throughout territorial waters. The Red Hind Bank Marine Conservation District off St. Thomas was closed seasonally in 1990 and permanently in 1999 to protect a red hind spawning aggregation (Nemeth 2005), as well as a Nassau grouper spawning aggregation that used the same site (García-Moliner and Sadovy 2007). In 2003, Nassau grouper were found aggregating to spawn in small numbers on Grammanik Bank off St. Thomas and numbers of spawning individuals in this area has been increasing since 2005 (Kadison et al. 2009). In a study of the St. Thomas East End Reserves, Nassau grouper was not documented in the various habitats that were part of the survey sites, though other juvenile grouper species were observed in very small numbers (Pait et al. 2016). Similarly, in a survey of data from two MPAs in USVI, large-bodied groupers were found to be very rare (Pittman et al. 2014). In Buck Island Reef National Monument, St. Croix, declines in mean adult grouper density were greater within the MPA than outside and Nassau grouper were not sighted within the MPA in 1,149 visual surveys over 10 years (Pittman et al. 2014). In the Virgin Islands National Park, St. John, Nassau grouper were not sighted in 417 visual surveys over 10 years (Pittman et al. 2014) nor were they sighted during assessments of habitats in Fish and Coral Bays, St. John (Menza et al. 2012). During TCRMP surveys, two Nassau grouper were sighted in St. Croix, which was the first time the species had been observed since coral monitoring began in 2000 (Smith et al. 2011). There have been anecdotal reports of juvenile Nassau grouper in nearshore nursery habitats, including sightings during surveys conducted for proposed in-water construction projects but no comprehensive surveys of these areas have been conducted to assess the potential abundance and distribution of juveniles of this species.

As spawning aggregations increase in size, numbers of juveniles in nearshore areas are expected to increase, which is essential for recovery of this species. Because the range of pH values allowed under the VIWQS include acidic values that could suppress growth in juvenile Nassau grouper, as well as impacting coral reef habitats used by grouper and distribution of prey species that are sensitive to more acidic pH, growth of juveniles could be affected. Chronic exposure to turbidity that is allowed under the “non-coral” standard could result in degradation of foraging habitat and declines or changes in prey species abundance and distribution. In addition to effects to individuals from exposure to acidic pH or chronic turbidity at levels that may result in effects to fish habitat and prey, with associated energetic costs to individuals associated with the need to spend greater effort foraging, there may be population-level effects. As historic spawning aggregations of Nassau grouper are restored, juvenile Nassau grouper are expected to become more common and need nursery habitat. Impacts to the fitness of juvenile Nassau grouper could affect the population as the species attempts to recover from historic effects of overfishing. The effects of the application of the pH and non-coral turbidity criteria on juvenile Nassau grouper are discussed further in Section 11.

9.4.3 ESA-Listed Corals and Elkhorn and Staghorn Coral Critical 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 Sladek Nowlis 2001; Smith et al. 2008b). Increases in acidification of seawater associated with climate change and terrestrial discharges in some local areas has led and is expected to continue to lead to degradation of reef structure and declines in coral health and cover (Hoegh-Guldberg et al. 2007). As noted in Section 9.1.3, ESA-listed species from the star coral complex are dominant in reefs and hard bottom habitats around the USVI despite declines in the number and size of colonies following events such as the 2005 mass bleaching (Smith et al. 2011). Additionally, hard bottom habitat containing the essential feature of elkhorn and staghorn coral is also present throughout USVI.

This risk analysis is related to the pH, “non-coral” turbidity (3 NTU), TP, enterococci, carbaryl, nonylphenol, and copper criterion included in the 2018 VIWQS.

NMFS concluded that the lower range of pH allowed under the VIWQS will affect the growth of ESA-listed corals as well as increasing their sensitivity to other stressors. The lower range of pH values (below 7.7) will also affect the ability of ESA-listed corals to survive and corals' ability to reproduce. An acidic pH will also lead to erosion of hard substrate composed of calcium carbonate that is the essential feature of elkhorn and staghorn coral critical habitat.

Similarly, NMFS concluded that the “non-coral”turbidity criterion, which does not clearly explain where the 3 NTU standard will be applied, is likely to result in chronic impacts to ESA-listed coral colonies. The 3 NTU turbidity criterion is also likely to result in chronic impacts to the essential feature of elkhorn and staghorn coral critical habitat, particularly associated with sediment cover impeding coral recruitment and survival.

The TP standard, which is a concentration that appears to be higher than levels found to affect skeletal density and reduce fertilization rates in some ESA-listed coral species, will affect growth and fitness of ESA-listed coral colonies. The TP concentration permitted under the standard may also result in increased macroalgal growth on hard substrate, which would affect the function of the essential feature of elkhorn and staghorn coral critical habitat.

In terms of enterococci bacteria, levels similar to those in the VIWQS standards have been found inconsistent with the much higher levels of bacteria found on coral heads. Fecal bacteria have been directly linked to disease in elkhorn coral, though enterococci as an indicator in the tropics is argued to be poor and other indicators using MST methods are thought to be more accurate in detecting fecal contamination from human sources.

The pesticides carbaryl and nonylphenol have been documented as affecting corals at concentrations less than the acute concentrations for both and the chronic concentration for nonylphenol included in the national standards adopted in the VIWQS. Other pesticides, particularly Irgarol, which is used in anti-fouling paint, have been found in high concentrations in some areas of the USVI with large concentrations of vessels. Irgarol is not regulated under the VIWQS but has been found to affect settlement and metamorphosis in corals.

The chronic exposure concentration of copper adopted in the VIWQS from national standards are at levels that may cause acute or chronic effects to ESA-listed corals in the form of reduced photosynthesis and skeletal growth.

Therefore, the application of the criteria for pH, “non-coral” turbidity, TP, enterococci, carbaryl (CMC), nonylphenol (CMC and CCC), and copper (CCC) are expected to result in reduced growth and reproduction of ESA-listed corals and reduced settlement and recruitment success, affecting both individual colonies and populations of ESA-listed corals around USVI. The application of the criteria for pH, “non-coral” turbidity, and TP are also expected to reduce the function of elkhorn and staghorn coral critical habitat, reducing the quality of habitat for recruitment and settlement of future elkhorn and staghorn coral colonies. These effects are discussed further in Section 11.

10 CUMULATIVE EFFECTS

“Cumulative effects” are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 C.F.R. §402.02). 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.

A number of activities affecting hawksbill and green sea turtles, Nassau grouper, ESA-listed corals and elkhorn and staghorn coral critical habitat such as fishing in federal waters, directed take for scientific research, federal vessel operations, and some coastal and in-water constructions are federally-regulated. Future federally-regulated activities within the action area will likely require ESA section 7 consultation. As those activities are unrelated to the proposed action, they are not considered as part of the cumulative effects analysis.

The fishing activities in territorial waters (which extend up to three nm from shore in USVI) of the action area are expected to continue into the foreseeable future. NMFS is not aware of any proposed or anticipated changes to fisheries in territorial waters discussed in the environmental baseline (Section 8) that would substantially change the impacts that the fisheries have on green and hawksbill sea turtles, Nassau grouper, ESA-listed corals, and elkhorn and staghorn coral critical habitat covered by this Opinion. Therefore, NMFS expects that the levels of interactions between fisheries and sea turtles described in the environmental baseline (Section 8) for fisheries activities will continue at similar levels into the foreseeable future.

The population of USVI increased dramatically between 1970 (approximate population of 64,726) and 2000, but has remained relatively constant at approximately 105,000 residents since 2000 (U.S. Census Bureau 2010, https://www.census.gov/programs-surveys/decennial-census/decade.2010.html). A portion of this change is due to seasonal residents and retirees from the U.S. mainland who have homes in the USVI, particularly in St. Thomas and St. John. Much of the seasonal development is in coastal areas.

USVI is also a top travel destination, including for cruise ships, with travel and tourism having a 31.8 percent total contribution to the gross domestic product of USVI in 2016 with leisure and travel spending contributing almost 95 percent of this (World Travel & Tourism Council 2017). This was projected to continue rising but the impacts of Hurricanes Irma and Maria on the islands in September 2017 had significant impacts on the USVI and many tourism facilities have still not reopened. However, the islands are rebuilding and cruise ship voyages to St. Thomas and St. Croix have been renewed. The majority of tourism development is in coastal areas around the islands.

Much of the development occurring in USVI that has the potential to affect water and in-water habitat quality for ESA-listed green and hawksbill sea turtles, Nassau grouper, the ESA-listed corals, and elkhorn and staghorn critical habitat, in particular through increases in sediment transport to nearshore waters, does not have a federal nexus and thus is not subject to the consultation requirements under section 7 of the ESA. These activities are part of the cumulative effects analysis. Depending on the number and location of development projects, sediment and nutrient loading to nearshore waters could become a chronic stressor to refuge and foraging habitats of hawksbill and green sea turtles and Nassau grouper such as coral habitats and seagrass beds, as well as growth and settlement habitat for ESA-listed coral species, including designated critical habitat for elkhorn and staghorn coral. Similarly, vessel operations and associated traffic are expected to continue into the future and potentially increase as coastal and marine development continues in the USVI and the Territory works to rebuild following the 2017 hurricanes. While some of the activities associated with vessel traffic, such as some USCG actions and construction of marine facilities are likely to have a federal nexus, other activities may not, particularly activities associated with recreational boating and regulation of these vessels, which is largely managed by the Territory.

In terms of natural disturbances, climate change predictions indicate that the magnitude and intensity of storms will increase, which may result not only in changes to the marine environment affecting green and hawksbill sea turtle, Nassau grouper, and ESA-listed corals and their habitat, but also potential changes in human activities. In particular, if the frequency and intensity of storms and other extreme events such as rainfall and drought increase, then human activities associated with coastal and marine development such as shoreline hardening and construction of irrigation systems, as examples, may also increase in scope. While some of these activities will have a federal nexus, requiring Section 7 consultations, others such as irrigation systems and other terrestrial activities may not but could result in impacts to ESA resources due to stormwater runoff and associated transport of contaminants to the marine environment.

At this time, NMFS is not aware of any specific projects or anticipated changes to human-related actions such as those leading to habitat degradation from development that would substantially change the impacts that each threat has on hawksbill and green sea turtles, Nassau grouper, ESA-listed corals, and elkhorn and staghorn critical habitat covered by this opinion. The impacts of Hurricanes Irma and Maria in August and September 2017 are likely to result in numerous coastal and nearshore projects to rebuild infrastructure and commercial, residential, public, and industrial properties. It is not possible for us to predict the number, type and scale of these projects in USVI at this time, but we believe the majority of projects would require federal authorization and ESA section 7 consultations. We also expect the majority of the projects to be for rebuilding rather than new construction. Therefore, NMFS expects that the effects to in-water habitat of green and hawksbill sea turtles, Nassau grouper, and ESA-listed corals, as well as effects to coral colonies themselves and designated critical habitat for elkhorn and staghorn corals from continued development and vessel operations will continue at similar levels into the foreseeable future.

11 Integration and Synthesis

The Integration and Synthesis section is the final step in our assessment of the risk posed to species and critical habitat as a result of implementing the proposed action. In this section, we add the *Effects of the Action* (Section 9) to the *Environmental Baseline* (Section 8) and the *Cumulative Effects* (Section 10) to formulate the agency's biological opinion as to whether the proposed action is likely to: (1) reduce appreciably the likelihood of both the survival and recovery of a ESA-listed species in the wild by reducing its numbers, reproduction, or distribution; or (2) reduce the value of designated or proposed critical habitat for the conservation of the species. These assessments are made in full consideration of the *Status of Endangered Species Act Protected Resources* (Section 6).

The following discussions separately summarize the probable risks the proposed action poses to threatened and endangered species and critical habitat that are likely to be exposed. These summaries integrate the exposure profiles presented previously with the results of our response analyses for each of the actions considered in this opinion.

The results of the risk analysis concluded the following:

- For the USVI's pH criteria:
  - Lower pH values within the range allowed under the standard will suppress growth of juvenile Nassau grouper and have indirect effects associated with acidification impacts to juvenile habitat and prey species.
  - pH values below 7.7 will result in decreased survival of ESA-listed corals, as well as reducing their growth and reproductive success and increasing their susceptibility to disease and other stressors.
  - Lower pH values will result in changes to the quality of hard substrate that is part of elkhorn and staghorn coral critical habitat due to structural erosion caused by acidification.
- For the USVI's “non-coral” turbidity criterion:
  - Turbidity levels in non-coral areas will lead to chronic effects to refuge and foraging habitat and prey species for juvenile green and hawksbill sea turtles, increasing the effort required to find food and shelter and reducing recruitment of juveniles to areas with chronic turbidity impacts.
  - Turbidity levels in non-coral areas will lead to chronic effects to refuge and foraging habitat and prey species, increasing the effort required to find food and reducing recruitment of juveniles to areas with chronic turbidity impacts.
  - The turbidity standard in areas considered non-coral will lead to reduced growth and settlement of ESA-listed corals, as well as causing physical impacts such as smothering and abrasion.
  - The “non-coral” turbidity will result in changes to the sediment cover on hard substrate containing the essential feature of elkhorn and staghorn coral critical habitat.
- For the USVI's TP criterion:
  - The TP concentration allowed under the standards will result in greater calcification and linear extension but lower skeletal density and reduced fertilization rates in ESA-listed corals.
  - The TP concentration, in combination with the TN concentration and the resulting N:P ratio, will lead to increased macroalgal growth, reducing the quality and function of the essential feature of elkhorn and staghorn coral critical habitat.
- For the USVI’s enterococci bacteria criteria:
  - The enterococci bacteria levels under the VIWQS will result in fecal contamination of ESA-listed coral colonies and disease outbreaks in species such as elkhorn coral.

- For the USVI's pesticide criteria:
  - o The carbaryl (CMC) and nonylphenol (CCC and CMC) criteria and the lack of standards for pesticides such as Irgarol will affect settlement and metamorphosis of coral recruits, affecting future colonies.
- For the USVI's copper criterion:
  - o The chronic exposure concentration allowed for copper will result in reductions in photosynthesis and growth of ESA-listed coral colonies.

11.1 Green and Hawksbill Sea Turtles

Based on our review of water quality monitoring data from in-water dredging and construction projects conducted from 2012 to 2016, baseline turbidity values and turbidity during rain events (ranging from 0.03 to 2.73 inches with runoff producing events typically being those over a quarter inch) rarely result in turbidity levels above 1 NTU outside construction footprints as long as sediment control measures associated with construction projects are adequate. On the other hand, as shown in Table 10, water quality impairment due to turbidity is common throughout USVI, indicating that stormwater and sediment control measures on terrestrial sites are either inadequate or poorly maintained, leading to frequent inputs of sediment-laden waters to marine areas around USVI. The VIWQS also do not contain clear language as to how non-coral areas will be determined and, if the NOAA benthic habitat maps are used, there are limitations in these maps due to pixel size and their age. A turbidity of three NTU appears to be high based on comparisons to the data from monitoring in nearshore waters around USVI for permitted projects for which compliance with WQS was required. As discussed previously, turbidity levels that affect habitat and prey distribution of juvenile green and hawksbill sea turtles are likely to have fitness consequences. We need to determine whether the indirect effects of the allowed turbidity standards on the fitness of juvenile green and hawksbill sea turtles will appreciably reduce the likelihood of survival and recovery of the species by decreasing their numbers, reproduction, or distribution.

The need to expend additional effort to find refuge and foraging habitat and prey species can result in reduced growth rates, older age to maturity, and lower lifetime fecundity. However, no reduction in the distribution of green and hawksbill sea turtles is expected and these turtle species will continue to be present throughout water surrounding the USVI.

Whether the potential effects to reproductive output would appreciably reduce the likelihood of survival of green and hawksbill sea turtles depends on the probable effect the changes in reproductive output would have relative to current population sizes and trends. The North Atlantic green turtle 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). Similarly, the South Atlantic DPS is large, estimated at over 63,000 nesting females, but 37 of the 51 identified nesting sites do not have sufficient data to estimate the number of nesters or trends (Seminoff et al. 2015). While the lack of data increases uncertainty, the overall trend of the South Atlantic DPS was not considered to be a major concern as some of the largest nesting beaches within the DPS appear to be increasing or stable. Nesting of green sea turtles from this DPS occurs on beaches of the USVI, primarily Sandy Beach and Buck Island, St. Croix. For hawksbill sea turtles, Mortimer and Donnelly (2008b) found that for nesting populations in the Atlantic (especially in the Insular Caribbean and Western Caribbean Mainland), nine of the ten sites with recent data (within the past 20 years) that show nesting increases were located in the Caribbean. Therefore, we believe the proposed action is not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of green and hawksbill sea turtles in the wild. The reduction in fitness of green and hawksbill sea turtles that could occur as a result of effects to juveniles related to the turbidity standards would not appreciably affect reproductive output of these sea turtle species.

The Atlantic Recovery Plan for the population of Atlantic green sea turtles (NMFS and USFWS 1991) 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 inhabiting coastal areas 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 (Lima et al. 2010; López-Barrera et al. 2012; Marcovaldi et al. 2008; Dow and Eckert 2007). Culebra Island, Puerto Rico, which borders the North and South Atlantic DPSs, is an important developmental habitat for juveniles and subadults based on capture data from 2000 to 2006 (Diez et al. 2007) and critical habitat was designated for green sea turtles around Culebra and its surrounding islands and cays in 1998.

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

- The adult female population is increasing, as evidenced by a statistically significant trend in the annual number of nests at five index beaches, including Mona Island (Puerto Rico) and Buck Island Reef National Monument (St. Croix).
- The number of adults, subadults, and juveniles is increasing, as evidenced by a statistically significant trend on at least five key foraging areas within Puerto Rico, USVI, 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 (USVI), all show increasing trends in the annual number of nests (USFWS/NMFS 2007b). In-water research projects at Mona Island, Buck Island, and the Marquesas, Florida, which involve the observation and capture of juvenile hawksbill turtles, are underway. Although there are over 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 (USFWS/NMFS 2007b).

The anticipated take of juvenile green and hawksbill sea turtles from indirect effects of the application of the VI “non-coral” turbidity criteria is not likely to reduce population numbers of these species over time given current population sizes and expected recruitment. Thus, the proposed action is not likely to impede the recovery objectives above for green and hawksbill sea turtles and will not result in an appreciable reduction in the likelihood of green and hawksbill sea turtles' recovery in the wild. We conclude the proposed action is not likely to jeopardize the continued existence of green and hawksbill sea turtles in the wild.

11.2 Nassau Grouper

As discussed previously, pH ranges allowed under the VIWQS include acidic values that will have fitness consequences on Nassau grouper associated with the suppression of growth and indirect effects of acidification on refuge and foraging habitat and prey availability. Turbidity levels that affect gill structure and habitat and prey distribution are also likely to have fitness consequences to Nassau grouper. Juvenile Nassau grouper are more likely to be affected by pH and “non-coral” turbidity criteria because of their use of nearshore habitats that serve as nursery areas because these areas are more likely to be exposed to decreased pH and increased turbidity than areas of deeper waters offshore. As shown in Table 10, there are areas with pH impairments and turbidity impairments are common throughout USVI. We need to determine whether the direct and indirect effects of the allowed pH and turbidity standards on the fitness of Nassau grouper will appreciably reduce the likelihood of survival and recovery of the species by decreasing their numbers, reproduction, or distribution.

Physical effects of acidic pH and chronic exposure to “non-coral” turbidity, along with the need to expend additional effort to find refuge and foraging habitat and prey species can result in reduced growth rates, physical impairment (if gills are damaged by sediment), and lower lifetime fecundity. However, no reduction in the distribution of Nassau grouper is expected and the species will continue to be present throughout water surrounding the USVI, though now in low numbers due to past overfishing.

Whether the potential effects to reproductive output would appreciably reduce the likelihood of survival of Nassau grouper depends on the probable effect the changes in reproductive output would have relative to current population sizes and trends. Population dynamic modeling by Goto and Wallace (2009) for the Bahamas found that fishing effort needs to be reduced significantly due to low spawning stock biomass. Nassau grouper were absent from 82 percent of shallow Caribbean reefs during a sampling period from 1997 to 2001 as part of underwater surveys conducted for a ReefCheck project (Hodgson and Liebeler 2002). Similarly, surveys conducted as part of NOAA CRCP projects have not observed Nassau grouper in various MPAs in USVI. However, there are few population analysis or stock assessments for the species that would enable us to determine population abundance. More recent surveys of spawning aggregations in USVI have documented some increases in adult Nassau grouper in one historic aggregation site (Kadison et al. 2009). Some countries, such as the Bahamas, have implemented spatial and seasonal protective measures to manage Nassau grouper and sightings of the species are still common in the Bahamas (NMFS 2013). Therefore, we believe the proposed action is not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of Nassau grouper in the wild. The reduction in reproduction of Nassau grouper that could occur as a result of effects to juveniles related to the application of the pH and "non-coral" turbidity criteria would not appreciably affect reproductive output of the species.

A recovery plan is not available for Nassau grouper but NMFS has developed a recovery outline (available at https://www.fisheries.noaa.gov/resource/document/nassau-grouper-recovery-outline). 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 Summary Assessment in the recovery outline concludes that the species shows little signs of recovery, requiring a two-pronged approach to conservation and recovery of the species. Specifically, spawning aggregations must continue to function throughout the species' range in order to provide larvae for future reproduction and recruitment and appropriate habitat must be available for settlement and growth across the Caribbean Sea.

To determine if the proposed action will appreciably reduce the likelihood of recovery for Nassau grouper, we evaluate the proposed action's impacts, if any, on the key elements of the recovery outline. The low abundance of the species, largely due to overfishing, combined with the need for nursery habitat, which is in nearshore waters that are prone to human impacts, exacerbates the vulnerability of Nassau grouper to extinction. Nassau grouper were historically present in large numbers around USVI based on fisheries catch records but are now rarely sighted. However, the proposed action will not affect the species' life history characteristics or vulnerability to overfishing. The proposed action will affect settlement and growth habitat due to the low pH values allowed under the VIWQS and turbidity. These effects will be chronic in those areas that regularly suffer impairment or where discharges regularly occur that are in compliance with the VIWQS but at levels that directly or indirectly affect the fitness of Nassau grouper juveniles. In addition, the reproductive potential of the species will be slightly reduced but is not expected to appreciably affect reproductive output of the species. Any effects to individuals from the proposed action will not affect the overall density and distribution of the species. Therefore, the impacts of the application of the pH and "non-coral" turbidity criteria under the proposed action are not likely to impede the recovery objectives identified in the recovery outline for Nassau grouper and will not result in an appreciable reduction in the likelihood of Nassau grouper's recovery in the wild. We conclude the proposed action is not likely to jeopardize the continued existence of Nassau grouper in the wild.

11.3 ESA-Listed Corals

As discussed in Section 9.3.3, ESA-listed corals are expected to be adversely affected by the application of the criteria for pH, "non-coral" turbidity, TP, carbaryl and nonylphenol pesticides, and copper, as well as the human health criteria for enterococci. These water quality constituents are expected to affect coral growth and reduce fertilization success and recruitment. We need to determine whether the direct and indirect effects of the allowed pH, "non-coral" turbidity, TP, enterococci, carbaryl, nonylphenol, and copper criteria on the fitness of ESA-listed coral species will appreciably reduce the likelihood of survival and recovery of the species by decreasing their numbers, reproduction, or distribution.

The abundance of elkhorn and staghorn coral is a fraction of what it was before the mass mortality in the 1970s and 80s and recent population models forecast the extirpation of elkhorn coral from some locations over the foreseeable future, including a site in Vieques, Puerto Rico, that was included in the Jackson et al. (2014b) report. Elkhorn coral abundance is at least hundreds of thousands of colonies but likely to decrease in the future with increasing threats. Staghorn coral abundance is at least tens of millions of colonies but likely to decrease in the future with increasing threats. The low density and cover of pillar coral and its natural rarity makes it difficult to extrapolate monitoring data in order to determine trends in abundance. Based on information in our project files from other consultations, pillar coral appears to be more common around Puerto Rico and USVI in general than in South Florida (NOAA NCRMP). Rough cactus coral is naturally uncommon to rare as stated in the listing rule, though the species may be more common in some sites around Puerto Rico and USVI based on data from EPA's bioassessments conducted in 2010 and 2011. Estimates of the populations of pillar and rough cactus corals are not possible due to the limited survey data for these species. The star coral complex (lobed star, mountainous star, and boulder star) has historically been dominant on coral reefs in the Caribbean, though examples from various countries and the U.S. (Florida Keys, Puerto Rico, and USVI) indicate the population has declined. Despite these declines, it is estimated that there are millions of colonies present throughout the range of this species' complex. Therefore, any reduction in numbers as a result of the direct and indirect effects of changes in water quality condition associated with the application of the pH, "non-coral" turbidity, TP, enterococci, carbaryl, nonylphenol, and copper criteria by the VIDPNR on ESA-listed corals is expected to be minimal, but will result in a loss of reproductive potential in addition to a loss of future recruits.

Despite the loss of some reproductive potential resulting from the proposed action, we do not believe that sexually mature colonies of ESA-listed corals will be affected to a degree that will cause short or long-term damage to the ability of each species to sexually reproduce. Therefore, although we believe there will be a small loss of reproductive potential and future colonies, we do not expect that the proposed action will alter the geographic range of the species. We also do not anticipate that the reduction in numbers and reproductive potential resulting from this project would represent a detectable reduction in reproduction of elkhorn, staghorn, pillar, rough cactus, lobed star, mountainous star, and boulder star corals in the action area. No change in distribution of lobed star, mountainous star, or boulder star corals is expected as a result of the proposed action and these species will continue to be present in waters around the USVI.

Whether the reduction in numbers and reproduction of ESA-listed coral species would appreciably reduce their likelihoods of survival depends on the probable effects these changes would have relative to current population levels and trends. Elkhorn and staghorn coral, pillar coral, rough cactus coral, and all three species from the star coral complex are reported on reefs in USVI and corals from the star coral complex are the dominant species in many areas. As noted previously, we are not able to determine the absolute abundance of pillar and rough cactus coral due to the natural rarity of these species but survey data from around Puerto Rico and USVI indicate that these species are likely more common in the U.S. Caribbean than in Florida. Elkhorn, staghorn, and all three species from the star coral complex are thought to have absolute abundances in the tens of millions based on available data from a few locations such as Florida and St. Croix. Any loss of reproductive potential due to the proposed action will not measurably impact the species' abundance around the USVI or throughout their ranges. Therefore, we believe any loss of colonies and future recruits associated with impacts from the application of the pH, “non-coral” turbidity, TP, carbaryl and nonylphenol, and copper criteria will not have any measurable effect on the overall population of ESA-listed coral species and will not appreciably reduce the ability of ESA-listed coral species to survive in the wild.

Now we evaluate whether the expected reduction in reproduction and future recruitment of ESA-listed corals will appreciably reduce the likelihood of the species' recovery in the wild. The recovery plan for elkhorn and staghorn corals notes that elkhorn and staghorn corals continue to decline and are at only a small percentage of their historic abundance throughout their ranges. The recovery plan outlines a recovery strategy for the species as populations large enough so that successfully reproducing individuals comprise numerous populations across the historic 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 (NMFS 2014). The most relevant recovery criteria to the impacts expected from the proposed action include:

- ***Objective 1: Ensure Population Viability***
    - **Criterion 1: Abundance**
        - **Elkhorn coral:** Thickets are present throughout approximately 10 percent of consolidated reef habitat in 1 – 5 m water depth within the forereef zone. Thickets are defined as either a) colonies $\geq$ 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 – 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.

  - **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.
- ***Objective 2: Eliminate or Sufficiently Abate Global, Regional, and Local Threats***
  - **Criterion 6: Loss of Recruitment Habitat**
    - Abundance (Criterion 1 above) addresses the threat of Loss of Recruitment Habitat because the criterion specifies the amount of habitat occupied by the 2 species.If Criterion 1 is met, then this threat is sufficiently abated; or
    - Throughout the ranges of these 2 species, at least 40 percent of the consolidated reef substrate in 1 – 20 m depth within the forereef remains free of sediment and macroalgal cover as measured on a broad reef to regional spatial scale.
  - **Criterion 7: Nutrients, Sediments, and Contaminants (Land-Based Sources of Pollution)**
    - Develop quantitative recovery criteria through research. Based on 5 years of data, criteria will be established to reduce sources of nutrients, sediments, and contaminants to levels appropriate for recovery.
  - **Criteria 8: Regulatory Mechanisms**
    - Adequate domestic and international regulations and agreements are adopted as necessary to ensure that all threat-based recovery criteria are met. For example, appropriate local, state/territorial, national, international, and multi-jurisdictional efforts, agreements, and regulations are necessary to abate the threats from LBSP, physical impacts to corals, and rising ocean temperatures and ocean acidification resulting from increasing atmospheric CO_2 concentrations.

Pillar, rough cactus, lobed star, mountainous star, and boulder star corals were listed in September 2014 and NMFS does not have an extensive consultation history for these species or a recovery plan. NMFS has developed a recovery outline for these species (available at http://sero.nmfs.noaa.gov/protected_resources/coral/documents/recovery_outline.pdf). As for Nassau grouper, 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. The Summary Assessment in the recovery outline concludes that population trends for rough cactus and pillar coral are unknown but abundance is very low and that populations of star coral species are on the decline. Thus, recovery will depend on successful sexual reproduction and reduction of mortality of existing populations. The key challenges to achieving recovery will be moderating the impacts of ocean warming associated with climate change and decreasing the species' susceptibility to disease, which may be furthered through reduction of local stressors with recovery requiring an ecosystem approach including habitat protection and a reduction in threats caused by human activity.

In terms of the recovery objectives, the proposed action is not expected to reduce the overall abundance of ESA-listed corals around the USVI though the project is expected to result in a small loss of numbers. The project is expected to result in decreases in reproductive potential and future recruits due to the effects of the application of the pH, “non-coral” turbidity, TP, carbaryl, nonylphenol, and copper criteria. Turbidity is also expected to affect settlement and recruitment habitat for ESA-listed corals. However, we do not believe there will be an appreciable reduction in the likelihood of recovery in the wild for ESA-listed corals. We base this conclusion on the fact that the VIWQS are meant to reduce LBSP through improving regulatory mechanisms to protect ESA resources and can continue to change in response to findings such as the data required to inform recovery criterion 7 for elkhorn and staghorn corals and will not increase the magnitude of threats that led to listing of all of these coral species. Therefore, we do not believe the proposed action will jeopardize the continued existence of ESA-listed corals in the wild.

11.4 Elkhorn and Staghorn Coral Critical Habitat

When determining the potential impacts to critical habitat for this Opinion, NMFS relies on the regulatory definition of “destruction or adverse modification” of critical habitat from the final rule issued by NMFS and USFWS (81 FR 7214) on February 11, 2016. Under the final rule, destruction or adverse modification means a direct or indirect alteration that appreciably diminishes the value of critical habitat for the conservation of a listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features.

Ultimately, we seek to determine if, with the implementation of the proposed actions, critical habitat would remain functional (or retain the current ability for the essential features to become 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 or because of significant delays in the development of these features. This analysis takes into account the geographic and temporal scope of the proposed actions, 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 characters of the critical habitat that will be required over time to support the successful recovery of the species.

Within the St. Thomas/St. John and St. Croix critical habitat units, 300 km^2 are likely to contain the essential feature of elkhorn and staghorn coral critical habitat based on the amount of coral, rock reef, colonized hard bottom, and other coralline communities mapped by NOAA in 2000 (Kendall et al. 2001). The key objective for the conservation and recovery of elkhorn and staghorn 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 reproduction success. Man-made stressors have the greatest impact on habitat quality for elkhorn and staghorn corals because degradation or loss of substrate for sexual and asexual recruits to settle impacts reproductive success, growth, and recovery of these species.

Therefore, the key conservation objective of designated elkhorn and staghorn coral critical habitat is to increase the potential for successful sexual and asexual reproduction, 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 Elkhorn and Staghorn Coral Critical Habitat* (Section 6.2.7), the *Environmental Baseline* (Section 8), the *Effects of the Action* (Section 9), and *Cumulative Effects* (Section 10).

The essential feature of critical habitat for elkhorn and staghorn coral is substrate of adequate quantity and quality to allow for settlement and growth where adequate quality refers to the need for hard substrate to be free of high macroalgal growth and sediment cover as these impede the settlement and growth of elkhorn and staghorn corals. Thus, we need to assess whether the “non-coral” turbidity standard and associated effects of turbidity on elkhorn and staghorn coral critical habitat rise to the level of adversely modifying or destroying the designated critical habitat when considered as a whole. Specifically, whether the “non-coral” turbidity standard will result in diminished function of the essential feature of elkhorn and staghorn coral critical habitat such that settlement and growth of sexual and asexual recruits are impaired, also affecting the recovery criteria for elkhorn and staghorn corals.

Based on the information in Table 10, a number of areas around USVI are impaired due to turbidity. While the VIDPNR has prioritized areas for the establishment of TMDLs due to turbidity impairment, sediment loading has not yet been measured and quantified to determine the relationship between turbidity and sediment loading. The VIDPNR has contracted a company to assess loading in some watersheds feeding a waterbody that has a TMDL (or TMDLs) for other constituents, the most common being fecal coliform and DO (DPNR 2016). Non-point sources of pollution, including those contributing to turbidity in waterbodies around the USVI, are expected to increase, particularly in St. Thomas and St. John where development has continued at a fast rate.

Our analysis indicated that “non-coral” turbidity is likely to have chronic effects on elkhorn and staghorn coral critical habitat, which are expected to be measurable in areas where impairment due to turbidity is common. On the other hand, the 2018 VIWQS have two allowed turbidity levels, one for areas containing corals and one for non-coral areas, with the one for coral areas more likely to be protective. It is not clear whether coral and non-coral areas can be clearly defined in a way that will ensure that elkhorn and staghorn coral critical habitat will not be affected by the higher allowed turbidity criterion of 3 NTU (for "non-coral" areas), leading to chronic sedimentation of hard substrate containing the essential feature. However, turbidity levels over background conditions, even those within the turbidity standards allowed under the 2018 VIWQS, are expected to be localized and are not expected to result in the loss or degradation of large areas containing the essential feature of elkhorn and staghorn coral critical habitat. We base this on the current presence of elkhorn and staghorn corals in areas containing the essential feature around USVI, including some of the areas where turbidity impairments have been documented. Therefore, we do not expect the effects of the application of the "non-coral" turbidity criterion allowed under the 2018 VIWQS to appreciably diminish the overall value of the designated critical habitat for the conservation of elkhorn and staghorn corals. Therefore, we conclude that the proposed action will not result in the destruction or adverse modification of elkhorn and staghorn coral critical habitat in the St. Thomas/St. John or St. Croix units.

12 CONCLUSION

After reviewing the current status of the ESA-listed species and designated critical habitat, the environmental baseline within the action area, the effects of the proposed action, any effects of interrelated and interdependent actions, and cumulative effects, it is NMFS biological opinion that the proposed action is not likely to jeopardize the continued existence of hawksbill sea turtles or the green sea turtle North or South Atlantic DPS; Nassau grouper; or elkhorn, staghorn, rough cactus, pillar, lobed star, mountainous star, or boulder star corals; or destroy or adversely modify elkhorn and staghorn coral designated critical habitat.

It is also NMFS opinion that the proposed action may affect, but is not likely to adversely affect blue, fin, sei, and sperm whales; scalloped hammerhead and oceanic whitetip sharks; leatherback and loggerhead sea turtles (Northwest Atlantic DPS); and giant manta ray; and will have no effect on leatherback sea turtle designated critical habitat.

13 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. "Take" is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by regulation to include significant habitat modification or degradation that results in death or injury to ESA-listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering (see 50 C.F.R. §222.102).

Harass is further defined as an act that “creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering” (NMFSPD 02-110-19).

Incidental take is defined as take that results from, but is not the purpose of, the carrying out of an otherwise lawful activity (see 50 C.FR. §402.02). Section 7(b) (4) and section 7(o) (2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this incidental take statement.

Green and hawksbill sea turtles, Nassau grouper, and ESA-listed corals analyzed in this opinion will be exposed to concentrations of certain water quality constituents in the action area that are likely to result in reductions in fitness of these species. For Nassau grouper and ESA-listed corals, reductions in fitness will be due to direct effects of water quality constituents such as pH and turbidity on growth and health of individuals of these species. Concentrations of the water quality constituents identified in this opinion as likely to adversely affect green and hawksbill sea turtles, Nassau grouper, and ESA-listed corals are also likely to cause habitat degradation that will impact these species by reducing the availability of suitable prey organisms, refuge and foraging habitat, and future recruitment habitat thereby impairing essential behavioral patterns of feeding and reproduction. Juvenile sea turtles and Nassau grouper are most likely to be affected as well as all life stages of ESA-listed corals.

13.1 Amount or Extent of Take

Section 7 regulations require NMFS to specify the impact of any incidental take of endangered or threatened species; that is, the amount or extent, of such incidental taking on the species (50 C.F.R. § 402.14(i) (1) (i)). The amount of take represents the number of individuals that are expected to be taken by actions while the extent of take or “the extent of land or marine area that may be affected by an action” may be used if we cannot assign numerical limits for animals that could be incidentally taken during the course of an action (51 FR 19953).

Where it is not practical to quantify the number of individuals that are expected to be taken by the action, a surrogate (e.g. similarly affected species or habitat or ecological conditions) may be used to express the amount or extent of anticipated take (50 C.F.R. §402.14(i) (1) (i)).

Incidental take caused by the direct and habitat-related effects of this action cannot be accurately quantified as the number of juvenile green and hawksbill sea turtles and juvenile Nassau grouper because the number of sea turtles and fish at a given location at a given time is affected by numerous biotic and abiotic factors such as habitat quality and availability, competition and predation, and interactions among these and other factors. These factors interact in ways that may be random or directional, and may operate across broader temporal and spatial scales than are affected by the proposed action. Thus, the distribution and abundance of juvenile green and hawksbill sea turtles and juvenile Nassau grouper within the action area cannot be attributed entirely to their response to water quality constituents or habitat conditions. In addition, NMFS cannot precisely predict the number of animals of each species that are reasonably certain to demonstrate behavioral and injurious effects due to the presence of certain water quality constituents in concentrations allowed under the VIWQS and associated impacts to refuge and foraging habitat of these species. Also, there is no feasible way to count, observe, or determine the number of individuals of each species that would be affected by exposure to turbidity and additionally to acidic pH values in the case of Nassau grouper. This is because the effects of the action will occur over a large geographic area (all waters of the USVI, including marine, coastal, and inland waters) and effects may occur in areas where animals are not likely to be observed due to water depth or remoteness of the area. In addition, even if affected animals are observed, it is unlikely that the exact cause of injury, mortality or behavioral effects could be determined. Further, sublethal effects of the proposed action could manifest later in time at locations where animals cannot be readily observed such as during travel from the USVI to other habitats in the Caribbean or beyond such as spawning aggregation sites and nesting beaches.

Similarly, incidental take caused by the direct and habitat-related effects of this action on ESA-listed coral colonies and future recruits cannot be accurately quantified as numbers of current and future ESA-listed coral colonies because there has not been an inventory of ESA-listed coral colonies completed for the USVI nor do we have an accurate assessment of the reproductive success of existing colonies. As for sea turtles and Nassau grouper, the presence of ESA-listed corals and the suitability of sites to attract recruits is affected by a number of biotic and abiotic interacting factors. The distribution and abundance of ESA-listed coral colonies and future recruits of these species cannot be attributed solely to their response to water quality constituents or habitat conditions. NMFS also cannot precisely predict the number of colonies of each species that are reasonably certain to demonstrate physiological responses or the number of future recruits that are reasonably certain to reject areas rather than settling due to the presence of certain water quality constituents in concentrations allowed under the VIWQS. Individual colonies of the same species of coral have been found to demonstrate varying responses to stressors due to genotypic variation, as well as variation in their symbionts (zooxanthellae). It is not feasible to easily determine the number of individuals of each species affected by exposure to water quality conditions resulting from application of the criteria by the VIDPNR for pH, "non-coral" turbidity, TP, pesticides (carbaryl and nonylphenol, which are included in the standards), and copper. Similarly, it is not feasible to easily determine the number of individuals of each species affected by a combination of these water quality constituents versus exposure to other stressors because of the size of the action area, the location where effects may occur, the likelihood of varying responses due to genotypic variation, and the lack of information on interactions among stressors. Further, the effects on future recruitment cannot be readily observed within extensive monitoring of coral spawning and settlement throughout the Territory or detailed laboratory experiments.

NMFS will use quantitative measurements of ambient concentrations of TP and turbidity in shallow water habitat suitable for green and hawksbill sea turtles, Nassau grouper, and ESA-listed corals as surrogates for incidental take of each species due to the proposed action. As discussed in Section 9.2.3, NOAA conducted benthic mapping around the USVI using aerial photography and identified 24 km^2 of unconsolidated sediment, 2 km^2 of mangroves, 161 km^2 of submerged aquatic vegetation, and 300 km^2 of coral reef and hard bottom in waters with depths to 30 m (Kendall et al. 2001). Of these habitats, submerged aquatic vegetation (dominated by seagrass) and coral reef and hard bottom are the ones that are used as habitat by the ESA-listed species discussed here. In terms of elkhorn and staghorn coral critical habitat, as discussed previously, of the area within the St. Thomas/St. John unit, approximately 67 km^2 are likely to contain the essential element of ESA-designated elkhorn and staghorn coral critical habitat based on the amount of mapped coral, rock reef, colonized hard bottom, and other coralline communities (Kendall et al. 2001). Of the area within the St. Croix unit, approximately 233 km^2 are likely to contain the essential feature of ESA-designated elkhorn and staghorn coral critical habitat based on the amount of coral habitats mapped (Kendall et al. 2001). These areas containing the essential feature are expected to contain colonies of elkhorn and staghorn coral, as well as colonies of the other ESA-listed coral species for which critical habitat has not been designated. Similarly, seagrass and coral areas are expected to contain resident populations of green and hawksbill sea turtles and Nassau grouper. In order to determine the location and extent of seagrass and coral areas, the following sources should be used by VIDPNR to create habitat maps in coordination with NMFS: NOAA benthic habitat maps developed in 2000 supplemented with data from NOAA's NCRMP; TCRMP data; data from site inspections by VIDPNR for proposed in-water construction projects and benthic surveys conducted for these projects; and other benthic habitat data that may be available from sources such as UVI.

Impacts to the habitat areas for green and hawksbill sea turtles, Nassau grouper, and ESA-listed corals as a result of criteria in the 2018 VIWQS are expected to result in impacts to the species. TP and turbidity are commonly discharged throughout the action area and are among the criteria most likely to cause adverse effects and thus contribute to incidental take, particularly of ESA-listed corals, which are the most sensitive of the species for which incidental take is anticipated. TP and turbidity are also readily measured and habitat type is easily observed. Chronic inputs of TP and levels of turbidity from discharges allowed under the VIWQSR will be monitored because they provide a more continuous environmental concentration that can be monitored. Acute concentrations are more likely to be exceeded in localized areas for short periods and as such do not allow for an analysis of trends. The use of ambient concentrations of TP and turbidity combined with affected habitat area as surrogates is quantifiable and may be monitored, serving the intended role as a reinitiation trigger. NMFS proposes to use the existing DPNR water quality monitoring program sites in areas containing coral and seagrass habitats to determine whether the extent of take is exceeded.

In order to comply with this ITS, EPA will need to ensure that monitoring for ambient concentrations of TP and turbidity occurs at DPNR sample sites in accordance with a revised monitoring plan for sites containing seagrass and coral. We recognize that TP is not part of the current monitoring program and turbidity measurements are often taken infrequently. Therefore, a revised monitoring plan will be developed and implemented. The extent of take for a given

ESA-listed species will be exceeded if: 25 percent or more of the sampling sites containing coral or seagrass around one or more islands (St. Croix, St. Thomas, St. John) have median values that exceed the thresholds of one or more of the water quality constituents selected for monitoring (specifically, TP and turbidity) over two consecutive sampling events.

NMFS recognizes that there will be a time lag between promulgation of the 2018 WQSR and the incorporation of the WQSR within the terms of all TPDES permits and other actions taken by the VIDPNR. To account for this lag period, the extent of take indicator will be triggered two years after the promulgation of the standards.

13.2 Reasonable and Prudent Measures

The measures described below are nondiscretionary, and must be undertaken by the EPA so that they become binding conditions for the exemption in section 7(o)(2) to apply. Section 7(b) (4) of the ESA requires that when a proposed agency action is found to be consistent with section 7(a)(2) of the ESA and the proposed action may incidentally take individuals of ESA-listed species, NMFS will issue a statement that specifies the impact of any incidental taking of endangered or threatened species. To minimize such impacts, RPMs, and term and conditions to implement the measures, must be provided. Only incidental take resulting from the agency actions and any specified reasonable and prudent measures and terms and conditions identified in the incidental take statement are exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA.

RPMs are nondiscretionary measures to minimize the amount or extent of incidental take (50 C.F.R. §402.02). NMFS believes the RPMs described below are necessary and appropriate to minimize the impacts of incidental take on ESA-listed species:

1. EPA shall ask that VIDPNR modify Territorial water quality and biological monitoring programs to collect data regarding physical and chemical water quality constituents, including pH, turbidity, TP, carbaryl, nonylphenol, and copper, and the effects of concentrations of these constituents on seagrass and coral habitats to determine the concentrations at which changes in habitat condition are observed to ensure the standards are protective of ESA-listed corals, green and hawksbill sea turtles, and Nassau grouper. The revised monitoring plan will be developed within 12 months and the plan finalized within two years of the date the VIWQS are promulgated and will be implemented within 6 months of approval of the final revised monitoring plan.
2. EPA shall ask VIDPNR to establish TMDLs (see discussion in Section 11.4 about TMDL development prioritization) in all areas identified as high and medium priority where water quality constituents are found to result in adverse impacts to seagrass and coral habitat conditions based on the availability of data to quantify anthropogenic pollutant loads and model their fate and effects, and pending availability of resources. TMDLs will be drafted within five years of promulgation of the 2018 VIWQS.

13.3 Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the EPA must comply with the following terms and conditions, which implement the Reasonable and Prudent Measures described above. These include the take minimization, monitoring and reporting measures required by the section 7 regulations (50 C.F.R. §402.14(i)). These terms and conditions are non-discretionary. If the EPA fails to ensure compliance with these terms and conditions and their implementing reasonable and prudent measures, the protective coverage of section 7(o) (2) may lapse.

The following terms and conditions implement RPM 1:

1. EPA shall ask that VIDPNR collect and summarize available water quality and biological monitoring data from existing federal and territorial monitoring programs, permits that required biological and/or water quality monitoring, literature reviews of work conducted in the USVI, data collected by UVI researchers, and data collected by federal, territorial, and non-governmental organizations over the past 10-20 years. Historic data will be collected and analyzed to determine whether there is a correlation between biological and water quality data based on the available information within 12 months of the date the 2018 VIWQS are promulgated, and the results will be shared with NMFS.
    a. These water quality and biological monitoring data will be grouped by location around the USVI. Data will then be grouped by sampling date to determine the extent to which water quality and biological monitoring data can be paired both spatially and temporally.
    b. In coordination with NMFS, the data will be compiled into a format that will allow for determination of whether information on acidic pH, turbidity levels, TP concentrations, pesticide concentrations (carbaryl and nonylphenol), and/or copper concentrations from different locations can be paired with information on the condition of seagrass and coral habitat from monitoring in the same locations.
    c. If available, rainfall data will also be incorporated into the dataset because samples collected during rain events could bias water quality monitoring results.
    d. The results of this data analysis process can be used to inform the selection of sampling sites and water quality constituents to be targeted under the revised monitoring plan.
2. EPA shall ask VIDPNR to modify their ambient monitoring plan in coordination with NMFS. The draft revisions to the monitoring plan will be prepared within 12 months and the plan finalized within 2 years of the date the 2018 VIWQS are promulgated. The proposed revisions will be shared with NMFS for review and comment and a copy of the final plan will be sent to NMFS. The implementation of the monitoring program under the revised plan will begin within 6 months of the plan being finalized.
    a. The revised plan should integrate existing water quality and biological monitoring programs in the USVI to collect measurements of physical and chemical water

quality constituents, including pH, turbidity, TP, carbaryl, nonylphenol, and copper, in the same locations as data on the cover, species, and condition of seagrass and coral habitats and ESA-listed coral colonies. This information will be used to determine the concentrations of water quality constituents at which changes in habitat condition and condition of ESA-listed corals are observed.

b. Seagrass condition monitoring should include determinations of the spatial extent of seagrass, seagrass abundance, species composition, and macroalgal distribution and cover (see, for example, Madden et al. 2009). Other indicators such as leaf width, leaf area index, shoot density by species, epiphyte cover, and sediment cover can also be included. Coral condition monitoring should include determinations of the percent live cover of hard and soft corals, algal distribution and cover, extent of bleaching or disease on individual coral colonies, species diversity and relative abundance, and species density and size.

c. Water quality and biological condition monitoring should be done simultaneously at sampling sites to the extent practicable. If data from other on-going monitoring programs, such as TCRMP, will be used to supplement data collected as part of the revised ambient monitoring program, the VIDPNR will develop a standard protocol for monitoring, data recording, and submission of data to VIDPNR.

d. In addition to data on water quality and biological indicators, the plan will require collection of location in decimal degrees, date, time, sea state[6], and rainfall in order to analyze patterns of water quality and biologic condition based on time of year, weather conditions, and other physical and climate factors. Sampling will be conducted on a regular schedule and will occur despite weather conditions as is done for the VIDPNR beach monitoring program. Based on notes of field conditions taken during sampling, decisions can be made as to whether or not to exclude data from certain sampling events due to weather conditions as not being representative of "normal" conditions.

e. The data collected under the revised monitoring plan will inform future triennial reviews and be used to assess whether the 2018 VIWQS standards are protective of ESA-listed corals, green and hawksbill sea turtles, and Nassau grouper.

The following terms and conditions implement RPM 2:

1. In order to focus efforts on waterbodies where water quality constituents are found to adversely affect ESA resources, EPA shall:
    a. Ask VIDPNR to develop clear definitions of high, medium, and low priority based on both CWA 303(d) requirements and the presence of ESA-listed corals, green and hawksbill sea turtles, and Nassau grouper in impaired waterbodies.

[6] The general condition of the sea surface with respect to wind waves and swell characterized by wave height, period, and power spectrum.

b. Collaborate with NMFS to prioritize the development of TMDLs for high and medium priority waterbodies where the results of the analysis of existing data and the monitoring program (RPM 1) indicate water quality constituents may be adversely affecting seagrass and coral habitat condition and/or ESA-listed corals. The final results of the prioritization process will also be sent to NMFS.
c. Support VIDPNR (e.g., provide technical assistance) in drafting TMDLs for TP and turbidity in priority waterbodies within 5 years of the date of final EPA approval and promulgation of the 2018 VIWQS by the VIDPNR given the required schedule for completion of the analysis of existing data and the implementation of additional sampling. The draft TMDLs for TP and turbidity in priority watersheds will be shared with NMFS. If VIDPNR does not initiate TMDL development and complete TMDLs in priority watersheds where impairment occurs due to turbidity and/or TP, EPA will pursue options for completion of TMDLs.
d. Support VIDPNR (e.g., provide technical assistance) in the establishment of water quality standards for any water quality constituents identified through the implementation of RPM 1 as causing adverse impacts to seagrass and coral habitat and/or ESA-listed corals followed by the drafting of TMDLs in priority waterbodies where impairment occurs due to these constituents. Because a schedule for this process cannot be determined at this time, EPA will collaborate with VIDPNR and NMFS to establish the timing for new standards and TMDLs. The establishment of revised or additional standards may require a new ESA section 7 consultation.

14 Conservation Recommendations

Section 7(a) (1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on ESA-listed species or critical habitat, to help implement recovery plans or develop information (50 C.F.R. §402.02).

We believe the following conservation recommendation would further the conservation of ESA-listed species and designated critical habitat in the USVI:

1. EPA, in partnership with the VIDPNR, should develop updated, detailed maps of Class A, Class B, and Class C waters in order to clearly define the boundaries of the different classes. Detailed mapping would also enable a clear determination of the embayments and other areas where less strict WQS are authorized under the 2018 VIWQS that can be used to establish sampling locations as part of the water quality monitoring plan. This will enable the EPA and VIDPNR to assess whether there need to be changes in classification of waters or areas where stricter WQS should be required, particularly in

areas containing the essential features of designated critical habitat and/or known to be used by ESA-listed species.

2. EPA should encourage the VIDPNR to request that new permit applicants conduct benthic surveys in the area or areas where discharges to estuarine, nearshore, or offshore waters of the USVI will occur as a result of their proposed actions to determine whether corals are present to ensure any permits issued have the appropriate standards incorporated. Benthic habitat maps available for USVI are limited because the pixel size of the maps is one acre, meaning many areas that may contain ESA-listed corals may not be accurately mapped. This information should be used to improve the definition of coral and non-coral areas included in the 2018 VIWQS to ensure criteria for water quality conditions such as turbidity are applied correctly.
3. EPA should encourage the VIDPNR to conduct water quality and biological sampling for pollutants of emerging concern, particularly endocrine disruptors, and other pollutants such as chlorine for which there is evidence that these compounds have some level of toxicity to marine organisms to determine whether the VIWQS should include standards to protect ESA resources from discharges of these pollutants.
4. EPA should work with the VIDPNR to include monitoring for TSS in the ambient water quality monitoring program in order to obtain data to allow a determination of whether new criteria for TSS should be included in a future revision of the VIWQS.
5. EPA should work with the VIDPNR to incorporate fecal bacteria monitoring, preferably using MST methods, with coral sampling to evaluate the link between human fecal bacteria contamination and coral disease. Information from this monitoring should be used to determine whether aquatic life criteria for fecal bacteria are needed to be protective of ESA-listed corals.
6. EPA should work with the VIDPNR to incorporate monitoring for pesticides commonly used in anti-fouling bottom paint (e.g., Irgarol) on vessels in the USVI to determine whether and in what concentrations these compounds are present, particularly in areas frequented by vessels, and assess the effect on ESA-listed corals, if present in areas with detectable concentrations. Information from this monitoring should be used to determine whether the VIWQS should include limits on concentrations of certain pesticides to be protective of ESA-listed corals.
7. EPA should work with the VIDPNR to ensure that all monitoring data collected are either included in one of EPA's existing databases or ecosystem tools (i.e., STORET, ESML, ENVIROATLAS), a database is created, or an existing database managed by VIDPNR or a partner such as the university is used to store monitoring data collected under the restructured program that will be available for use by NMFS and others to view and analyze monitoring results.
8. We request that NMFS OPR ESA Interagency Cooperation Division be provided with copies of all monitoring report generated as a result of the implementation of

recommendations 2, 3, 4, 5, and 6 above unless these are included in the database suggested in recommendation 7.

In order for NMFS OPR Endangered Species Act Interagency Cooperation Division to be kept informed of actions minimizing or avoiding adverse effects on, or benefiting, ESA-listed species or their critical habitat, EPA should notify the Endangered Species Act Interagency Cooperation Division of any conservation recommendations they implement in their final action.

15 Reinitiation Notice

This concludes formal consultation for EPA, Region 2 for the proposed approval of the 2018 VIWQS. As 50 C.F.R. §402.16 states, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if:

(1) The amount or extent of taking specified in the incidental take statement is exceeded.
(2) New information reveals effects of the agency action that may affect ESA-listed species or designated critical habitat in a manner or to an extent not previously considered.
(3) The identified action is subsequently modified in a manner that causes an effect to ESA-listed species or designated critical habitat that was not considered in this opinion.
(4) A new species is listed or critical habitat designated under the ESA that may be affected by the action.

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17 APPENDICES

17.1 Appendix A. 2018 Virgin Islands Water Quality Standards

UNITED STATES VIRGIN ISLANDS
WATER QUALITY MANAGEMENT PROGRAM
WATER QUALITY STANDARDS
RULES AND REGULATIONS

Title 12, Chapter 7, Subchapter 186

GOVERNMENT OF THE US VIRGIN ISLANDS
DEPARTMENT OF PLANNING AND NATURAL RESOURCES
DIVISION OF ENVIRONMENTAL PROTECTION

UNITED STATES VIRGIN ISLANDS
WATER QUALITY MANAGEMENT PROGRAM
WATER QUALITY STANDARDS
RULES AND REGULATIONS
TITLE 12, CHAPTER 7, SUBCHAPTER 186

APPROVED:

Dated: ____________________, 2018

____
Dawn L. Henry, Esq., Commissioner
Department of Planning and
Natural Resources

APPROVED:

Dated: _____________________, 2018

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Kenneth E. Mapp,
Governor of the
United States Virgin Islands

I, Osbert E. Potter, Lieutenant Governor of the United States Virgin Islands, have reviewed the foregoing Rules and Regulation, Title 12, Chapter 7, Subchapter 186, find them to be in compliance with Title 3, Chapter 35, Virgin Island Rules and Regulation, and hereby approve the same in accordance with Title 3, Section 936.

Dated: _____________________, 2018

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Osbert E. Potter
Lieutenant Governor of the
United States Virgin Islands

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§ 186 Legal Basis

This Regulation is promulgated in accordance with 12 V.I.C. § 186, as amended, known as the Standards of Water Quality VI Code, and this version supersedes and nullifies any previous provision, resolution, agreement or regulation of the Government of United States Virgin Islands relating to USVI Water Quality Standards.

§ 186 - 1: Definitions

The following words and terms, when used in these standards, shall have the following meanings, unless the context clearly indicates otherwise.

(a) Abiotic: A nonliving physical and chemical attribute of a system such as light, temperature, wind patterns, rocks, soil, pH, and pressure in an environment.

(b) Acute: Refers to a stimulus severe enough to rapidly induce an effect; in aquatic toxicity tests, an effect observed in 96-hours or less is typically considered acute. When referring to aquatic toxicology or human health, an acute affect is not always measured in terms of lethality.

(c) Aesthetic qualifications: Criteria that by definition preserves one's ability to experience natural beauty.

(d) Aquatic Nuisance Species: A nonindigenous species that threatens the diversity or abundance of native species or the ecological stability of infested waters, or commercial, agricultural, aquacultural or recreational activities dependent on such waters.

(e) Assimilative capacity: The natural capacity of a water body to dilute and absorb pollutants and prevent harmful effects (e.g., damage to public health or physical, chemical, biological quality of the water).

(f) Best management practices or BMPs: Schedules of activities, prohibitions of practices, maintenance procedures, and other management practices to prevent or reduce point and non-point source pollution to “waters of the United States” or “territorial waters.” BMPs also include treatment requirements, operating procedures, and practices to control plant site runoff, spillage or leaks, sludge or waste disposal, or drainage from raw material storage.

(g) Bioaccumulation factor or BAF: The ratio of a substance's concentration in tissue versus its concentration in ambient water, in situations where the organism and the food chain are exposed.

(h) Biological criteria or biocriteria: Narrative expressions or numeric values of the biological characteristics of aquatic communities based on appropriate reference conditions. As such, biological criteria serve as an index of aquatic community health.

(i) Biological Integrity: The capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region

(j) Biotic: Living or of life.

(k) Brackish Water: A mixture of seawater and fresh water whose salinity range is between 0.5 to 35 ppt.

(l) CFR: The Code of Federal Regulations.

(m) Chronic: A stimulus that lingers or continues for a relatively long period of time, often one-tenth of the life span or more. Chronic should be considered a relative term depending on the life span of an organism. The measurement of a chronic effect can be reduced growth, reduced reproduction, etc., in addition to lethality.

(n) Class A Waters: Marine and coastal waters designated as Outstanding National Resource Waters that have unique characteristics to be preserved (e.g., waters of exceptional recreational, environmental, or ecological significance). Class A Waters are designated for the maintenance and propagation of desirable species of wildlife and aquatic life, for primary contact recreation and as a potable water source where applicable.

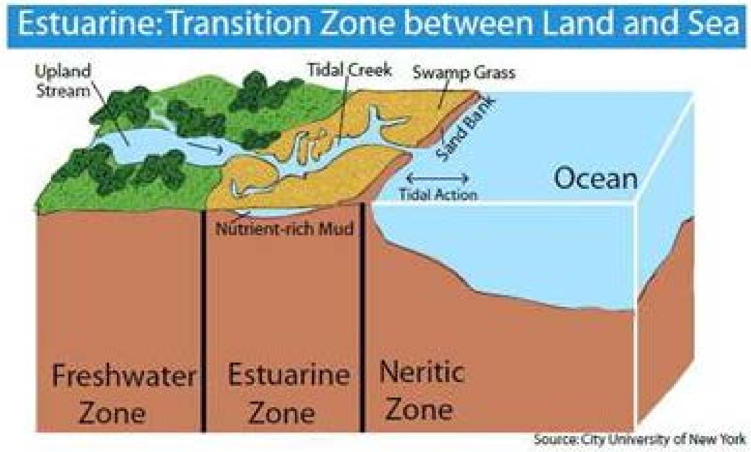
(o) Class B Waters: Marine and coastal waters designated for the maintenance and propagation of desirable species of wildlife and aquatic life, for primary contact recreation and as a potable water source where applicable.

(p) Class C Waters: Marine and coastal waters designated for the maintenance and propagation of desirable species of wildlife and aquatic life, for primary contact recreation and as a potable water source where applicable. Class C waters are those waters which are located in industrial harbors and ports and have less stringent water quality standards for certain parameters than Class B waters.

(q) Class I Waters or Inland Waters: Designated aquatic-influenced environments located within land boundaries. Inland water systems are designated for the maintenance and propagation of desirable species of wildlife and aquatic life, for primary contact recreation and as a potable water source where applicable. Waters included in this class can be either inland groundwaters (Subclass IG waters) or inland surface waters. Inland surface waters can be fresh (Subclass IF Waters) as well as saline or brackish (Subclass IBS Waters).

(r) Commissioner: The Commissioner of the Department of Planning and Natural Resources, or his/her designee.

(s) Coastal Estuary: All or part of the mouth of a river or stream or other body of water having a direct natural connection with open sea *which is located along the coast* and within which the sea water is measurably diluted with fresh water derived from land drainage (see diagram below).



(t) Community diversity: The variety and type of species present in a community, the complexity of their interactions, and the age and stability of the community. The community diversity of a region is influenced by the number of communities present, the degree of difference among the communities, and how the communities are distributed across the region.

(u) Consistent sampling: Used to ensure that results can be aggregated and compared over time.

(v) Coral habitat: A place where the physical and biological elements of coral reef ecosystems (including any threatened and endangered species) provide a suitable environment, including the food, cover, and space resources, needed for plant and animal livelihood.

(w) Coral reef: Any reef or shoal composed primarily of corals. A wave-resistant structure resulting from cementation processes and the skeletal construction of hermatypic corals, calcareous algae, and other calcium carbonate-secreting organisms

(x) Coral reef ecosystems: Corals and other species of reef organisms (including plants) associated with coral reefs, seagrasses and the nonliving environmental factors that directly affect coral reefs, that together function as an ecological unit in nature.

(y) Coral reef ecosystems areas: areas, where a dynamic complex of plants, animals and microorganism communities and their nonliving environment interact as a functional unit. Areas that can be measured in square miles/kilometers or as a percent of area with coral cover. These areas include but are not limited to those areas defined by the National Oceanic and Atmospheric Administration (NOAA).

(z) Criteria: Elements of State water quality standards, expressed as constituent concentrations, levels, or narrative statements, representing a quality of water that supports a particular use. When criteria are met, water quality will generally protect the designated use.

(aa) Criteria Continuous Concentration or CCC: The EPA national water quality criteria recommendation for the highest instream concentration of a toxicant or an effluent to which organisms can be exposed indefinitely without causing unacceptable effect.

(bb) Criteria Maximum Concentration or CMC: The EPA national water quality criteria recommendation for the highest in stream concentration of a toxicant or an effluent to which organisms can be exposed for a brief period of time without causing an acute effect.

(cc) DPNR or Department: The Department of Planning and Natural Resources.

(dd) Designated Use: A use specified in water quality standards for each water body or segment, whether or not it is being attained.

(ee) Desirable Species: Species indigenous to the area or introduced to the area because of ecological or commercial value.

(ff) Dissolved oxygen: The concentration of free molecular oxygen dissolved in water; expressed in milligrams/liter (mg/L) saturation and measured 1 meter below the surface and 1 meter above the sea floor (or at the max depth of the instrument (~30 m)) with an EPA approved field instrument.

(gg) Effluent: The discharge of used waters, sanitary wastes, other wastewaters, or any liquid substances treated or untreated, proceeding from sanitary treatment plants, industrial wastewater treatment plants, manufacturing processes, storage tanks, ponds, sewers or any water pollution source.

(hh) Embayment: Consists of alluvial valley-fill deposits that grade into beach sands as the bedrock valleys open onto embayments. The alluvium, which commonly ranges in thickness from 30 to 50 feet, generally is fine grained and consists of clay, silt, and fine sand eroded primarily from volcanic rocks. Where they contain mostly fine-grained sediments, the aquifers yield only small amounts of water and are semi confined. Locally, the alluvium is coarse sand and gravel, and the aquifer is unconfined. The alluvial deposits interfinger and grade into beach deposits that consist primarily of coarse coral sand. These deposits are permeable and yield only a few gallons per minute to wells. However, water in the embayment is generally brackish to saline.

(ii) Enterococci Bacteria: Are commonly found in the feces of humans and other warm-blooded animals. The presence of enterococci in water is an indication of fecal pollution and the possible presence of enteric pathogens. Enterococci have a greater correlation with swimming-associated gastrointestinal illness in both marine and fresh waters than other bacterial indicator organisms and are less likely to "die off" in saltwater.

(jj) EPA: The United States Environmental Protection Agency.

(kk) ESA: Endangered Species Act.

(ll) Existing uses: Those uses actually attained in the water body on or after November 28, 1975 (the date of EPA's initial water quality standards regulation), whether or not they are included in water quality standards 40 CFR 131.3(e).

(mm) Exotic Species: A non-native plant or animal deliberately or accidentally introduced into a new habitat.

(nn) Federal Clean Water Act or CWA or Federal Water Pollution Control Act or Federal Act or FWPCA: The Federal Clean Water Act, 33 U.S.C., section 1251 et seq. as amended, and the rules and regulations promulgated there under.

(oo) Fresh water(s): All nontidal and tidal waters generally having a salinity, due to natural sources, of less than or equal to 0.5 parts per thousand at mean high tide.

(pp) Freshwater ponds: Refer to ponds that have waters that are entirely non-marine.

(qq) Groundwater: Water that fills all of the unblocked voids of underlying material below the ground surface which is the upper limit of saturation, or water which is held in the unsaturated zone by capillarity.

(rr) Gut (/Ghut) or Stream: A natural or constructed waterway or any permanent or intermittent stream.

(ss) Impaired Water body: A water body classified under Category 4, 4A, 4B, 4C or 5 of the most current 303(d) list.

(tt) Indicator Species or Indicator Communities: Unique environmental indicators which provide insight into the biological condition in a watershed.

(uu) Inland Estuary: All or part of the mouth of a river or stream or other body of water having a direct natural connection with open sea *which is located inland* and within which the sea water is measurably diluted with fresh water derived from land drainage.

(vv) Inlet: A narrow water passage between peninsulas or through a barrier island leading to a bay or lagoon.

(ww) Mangrove Wetlands: A wetland area where mangroves are the dominant plant species. See definition of *wetlands* (xxxx).

(xx) Marine and Coastal Waters: Consist of the Atlantic Ocean, the Caribbean Sea, and all contiguous saline bays, inlets and harbors within the jurisdiction of Virgin Islands.

(yy) Metric: An attribute that shows a quantitative change in value along a gradient of human influence.

(zz) Mixing Zone: An area where an effluent discharge undergoes initial dilution and is extended to cover the secondary mixing in the ambient waterbody. A mixing zone is an allocated impact zone where water quality criteria can be exceeded as long as acutely toxic conditions are prevented.

(aaa) NPDES: National Pollutant Discharge Elimination System.

(bbb) Natural Condition / Natural State: Describes the quality of surface and marine water untouched by human-caused pollution or disturbance. Natural conditions or a natural state are rare and exist in limited settings and must be demonstrated to the satisfaction of the DPNR-DEP, through extensive sampling and field investigations.

(ccc) Natural forces: Refers to chemical, biological, geological, ecological or any other conditions existing at specific sites, not resulting from, or as a consequence of, human intervention, that may influence a particular parameter at those sites.

(ddd) Non-point Source: A source of pollutant which is caused by rainfall moving over and through the ground. As runoff moves, it picks up and carries away both natural pollutants and pollutants resulting from human activities. These pollutants include sediments, nutrients, pesticides, and toxic substances such as hydrocarbons and heavy metals. Eventually these pollutants are deposited in wetlands, coastal waters and ground water.

(eee) Nuisances: Any plant/animal species, material, or substance which is found in water to cause damage or interference with attainment of designated uses.

(fff) Nutrients: Refer to Total Nitrogen and Total Phosphorus concentrations, which are expressed in mg/L.

(ggg) Other Discharges or Wastes: Includes but is not limited to garbage, refuse, decayed wood, sawdust, shavings, bark, sand, lime, cinders, ashes, offal, oil, tar, dyestuffs, acids, chemicals, leachate, sludge, salt and all other discarded matter not sewage or industrial waste that may cause or might reasonably be expected to cause pollution of the waters of the US Virgin Islands.

(hhh) Outstanding National Resource Waters or ONRW: Waters that have unique characteristics to be preserved (e.g., waters of exceptional recreational, environmental, economic, or ecological significance) (Class A Waters).

(iii) Parameter of Concern: The parameter that is being assessed analyzed or assumed to cause impairment.

(jjj) Passageway: A continuous stretch where water characteristics are affected only by natural conditions in such a manner that the free movement, flow or continuous drifting of biota is possible.

(kkk) Permit: An authorization, license, or equivalent control document to discharge pollutants into United States Virgin Islands waters issued under 12 V.I.C., section 185 and regulations promulgated pursuant thereto.

(lll) Permittee: The holder of a TPDES or NPDES permit.

(mmm) Person: An individual, corporation, partnership, association, municipality, territory, or territorial agency, the Government of the United States Virgin Islands, the Government of the United States, and any board, commission, authority, or independent instrumentality of the Government of the Virgin Islands and the United States Government and any officer, agent, or employee thereof, including those having regulatory authority over the discharge of pollutants.

(nnn) pH: A measure of the concentration of hydrogen ions in the water expressed in Standard Units (SU) measured 1 meter below the surface and 1 meter above the sea floor (or at the max depth of the instrument (~30 m)) with an EPA approved field instrument.

(ooo) Point Source: Includes but is not limited to any discernible, confined and discrete conveyance, including, but not limited to, any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, concentrated animal feeding operation, or vessel or other floating craft, or landfill leachate collection system from which pollutants are or may be discharged.

(ppp) Pollution: The man-made or man-induced alteration of the chemical, physical, biological and radiological integrity of any Waters of the United States Virgin Islands.

(qqq) Pollutant or Waste: Dredged spoil, solid waste, incinerator residue, filter backwash, sewage, garbage, sewage sludge, munitions, chemical wastes, biological materials, radioactive materials, heat, wrecked or discarded equipment, rock, sand, cellar dirt and industrial, municipal and agricultural waste discharged into water.

(rrr) Potable Water (Potable Water where applicable): Inland and Marine Water that is either currently intended or could potentially be used for drinking, cooking, or domestic purposes, subject to compliance with Territorial or Federal drinking water standards after treatment via available technology. In order for a Water of the USVI to be considered "applicable as Potable Water", it must have undergone standard drinking water testing and the results submitted to DPNR for review and approval.

(sss) Potable Water Source: Water, whether it be from an inland or marine source, to include but is not limited to springs, artisan wells, drilled wells, public or community water systems or any other source that has been tested and identified to be of a quality that will, after treatment via available technology, be able to meet Territorial or Federal drinking water standards at the tap.

(ttt) Primary Contact Recreation: Activities where the human body may come in direct contact with raw water to the point of complete body submergence. Primary contact recreation includes, but is not limited to, swimming, diving, water skiing, skin diving and surfing.

(uuu) Reference Conditions: The characteristics of water body segments least impaired by human activities. As such, reference conditions can be used to describe attainable biological or habitat conditions for water body segments with common watershed/catchment characteristics within defined water body classes.

(vvv) Reliable Measure: A measure that is reliable if it consistently produces the same result.

(www) Saline Water: Saline water(s) means waters having salinities generally greater than 3.5 parts per thousand at mean high tide.

(xxx) Salinity: An estimate of the concentration of dissolved salts in seawater expressed in parts per thousand and measured 1 meter below the surface and 1 meter above the sea floor (or at the max depth of the instrument (~30 m)).

(yyy) Salt Flats: Refers to a salt-encrusted flat area resulting from evaporation of a former body of water.

(zzz) Salt Pond: A salt water embayment or lagoon separated from coastal waters by any barrier.

(aaaa) Secchi Disc: Provides a method for assessing the water clarity expressed in meters by a secchi depth recording light transparency.

(bbbb) Site: The land or water area where any facility or activity is physically located or conducted, including adjacent land used in connection with the facility or activity.

(cccc) Stream or Gut (/Ghut): A natural or constructed waterway or any permanent or intermittent stream.

(dddd) Subclass IBS Waters: All inland brackish or saline waters.

(eeee) Subclass IF Waters: All inland fresh waters.

(ffff) Subclass IG Waters: All inland groundwaters that are current or potential supplies of potable water and their associated recharge areas. They shall be protected as potable water supplies. Unless otherwise identified, Subclass IG include all ground water with a naturally occurring salinity of less than 10,000 mg/l.

(gggg) Sufficient Quality: The level of water quality providing acceptable conditions to support aquatic life.

(hhhh) Surface Water: Includes all waters other than groundwater within the jurisdiction of the United States Virgin Islands including all streams and/or guts (permanent or intermittent), freshwater ponds, wells, estuaries, and wetlands which includes: swamps, salt flats, salt ponds, and mangrove wetlands situated wholly or partly within or bordering upon the United States Virgin Islands, including the Territorial seas, contiguous zone, and oceans.

(iiii) Swamp: Wetland often partially or intermittently covered with water; especially, one dominated by wooded vegetation.

(jjjj) Temperature: A measure of the energy of molecular motion expressed in degrees Centigrade measured 1 meter below the surface and 1 meter above the sea floor (or at the max depth of the instrument (~30 m)) with an EPA approved field instrument.

(kkkk) Territorial Waters (or Waters of the United States Virgin Islands): All inland and marine and coastal waters within the jurisdiction of the United States Virgin Islands.

(llll) TPDES: Territorial Pollutant Discharge Elimination System, a permitting program under CWA Section 402 that addresses water pollution by regulating point sources that discharge pollutants to waters of the USVI, and implemented in the USVI through 12VIC§185 and 12VIRR§184.

(mmmm) Thermal Discharge: A discharge that results or would result in a significant or measurable temperature change of the receiving water.

(nnnn) Thermal Pollution: The change in the water temperature of any Territorial Water caused by man-made practices that may adversely affect fish, aquatic life, animals, and human health.

(oooo) Total Maximum Daily Load or TMDL: The maximum amount of a pollutant that a waterbody can receive while still meeting water quality standards.

(pppp) Toxic Pollutant: Any pollutant listed as toxic under section 307(a)(1) of the CWA.

(qqqq) Turbidity: A measure of the degree to which light is scattered by suspended particulate material and soluble colored compounds in the water. Expressed in Nephelometric Turbidity Units (NTU's) measured 1 meter below the surface and 1 meter above the sea floor (or at the max depth of the instrument (~30 m)) using an EPA approved field instrument.

(rrrr) WQS Variance: A time-limited designated use and criterion for a specific pollutant(s) or water quality parameter(s) that reflect the highest attainable condition during the term of the water quality standard variance. Variances are different from changes to the designated use and associated criteria in that they are intended as a mechanism to provide time for states and stakeholders to implement adaptive management approaches that will improve water quality where the designated use and criterion currently in place are not being met, but still retain the designated use as a long-term goal.

(ssss) Wastewaters: Waters containing dissolved, suspended, agglomerated, emulsified or floating substances or solid pollutants resulting from industrial, commercial, residential, agricultural, and recreational or any other type of establishment or man induced activity.

(tttt) Water Quality Standards or WQS: Any water quality standards adopted and effective under United States Virgin Islands or Federal laws applicable to waters of the United States Virgin Islands, including the designated use or uses of a water body, the numeric and narrative water quality criteria that are necessary to protect the use or uses of that particular water body, and an anti-degradation policy.

(uuuu) Water Quality Criteria: Any criteria describing the required quality supporting a particular designated use of United States Virgin Islands waters, as adopted under United States Virgin Islands laws or Federal laws applicable to waters of the United States Virgin Islands.

(vvvv) Waters of the United States Virgin Islands (or Territorial Waters) : All waters within the jurisdiction of the United States Virgin Islands including all inland and marine waters, including but not limited to harbors, streams, lakes, ponds, impounding reservoirs, marshes, water-courses, water-ways, wells, springs, irrigation systems, drainage systems and all other bodies or accumulations of water, surface and underground, natural or artificial, public or private, situated wholly or partly within or bordering upon the United States Virgin Islands, including the territorial seas, contiguous zones, and oceans.

(wwww) Well: A pit or hole sunk into the earth to reach a resource of potable water supply to be used for domestic purposes.

(xxxx) Wetlands: Those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include salt ponds, marshes, swamps, and similar areas, which may be freshwater or brackish or saline.

§ 186 - 2: Classification of Territorial Waters

Territorial Waters are classified as either Inland Waters or Marine and Coastal Waters.

(a) Class of Inland Waters - Class I Waters - include groundwaters (IG waters) and surface waters: fresh (IF) or brackish saline (IBS) waters.

(1) Subclass of Inland Fresh Surface Waters - Subclass IF Waters: Based on their ecological characteristics and other natural attributes, all inland fresh waters are classified as follows:

(A) Streams and/or guts (Permanent or Intermittent)

(B) Freshwater Wetlands:

(i) Freshwater Ponds

(2) Subclass of Inland Brackish or Saline Surface Waters - Subclass IBS Waters: All inland brackish or saline waters are classified as follows, based on their ecological characteristics and other natural attributes:

(A) Inland Estuaries (not designated as Classes A, B or C)

(B) Brackish or Saline Wetlands:

(i) Swamps
(ii) Salt Flats
(iii) Salt Ponds
(iv) Mangrove Wetlands
(v) Marshes

(3) Subclass of Groundwaters – Subclass IG Waters:

(A) Wells

(b) Class of Marine and Coastal Waters – Class A, B and C Waters: All marine and coastal waters are either embayments, open coastal, or oceanic waters to include all contiguous saline bays, inlets, coastal estuaries and harbors within the jurisdiction of US Virgin Islands.

(1) Class A Waters - Outstanding National Resource Waters

(2) Class B - All other coastal or marine waters not classified as Class A or Class C

(3) Class C - All other coastal or marine waters not classified as Class A or Class B. Class C waters have less stringent water quality standards than Class B.

§ 186 - 3: Legal Limits

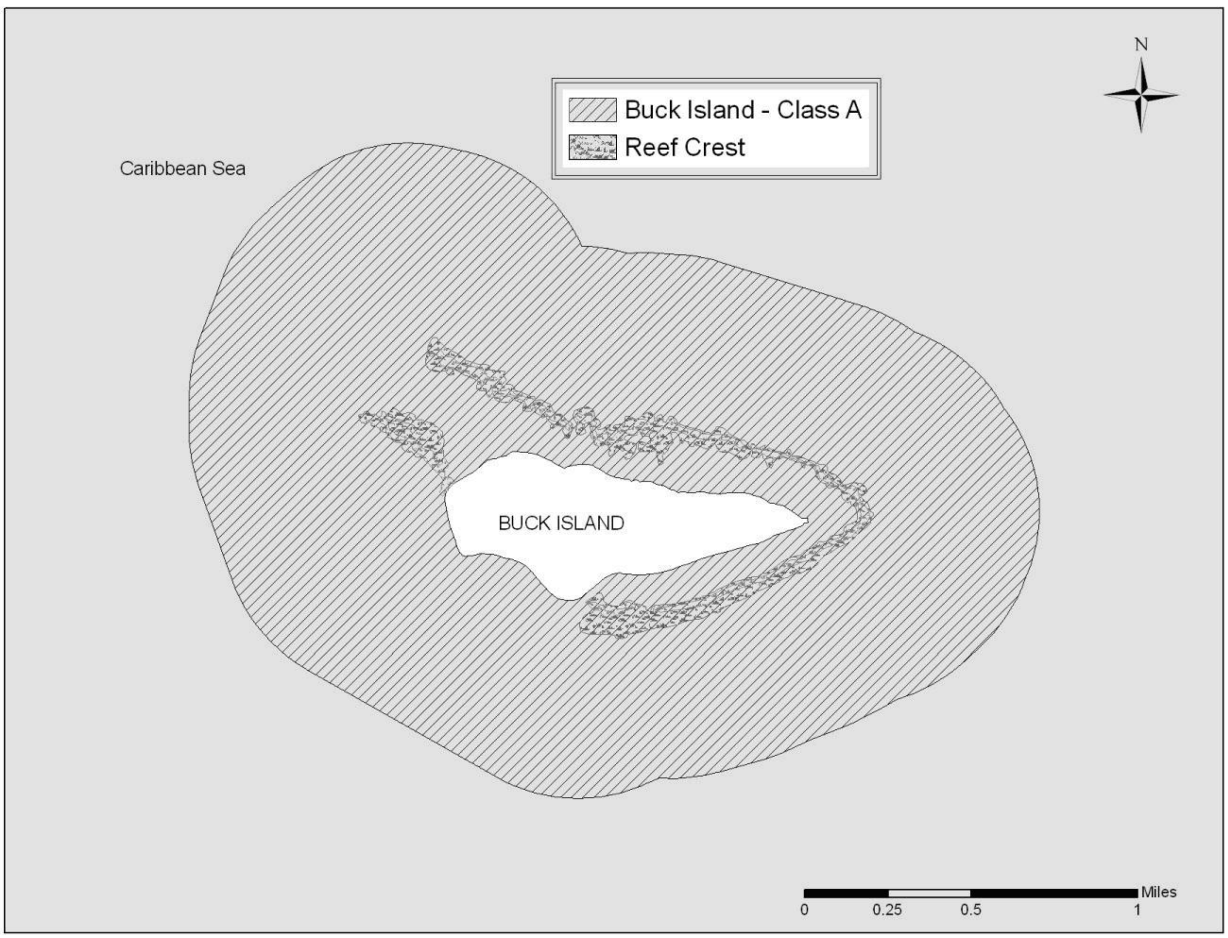
The following serves as the legal description and boundaries for the Territorial Waters:

(a) Class I (Inland Waters): Designated aquatic-influenced environments located within land boundaries. Waters included in this class can be either inland groundwaters (Subclass IG waters) or inland surface waters. Inland surface waters can be fresh (Subclass IF Waters), as well as saline or brackish (Subclass IBS Waters).

(b) Class A Marine and Coastal Waters (Outstanding National Resource Waters):

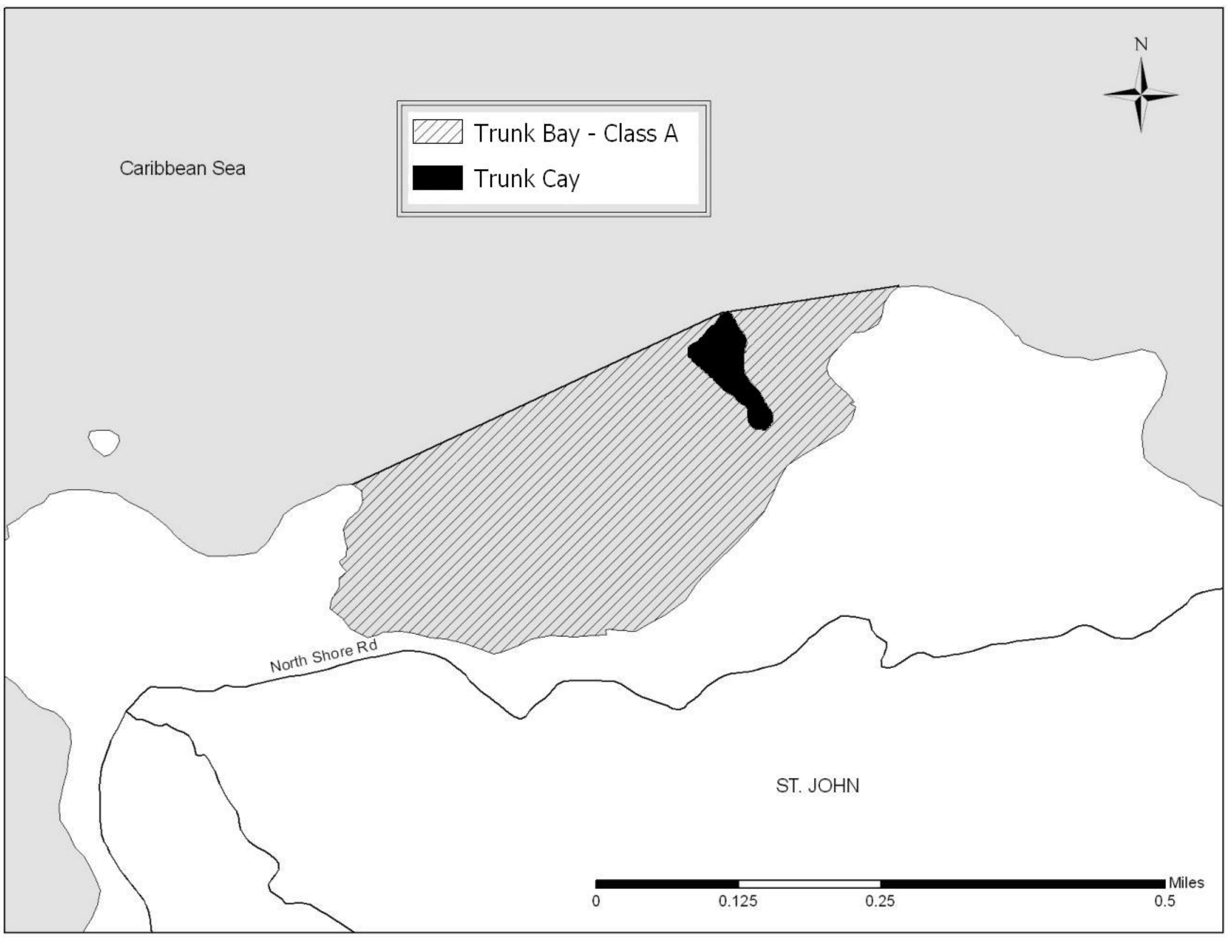
(1) Within 0.5 miles of the boundaries of Buck Island's Natural Barrier Reef, St. Croix.

Figure 1. Class A - Buck Island, St. Croix



(2) Trunk Bay, St. John.

Figure 2. Class A - Trunk Bay, St. John



(c) Class B Marine and Coastal Waters:

(1) All other coastal or marine waters not classified as Class A or Class C.

Figure 3. Class B - St. Croix

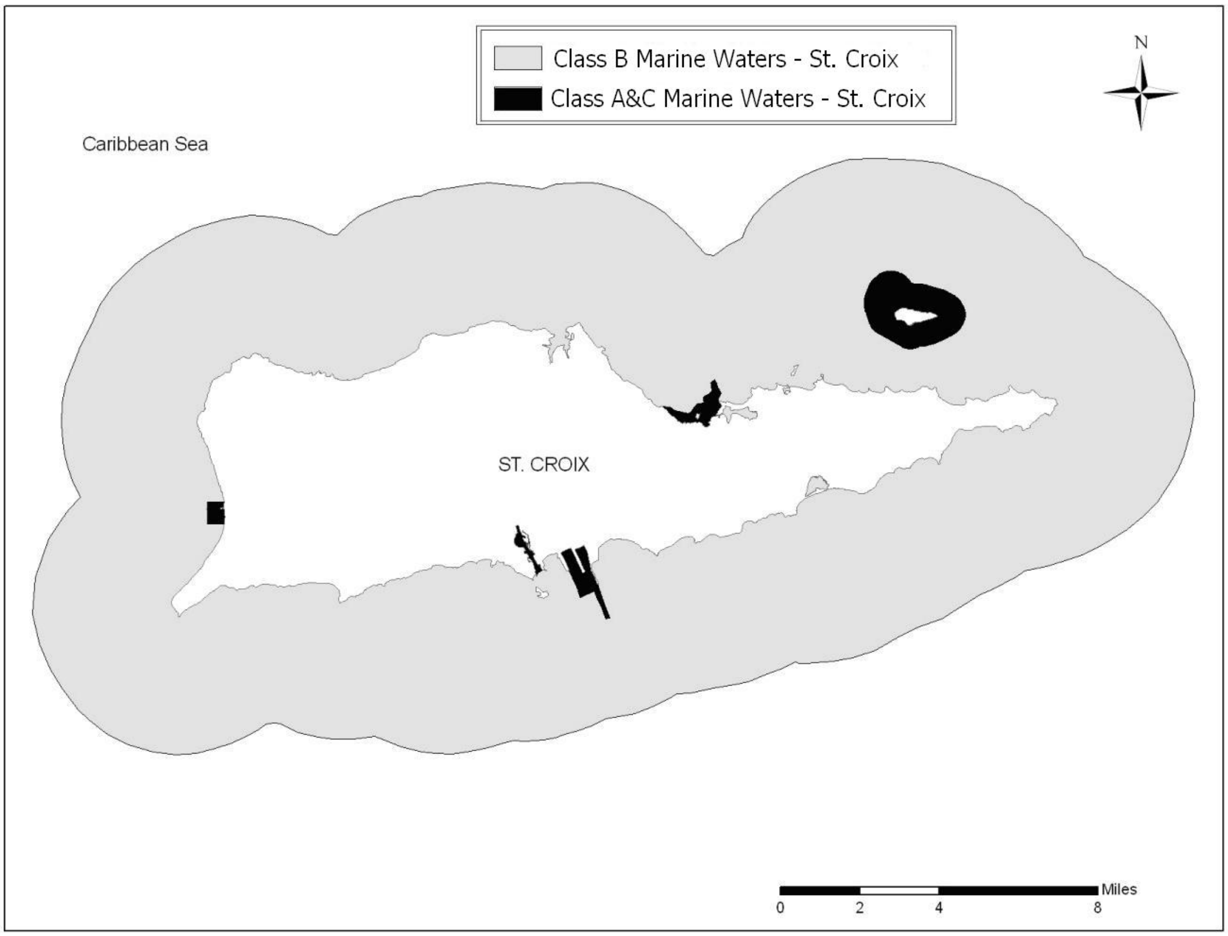
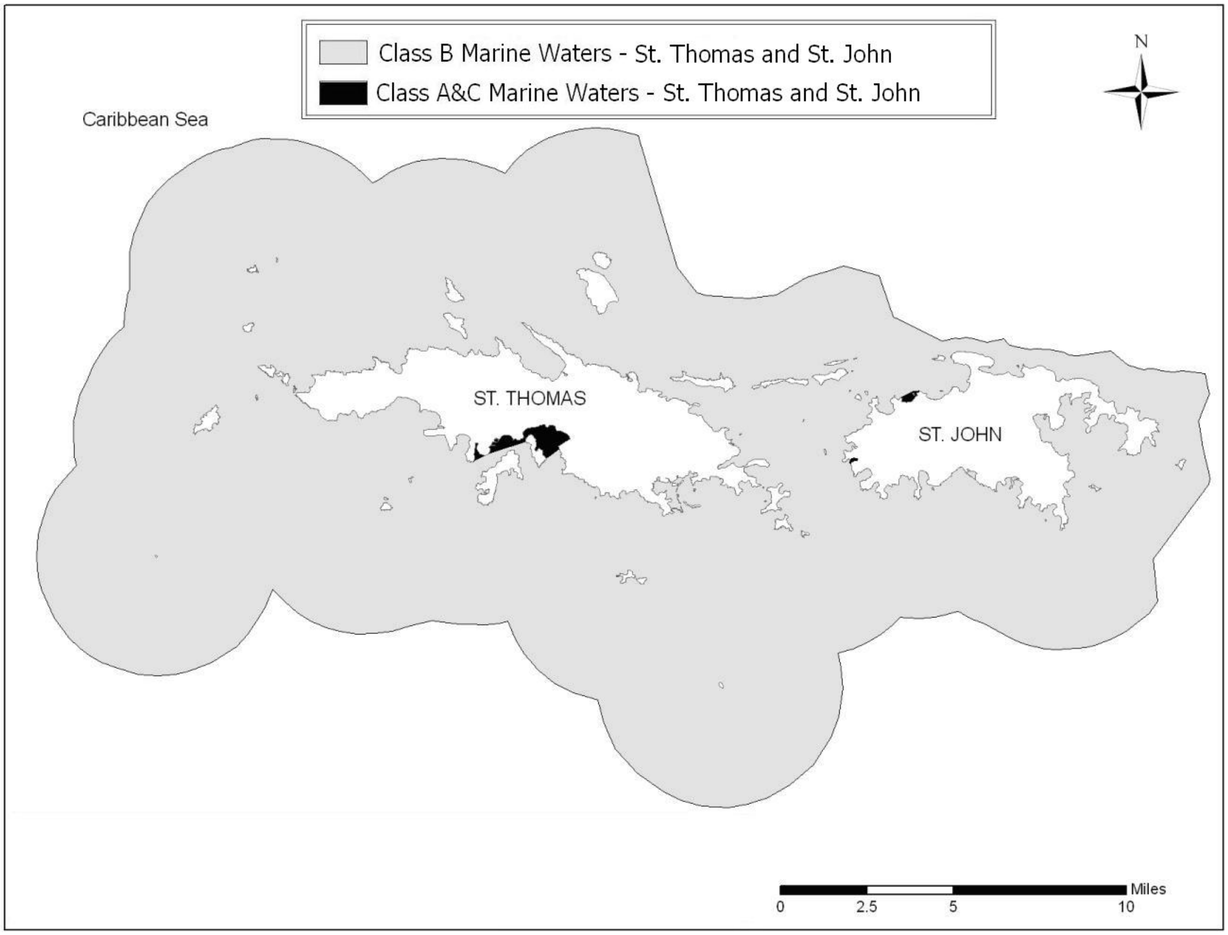


Figure 4. Class B - St. Thomas and St. John



(d) Class C Marine and Coastal Waters:

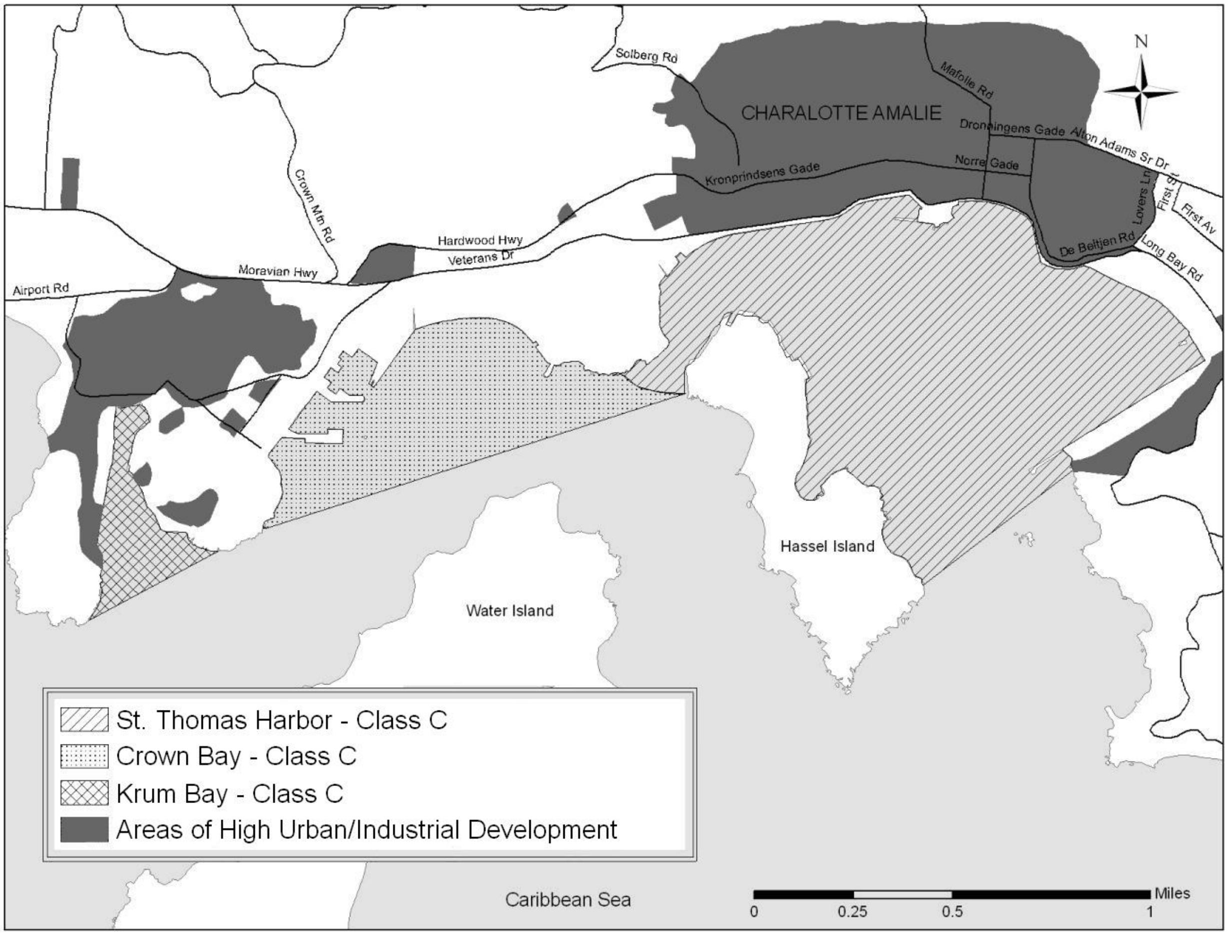
(1) St. Thomas:

(A) St. Thomas Harbor beginning at Rupert Rock and extending to Haulover Cut.

(B) Crown Bay enclosed by a line from Hassel Island at Haulover Cut to Regis Point at West Gregerie Channel.

(C) Krum Bay.

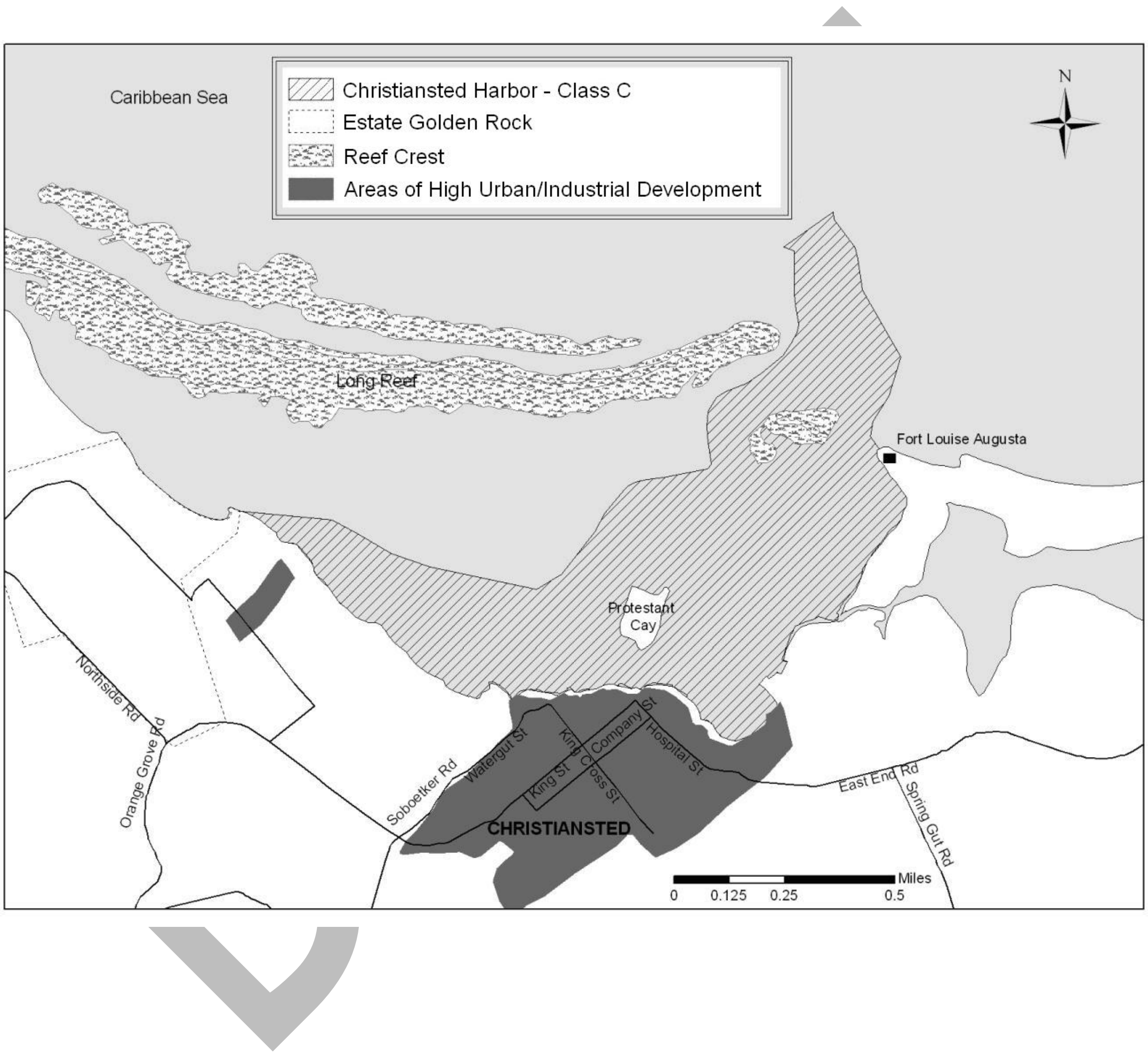
Figure 5. Class C - St. Thomas Harbor, Crown Bay and Krum Bay, St. Thomas



(2) St. Croix:

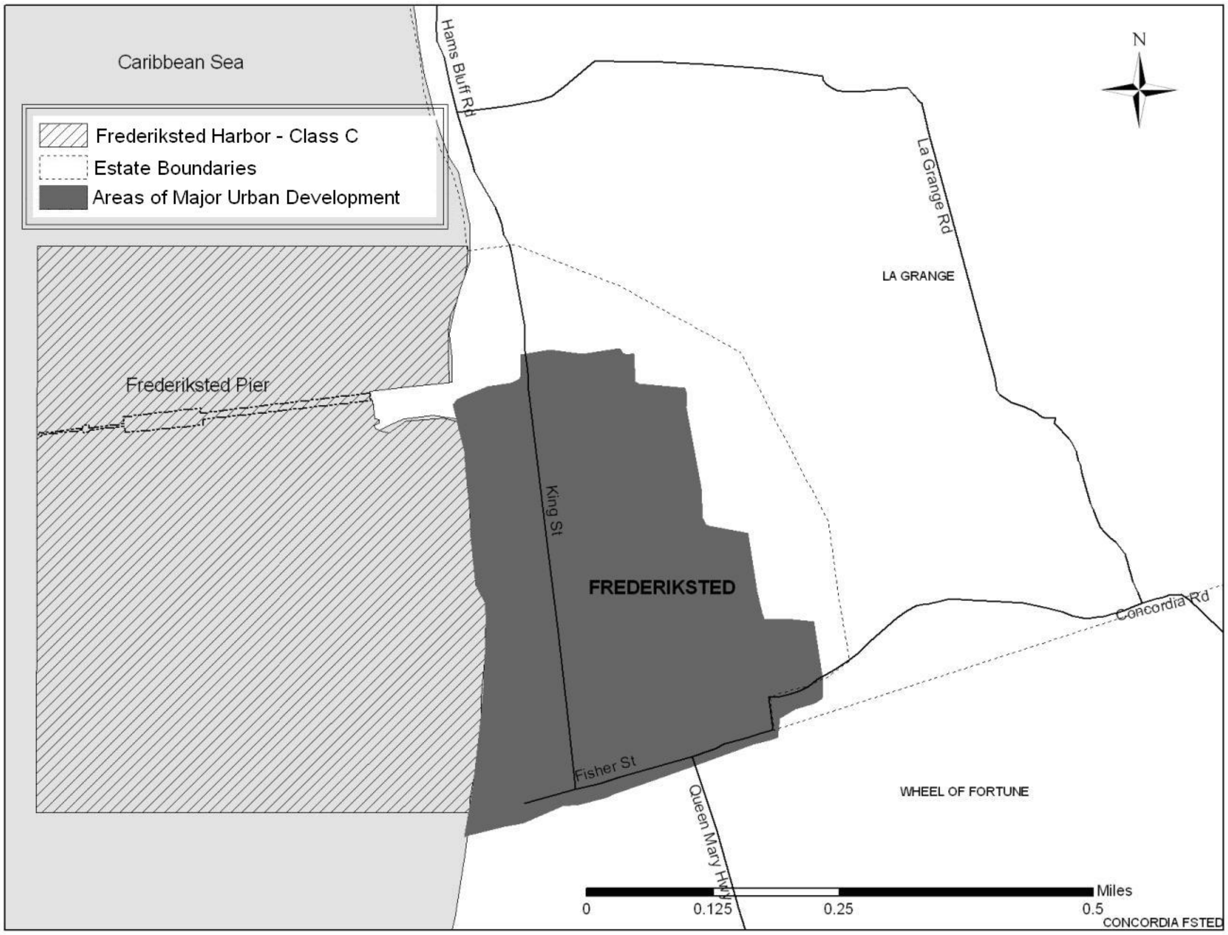
(A) Christiansted Harbor from Fort Louise Augusta to Golden Rock, along the waterfront and seaward to include the navigational channels and mooring areas.

Figure 6. Class C - Christiansted Harbor, St. Croix



(B) Frederiksted Harbor from La Grange to Fisher Street and seaward to the end of the Frederiksted Pier.

Figure 7. Class C - Frederiksted Harbor, St. Croix



(C) Hess Oil Virgin Islands Harbor (alternatively named HOVENSA Harbor).

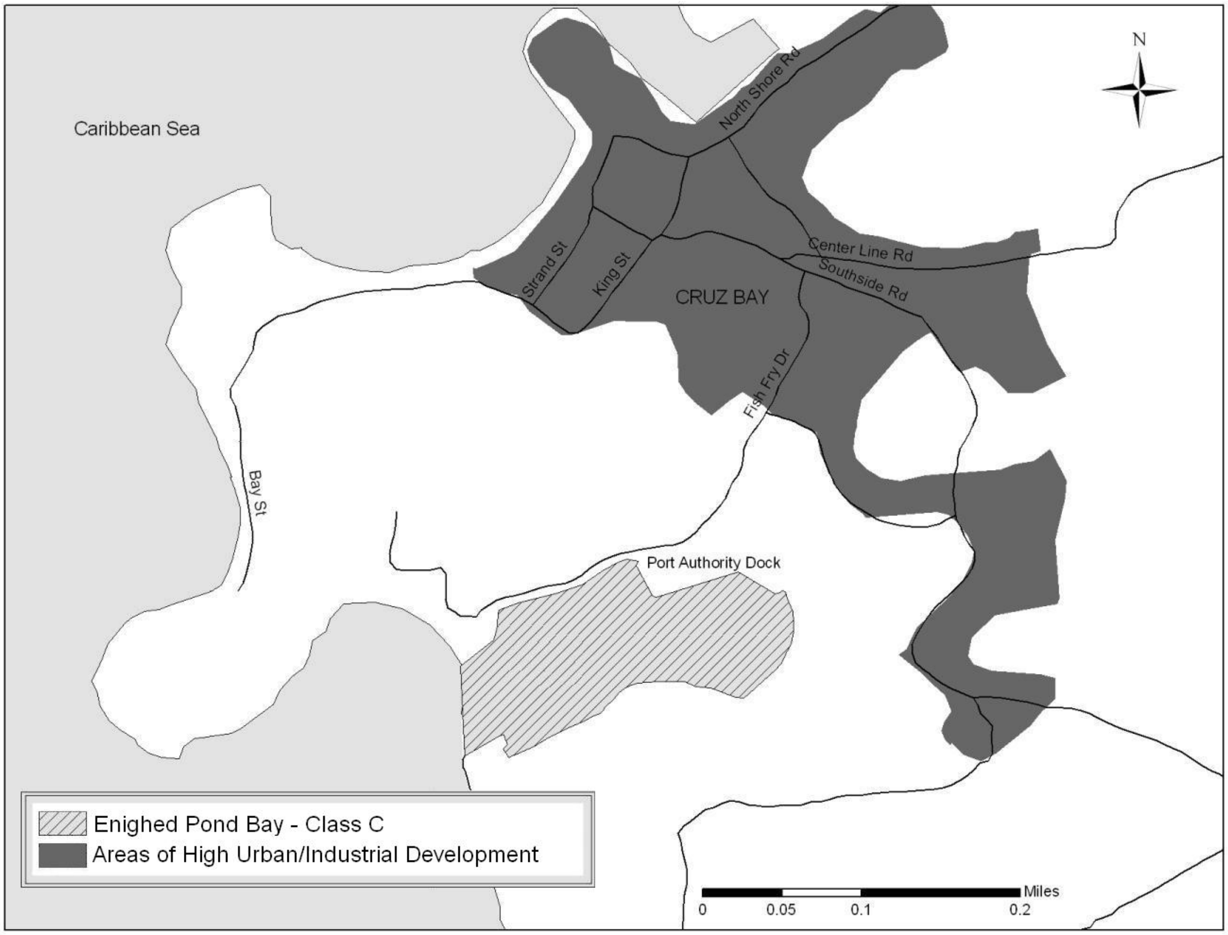
(D) Martin-Marietta Alumina Harbor (alternatively named Port Alucroix or St. Croix Renaissance Group Harbor).

Figure 8. Class C - HOVENSA Harbor and St. Croix Renaissance Group Harbor, St. Croix

(3) St. John:

(A) Enighed Pond Bay

Figure 9. Class C - Enighed Pond, St. John



§ 186 - 4: Classification of Water Designated Uses

The following Territorial Waters classifications specify the designated uses to be protected and the applicable criteria to protect those uses.

(a) Class of Inland Waters - Class I Waters (Inland Fresh Waters (IF Waters), Inland Brackish or Saline Waters (IBS Waters) and Inland Groundwaters (IG Waters)

(1) Subclass IF and IBS

(A) Designated uses: Maintenance and propagation of desirable species of wildlife and aquatic life (including threatened, endangered species listed pursuant to section 4 of the *Federal Endangered Species Act* and threatened, endangered and indigenous species listed pursuant to Title 12, Chapter 2 of the Virgin Islands Code), primary contact recreation and for use as a potable water source (with the understanding that such water sources are declared to be public waters belonging to the people of the United States Virgin Islands, subject to appropriation for beneficial use in the manner set forth in 12 VIC §151 and not otherwise), where applicable.

(B) Water Quality Criteria: Waters shall remain in their natural state to the maximum extent possible with an absolute minimum of pollution from any human-caused source. To the extent possible, the ecological character of these areas shall be maintained and protected. The following water quality standards apply:

(i) Narrative Criteria: Criteria listed in §186-5(a)(1) are applicable.

(ii) Numeric Criteria:

(a) Toxic Pollutants: Numeric criteria listed in §186-5(b) including Aquatic Life Criteria in Tables I, Human Health Criteria (for the consumption of Water & Organisms) in Table II and Organoleptic Criteria in Table III are applicable.

(b) Bacteria:

(1) The 30-day geometric mean for enterococci shall not exceed 30 CFU/100 mL and no more than 10

percent of the samples collected in the same 30 days shall exceed 110 CFU/100 mL.

(2) For waters used as potable water sources, concentration of total coliforms shall be sufficient to meet applicable USVI Drinking Water Regulations after treatment with available technology.

(c) Dissolved Oxygen: Not less than 5.5 mg/l except when due to natural forces.

(2) Subclass IG Waters

(A) Designated uses: For use as a potable water source.

(B) Water Quality Criteria: Waters shall remain in their natural state to the maximum extent possible with an absolute minimum of pollution from any human-caused source. To the extent possible, the ecological character of these areas shall be maintained and protected. The following water quality standards apply:

(i) Narrative Criteria: Criteria listed in §186-5(a)(1) are applicable.

(ii) Numeric Criteria:

(a) Toxic Pollutants: Numeric criteria listed in §186-5(b) including Aquatic Life Criteria in Tables I, Human Health Criteria (for the consumption of Water & Organisms) in Table II and Organoleptic Criteria in Table III are applicable.

(b) Bacteria:

(1) For waters used as potable water sources, concentration of total coliforms shall be sufficient to meet applicable USVI Drinking Water Regulations after treatment with available technology.

(b) Class of Marine and Coastal Waters

(1) Class A Waters:

(A) Designated uses: Maintenance and propagation of desirable species of wildlife and aquatic life (including threatened, endangered species listed pursuant to section 4 of the Federal *Endangered Species Act* and threatened, endangered and indigenous species listed pursuant Title 12, Chapter 2 of the Virgin Islands Code) primary contact recreation and for use as a potable water source, where applicable. Preservation of the unique characteristics of the waters designated as Outstanding Natural Resource Waters (e.g., Natural Barrier Reef at Buck Island, St. Croix and the Under Water Trail at Trunk Bay, St. John), waters of exceptional recreational, environmental, or ecological significance. No new or increased dischargers shall be permitted.

(B) Water Quality Criteria: Natural conditions shall not be altered.

(i) Narrative Criteria:

(a) Criteria listed in § 186-5(a) are applicable.

(b) Biocriteria: The biological condition shall be similar or equivalent to reference condition established for biological integrity within Class A waters.

(ii) Numeric Criteria: In no case shall Class B water quality standards be exceeded.

(a) Toxic Pollutants: Numeric criteria listed in § 186-5(b) including Aquatic Life Criteria in Table I, Human Health Criteria in Table II and Organoleptic Criteria in Table III are applicable.

(b) Dissolved oxygen: Not less than 5.5 mg/l except when due to natural forces.

(c) pH: Natural Conditions of pH must not be extended at any location by more than +/- 0.1 pH unit. At no time shall the pH be less than 7.0 or greater than 8.3.

(d) Temperature:

(1) Except by natural conditions, not to exceed 32˚C at any time, nor as a result of waste discharge to be greater than 1.0˚C above natural conditions. Thermal policies (Section 186-6) shall also apply.

(2) Areas where coral reef ecosystems are located shall not exceed 25-29˚C at any time, nor as a result of waste discharge to be greater than 1.0˚C above natural. Thermal policies (Section 186-6) shall also apply.

(e) Bacteria:

(1) The 30-day geometric mean for enterococci shall not exceed 30 CFU/100 mL and no more than 10 percent of the samples collected in the same 30 days shall exceed 110 CFU/100 mL.

(2) For areas used as potable water sources, concentration of total coliforms shall be sufficient to meet applicable USVI Drinking Water Regulations after treatment with available technology.

(f) Phosphorus: Phosphorus as total P shall not exceed 50 µg/L in marine and coastal waters.

(g) Nitrogen: Nitrogen as total N shall not exceed concentration of 207 µg/L in more than 10% of samples over a three-year period in estuarine, marine and coastal waters.

(h) Radioactivity:

(1) Gross beta: 1000 picocuries per liter, in the absence of Sr 90 and alpha emitters.
(2) Radium-226: 3 picocuries per liter.
(3) Strontium-90: 10 picocuries per liter.

(i) Clarity:

(1) A Secchi disc shall be visible at a minimum depth of one (1) meter. For waters where the depth does not exceed one (1) meter, the bottom must be visible.

(2) In areas where coral reef ecosystems are located, a secchi disc shall be visible at a minimum depth of fifteen (15) meters. For such waters where the depth does not exceed fifteen (15) meters, the bottom must be visible.

(j) Turbidity:

(1) A maximum nephelometric turbidity unit reading of three (3) shall be permissible.

(2) For areas where coral reef ecosystems are located, a maximum nephelometric turbidity unit reading of one (1) shall be permissible.

(2) Class B Waters:

(A) Designated uses: Maintenance and propagation of desirable species of wildlife and aquatic life (including threatened, endangered species listed pursuant to section 4 of the Federal *Endangered Species Act* and threatened, endangered and indigenous species listed pursuant to Title 12, Chapter 2 of the Virgin Islands Code), primary contact recreation (swimming, water skiing, etc.) and for use as potable water source, where applicable.

(B) Water Quality Criteria:

(i) Narrative Criteria: Criteria listed in § 186-5(a) are applicable.

(a) Biocriteria: The biological condition shall reflect no more than a minimal departure from reference condition for biological integrity within Class B waters. Class B allows minimal changes in structure of the biotic community and minimal changes in ecosystem function. Virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability.

(ii) Numerical Criteria: The following criteria apply at and beyond the boundary of the applicable mixing zone as specified in section 186-7 or variance adopted in accordance with section 186-14, as the case may be.

(a) Toxic Pollutants: Numeric criteria listed in § 186-5(b) including Aquatic Life Criteria in Table I, Human Health Criteria in Table II and Organoleptic Criteria in Table III are applicable.

(b) Dissolved oxygen: Not less than 5.5 mg/l except when due to natural forces.

(c) pH: Natural Conditions of pH must not be extended at any location by more than +/- 0.1 pH unit. At no time shall the pH be less than 7.0 or greater than 8.3.

(d) Temperature:

(1) Except due to natural conditions, not to exceed 32˚C at any time, nor as a result of waste discharge to be greater than 1.0˚C above natural conditions. Thermal policies (Section 186-6) shall also apply.

(2) Areas where coral reef ecosystems are located shall not exceed 25-29˚C at any time, nor as a result of waste discharge to be greater than 1.0˚C above natural conditions. Thermal Policies (Section 186-6) shall also apply.

(e) Bacteria:

(1) The 30-day geometric mean for enterococci shall not exceed 30 CFU/100 mL and no more than 10 percent of the samples collected in the same 30 days shall exceed 110 CFU/100 mL.

(2) For waters used as potable water sources, concentration of total coliforms shall be sufficient to meet applicable USVI Drinking Water Regulations after treatment with available technology.

(f) Phosphorus: Phosphorus as total P shall not exceed 50 µg/L in marine and coastal waters.

(g) Nitrogen: Nitrogen as total N shall not exceed concentration of 207 µg/L in more than 10% of samples over a three-year period in estuarine, marine and coastal waters.

(h) Radioactivity:

(1) Gross beta: 1000 picocuries per liter, in the absence of Sr 90 and alpha emitters.
(2) Radium-226: 3 picocuries per liter.
(3) Strontium-90: 10 picocuries per liter.

(i) Clarity:

(1) A Secchi disc shall be visible at a minimum depth of one (1) meter. For waters where the depth does not exceed one (1) meter, the bottom must be visible.

(2) In areas where coral reef ecosystems are located, a secchi disc shall be visible at a minimum depth of fifteen (15) meters. For such waters where the depth does not exceed fifteen (15) meters, the bottom must be visible.

(j) Turbidity: The following turbidity criteria are applicable to all Class B waters, except for those Class B Waters listed below in §186-4 (b)(2)(B)(ii)(j)(3).

(1) A maximum nephelometric turbidity unit reading of three (3) shall be permissible.

(2) For areas where coral reef ecosystems are located, a maximum nephelometric turbidity unit reading of one (1) shall be permissible.

(3) The following Class B waters, based on §186-11 (Natural Conditions), are not covered by the narrative turbidity criteria found in §186-

5(a)(1)(C) and therefore shall be also excluded from the above ((j) 1 and 2) requirements:

(i) St. Thomas Waters-Mandahl Bay (Marina), Vessup Bay, Water Bay, Benner Bay, and the Mangrove Lagoon.

(ii) St. Croix Waters-Carlton Beach, Good Hope Beach, Salt River Lagoon (Marina), Salt River Lagoon (Sugar Bay), Estate Anguilla Beach, Buccaneer Beach, Tamarind Reef Lagoon, Green Cay Beach and Enfield Green Beach.

(3) Class C Waters

(A) Designated uses: Maintenance and propagation of desirable species of wildlife and aquatic life (including threatened and endangered species listed pursuant to section 4 of the Federal *Endangered Species Act* and threatened, endangered and indigenous species listed pursuant Title 12, Chapter 2 of the Virgin Islands Code), primary contact recreation (swimming, water skiing, etc.), industrial water supplies, shipping, navigation and for use as potable water source, where applicable.

(B) Water Quality Criteria:

(i) Narrative Criteria: Criteria listed in § 186-5(a) are applicable.

(a) Biocriteria: Class C allows for evident changes in structure of the biotic community and minimal changes in ecosystem function. Evident changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa (community structure) are allowed but sensitive-ubiquitous taxa remain common and abundant; ecosystem functions are fully maintained through redundant attributes of the system. Ecosystem function shall be similar or equivalent to reference condition established at the least disturbed reference site(s) within Class C waters.

(ii) Numerical Criteria: The following criteria apply at and beyond the boundary of the applicable mixing zone as specified in section 186-7 or variance adopted in accordance with section 186-14.

(a) Toxic Pollutants: Numeric criteria listed in § 186-5(b) including Aquatic Life Criteria in Table I, Human Health Criteria in Table II and Organoleptic Criteria in Table III are applicable.

(b) Dissolved Oxygen: Not less than 5.0 mg/l except when due to natural forces.

(c) pH:

(1) Natural Conditions of pH must not be extended at any location by more than +/- 0.1 pH unit. At no time shall the pH be less than 6.7 or greater than 8.5.

(2) Temperature:

a. Except due to natural conditions, not to exceed 32˚C at any time, nor as a result of waste discharge to be greater than 1.0˚C above natural conditions. Thermal policies (Section 186-6) shall also apply.

b. Areas where coral reef ecosystems are located shall not exceed 25-29˚C at any time, nor as a result of waste discharge to be greater than 1.0˚C above natural conditions. Thermal Policies (Section 186-6) shall also apply.

(c) Bacteria:

(1) The 30-day geometric mean for enterococci shall not exceed 30 CFU/100 mL and no more than 10 percent of the samples collected in the same 30 days shall exceed 110 CFU/100 mL.

(2) For waters used as potable water sources, concentration of total coliforms shall be sufficient to meet applicable USVI Drinking Water Regulations after treatment with available technology.

(d) Phosphorus: Phosphorus as total P shall not exceed 50 µg/L in marine and coastal waters.

(e) Nitrogen: Nitrogen as total N shall not exceed concentration of 207 µg/L in more than 10% of samples over a three-year period in estuarine, marine and coastal waters.

(f) Radioactivity:

(1) Gross beta: 1000 picocuries per liter, in the absence of Sr 90 and alpha emitters.
(2) Radium-226: 3 picocuries per liter.
(3) Strontium-90: 10 picocuries per liter.

(g) Clarity:

(1) A Secchi disc shall be visible at a minimum depth of one (1) meter. For waters where the depth does not exceed one (1) meter, the bottom must be visible.

(2) In areas where coral reef ecosystems are located, a secchi disc shall be visible at a minimum depth of fifteen (15) meters. For such waters where the depth does not exceed fifteen (15) meters, the bottom must be visible.

(h) Turbidity:

(1) A maximum nephelometric turbidity unit reading of three (3) shall be permissible.

(2) For areas where coral reef ecosystems are located, a maximum nephelometric turbidity unit

reading of one (1) shall be permissible.

§ 186 - 5: General Water Quality Criteria

The following is the narrative water quality criteria and numeric water quality criteria for toxic pollutants applicable to all Territorial Waters – Inland, Marine and Coastal Waters (unless otherwise stated) in all classes.

(a) Narrative Water Quality Criteria: All Territorial Waters shall meet generally accepted aesthetic qualifications and shall be capable of supporting diversified aquatic life. Refer to section 186-3 above for a complete list of these waters.

(1) All Territorial Waters shall be free of substances attributable to municipal, industrial, or other discharges or wastes as follows:

(A) Deposits - materials that will settle to form objectionable deposits,
(B) Matter - floating debris, oils, scum, and other nuisance matter,
(C) Turbidity - substances producing objectionable turbidity, such as sediment, floating debris, scum and other floating materials attributable to discharges in amounts sufficient to be unsightly, deleterious, create a nuisance, or be detrimental to the existing or designated uses of the waterbody,
(D) Materials - including radionuclides, in concentrations or combinations which are toxic or which produce undesirable physiological responses in human, fish and other animal life, and plants,
(E) Color - virtually free from substances producing objectionable color for aesthetic purposes,
(F) Suspended, colloidal, or settleable solids - from wastewater sources which will cause disposition or be detrimental to existing or designated uses,
(G) Oil and floating substances - residue attributable to wastewater or visible oil film or globules of grease,
(H) Taste and odor producing substances - in amounts that will interfere with the use for primary contact recreation, potable water supply or will render any undesirable taste or odor to edible aquatic life, except where these substances are primarily the result of natural conditions, states or forces (e.g. decomposition) rather than anthropogenic factors or processes.
(I) Substances and/or conditions - in concentrations which produce undesirable aquatic life,
(J) Nuisance species - Exotic or aquatic, and
(K) Downstream Protection – All waters shall attain and maintain a level

of water quality that provides for the attainment and maintenance of the water quality standards of downstream waters.

(2) Biocriteria: These narrative biological criteria shall apply to Class A, B, C waters, wetlands, estuarine, mangrove, seagrass, coral reef and other marine ecosystems based upon their respective reference conditions and metrics. The Territory shall preserve, protect, and restore these water resources to their most natural condition. The condition of these waterbodies shall be determined from measures of physical, chemical, and biological characteristics of each waterbody class, according to its designated use. As a component of these measures, the Territory may consider the biological integrity of the benthic communities living within waters. These communities shall be assessed by comparison to reference conditions with similar abiotic and biotic environmental settings that represent the optimal or least disturbed condition for that system. Such reference conditions shall be those observed to support the greatest community diversity, and abundance of aquatic life as is expected to be or has been historically found in natural settings essentially undisturbed or minimally disturbed by human impacts, development, or discharges. This condition shall be determined by consistent sampling and reliable measures of selected indicator communities of flora and/or fauna and may be used in conjunction with other measures of water quality.

In utilizing the following criteria, the Virgin Islands shall preserve, protect, and restore marine and coastal Territorial Waters to their most natural condition.

(A) Determining Conditions

(i) The condition of these waterbodies shall be determined from measures of physical, chemical, and biological characteristics of each waterbody class, according to its designated use.

(ii) As a component of these measures, the Virgin Islands may consider the biological integrity of the benthic communities living within waters. These communities shall be assessed by comparison to reference conditions(s) with similar abiotic and biotic environmental settings that represent the optimal or least disturbed condition for that system. Such reference conditions shall be those observed to support the greatest community diversity, and abundance of aquatic life as is expected to be or has been historically found in natural settings essentially undisturbed or minimally disturbed by human impacts, development, or discharges. This condition shall be determined by consistent sampling and reliable measures of selected indicator communities of flora and/or fauna and may be used in conjunction with other measures of water quality.

(B) Sufficient Quality: Waters shall be of a sufficient quality to support a resident biological community defined by metrics based upon reference conditions.

(b) Numeric Criteria: For all waters used as potable water source (within Class I, A, B and C), the applicable water quality standard is the Drinking Water criteria, Human Health criteria or Aquatic Life criteria, whichever is more stringent.

Numeric Water Quality Criteria for Toxic Pollutants.

(1) The applicable numeric water quality criteria for the toxic pollutants to protect the designated uses of the Territorial Waters shall be the United States Environmental Protection Agency's (EPA) national recommended Clean Water Act section 304(a) water quality criteria (http://www.epa.gov/wqc/national-recommended-water-quality-criteria, accessed on August 17, 2018), EPA's Office of Water, Office of Science and Technology (4304T), adopted for the protection of freshwater and saltwater aquatic life from acute (criterion maximum concentration) and chronic (criterion continuous concentration) effects; and, the protection of human health.

(A) The applicable criteria are as follows:

Table I. Aquatic Life Criteria

Pollutant	CAS Number	**Freshwater** *(apply to IF & IG waters)* CMC (acute) (A1) (µg/L)	CCC (chronic) (A2) (µg/L)	**Saltwater** *(apply to IBS, A, B, and C waters)* CMC (acute) (A1) (µg/L)	CCC (chronic) (A2) (µg/L)
Acrolein	107028	3	3		
Aldrin (H)	309002	3.0		1.3	
Alkalinity (I)	—		20000		
alpha-Endosulfan (H)	959988	0.22	0.056	0.034	0.0087
Aluminum pH 6.5 – 9.0	7429905	750	87		

Pollutant	**CAS Number**	**Freshwater** *(apply to IF & IG waters)*		**Saltwater** *(apply to IBS, A, B, and C waters)*	
		CMC (acute) (A1) (µg/L)	**CCC (chronic) (A2) (µg/L)**	**CMC (acute) (A1) (µg/L)**	**CCC (chronic) (A2) (µg/L)**
Ammonia (A.3)	7664417	Freshwater Criteria Are pH, Temperature and Life-stage Dependent (B1) Saltwater Criteria Are pH and Temperature Dependent (B2)			
Arsenic (C)	7440382	340	150	69	36
beta-Endosulfan (H)	33213659	0.22	0.056	0.034	0.0087
Carbaryl	63252	2.1	2.1	1.6	
Cadmium (C) (D) (J)	7440439	1.8	0.72	33	7.9
Chlordane (H)	57749	2.4	0.0043	0.09	0.004
Chloride	16887006	860000	230000		
Chlorine	7782505	19	11	13	7.5
Chloropyrifos	2921882	0.083	0.041	0.011	0.0056
Chromium (III) (C) (D) (J)	16065831	570	74		
Chromium (VI) (C)	18540299	16	11	1,100	50
Copper (C)	7440508	Freshwater criteria calculated using the Biotic Ligand Model		4.8	3.1
Cyanide (F)	57125	22	5.2	1	1
Demeton	8065483		0.1		0.1
Diazinon	333415	0.17	0.17	0.82	0.82
Dieldrin	60571	0.24	0.056	0.71	0.0019
Endrin	72208	0.086	0.036	0.037	0.0023
gamma-BHC (Lindane)	58899	0.95		0.16	
Gases, Total Dissolved	—	To protect freshwater and marine aquatic life, the total dissolved gas concentrations in water should not exceed 100 percent of the saturation value for gases at the existing atmospheric and hydrostatic pressures.			
Guthion (I)	86500		0.01		0.01
Heptachlor (H)	76448	0.52	0.0038	0.053	0.0036
Heptachlor Epoxide (H)	1024573	0.52	0.0038	0.053	0.0036
Iron (I)	7439896		1000		
Lead (C) (D) (J)	7439921	65	2.5	210	8.1
Malathion (I)	121755		0.1		0.1
Mercury Methylmercury (C)	7439976 22967926	1.4	0.77	1.8	0.94
Methoxychlor (I)	72435		0.03		0.03
Mirex (I)	2385855		0.001		0.001

Pollutant	CAS Number	Freshwater (apply to IF & IG waters) CMC (acute) (A1) (µg/L)	Freshwater (apply to IF & IG waters) CCC (chronic) (A2) (µg/L)	Saltwater (apply to IBS, A, B, and C waters) CMC (acute) (A1) (µg/L)	Saltwater (apply to IBS, A, B, and C waters) CCC (chronic) (A2) (µg/L)
Nickel (C) (D) (J)	7440020	470	52	74	8.2
Nonylphenol	84852153	28	6.6	7	1.7
Parathion	56382	0.065	0.013		
Pentachlorophenol	87865	19	15	13	7.9
Polychlorinated Biphenyls (PCBs) (E)			0.014		0.03
Selenium (C)	7782492	Freshwater criteria applied as a four-part criterion (K)		290	71
Silver (C) (D)	7440224	3.2		1.9	
Sulfide-Hydrogen Sulfide	7783064		2.0		2.0
Toxaphene	8001352	0.73	0.0002	0.21	0.0002
Tributyltin (TBT)	—	0.46	0.072	0.42	0.0074
Zinc (C) (D)	7440666	120	120	90	81
4,4'-DDT (G)	50293	1.1	0.001	0.13	0.001

Table 1 Footnotes:

A - Frequency and Duration of Criteria Exceedance:

A1. Acute aquatic life protection criteria are expressed as one-hour average not to be exceeded more than once over a three-year period.

A2. Chronic aquatic life protection criteria are expressed as four-day average not to be exceeded more than once over a three-year period.

A3. For ammonia, the highest four-day average within the 30-day period should not exceed 2.5 times the CCC.

B – Ammonia criteria calculations:

B1. Freshwater Ammonia Criteria (for Class IF waters):

i. Acute Criterion:

The one-hour average concentration of total ammonia nitrogen (in mg TAN/L) is not to exceed, more than once every three years on the average, the CMC (acute criterion magnitude) calculated using the following equation:

$$CMC = MIN\left(\left(\frac{0.275}{1 + 10^{7.204-pH}} + \frac{39.0}{1 + 10^{pH-7.204}}\right), \left(0.7249 \times \left(\frac{0.0114}{1 + 10^{7.204-pH}} + \frac{1.6181}{1 + 10^{pH-7.204}}\right) \times \left(23.12 \times 10^{0.036\times(20-T)}\right)\right)\right)$$

ii. Chronic Criterion Calculations

The thirty-day rolling average concentration of total ammonia nitrogen (in mg TAN/L) is not to exceed, more than once every three years on the average, the chronic criterion magnitude (CCC) calculated using the following equation:

$$CCC = 0.8876 \times \left(\frac{0.0278}{1 + 10^{7.688-pH}} + \frac{1.1994}{1 + 10^{pH-7.688}}\right) \times \left(2.126 \times 10^{0.028\times(20-MAX(T,7))}\right)$$

In addition, the highest four-day average within the 30-day averaging period should not be more than 2.5 times the CCC (e.g., 2.5 x 1.9 mg TAN/L at pH 7 and 20°C or 4.8 mg TAN/L) more than once in three years on average.

B2. Saltwater Ammonia Criteria (for Class A, B, C, and IBS Waters):

Concentrations based on total ammonia for the pH range of 7.0 to 9.0, temperature range of 0 to 35°C, and salinities of 10, 20 and 30 g/kg are provided in Tables 1 and 2 below:

Table 1. Water quality criteria for saltwater aquatic life based on total ammonia (mg/L) – Criteria Maximum Concentrations.

	Temperature (C deg)					
	10	15	20	25	30	35
pH	**Salinity = 10 g/kg**					
7.0	131	92	62	44	29	21
7.2	83	58	40	27	19	13
7.4	52	35	25	17	12	8.3
7.6	33	23	16	11	7.7	5.6
7.8	21	15	10	7.1	5.0	3.5
8.0	13	9.4	6.4	4.6	3.1	2.3
8.2	8.5	5.8	4.2	2.9	2.1	1.5
8.4	5.4	3.7	2.7	1.9	1.4	1.0
8.6	3.5	2.5	1.8	1.3	0.98	0.75
8.8	2.3	1.7	1.2	0.92	0.71	0.56
9.0	1.5	1.1	0.85	0.67	0.52	0.44
pH	**Salinity = 20 g/kg**					
7.0	137	96	64	44	31	21
7.2	87	60	42	29	20	14
7.4	54	37	27	18	12	8.7
7.6	35	23	17	11	7.9	5.6
7.8	23	15	11	7.5	5.2	3.5
8.0	14	9.8	6.7	4.8	3.3	2.3
8.2	8.9	6.2	4.4	3.1	2.1	1.6
8.4	5.6	4.0	2.9	2.0	1.5	1.1
8.6	3.7	2.7	1.9	1.4	1.0	0.77
8.8	2.5	1.7	1.3	0.94	0.73	0.56
9.0	1.6	1.2	0.87	0.69	0.54	0.44
pH	**Salinity = 30 g/kg**					
7.0	148	102	71	48	33	23
7.2	94	64	44	31	21	15
7.4	58	40	27	19	13	9.4
7.6	37	25	21	12	8.5	6.0
7.8	23	16	11	7.9	5.4	3.7
8.0	15	10	7.3	5.0	3.5	2.5
8.2	9.6	6.7	4.6	3.3	2.3	1.7
8.4	6.0	4.2	2.9	2.1	1.6	1.1
8.6	4.0	2.7	2.0	1.4	1.1	0.81
8.8	2.5	1.8	1.3	1.0	0.75	0.58
9.0	1.7	1.2	0.94	0.71	0.56	0.46

Table 2. Water quality criteria for saltwater aquatic life based on total ammonia (mg/L) – Criteria Continuous Concentrations.

Temperature (C deg)						
	10	15	20	25	30	35
pH	**Salinity = 10 g/kg**					
7.0	20	14	9.4	6.6	4.4	3.1
7.2	12	8.7	5.9	4.1	2.8	2.0
7.4	7.8	5.3	3.7	2.6	1.8	1.2
7.6	5.0	3.4	2.4	1.7	1.2	0.84
7.8	3.1	2.2	1.5	1.1	0.75	0.53
8.0	2.0	1.4	0.97	0.69	0.47	0.34
8.2	1.3	0.87	0.62	0.44	0.31	0.23
8.4	0.81	0.56	0.41	0.29	0.21	0.16
8.6	0.53	0.37	0.27	0.2	0.15	0.11
8.8	0.34	0.25	0.18	0.14	0.11	0.08
9.0	0.23	0.17	0.13	0.1	0.08	0.07
pH	**Salinity = 20 g/kg**					
7.0	21	14	9.7	6.6	4.7	3.1
7.2	13	9.0	6.2	4.4	3.0	2.1
7.4	8.1	5.6	4.1	2.7	1.9	1.3
7.6	5.3	3.4	2.5	1.7	1.2	0.84
7.8	3.4	2.3	1.6	1.1	0.78	0.53
8.0	2.1	1.5	1.0	0.72	0.5	0.34
8.2	1.3	0.94	0.66	0.47	0.31	0.24
8.4	0.84	0.59	0.44	0.3	0.22	0.16
8.6	0.56	0.41	0.28	0.2	0.15	0.12
8.8	0.37	0.26	0.19	0.14	0.11	0.08
9.0	0.24	0.18	0.13	0.1	0.08	0.07
pH	**Salinity = 30 g/kg**					
7.0	22	15	11	7.2	5.0	3.4
7.2	14	9.7	6.6	4.7	3.1	2.2
7.4	8.7	5.9	4.1	2.9	2.0	1.4
7.6	5.6	3.7	3.1	1.8	1.3	0.9
7.8	3.4	2.4	1.7	1.2	0.81	0.56
8.0	2.2	1.6	1.1	0.75	0.53	0.37
8.2	1.4	1.0	0.69	0.5	0.34	0.25
8.4	0.9	0.62	0.44	0.31	0.23	0.17
8.6	0.59	0.41	0.3	0.22	0.16	0.12
8.8	0.37	0.27	0.2	0.15	0.11	0.09
9.0	0.26	0.19	0.14	0.11	0.08	0.07

C - Freshwater and saltwater criteria for metals are expressed in terms of the dissolved metal in the water column.

Table 3. Conversion Factors for Dissolved Metals

Metal	**Conversion Factor**			
	freshwater CMC	**freshwater CCC**	**saltwater CMC**	**saltwater CCC**
Arsenic	1.000	1.000	1.000	1.000
Cadmium	1.136672-[(Inhardness)(0.041838)]	1.101672-[(Inhardness)(0.041838)]	0.994	0.994
Chromium III	0.316	0.860	—	—
Chromium VI	0.982	0.962	0.993	0.993
Copper	0.960	0.960	0.83	0.83
Lead	1.46203-[(Inhardness)(0.145712)]	1.46203-[(Inhardness)(0.145712)]	0.951	0.951
Mercury	0.85	0.85	0.85	0.85
Nickel	0.998	0.997	0.990	0.990
Selenium	—	—	0.998	0.998
Silver	0.85	—	0.85	—
Zinc	0.978	0.986	0.946	0.946

D - Parameters for Calculating Freshwater Dissolved Metals Criteria That Are Hardness-Dependent

Table 4.

Chemical	**m_A**	**b_A**	**m_C**	**b_C**	**Freshwater Conversion Factors (CF)**	
					CMC	**CCC**
Cadmium	1.0166	-3.924	0.7409	-4.719	1.136672-[(Inhardness)(0.041838)]	1.101672-[(Inhardness)(0.041838)]
Chromium III	0.8190	3.7256	0.8190	0.6848	0.316	0.8600
Lead	1.273	-1.460	1.273	-4.705	1.46203-[(Inhardness)(0.145712)]	1.46203-[(Inhardness)(0.145712)]
Nickel	0.8460	2.255	0.8460	0.0584	0.998	0.997
Silver	1.72	-6.59	—	—	0.85	—
Zinc	0.8473	0.884	0.8473	0.884	0.978	0.986

Hardness-dependant metals' criteria may be calculated from the following:
CMC (dissolved) = $\exp\{m_A [\ln(\text{hardness})] + b_A\}$ (CF)
CCC (dissolved) = $\exp\{m_C [\ln(\text{hardness})] + b_C\}$ (CF)

E - This criterion applies to total PCBs, (e.g., the sum of all congener or all isomer or homolog or Aroclor analyses.)

F - This recommended water quality criterion is expressed as µg free cyanide (as CN)/L.
G - This criterion applies to DDT and its metabolites (i.e., the total concentration of DDT and its metabolites should not exceed this value).

H - These criteria are based on the 1980 criteria which used different Minimum Data Requirements and derivation procedures from the 1985 Guidelines. If evaluation is to be done using an averaging period, the acute criteria values given should be divided by 2 to obtain a value that is more comparable to a CMC derived using the 1985 Guidelines.

I - The CCC of 20mg/L is a minimum value except where alkalinity is naturally lower, in which case the criterion cannot be lower than 25% of the natural level.

J - The freshwater criterion for this metal is expressed as a function of hardness (mg/L). The value given here corresponds to a hardness of 100 mg/L.

K - The freshwater criterion for Selenium is summarized in the table below:

Table 5.

Media Type	Fish Tissue[1]		Water Column[4]	
Criterion Element	**Egg/Ovary[2]**	**Fish Whole Body or Muscle[3]**	**Monthly Average Exposure**	**Intermittent Exposure[5]**
Magnitude	15.1 mg/kg dw	8.5 mg/kg dw whole body **or** 11.3 mg/kg dw muscle (skinless, boneless filet)	1.5 µg/L in lentic aquatic systems 3.1 µg/L in lotic aquatic systems	$WQC_{int} = \dfrac{WQC_{30-day} - C_{bkgrnd}(1 - f_{int})}{f_{int}}$
Duration	Instantaneous measurement[6]	Instantaneous measurement[6]	30 days	Number of days/month with an elevated concentration
Frequency	Not to be exceeded	Not to be exceeded	Not more than once in three years on average	Not more than once in three years on average

1. Fish tissue elements are expressed as steady-state.
2. Egg/Ovary supersedes any whole-body, muscle, or water column element when fish egg/ovary concentrations are measured.
3. Fish whole-body or muscle tissue supersedes water column element when both fish tissue and water concentrations are measured.
4. Water column values are based on dissolved total selenium in water and are derived from fish tissue

values via bioaccumulation modeling. Water column values are the applicable criterion element in the absence of steady-state condition fish tissue data.

5. Where $WQC_{30\text{-day}}$ is the water column monthly element, for either a lentic or lotic waters; C_{bkgrnd} is the average background selenium concentration, and f_{int} is the fraction of any 30-day period during which elevated selenium concentrations occur, with f_{int} assigned a value ≥ 0.033 (with 0.033 corresponding to 1 day).
6. Fish tissue data provide instantaneous point measurements that reflect integrative accumulation of selenium over time and space in fish population(s) at a given site.

Table II. Human Health Criteria

Pollutant	CAS Number	Human Health for the consumption of **Water+Organism (A)** (µg/L) (*apply to waters designated as potable water sources only Class I, A, B & C (where applicable based on 186-4) waters*)	**Organism Only (A)** (µg/L) *(apply to all waters except ones designated as potable water sources Class A,B,C waters)*
Acenaphthene (E)	83329	70	90
Acrolein	107028	3	400
Acrylonitrile (C)	107131	0.061	7.0
Aldrin (C)	309002	0.00000077	0.00000077
alpha-BHC	319846	0.0036	0.00039
alpha-Endosulfan	959988	20	30
Anthracene	120127	300	400
Antimony (B) (D)	7440360	5.6	640
Arsenic (C) (D)	7440382	0.018	0.14
Asbestos (D)	1332214	7 million fibers/L	-
Barium (D)	7440393	1,000	-
Benzene (C) (D)	71432	0.58-2.1	16-58
Benzidine (C)	92875	0.00014	0.011
Benzo(a) Anthracene (C)	56553	0.0012	0.0013
Benzo(a) Pyrene (C) (D)	50328	0.00012	0.00013
Benzo(b) Fluoranthene (C)	205992	0.0012	0.0013
Benzo(k) Fluoranthene (C)	207089	0.012	0.013
beta-BHC	319857	0.0080	0.014
beta-Endosulfan	33213659	20	40
Bis(2-Chloroethyl) Ether (C)	111444	0.030	2.2
Bis(2-Chloroisopropyl) Ether	108601	200	4,000
Bis(2-Ethylhexyl) Phthalate[X] (C) (D)	117817	0.32	0.37
Bis(Chloromethyl) Ether (C)	542881	0.00015	0.017
Bromoform (C) (D)	75252	7.0	120
Butylbenzyl Phthalate[W]	85687	0.10	0.10

Carbon Tetrachloride (C) (D)	56235	0.4	5
Chlordane (D)	57749	0.00031	0.00032
Chlorobenzene (D) (E)	108907	100	800
Chlorodibromomethane (C) (D)	124481	0.8	21
Chloroform (C) (D)	67663	60	2,000
Chlorophenoxy Herbicide (2,4-D) (D)	94757	1,300	12,000
Chlorophenoxy Herbicide (2,4,5-TP) [Silvex] (D)	93721	100	400
Chrysene (C) (D)	218019	0.12	0.13
Copper (C) (D)	7440508	1,300	-
Cyanide (D) (F)	57125	4	400
Dibenzo(a,h)Anthracene (C)	53703	0.00012	0.00013
Dichlorobromomethane (C) (D)	75274	0.95	27
Dieldrin (C)	60571	0.0000012	0.0000012
Diethyl Phthalate[W]	84662	600	600
Dimethyl Phthalate[W]	131113	2,000	2,000
Di-n-Butyl Phthalate[W]	84742	20	30
Dinitrophenols	25550587	10	1,000
Endosulfan Sulfate	1031078	20	40
Endrin (D)	72208	0.03	0.03
Endrin Aldehyde	7421934	1	1
Ether, Bis (Chloromethyl)	542881	0.00010	0.00029
Ethylbenzene (D)	100414	68	130
Fluoranthene	206440	20	20
Fluorene	86737	50	70
gamma-BHC (Lindane) (D)	58899	4.2	4.4
Heptachlor (C) (D)	76448	0.0000059	0.0000059
Heptachlor Epoxide (C) (D)	1024573	0.000032	0.00032
Hexachlorobenzene (C) (D)	118741	0.000079	0.000079
Hexachlorobutadiene (C) (D)	87683	0.01	0.01
Hexachlorocyclo-hexane-Technical	608731	0.0066	0.010
Hexachlorocyclopentadiene (D) (E)	77474	4	4
Hexachloroethane (C)	67721	0.1	0.1
Indeno(1,2,3-cd) Pyrene (C)	193395	0.0012	0.0013
Isophorone (C)	78591	34	1,800
Manganese (E)	7439965	50	100
Mercury Methylmercury	7439976 22967926	-	0.3 mg/kg
Methoxychlor (D)	72435	0.02	0.02

Methyl Bromide	74839	100	10,000
Methylene Chloride (C) (D)	75092	20	1,000
Nickel (B)	7440020	610	4,600
Nitrates (D)	14797558	10,000	-
Nitrobenzene (E)	98953	10	600
Nitrosamines	—	0.0008	1.24
Nitrosodibutylamine, T (C)	924163	0.0063	0.22
Nitrosodiethylamine, T (C)	55185	0.0008	1.24
Nitrosopyrrolidine, T (C)	930552	0.016	34
N-Nitrosodimethylamine (C)	62759	0.00069	3.0
N-Nitrosodi-n-Propylamine (C)	621647	0.0050	0.51
N-Nitrosodiphenylamine (C)	86306	3.3	6.0
Pentachlorobenzene	608935	0.1	0.1
Pentachlorophenol (C) (D) (E)	87865	0.03	0.04
Phenol (E) (G)	108952	4,000	300,000
Polychlorinated Biphenyls (PCBs) (C) (D)		0.000064	0.000064
Pyrene	129000	20	30
Selenium (D)	7782492	170	4,200
Solids Dissolved and Salinity	—	250,000	
Tetrachlorobenzene,1,2,4,5-	95943	0.97	1.1
Tetrachloroethylene (C) (D)	127184	10	29
Thallium	7440280	0.24	0.47
Toluene (D)	108883	57	520
Toxaphene (C) (D)	8001352	0.00070	0.00071
Trichloroethylene (C) (D)	79016	0.6	7
Trichlorophenol,2,4,5-	95954	1,800	3,600
Vinyl Chloride (C) (D)	75014	0.022	1.6
Zinc (E)	7440666	7,400	26,000
1,1,1-Trichloroethane (D)	71556	10,000	200,000
1,1,2,2-Tetrachloroethane (C)	79345	0.2	3
1,1,2-Trichloroethane (C) (D)	79005	0.55	8.9
1,1-Dichloroethylene (C) (D)	75354	300	20,000
1,2,4-Trichlorobenzene (D)	120821	0.071	0.076
1,2-Dichlorobenzene (D)	95501	1,000	3,000
1,2-Dichloroethane (C) (D)	107062	9.9	650
1,2-Dichloropropane (C) (D)	78875	0.90	31
1,2-Diphenylhydrazine (C)	122667	0.03	0.2
1,2-Trans-Dichloroethylene (D)	156605	100	4,000
1,3-Dichlorobenzene	541731	7	10

1,3-Dichloropropene (C)	542756	0.27	12
1,4-Dichlorobenzene (D)	106467	300	900
2,3,7,8-TCDD (Dioxin) (C) (D)	1746016	5.0E-9	5.1E-9
2,4,5-Trichlorophenol (E)	95954	300	600
2,4,6-Trichlorophenol (E)	88062	1.5	2.8
2,4-Dichlorophenol (E)	120832	10	60
2,4-Dimethylphenol (E)	105679	100	3,000
2,4-Dinitrophenol	51285	10	300
2,4-Dinitrotoluene (C)	121142	0.049	1.7
2-Chloronaphthalene	91587	800	1,000
2-Chlorophenol (E)	95578	30	800
2-Methyl-4,6-Dinitrophenol	534521	2	30
3,3'-Dichlorobenzidine (C)	91941	0.049	0.15
3-Methyl-4-Chlorophenol (E)	59507	500	2000
4,4'-DDD (C)	72548	0.00012	0.00012
4,4'-DDE (C)	72559	0.000018	0.000018
4,4'-DDT (C)	50293	0.000030	0.000030

Table II. Footnotes:

A - Frequency and Duration of Criteria Exceedance:

A1. Human health noncarcinogenic effect-based criteria are expressed as a 30-day average with no frequency of exceedance.

A2. Human health carcinogenic effect-based criteria are expressed as a 70-year average with no frequency of exceedance.

B - This criterion has been revised to reflect The United States Environmental Protection Agency's q1* or reference dose RfD, as contained in the Integrated Risk Information System (IRIS) as of May 17, 2002. The fish tissue bioconcentration factor (BCF) from the 1980 Ambient Water Quality Criteria document used to derive the original criterion was retained in each case.

C - This criterion is based on carcinogenicity of 10^{-6} risk.

D - EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.

E - The criterion for organoleptic (taste and order) effects may be more stringent. The organoleptic criteria can be found in Table III below for both fresh and marine waters.

F - This recommended water quality criterion is expressed as total cyanide, even though the IRIS RFD we used to derive the criterion is based on free cyanide. The multiple forms of cyanide that are present in ambient water have significant differences in toxicity due to their differing abilities to liberate the CN-moiety. Some complex cyanides require even more extreme conditions than refluxing with sulfuric acid to liberate the CN-moiety. Thus, these complex cyanides are expected to have little or no 'bioavailability' to humans. If a substantial fraction of the cyanide present in a water body is present in a complexed form (e.g., $Fe_4[Fe(CN)_6]_3$), this criterion may be over conservative.

G - This criterion has been revised to reflect the United States Environmental Protection Agency's cancer slope factor (CSF) or reference dose (RfD), as contained in the Integrated Risk Information System (IRIS) as of (date of publication of Final FR Notice). The fish tissue bioconcentration factor (BCF) from the 1980 Ambient Water Quality Criteria document was retained in each case.

Table III - Organoleptic Effects (e.g., taste and odor)

Pollutant	CAS Number	Organoleptic Effect Criteria (µg/L)
Acenaphthene	83329	20
Monochlorobenzene	108907	20
3-Chlorophenol	—	0.1
4-Chlorophenol	106489	0.1
2,3-Dichlorophenol	—	0.04
2,5-Dichlorophenol	—	0.5
2,6-Dichlorophenol	—	0.2
3,4-Dichlorophenol	—	0.3
2,4,5-Trichlorophenol	95954	1
2,4,6-Trichlorophenol	88062	2
2,3,4,6-Tetrachlorophenol	—	1
2-Methyl-4-Chlorophenol	—	1800
3-Methyl-4-Chlorophenol	59507	3000
3-Methyl-6-Chlorophenol	—	20
2-Chlorophenol	95578	0.1
Copper	7440508	1000
2,4-Dichlorophenol	120832	0.3
2,4-Dimethylphenol	105679	400
Hexachlorocyclopentadiene	77474	1
Nitrobenzene	98953	30
Pentachlorophenol	87865	30
Phenol	108952	300
Zinc	7440666	5000

§ 186 - 6: Thermal Policy

In order to protect the Territorial Waters from thermal pollution, the following criteria shall apply:

(a) Fish and other aquatic life shall be protected from thermal blocks, providing for a minimum of seventy-five percent (75%) stream or estuarine cross-section and/or volumetric passageway, including a minimum of one half of the surface as measured from water edge to water edge at any stage of tide.

(b) In non-passageway the surface water temperature shall not exceed 32°C.

(c) No heat may be added except in designated mixing zones which would cause temperatures to exceed 32°C, or which would cause the monthly mean of the maximum daily temperature at any site, prior to the addition of any heat, to be exceeded by more than 1.0°C.

(d) No discharge or combination of discharges shall be injurious to aquatic life (including threatened and endangered species listed pursuant to section 4 of the Federal *Endangered Species Act* and Title 12, Chapter 2 of the Virgin Islands Code) or the culture or propagation of a balanced indigenous population thereof.

(f) Rate of temperature change outside the mixing zone shall not be more that 0.5°C per hour nor to exceed 3°C in any 24-hour period except when natural phenomena cause these limits to be exceeded.

(g) Unless specific conditions, such as spawning ground, migratory routes, or other sections of conditions from these regulations are applicable, the mixing zone should be defined by a sphere with a specified point as the center (not necessarily the outfall but limited to one point for each installation) and a radius equal to the square root of the volume of discharge (A) expressed as millions of gallons per day, times 200 feet; and in no case exceed 3/8 mile. The formula is:
Radius (mixing zone) = $(\sqrt{A}) * 200$ feet $\leq 3/8$ mile.

§ 186 - 7: Mixing Zones

DPNR-DEP may establish mixing zones that apply to the discharge of treated wastewater to surface waters of the Territorial Waters.

(a) DPNR-DEP, in determining whether to establish/grant a mixing zone, shall apply the following criteria:

1. Mixing zones shall be limited to an area or volume as small as feasible;
2. There shall be prompt mixing of the discharge with receiving waters;
3. Mixing zones shall not be used for, or considered as a substitute for, minimum treatment technology;
4. Mixing zones shall not create nuisance conditions, accumulate pollutants in sediments or biota in toxic amounts, or diminish existing or best usages of surface waters disproportionately;
5. There shall be no mixing zones for pathogens or indicators of pathogens;
6. Mixing zones shall not encroach upon intakes for potable water supplies;
7. Mixing zones shall not encroach upon areas used for harvesting of stationary species such as shellfish;
8. There shall be no lethality to organisms passing through the mixing zone;
9. There shall be safe and adequate passage for swimming and drifting organisms;
10. The location, design, and operation of the discharge shall minimize impacts on aquatic life, and shall not interfere with biological communities, including coral reefs and all their habitats, spawning areas, nursery areas, and fish migration routes to a degree that is damaging to the ecosystem;
11. There shall be no mixing zones for discharges that would likely jeopardize the continued existence of any threatened or endangered species or all their habitats listed pursuant to section 4 of the Federal *Endangered Species Act* and threatened, endangered and indigenous species listed pursuant Title 12, Chapter 2 of the Virgin Islands Code, or result in the destruction or adverse modification of such species' critical habitat.

§ 186 - 8: Anti-degradation Policy and Implementation Procedures

DPNR-DEP shall maintain and protect existing water uses, including those that protect threatened and endangered species listed pursuant to section 4 of the *Federal Endangered Species Act* and Title 12, Chapter 2 of the Virgin Islands Code, as well as the level of water quality necessary to protect existing uses.

(a) Anti-degradation Policy

(1) In those water bodies where the quality exceeds levels necessary to support the protection and propagation of fish, wildlife, desirable species, including threatened or endangered species and recreation in and on the water, that quality shall be maintained and protected.

(2) A lower water quality may be allowed when the Territory's Water Quality Management Program determines, after full satisfaction, and in accordance with the public review process (§ 186- 15 herein), that allowing lower water quality is necessary to accommodate important economic or social development and will not interfere with or become injurious to any assigned uses made of, or presently possible in such waters. In allowing such lower water quality, the Territory's Water Quality Management Program shall require a water quality level adequate to fully protect existing and designated uses. Further, the Territory's Water Quality Management Program will require that:

- The highest statutory and regulatory requirements for all new and/or existing point sources be achieved, and
- The cost-effective and reasonable best management practices for non-point sources control be implemented.

A. The Territory's Water Quality Management Program identify waters for the protections describe in (a)(2) of this section on a parameter-by-parameter basis

B. Before allowing any lowering of high water quality, pursuant to (a)(2) of this section, the Territory's Water Quality Management Program shall conduct an analysis of alternatives. If found that such a lowering is necessary to accommodate important economic or social development in the area in which the waters are located, it may be allowed provided that water quality remains sufficient to protect and maintain the existing and designated uses of that water body. The analysis of alternatives shall evaluate a range of practicable solutions/best management practices that would prevent or lessen the degradation associated with the proposed activity. When the analysis identifies one or more practicable alternatives, the Territory's Water Quality Management Program shall only find that a lowering is necessary if one such alternative is selected for implementation.

(3) Where high quality waters constitute an outstanding national resource, such as waters of National and State parks and wildlife refuges and waters of exceptional recreational or ecological significance, the water quality shall be maintained and protected.

(4) Where potential water quality impairment is associated with a thermal discharge this thermal discharge must comply with Section 316 of the Clean Water Act, as amended, 33 U.S.C. §1326.

(b) Antidegradation Implementation Procedures

(1) General: In conducting an antidegradation review, the DPNR-DEP will sequentially apply the following steps:

(A) Determine which level of antidegradation applies:

(i) Tier 1 – Protection of Existing and Designated Uses
(ii) Tier 2 – Protection of High Quality Waters
(iii) Tier 3 – Protection of Outstanding National Resource Waters (ONRWs)

These three tiers differ from the classification system of the three classes of Marine and Coastal Waters (Class A, B and C) described in § 186 – 4. For Tiered waters in this section, a higher Tier correlates to a higher quality of water designation. Class B & C waters are generally Tier 1 and 2, while Class A waters are almost always Tier 3 for the purposes of this section.

(B) Review existing water quality data and other information submitted by the applicant. The applicant shall provide to the DPNR the information regarding the discharge required by the WQS including, but not limited to the following:

(i) Description of the nature of the pollutants to be discharged,

(ii) Treatment technologies applied to the pollutants to be discharged,

(iii) Nature of the petitioner's business,

(iv) Daily maximum and average flow to be discharged,

(v) Effluent characterization,

(vi) Effluent limitations requested to be applied to the discharge according to the TPDES regulations,

(vii) Location of the point of discharge,

(viii) Receiving water body name,

(ix) Water quality data of the receiving water body,

(x) Receiving water body minimum flow for stream waters,

(xi) Location of water intakes within the water body, and

(xii) In the event that the proposed discharge will result in the lowering of water quality, data and information demonstrating that the discharge is necessary to accommodate important economic or social development in the area where the receiving waters are located

(C) Determine if additional information or assessment is necessary to make the decision.

(D) Prepare an intent to issue or deny the request for an increased loading and publish a notice in a newspaper of wide circulation in the island where the water body is located informing the public of DPNR's preliminary decision and granting a public participation period of at least thirty (30) days.

(E) Address the comments received from the interested parties and consider such comments as part of the decision making process.

(F) Make the final determination to approve or deny the request for an increased loading.

(2) **Tier 1 - Existing uses protection:** All existing uses and the water quality necessary to protect the existing uses shall be maintained and protected.

(A) Tier 1 waters are:

(i) Those Waters of the US Virgin Islands identified as impaired and that have been included in the list required by Section 303(d) of the CWA; and

(ii) Those Territorial Waters for which attainment of applicable water quality standards has been or is expected to be, achieved through implementation of effluent limitations more stringent than technology-based controls.

(B) To implement Tier 1 anti-degradation, DPNR must determine if a discharge would lower the water quality to the extent that it would no longer be sufficient to protect and maintain the existing and designated uses of that water body.

(C) When a water body has been affected by a parameter of concern causing it to be included on the 303(d) List, then the DPNR will not allow an increase of the concentration of the parameter of concern or pollutants affecting the parameter of concern in the water body. This “no increase” will be achieved by meeting the applicable water quality standards at the end of the pipe. Until such time that a Total Maximum Daily Load (TMDL) is developed for the parameter of concern for the water body, no discharge will be allowed to cause or contribute to further degradation of the water body.

(D) When the assimilative capacity of a water body is not sufficient to ensure maintenance of the water quality standard for a parameter of concern with an additional load to the water body, then the DPNR will not allow an increase of the concentration of the parameter of concern, or pollutants affecting the parameter of concern, in the water body. This “no increase” will be achieved by meeting the applicable water quality standards at the end of the pipe. Until such time that a TMDL is developed for the parameter of concern for the water body, no discharge will be allowed to cause or contribute to further degradation of the water body.

(3) **Tier 2 - High quality water protection:**

(A) Identification of high quality water shall be performed on a parameter-by-parameter basis. Waters shall not be excluded from Tier 2 protection based solely on the impairment of a single parameter or group of parameters if any of the uses specified in CWA section 101(a)(2) is attained –

(B) To verify that a water body is a high quality water for a parameter of concern, which initiates a Tier 2 antidegradation review, the DPNR must evaluate and determine:

(i) The existing water quality of the water body,

(ii) The projected water quality of the water body, and

(iii) If the existing and designated uses of the water body will be fully maintained and protected in the event of a lowering of water quality.

(C) In multiple discharge situations, the effects of all discharges shall be evaluated.

(D) Alternative analysis and social/economic analysis shall be conducted consistent with the requirements of § 186 - 8(a)(2)(B) above.

(E) An antidegradation demonstration submittal is required for any person seeking to lower the water quality in a High Quality Water. The antidegradation demonstration submittal to DPNR must include the following:

(i) Pollution Prevention Alternatives Analysis: Identify any cost-effective pollution prevention alternatives and techniques that are available to person that would eliminate or significantly reduce the extent to which the increased loading results in a lowering of the water quality.

(ii) Alternative or Enhanced Treatment Analysis: Identify alternative or enhanced treatment techniques that are available to the person that would eliminate the lowering of the water quality and their costs relative to the cost of treatment necessary to achieve applicable effluent limitations.

(iii) Important Social or Economic Development Analysis: Identify the social or economic development and the benefits to the area in which the waters are located that will be foregone if the lowering of water quality is not allowed.

(F) In order to allow the lowering of water quality in high quality waters, the applicant must show and justify the necessity for such lowering of the water quality. DPNR will not allow the entire assimilative capacity of a water body for a parameter of concern to be allocated to a discharger, if the necessity of the requested effluent limitation for the parameter of concern is not demonstrated to the full satisfaction of DPNR.

(G) The public review process in § 186-15 shall be fully satisfied in any finding that will allow a lower water quality.

(H) Requirements for point and nonpoint sources when allowing a lowering of water quality shall be the highest statutory and regulatory requirements for all new and existing point sources and all cost-effective and reasonable best management practices for nonpoint source control.

(I) DPNR's Antidegradation decision process is as follows:

(i) Once DPNR determines that the information provided by the person proposing to increase loadings is administratively complete, DPNR shall use that information to determine whether or not the lowering of the water quality is necessary.

(ii) If DPNR determines that the lowering of the water quality is necessary, DPNR must then determine whether or not the lowering of the water quality will support important social and economic development in the area.

(iii) If the proposed lowering of water quality is either not necessary, or will not support important social and economic development, DPNR shall deny the request to lower the water quality.

(iv) If the lowering of the water quality is necessary, and will support important social and economic development, DPNR may allow all or part of the proposed lowering to occur as necessary.

(4) **Tier 3 - ONRW Protection:** Waters identified as ONRWs shall be maintained and protected.

(A) The DPNR may designate a specific water as Class A which identifies it as an ONRW.

(B) Any interested party may nominate a specific water to be classified as an ONRW and the DPNR will make the final determination.

(C) The natural conditions of Class A waters shall not be changed. No new or increased point source dischargers will be permitted in

ONRWs.

(5) Thermal Discharges: Consistency with Section 316 of the CWA shall be ensured in cases that involve potential water quality impairment associated with thermal discharges.

§ 186 - 9: Analytical Procedure

(a) The analytical procedures used as methods of analysis to determine the chemical, bacteriological, biological, and radiological quality of waters sampled shall be in accordance with those specified in or approved under Title 40 of the Code of Federal Regulations (CFR) Part 136 or other methods approved by the DPNR and the EPA.

§ 186 - 10: Applicability of Standards

(a) If a requirement established by any provision of this Regulation is either more restrictive or less restrictive than a requirement established by any other provision of this Regulation, or by any other law, regulation, standard, or limit established by any duly constituted governmental authority having jurisdiction, the requirement which is more restrictive shall apply.

§186 - 11: Natural Conditions

(a) Natural waters may have characteristics outside of the limits prescribed by these regulations. The criteria contained herein do not relate to violations of standards resulting from natural forces.

§ 186 - 12: Schedules of Compliance for limits in TPDES Permits

(a) Persons, who are authorized to discharge pollutants into the Waters of the United States Virgin Islands at the time these WQS are amended to add or make more stringent any water quality standards, shall meet such newly adopted or more stringent water quality standards within three (3) years of the effective date of the amendment.

(b) The Commissioner shall upon the expiration of the three (3) years revoke or modify any discharge permit previously issued which result in reducing the quality of such waters below the newly established standards.

(3) Nothing in this Section shall limit any authority of the Commissioner to set or revise schedules of compliance pursuant to the statues and regulations referred to herein.

(4) Any schedule of compliance issued by the Commissioner shall be in accordance with sections 502(17) and 301(b)(1)(C) of the Act.

§ 186 - 13: Site-specific Criteria

DPNR may allow site-specific modifications to criteria on a site-specific basis in order to reflect local environmental conditions.

(a) Requirements for Site-specific Modifications to Criteria: Any modification must comply with the following:

(1) Modifications must be protective of designated uses and aquatic life, wildlife or human health.

(2) Modifications that result in less stringent criteria must be based on a sound scientific rationale and shall not be likely to jeopardize the continued existence of endangered or threatened species listed or proposed under section 4 of the Federal *Endangered Species Act* and Title 12, Chapter 2 of the Virgin Islands Code or result in the destruction or adverse modification of such species' critical habitat.

(3) More stringent modifications shall be developed to protect endangered or threatened species listed or proposed under section 4 of the federal ESA and Title 12, Chapter 2 of the Virgin Islands Code, where such modifications are necessary to ensure that water quality is not likely to jeopardize the continued existence of such species or result in the destruction or adverse modification of such species' critical habitat.

(4) Modification that result in less stringent criteria must comply with the public review process in § 186 – 15 herein.

(5) Modifications must be submitted by DPNR to EPA for approval.

(b) Aquatic Life Criteria:

(1) Aquatic life criteria may be modified on a site-specific basis to provide an

additional level of protection.

(2) Less stringent site-specific modifications to chronic or acute aquatic life criteria may be developed when:

(A) The local water quality characteristics such as pH, hardness, temperature, color, etc., alter the biological availability or toxicity of a pollutant; or

(B) The sensitivity of the aquatic organism species that "occur at the site" differs from the species actually tested in developing the criteria. The phrase "occur at the site" includes the species, genera, families, orders, classes, and phyla that: are usually present at the site; are present at the site only seasonally due to migration; are present intermittently because they periodically return to or extend their ranges into the site; were present at the site in the past, are not currently present at the site due to degraded conditions, and are expected to return to the site when conditions improve; are present in nearby bodies of water, are not currently present at the site due to degraded conditions, and are expected to be present at the site when conditions improve. The taxa that "occur at the site" cannot be determined merely by sampling downstream and/or upstream of the site at one point in time. "Occur at the site" does not include taxa that were once present at the site but cannot exist at the site now due to permanent physical alteration of the habitat at the site resulting, for example, from dams, etc.

(3) Less stringent modifications also may be developed to acute and chronic aquatic life criteria to reflect local physical and hydrological conditions.

(c) Human Health Criteria:

(1) Human health criteria may be modified on a site-specific basis to provide and additional level of protection. Human health criteria shall be modified on a site-specific basis to provide additional protection appropriate for highly exposed subpopulations.

(2) Less stringent site-specific modifications to human health criteria may be developed when: i. local fish consumption rates are lower than the rate used in deriving the human health criteria in 186-5(c) and/or ii. a site-specific bioaccumulation factor is derived which is lower than that used in deriving human health criteria in 186-5(c).

§ 186 - 14: Water Quality Standards Variances

It is DPNR's policy that a WQS variance is only appropriate when a designated use is not attainable in the short-term but might be attainable in the long-term. DPNR-DEP may consider a temporary modification to a designated use and associated water quality criteria that would otherwise apply.

(a) Applicability: A variance from any WQS that is the basis of a water quality-based effluent limitation included in a TPDES Permit is based on the following:

(1) A variance from WQS applies only to the permittee requesting the WQS variance, the water body/waterbody segment(s) specified in the WQS variance and only to the pollutant or pollutants specified in the WQS variance.

(2) A WQS variance does not affect, or require DPNR to modify, in its standards, the underlying designated use and criterion address by the WQS variance, unless the Territory adopts and EPA approves a revision to the underlying designated use and criterion consistent with §131.10 and §131.11. All other applicable standards not specifically addressed by the WQS variance remain applicable.

(3) A variance does not affect, or require DPNR to modify, the corresponding water quality standard for the waterbody as a whole.

(4) A WQS variance, once adopted by the Territory and approved by EPA, shall be the applicable standard for purposes of the CWA under 40 CFR 131.21(d)-(e), for the following limited purposes. An approved WQS variance applies for the purposes of developing TPDES permit limits and requirements under 301(b)(1)(C), where appropriate, consistent with paragraph (a)(1) of this section. DPNR and other certifying entities may also use an approved WQS variance when issuing certifications under section 401 of the CWA.

(5) A variance from a water quality standard shall not be adopted that would likely jeopardize the continued existence of any endangered or threatened species listed under Section 4 of the Federal *Endangered Species Act* (ESA) Act and Title 12, Chapter 2 of the Virgin Islands Code or result in the destruction or adverse modification of such species' critical habitat.

(6) A variance from WQS shall not be adopted if standards will be attained by implementing effluent limits required under sections 301(b) and 306 of the Clean Water Act (CWA) and by the permittee implementing cost-effective and reasonable best management practices for nonpoint source control.

(b) The maximum timeframe: A variance from the WQS shall not exceed five (5) years or the term of the TPDES permit, whichever is less. DPNR will review, and modify as necessary, variances from WQS as part of each water quality standards review pursuant to section 303(c) of the CWA.

(c) Conditions to adopt: A variance from the WQS may be adopted if, and only if:

(1) The permittee demonstrates to DPNR that attaining the WQS is not feasible because:

(A) Naturally occurring pollutant concentrations prevent the attainment of the WQS;

(B) Natural, ephemeral, intermittent or low flow conditions or water levels prevent the attainment of the WQS, unless these conditions may be compensated for by the discharge of sufficient volume of effluent to enable WQS to be met without violating Territorial water conservation requirements;

(C) Human-caused conditions or sources of pollution prevent the attainment of the WQS and cannot be remedied, or would cause more environmental damage to correct than to leave in place;

(D) Dams, diversions or other types of hydrologic modifications preclude the attainment of the WQS, and it is not feasible to restore the waterbody to its original condition or to operate such modification in a way that would result in the attainment of the WQS;

(E) Physical conditions related to the natural features of the waterbody, such as the lack of a proper substrate cover, flow, depth, pools, riffles, and the like, unrelated to chemical water quality, preclude attainment of WQS; or

(F) Controls more stringent than those required by sections 301(b) and 306 of the CWA would result in substantial and widespread economic and social impact.

(2) The permittee shall also:

(A) Show that the WQS variance requested conforms to the requirements of the antidegradation procedures in §186-8; and

(B) Characterize the extent of any increased risk to human health and the environment associated with adoption of the WQS variance compared with compliance with WQS absent the variance, such that DPNR is able to conclude that any such increased risk is consistent with the protection of the public health, safety and welfare.

(d) Requirements for Submission of Application to DPNR

(1) An application for a WQS variance must include:

(A) Identification of the pollutant(s) or water quality parameter(s), and the water body/waterbody segment(s) to which the WQS variance applies. Discharger(s)-specific WQS variances must also identify the permittee(s) subject to the WQS variance and the anticipated degree of variance from WQS.

(B) The requirements that apply throughout the term of the WQS variance. The requirements shall represent the highest attainable condition of the water body or waterbody segment applicable throughout the term of the WQS variance based on the documentation required in (d)(2) of this section. The requirements shall not result in any lowering of the currently attained ambient water quality, unless a WQS variance is necessary for restoration activities, consistent with paragraph (d)(2)(A)(i)(b) of this section. DPNR must specify the highest attainable condition of the water body or waterbody segment as a quantifiable expression that is one of the following:

(i) For discharger(s)-specific WQS variances:

(a) The highest attainable interim criterion, or

(b) The interim effluent condition that reflects the greatest pollutant reduction achievable, or

(c) If no additional feasible pollutant control technology can be identified, the interim criterion or interim effluent condition that reflects the greatest pollutant reduction achievable with the pollutant control technologies installed at the time the Territory adopts the WQS variance, and the adoption and implementation of a Pollutant Minimization Program.

(ii) For WQS variances applicable to a water body or waterbody segment:

(a) The highest attainable interim use and interim criterion, or

(b) If no additional feasible pollutant control technology can be identified, the interim use and interim criterion that reflects the greatest pollutant reduction achievable with the pollutant control technologies installed at the time the Territory adopts the WQS variance, and the adoption and implementation of a Pollutant Minimization Program.

(C) A statement providing that the requirements of the WQS variance is the highest attainable condition identified at the time of the adoption of the WQS variance

(D) The term of the WQS variance, expressed as an interval of time from the date of DPNR approval or a specific date. The term of the WQS variance must only be as long as necessary to achieve the highest attainable condition and consistent with the demonstration provided in paragraph (d)(2) of this section. The Territory may adopt a subsequent WQS variance consistent with this section.

(2) The supporting documentation must include:

(A) Documentation demonstrating the need for a WQS variance.

(i) For a WQS variance to a use specified in section 101(a)(2) of the CWA or a sub-category of such a use, DPNR must demonstrate that attaining the designated use and criterion is not feasible throughout the term of the WQS variance because:

(a) All relevant information demonstrating that attaining the WQS is not feasible based on one or more of the conditions in §186-14 (c)(1) herein; or

(b) Actions necessary to facilitate lake, wetland, or stream restoration through dam removal or other significant reconfiguration activities preclude attainment of the designated use and criterion while the actions are being implemented.

(ii) For a WQS variance to a non-101(a)(2) use, DPNR must submit documentation justifying how its consideration of the use and value of the water for those uses listed in § 131.10(a) appropriately supports the WQS variance and term. A demonstration consistent with (d)(2)(A)(i) of this section may be used to satisfy this requirement.

(B) Documentation demonstrating that the term of the WQS variance is only as long as necessary to achieve the highest attainable condition. Such documentation must justify the term of the WQS variance by describing the pollutant control activities to achieve the highest attainable condition, including those activities identified through a Pollutant Minimization Program, which serve as milestones for the WQS variance.

(C) In addition to (A) and (B) of this section, for a WQS variance that applies to a water body or waterbody segment:

(i) Identification and documentation of any cost-effective and reasonable best management practices for nonpoint source controls related to the pollutant(s) or water quality parameter(s) and water body or waterbody segment(s) specified in the WQS variance that could be implemented to make progress towards attaining the underlying designated use and criterion. DPNR must provide public notice and comment for any such documentation.

(ii) Any subsequent WQS variance for a water body or waterbody segment must include documentation of whether and to what extent best management practices for nonpoint source controls were implemented to address the pollutant(s) or water quality parameter(s) subject to the WQS variance and the water quality progress achieved.

(D) All relevant information demonstrating compliance with the conditions in §186-14 (c)(2) herein.

(e) Implementing WQS variances in TPDES permits: A WQS variance serves as the applicable water quality standard for implementing TPDES permitting requirements pursuant to 12 VIRR §184-54(c) and 40 CFR § 122.44(d) for the term of the WQS variance. Any limitations and requirements necessary to implement the WQS variance shall be included as enforceable conditions of the TPDES permit for the permittee(s) subject to the WQS variance.

(f) Public notice of preliminary decision: Upon receipt of a complete application for a variance from the WQS, and upon making a preliminary decision regarding the WQS variance, DPNR-DEP shall public notice the request and preliminary decision for public comment. This public notice will be satisfied by including the supporting information for the variance from the WQS and the preliminary decision in the public notice of a draft TPDES permit.

(g) Final decision: DPNR-DEP will issue a final decision on a WQS variance request within 90 days of the expiration of the public comment period required in accordance with the TPDES permit. If DPNR-DEP approves all or part of the variance from the WQS, the decision shall include all permit conditions needed to implement those parts of the WQS variance as approved. Such permit conditions shall, at a minimum, require:

(1) Compliance with an initial effluent limitation which, at the time the variance from the WQS is granted, represents the level currently achievable by the permittee, and which is no less stringent than that achieved under the previous permit;

(2) Achieving reasonable progress toward attaining the water quality standards for the waterbody as a whole through appropriate conditions;

(3) When the duration of a variance from the WQS is shorter than the duration of a permit, compliance with an effluent limitation sufficient to meet the underlying water quality standard, upon the expiration of said WQS variance; and

(4) A provision that allows DPNR to reopen and modify or revoke any condition granted in a WQS variance due to the permittee not providing relevant information that reasonable would affect the decision process.

(h) Incorporating the WQS variance: DPNR-DEP will establish and incorporate into the permittee's TPDES permit all conditions needed to implement the variance from the WQS as determined in §186-14 (g) herein.

(i) Renewal of WQS variance: A WQS variance may be renewed, subject to the requirements of §186-14 (a) through §186-14 (h) herein. As part of any renewal application, the permittee shall again demonstrate that attaining the WQS is not feasible based on the requirements of §186-14 (c). The permittee's application shall also contain information concerning its compliance with the conditions incorporated into its permit as part of the original variance from the WQS pursuant to §186-14 (g) through §186-14 (h) herein. Renewal of a WQS variance may be denied if the permittee did not comply with the conditions of the original WQS variance.

(i) EPA Approval: DPNR shall submit all variances from the WQS and supporting information to EPA Region 2 for approval. The submittal shall include

(1) Relevant permittee applications pursuant to §186-14 (d),

(2) Public comments and records of any public hearings pursuant to §186-14 (f),

(3) The final decision pursuant to §186-14 (g) of this procedure, and

(4) TPDES permits issued pursuant to §186-14 (h) of this procedure.

§ 186 - 15: Public Review Process

(a) Public Notice: Public notice shall be published on each island in one (1) newspaper of wide circulations within that island informing of the DPNR-DEP's intention to amend the VI Water Quality Standards Regulations. Such notice shall also:

(1) Inform the public and interested parties that comments related to the proposed WQS amendments can be submitted to the DPNR-DEP within sixty (60) days after publication of the notice;

(2) Include the location and times in which the amended draft of the VI's Water Quality Standards Regulations, "Background and Basis" document and other relevant documents are available for public review; and

(3) Include other relevant information determined by the DPNR-DEP.

(b) Public Hearing: The DPNR-DEP shall hold public hearings, within a reasonable time, after the expiration of the sixty (60) day public comment period described above in Section A. The DPNR-DEP shall publish public notice of the public hearing at least thirty (30) days prior to the set public hearing date, in one (1) newspaper of wide circulation within each island, specifying the following:

(1) The day(s), the time(s) and the place(s) of the public hearing,
(2) The waters for which standards are sought to be amended, and
(3) Include any other pertinent information specified by the DPNR-DEP.

(c) Response to Comments: The DPNR-DEP shall review all public comments submitted during the public review process, including comments received during the public hearings. The DPNR-DEP shall complete and make available to the public the "Response to Comments" document prior to the adoption of proposed amendments into the VIWQS Regulations.

§ 186 - 16: Enforcement

The Commissioner shall enforce these Water Quality Standards Rules and Regulations in accordance with 12 VIC § 186 (d) and as amended.

17.2 Appendix B. Basis and Background for the 2018 Water Quality Standards Revisions for the U.S. Virgin Islands

Virgin Islands Department of Planning and Natural Resources

February 22, 2018

BASIS AND BACKGROUND FOR THE
2018 WATER QUALITY STANDARDS REVISION
FOR THE U.S. VIRGIN ISLANDS

1. BACKGROUND:

The water quality standards for the U.S. Virgin Islands (USVI) were first adopted in 1973, and revised in 1985, 2004, 2010 and 2015. Subsequent information concerning biological resources, the state of the coastal waters in the USVI, the identification of the existence of inland systems (both fresh and saline) in the USVI and the associated need to maintain and protect these inland systems, and new national guidelines and regulations make it necessary to review the current water quality standards. Additionally, Title 12, Chapter 7, Section 186(a) of the Virgin Islands Code requires the periodic review of the water quality standards.

The process of reviewing the current water quality standards and recommending the adoption of new and/or revised standards was performed by the Department of Planning and Natural Resources, Division of Environmental Protection and Office of the Commissioner.

2. LIST OF REVISIONS MADE TO THE 2018 WATER QUALITY STANDARDS REGULATIONS:

A. <u>SUBSTANTIVE REVISIONS:</u>

The 2018 Water Quality Standards Regulations (WQSR) include the following substantive revisions:

(1) Section 186-1 “Definitions”:

- (a) The following new definitions have been added: Coastal Estuary, Coral Habitat, Fresh Water, Impaired Water Body, Inland Estuary, Passageway, Potable Water Source, Saline Water, and Thermal Pollution.

(b) The following definitions have been revised: Class A Waters, Class B Waters, Class C Waters, Class I Waters.

(c) The following definition has been removed: Estuary.

(2) Section 186-4 "Classification of Water Designated Uses":

(a) Water quality standards for all Class Waters have been revised to include a new designated use for potable water source, where applicable, and to include corresponding criteria to protect this new use,

(b) Total coliform criterion adopted for waters used as potable water sources,

(c) Dissolved Oxygen criterion adopted for Class I waters, Subclass IF & IBS,

(d) Temperature criterion revised to incorporate "natural conditions" provision,

(e) More stringent clarity criterion adopted for areas where coral reefs are located,

(f) Total Nitrogen criterion adopted for all of marine waters,

(g) Turbidity criterion revised to provide more details on exemptions to the established criteria, specifically in support of existing practices and permitting programs already in place for the conditions that trigger these exemptions, such as for in-water projects and maintenance work.

(3) Section 186-5 "General Water Quality Criteria":

(a) Revision to the narrative criterion for Turbidity

(b) Revision to the narrative criterion for Downstream Protection

(c) Revisions of freshwater and saltwater Aquatic Life criteria for cadmium

(d) Revision of freshwater Aquatic Life criterion for selenium

(e) Revisions to Human Health water quality criteria, per EPA 304(a) criteria recommendations published in 2015.

(4) Section 186-8 "Anti-degradation Policy and Implementation Procedures" section was revised to reflect EPA's Final Rule Changes to 40 CFR §131 (published in 2015).

(5) Section 186-12 "Schedules of Compliance" section was revised to reflect EPA's Final Rule Changes to 40 CFR §131 (published in 2015).

(6) Section 186-14 “Water Quality Standards Variances” section was revised to reflect EPA’s Final Rule Changes to 40 CFR §131 (published in 2015).

B. CLARIFICATIONS:

(1) Section 186-1 “Definitions”: The following definitions have been clarified: Biological Integrity, Coral Reef, Coral Reef Ecosystem Areas, Mangrove Wetlands, Potable Water, Territorial Waters, WQS Variance, and Waters of the United States Virgin Islands. Please refer to the Attachment A for a complete list of revised terms and sources of definitions.

(2) Section 186-2 “Classification of Territorial Waters”: Identification of different types of Inland Waters (Class I) was clarified.

(3) Section 186-3 “Legal Limits”: Legend information has been added to clarify map delineations to prevent confusion between the different shaded areas, in response to public comments generated during 2015 WQSR triennial review process.

(4) Section 186-4 “Classification of Water Designated Uses”:

(a) Clarification of biocriteria standard to further condense the requirements,

(b) Clarification of pH criterion to better address “natural conditions” provision for all Water Classes. The pH criterion was clarified to eliminate the reference to the “normal range of pH must not be extended at any location by more than ±0.1 pH unit, except where due to natural forces” and replaced with “Natural Conditions of pH must not be extended at any location by more than +/- 0.1 pH unit.”

(5) Section 186-6: “Thermal Policy” – Originally misplaced sub-section (5) related to the Mixing Policy has been removed.

C. EDITORIAL REVISIONS:

(1) Correction of term Outstanding *Natural* Resource Waters to Outstanding *National* Resource Waters (or ONRW) throughout the WQSR document,

(2) Section 186-4 “Classification of Water Designated Uses”: Editorial revisions to the biocriteria standard language to eliminate redundancy,

(3) Section 186-5 “General Water Quality Criteria”: Revision to Tables, charts and associated footnotes relating to Numeric Water Quality Criteria for Toxic Pollutants in Section 186-5(b) to address formatting and layout.

3. JUSTIFICATION FOR SUBSTANTIVE REVISIONS TO THE WQSR

(1) Section 186-1: Definitions:

(d) The following new definitions have been added to provide the additional clarity to the WQSR document: Coastal Estuary, Coral Habitat, Fresh Water, Impaired Water Body, Inland Estuary, Passageway, Potable Water Source, Saline Water, and Thermal Pollution. Please refer to the Attachment A for a complete list of terms and sources of definitions. New definitions have been added to provide clarification/definitions for important terms and phrases used throughout the VI's Water Quality Standards Regulations document. Some of the definitions were added or revised to address public comments generated during the 2015 triennial WQSR review process.

(e) The following definitions have been revised to provide further clarity to the WQSR document: Class A Waters, Class B Waters, Class C Waters, Class I Waters.

(f) The definition for "Estuary" has been removed and replaced with new definitions for "Inland Estuary" and "Coastal Estuary".

(2) Section 186-4 "Classification of Water Designated Uses":

(a) Water quality standards for all Class Waters have been revised to include a new designated use for potable water source, where applicable, and to include corresponding criteria to protect this new use. The new Class of Inland Waters (Class I Waters) was added to the WQSR in 2015 to acknowledge the existence of various inland waterbodies throughout the Islands, to evaluate their existing uses and to adopt water quality criteria to protect their designated uses. DPNR determined that it was necessary to have criteria by which they can assess these waters. These inland waters represent waters that previously had no water quality standards by which to determine compliance. As a result, criteria for Total Coliform and Dissolved Oxygen were adopted for Class I Waters, as listed in (b) and (c) subsections below.

(b) Total coliform criterion adopted for waters used as potable water sources is based on USVI Drinking Water Standards (§1303-41), which adopted Federal standards (40CFR§141.21) for Total Coliform Rule (TCR)

(c) Dissolved Oxygen criterion adopted for Class I waters, Subclass IF & IBS is based on existing DO criteria established for Class A & B waters in section §186-4(b)(1) & (2)

(d) More stringent clarity criterion was adopted for areas where coral reefs are located. Criterion is based on recommended clarity criterion for areas of coral reef ecosystems from a 2001 study on Sunlight and water transparency by C.S. Yentsch et al.

(e) Total Nitrogen criterion was adopted for all marine waters. Total Nitrogen (TN) criterion is proposed as "shall not exceed 207 µg/L more than once in a three-year period in marine and coastal waters". Proposed criterion is based on the Best Professional Judgement and derived based on all locally available data. Review and statistical analysis of available TN data generated around the USVI suggest the proposed criterion to be protective of aquatic life designated use. Review of published literature supports criteria as protective for coral reefs. Please refer to Appendix C for detailed information related to criteria derivation.

(f) Turbidity criterion was revised to provide more details on exemptions to the established criteria, specifically in support of existing practices and permitting programs already in place for the conditions that trigger these exemptions, such as for in-water projects and maintenance work.

(3) Section 186-5 "General Water Quality Criteria":

a. Revision to the narrative criterion for Turbidity

b. Revision to the narrative criterion for Downstream Protection

Additional limitations on substances that can be found in all Territorial Waters have been added to Section 186-5(a)(1). Specifically, limitations on narrative criteria as it relates to Turbidity and Downstream Protection of Waters. Details on what substances produce objectionable turbidity was expounded upon, allowing for a more robust definition of these types of substances that are prohibited. Limitations on substances that would prevent waters downstream of discharge locations meeting designated uses were added to this section, further expanding limitations on the discharge of various pollutants. Language has been adopted from PRWQSR. This was done to provide a more robust definition of substances that are prohibited from being discharged in order to protect those waterbodies, and those downstream, from degrading or not meeting designated uses.

c. Revisions of freshwater and saltwater Aquatic Life criteria for cadmium. Criteria are being revised based on 2016 Environmental Protection Agency's (EPA) national recommended ambient water quality criteria for cadmium, which reflect the latest scientific information, and current EPA policies and methods. For details, please refer to USEPA's Report EPA-820-R-16-002, published in March of 2016 entitled: "Aquatic Life Ambient Water Quality Criteria - Cadmium – 2016." As stated in the Report, EPA recommends the:

- One-hour freshwater acute criterion maximum concentration not exceed 1.8 µg/L at a total hardness of 100 mg/L as $CaCO_3$.
- Four-day average freshwater chronic criterion concentration not exceed 0.72 µg/L at a total hardness of 100 mg/L as $CaCO_3$.
- One-hour estuarine/marine acute criterion maximum concentration not exceed 33 µg/L.
- Four-day average estuarine/marine chronic criterion concentration not exceed 7.9 µg/L.

The recommended frequency of exceedance for the above is no more than once every three years.

The 2016 criteria reflect data for 75 new species and 49 new genera. The 2016 freshwater acute criterion (1.8 micrograms per liter) for dissolved cadmium is slightly lower than the 2001 acute criterion (2.0 micrograms per liter). The 2016 freshwater chronic criterion (0.72 micrograms per liter) for dissolved cadmium is slightly higher (less stringent) compared to the 2001 criterion (0.25 micrograms per liter). These modest changes are primarily due to the inclusion of new toxicity studies. As in the 2001 criteria, the 2016 freshwater acute criterion was derived to be protective of aquatic species and was lowered further to protect the commercially and recreationally important rainbow trout. In addition, the duration of the 2016 acute criterion was changed to one-hour. Both changes are consistent with EPA's current aquatic life criteria guidelines.

The 2016 estuarine/marine acute criterion for dissolved cadmium (33 micrograms per liter) is lower (more stringent) than the 2001 acute criterion (40 micrograms per liter), which is primarily due to the addition of new toxicity studies for sensitive genera. The 2016 estuarine/marine chronic criterion (7.9 micrograms per liter) is also slightly more stringent than the 2001 chronic criterion (8.8 micrograms per liter), due the consideration of more species in the chronic criterion development.

d. Revision of freshwater Aquatic Life criterion for selenium. Criteria are being revised based on 2016 Environmental Protection Agency's (EPA) national recommended ambient water quality criteria for selenium, which reflect the latest scientific information, and current EPA policies and methods.

For details, please refer to USEPA’s Report EPA 822-R-16-006, published in June of 2016 entitled: “Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater 2016.”

As stated in the Report, the 2016 criterion document is the final update of EPA's 1999 recommended national chronic aquatic life criterion for selenium, developed per Clean Water Act section 304(a). The 2016 criterion reflects the latest scientific knowledge, which indicates that selenium toxicity to aquatic life is primarily based on organisms consuming selenium contaminated food rather than by being exposed only to selenium dissolved in water. The final criterion is expressed both in terms of fish tissue concentration (egg/ovary, whole body, muscle) and water concentration (lentic, lotic).

The 2016 selenium criterion document recommends that states and authorized tribes adopt a multi-media criterion into their water quality standards. The criterion has four elements and EPA recommends that states includes all four elements in their standards. Two elements are based on the concentration of selenium in fish tissue (eggs and ovaries, and whole-body or muscle) and two elements are based on the concentration of selenium in the water-column (two 30-day chronic values and an intermittent value).

e. Table and associated footnotes related to Numeric Water Quality Criteria for Toxic Pollutants to protect aquatic life in Section 186-5(b) have been revised to reflect EPA's current nationally recommended Clean Water Act section 304(a) water quality criteria for cadmium and selenium.

f. Revisions to Human Health water quality criteria, per EPA 304(a) criteria recommendations published in 2015. Table and associated footnotes related to Numeric Water Quality Criteria for Toxic Pollutants to protect human health in Section 186-5(b) have been revised to reflect EPA's current nationally recommended Clean Water Act section 304(a) water quality criteria. In this 2015 update, EPA revised 94 of the existing human health criteria to reflect the latest scientific information, including updated exposure factors (body weight, drinking water consumption rates, fish consumption rate), bioaccumulation factors, and toxicity factors (reference dose, cancer slope factor). The criteria have also been updated to follow the current EPA methodology for deriving human health criteria (USEPA 2000). EPA also developed chemical-specific science documents for each of the 94 chemical pollutants. The science documents detail the latest scientific information supporting the updated final human health criteria, particularly the updated toxicity and exposure input values.

- Body Weight - the default body weight for human health criteria was updated to 80 kilograms based on National Health and Nutrition Examination Survey (NHANES) data from 1999 to 2006 (USEPA 2011). This represents the mean body weight for adults ages 21 and older. EPA's previously recommended default body weight was 70 kilograms, which was based on the mean body weight of adults from the NHANES III database (1988-1994).

- Drinking Water - the default drinking water consumption rate was updated to 2.4 liters per day based on NHANES data from 2003 to 2006 (USEPA 2011). This represents the per capita estimate of community water ingestion at the 90th percentile for adults ages 21 and older. EPA previously recommended a default drinking water consumption rate of 2 liters per day, which represented the per capita community water ingestion rate at the 86th percentile for adults surveyed in the US Department of Agriculture's 1994-1996 Continuing Survey of Food Intake by Individuals (CSFII) analysis and the 88th percentile of adults in the National Cancer Institute study of the 1977-1978 Nationwide Food Consumption Survey.

- Fish Consumption - the default fish consumption rate was updated to 22 grams per day. This rate represents the 90th percentile consumption rate of fish and shellfish from inland and nearshore waters for the U.S. adult population 21 years of age and older, based on NHANES data from 2003 to 2010 (USEPA 2014). EPA's previously recommended rate of 17.5 grams per day was based on the 90th percentile consumption rate of fish and shellfish from inland and nearshore waters for the U.S. adult population and was derived from 1994-1996 CSFII data.

For additional details, please refer to EPA's Report EPA 820-F-15-001, entitled: "Health Ambient Water Quality Criteria: 2015 Update", published in June 2015. Please refer to Appendix B for a complete list of revised human health criteria and corresponding background documents published by EPA.

(6) Section 186-8: "Anti-degradation Policy and Implementation Procedures": section has been revised to reflect EPA's Final Rule Changes to 40 CFR §131 (published in 2015). For details, please refer to EPA's Report EPA 820-F-15-00, entitled: "Water Quality Standards Regulatory Revisions (Final Rule)" published in July of 2015. In general, the Final Rule established stronger antidegradation requirements by creating a more structured process for identifying high quality waters and specifying the type of analysis that is

required before a state or authorized tribe allows degradation of high water quality, resulting in enhanced protection of high quality waters and promoting public transparency.

(7) Section 186-12: "Schedules of Compliance": section has been revised to reflect EPA's Final Rule Changes to 40 CFR §131 (published in 2015). For details, please refer to EPA's Report EPA 820-F-15-00, entitled: "Water Quality Standards Regulatory Revisions (Final Rule)" published in July of 2015. In general, this Final Rule clarified that a state or authorized tribe must adopt, and EPA must approve, a provision authorizing the use of permit compliance schedules prior to legally using schedules of compliance for WQBELs in NPDES permits.

(8) Section 186-14: "Water Quality Standards Variances": section has been revised to reflect July 2015 Final Rule Changes to 40 CFR §131). For details, please refer to EPA's Report EPA 820-F-15-00, entitled: "Water Quality Standards Regulatory Revisions (Final Rule)" published in July of 2015. In general, this Final Rule outlined a comprehensive regulatory structure for WQS variances, promoting the appropriate use of this Clean Water Act tool and providing regulatory certainty to states, authorized tribes, the regulated community, stakeholders, and the public in making progress toward attaining designated uses.

Appendix A

List of definitions and their sources

The following terms have been listed in "Definitions" section 186-1:

Abiotic, Acute, Aesthetic qualifications, Aquatic Nuisance Species, Assimilative Capacity, Best Management Practices, Bioaccumulation Factor, Biological Criteria or Biocriteria, Biological Integrity, Biotic, Code of Federal Regulations (CFR), Chronic, Class A Waters, Class B Waters, Class C Waters, Class I Waters, Commissioner, Coastal Estuary, Community diversity, Consistent sampling, Coral Habitat, Coral reef, Coral Reef Ecosystems, Coral Reef Ecosystems Areas, Criteria, Criteria Continuous Concentration (CCC), Criteria Maximum Concentration (CMC), DPNR or Department, Designated Use, Desirable Species, Dissolved Oxygen, Effluent, Embayment, Enterococci Bacteria, Environmental Protection Agency (EPA), Endangered Species Act (ESA), Existing uses, Exotic Species, Federal Clean Water Act or Federal Water Pollution Control Act (CWA), Freshwater, Freshwater ponds, Groundwater, Gut or Stream, Impaired Waterbody, Indicator Species or Indicator Community, Inland Estuary, Inlet, Mangrove Wetlands, Marine and Coastal Waters, Metric, Mixing Zone, NPDES, Natural Condition, Natural forces, Non-point Source, Nuisances, Nutrients, Other Discharges or Wastes, Outstanding Natural Resource Waters (ONRW), Parameter of Concern, Passageway, Permit, Permittee, Person, pH, Point Source, Pollution, Pollutant or Waste, Potable Water, Potable Water Source, Primary Contact Recreation, Reference Conditions, Reliable Measure, Saline Water, Salinity, Salt Flats, Salt Pond, Secchi Disc, Site, Stream or Gut, Subclass IBS Waters, Subclass IF Waters, Subclass IG Waters, Sufficient Quality, Surface Water, Swamp, Temperature, Territorial Waters, Territorial Pollutant Discharge Elimination System (TPDES), Thermal Discharge, Thermal Pollution, Total Maximum Daily Load (TMDL), Toxic Pollutant, Turbidity, Wastewaters, Water Quality Standards (WQS), Water Quality Criteria, WQS Variance, Waters of the US Virgin Islands, Well, and Wetlands.

The following sources have been used to define terms listed in "Definitions" section:

- 12 V.I.C
- Biology-Online.Org Online Dictionary 2011
- Code of Federal Regulations - 40 CFR
- Commonwealth of the Northern Mariana Islands. Commonwealth of the Northern Mariana Islands Water Quality Standards. 2004.
- Puerto Rico Water Quality Standards Regulation. Commonwealth of Puerto Rico, Office of the Governor, Environmental Quality Board, March 2010.
- US Environmental Protection Agency. Water Quality Standards Handbook. EPA-823-B-12-002, March 2012.
- US Environmental Protection Agency. Glossary of Water Resource Terms, Chicago, IL, April 1970

- US Environmental Protection Agency. Environmental Compliance Workshop: Clean Water. April 16, 2009
- US Environmental Protection Agency. Land by the Lakes: Nearshore Terrestrial Ecosystems Glossary. http://www.epa.gov. Date Accessed: April 04, 2013.
- US Environmental Protection Agency. Terms of Environment: Glossary, Abbreviations and Acronyms. EPA 175-B-94-015. April 1994.
- US Environmental Protection Agency. Drinking Water Glossary: A Dictionary of Technical and Legal Terms Related to Drinking Water. http://water.epa.gov/drink/. Date Accessed: April 04, 2013.
- Virgin Islands Code Title 12: Conservation. Government of the US Virgin Islands, Department of Planning and Natural Resources.
- Virgin Islands Territorial Pollutant Discharge Elimination System Rules and Regulations Title 12, Chapter 7, Subchapter 184. Government of the US Virgin Islands, Department of Planning and Natural Resources, Division of Environmental Protection, June 2007.
- Virgin Islands Water Quality Standards Title 12, Chapter 7, Subchapter 186. Government of the US Virgin Islands, Department of Planning and Natural Resources, Division of Environmental Protection, June 2010.
- Water Encyclopedia (online). April 04, 2013.

Other References:

Yentsch, C.S., Yentsch, C.M. et al. 2001. Sunlight and water transparency : cornerstones in coral research. *Journal of Experimental Marine Biology and Ecology.*, 268, pp. 171-183.

Appendix B

List of revised human health criteria and corresponding EPA Reports published in June of 2015

Parameter:	EPA REPORT #:	"Update of Human Health Ambient Water Quality Criteria:
Acenaphthene	820-R-15-002	Acenaphthene 83-32-9"
Acrolein	820-R-15-003	Acrolein 107-02-8"
Acrylonitrile	820-R-15-004	Acrylonitrile 107-13-1
Aldrin	820-R-15-005	Aldrin 309-00-2
alpha-BHC	820-R-15-006	alpha-Hexachlorocyclohexane (HCH) 319-84-6
alpha-Endosulfan	820-R-15-007	alpha-Endosulfan 959-98-8
Anthracene	820-R-15-008	Anthracene 120-12-7
Benzene	820-R-15-009	Benzene 71-43-2
Benzidine	820-R-15-010	Benzidine 92-87-5
Benzo(a) Anthracene	820-R-15-011	Benzo(a)anthracene 56-55-3
Benzo(a) Pyrene	820-R-15-012	Benzo(a)pyrene 50-32-8
Benzo(b) Fluoranthene	820-R-15-013	Benzo(b)fluoranthene 205-99-2
Benzo(k) Fluoranthene	820-R-15-014	Benzo(k)fluoranthene 207-08-9
beta-BHC	820-R-15-015	beta-Hexachlorocyclohexane (HCH) 319-85-7
beta-Endosulfan	820-R-15-016	beta-Endosulfan 33213-65-9
Bis(2-Chloroethyl) Ether	820-R-15-018	Bis(2-chloroethyl) Ether 111-44-4
Bis(2-chloro-1-methylethyl) Ether	820-R-15-019	Bis(2-chloro-1-methylethyl) Ether 108-60-1
Bis(2-Ethylhexyl) Phthalate	820-R-15-020	Bis(2-ethylhexyl) Phthalate 117-81-7
Bromoform	820-R-15-021	Bromoform 75-25-2
Butylbenzyl Phthalate	820-R-15-022	Butylbenzyl Phthalate 85-68-7
Carbon Tetrachloride	820-R-15-023	Carbon Tetrachloride 56-23-5
Chlordane	820-R-15-024	Chlordane 57-74-9
Chlorobenzene	820-R-15-025	Chlorobenzene 108-90-7
Chlorodibromomethane	820-R-15-026	Chlorodibromomethane 124-48-1
Chloroform	820-R-15-027	Chloroform 67-66-3
Chlorophenoxy, Herbicide (2,4-D)	820-R-15-028	Chlorophenoxy Herbicide (2,4-D) 94-75-7
Chlorophenoxy Herbicide (2,4,5-TP) [Silvex]	820-R-15-029	Chlorophenoxy Herbicide (2,4,5-TP) 93-72-1
Chrysene	820-R-15-030	Chrysene 218-01-9
Cyanide	820-R-15-031	Cyanide 57-12-5
Dibenzo(a,h)Anthracene	820-R-15-032	Dibenzo(a,h)anthracene 53-70-3
Dichlorobromomethane	820-R-15-033	Dichlorobromomethane 75-27-4
Dieldrin	820-R-15-034	Dieldrin 60-57-1

Diethyl Phthalate	820-R-15-035	Diethyl Phthalate 84-66-2
Dimethyl Phthalate	820-R-15-036	Dimethyl Phthalate 131-11-3
Di-n-Butyl Phthalate	820-R-15-037	Di-n-butyl Phthalate 84-74-2
Dinitrophenols	820-R-15-038	Dinitrophenols 25550-58-7
Endosulfan Sulfate	820-R-15-039	Endosulfan Sulfate 1031-07-8
Endrin	820-R-15-040	Endrin 72-20-8
Endrin Aldehyde	820-R-15-041	Endrin Aldehyde 7421-93-4
Ethylbenzene	820-R-15-042	Ethylbenzene 100-41-4
Fluoranthene	820-R-15-043	Fluoranthene 206-44-0
Fluorene	820-R-15-044	Fluorene 86-73-7
gamma-BHC (Lindane)	820-R-15-045	Gamma-HCH 58-89-9
Heptachlor	820-R-15-046	Heptachlor 76-44-8
Heptachlor Epoxide	820-R-15-047	Heptachlor Epoxide 1024-57-3
Hexachlorobenzene	820-R-15-048	Hexachlorobenzene 118-74-1
Hexachlorobutadiene	820-R-15-049	Hexachlorobutadiene 87-68-3
Hexachlorocyclo-hexane-Technical	820-R-15-050	Hexachlorocyclohexane (HCH)-Technical 608-73-1
Hexachlorocyclopentadiene	820-R-15-051	Hexachlorocyclopentadiene 77-47-4
Hexachloroethane	820-R-15-052	Hexachloroethane 67-72-1
Ideno(1,2,3-cd)Pyrene	820-R-15-053	Indeno(1,2,3-cd)pyrene 193-39-5
Isophorone	820-R-15-054	Isophorone 78-59-1
Methoxychlor	820-R-15-055	Methoxychlor 72-43-5
Methyl Bromide	820-R-15-056	Methyl Bromide 74-83-9
Methylene Chloride	820-R-15-057	Methylene Chloride 75-09-2
Nitrobenzene	820-R-15-058	Nitrobenzene 98-95-3
Pentachlorobenzene	820-R-15-059	Pentachlorobenzene 608-93-5
Pentachlorophenol	820-R-15-060	Pentachlorophenol 87-86-5
Phenol	820-R-15-061	Phenol 108-95-2
Pyrene	820-R-15-062	Pyrene 129-00-0
Tetrachlorobenzene,1,2,4,5-		
Tetrachloroethylene	820-R-15-063	Tetrachloroethylene (Perchloroethylene) 127-18-4
Toluene	820-R-15-064	Toluene 108-88-3
Toxaphene	820-R-15-065	Toxaphene 8001-35-2
Trichloroethylene	820-R-15-066	Trichloroethylene (TCE) 79-01-6
Trichlorophenol,2,4,5-		
Vinyl Chloride	820-R-15-067	Vinyl Chloride 75-01-4
1,1,1-Trichloroethane	820-R-15-068	1,1,1-Trichloroethane 71-55-6
1,1,2,2-Tetrachloroethane	820-R-15-069	1,1,2,2-Tetrachloroethane 79-34-5
1,1,2-Trichloroethane	820-R-15-070	1,1,2-Trichloroethane 79-00-5
1,1-Dichloroethylene	820-R-15-071	1,1-Dichloroethylene 75-35-4
1,2,4-Trichlorobenzene	820-R-15-072	1,2,4-Trichlorobenzene 120-82-1
1,2-Dichlorobenzene	820-R-15-074	1,2-Dichlorobenzene 95-50-1
1,2-Dichloroethane	820-R-15-075	1,2-Dichloroethane 107-06-2

1,2-Dichloropropane	820-R-15-076	1,2-Dichloropropane 78-87-5
1,2-Diphenylhydrazine	820-R-15-077	1,2-Diphenylhydrazine 122-66-7
1,2-Trans-Dichloroethylene	820-R-15-078	trans-1,2-Dichloroethylene (DCE) 156-60-5
1,3-Dichlorobenzene	820-R-15-079	1,3-Dichlorobenzene 541-73-1
1,3-Dichloropropene	820-R-15-080	1,3-Dichloropropene 542-75-6
1,4-Dichlorobenzene	820-R-15-081	1,4-Dichlorobenzene 106-46-7
2,4,6-Trichlorophenol	820-R-15-083	2,4,6-Trichlorophenol 88-06-2
2,4-Dichlorophenol	820-R-15-084	2,4-Dichlorophenol 120-83-2
2,4-Dimethylphenol	820-R-15-085	2,4-Dimethylphenol 105-67-9
2,4-Dinitrophenol	820-R-15-086	2,4-Dinitrophenol 51-28-5
2,4-Dinitrotoluene	820-R-15-087	2,4-Dinitrotoluene 121-14-2
2-Chloronaphthalene	820-R-15-088	2-Chloronaphthalene 91-58-7
2-Chlorophenol	820-R-15-089	2-Chlorophenol 95-57-8
2-Methyl-4,6-Dinitrophenol	820-R-15-090	2-Methyl-4,6-dinitrophenol 534-52-1
3,3'-Dichlorobenzidine	820-R-15-091	3,3′-Dichlorobenzidine 91-94-1
4,4'-DDD	820-R-15-093	p,p′-Dichlorodiphenyldichloroethane (DDD) 72-54-8
4,4'-DDE	820-R-15-094	p,p′-Dichlorodiphenyldichloroethylene (DDE) 72-55-9
4,4'-DDT	820-R-15-095	p,p′-Dichlorodiphenyltrichloroethane (DDT) 50-29-3

Appendix C

Basis and Background Document
Derivation of Total Nitrogen Criterion
to protect marine waters around the US Virgin Islands

Coral Reefs around the US Virgin Islands

The US Virgin Islands (USVI) in the northeastern Caribbean, consists of St. Croix (207 km2), St. Thomas (83 km2), St. John (52 km2) and numerous smaller islands (Rogers et al 2008). Fringing, bank-barrier, patch, spur and groove reefs, algal ridges, and a submarine canyon are all present in the USVI. Coral communities, as opposed to true coral reefs, are found growing on boulders and mangrove prop roots in shallow water around most of the island shorelines (St. Croix, St. Thomas, and St. John have 113, 85 and 80 km of shoreline, respectively). Some reefs have grown off of rocky points and across the mouths of bays, creating salt ponds, for example, in Newfound Bay, St. John. The most developed reefs in general are found off the eastern, windward ends of the islands. Algal ridges occur off the eastern end of St. Croix. The steep, lower forereefs of the fringing reefs around the islands tend to have higher coral cover than other habitats at depths less than 20 m around the islands, although high coral cover is found on deeper offshore reefs such as those that are part of the Mid-Shelf Reef complex and the Red Hind Bank which lie south of St. Thomas and St. John. Well-developed reefs occur at depths of 33 to 47 m south of St. Thomas. As reported by Rogers e al (2008), some reefs are close to seagrass beds and mangroves, e.g., Salt River Submarine Canyon and Teague Bay Reef (St. Croix), reefs in Benner Bay and around Cas Cay (St. Thomas), and reefs in Great Lameshur Bay (St. John), although mangroves are not extensive in the USVI.

Many Marine Protected Areas including some marine (“no-take”) reserves, are found in the USVI. Buck Island Reef National Monument was established in 1961 and consisted of 356 ha. Virgin Islands National Park was established in 1956, with the marine portions (2,286 ha) added in 1962. In 1999 the Marine Conservation District (also known as the Red Hind Bank) was established to protect 41 km2 of deep reef habitats south of St. Thomas.

Proposed TN Criterion

The USVI Department of Planning and Natural Resources (DPNR) is proposing to adopt the TN criterion of not to exceed 207 µg/L expressed as an annual geometric mean that shall not be exceeded more than once in a three-year period in marine and coastal waters. Because the coral reef ecosystems are very sensitive to environmental impacts and require very high water quality, the DPNR believes that criteria protective of corals will also be as protective (if not more) of other marine life.

Why is the adoption of TN criterion important for coral reef protection around the USVI?

Tropical reef-building corals commonly flourish in nutrient-poor oligotrophic environments (Bythell 1990, Radecker et al 2015, Zhou et al 2017). Coral reef ecosystems appear to be able to outcompete other ecosystems when the surrounding water environment is poor in nutrients, having been described as oases in the oceanic desert (Salin, 1983; Sorokin, 1973, p. 17; Odum and Odum, 1955). Their ability to do this is not fully understood but it is known that nitrogen-fixing organisms are often prolific in coral reef environs and that various symbiotic relationships occur within the biological community structure which result in very efficient recycling of nutrients. This recycling of nutrients provides longer residence times for the nutrients, or in other words, a storage capacity or capacitance for nutrients. This inbuilt capacitance means that the coral reef ecosystems are able to flourish with only periodic inputs of nutrients, e.g. from upwellings or river run-off (Bell, 1992). For corals, the close association between the coral animal host and its endosymbiotic dinoflagellate algae (zooxanthellae) enables an effective use and retention of nutrients and photosynthates (photosynthetically fixed carbon). Given that symbiont production in corals is highly dependent on nitrogen availability, nitrogen cycling in the coral is an important component for the acquisition and retention of nitrogen to sustain primary productivity (i.e., photosynthesis). The availability of nitrogen sources in coral reefs, however, underlies strong seasonal and diel variations, and can be affected by anthropogenic activities. As it will be discussed in more details below, Wiedenmann et al (2013) reported that a shift away from nitrogen limitation (by excess nitrogen provision) can ultimately result in phosphate starvation, which can increase the susceptibility of corals to heat and light induced loss of their algal symbionts, resulting in coral bleaching. As a result, low internal nutrient availability, specifically of nitrogen, appears to be crucial for maintaining high primary production, while simultaneously controlling algae growth. In addition, an imbalanced nutrient availability (e.g., elevated inorganic nitrogen concentrations in combination with phosphate depletion), rather than enrichment of both nitrogen and phosphate, can destabilize the coral–algae symbiosis.

In addition, it is important to point out that the USVI already adopted TP criterion of 0.05 mg/L into its Water Quality Standard Regulations to protect all of the USVI's marine waters. The adoption of dual nutrient criteria is also consistent with EPA's recommendations most recently highlighted in EPA's Fact Sheet "Preventing Eutrophication: Scientific Support for Dual Nutrient Criteria (February 2015)."

Basis for derivation of proposed TN criteria

As recommended by USEPA, DPNR developed a Nutrients Standards Plan, which outlines the steps for adopting numeric nutrient criteria in USVI WQS. Since 2003, USVI DPNR has completed multiple projects in relation to nutrients and coral health. DPNR's overall goal was to use this local data to possibly adopt TN criteria for all estuarine/marine waters and revise their existing TP criterion, if needed. The USVI DPNR sought support from the USEPA Nutrient Scientific Technical Exchange

Partnership Support (NSTEPS) program with help developing nutrient criteria for their estuarine/marine waters.

In order to include other Federal and local Agencies in the process of TN /TP derivation, DPNR established Water Quality Work Group consisting of the representatives of the following entities: USVIDPNR, USEPA, NPS, FWS, UVI, NOAA, NRCS and USDA. Work Group members, as local coral reef experts, participated in monthly calls during which DPNR introduced and asked for comments and suggestions on the proposed TN criterion and discussed its potential implementation.

TN derivation process – 3 lines of evidence:

(1) Statistical analysis of Local data – reference approach used to derive proposed criterion

(2) Proposed criteria compared to criteria adopted by other states for similar ecosystems (FL, HI)

(3) Proposed criteria compared to the protective values reported in the peer-reviewed Literature

Data used in the process of TN derivation was collected through three main projects:

(1) the RARE (Regional Applied Research Efforts) project which EPA's Office of Research and Development's (ORD) conducted with DPNR's assistance between 2007 and 2011. During this project a total of 60 sites around both districts were selected, and coral assessment was done by ORD with DPNR. Concurrently, water quality monitoring was also done at these RARE sites by DPNR.

(2) the Supplemental 106 Phase 1 project, which had a more specific goal, looking at how water quality in 5 nearshore water body types varied spatially as well as over time. Data collection followed similarly to RARE, using (ORD) Stony Coral Rapid Bioassessment Protocol (SCRB Protocol) for coral assessment with concurrent water quality samples and readings.

(3) the Supplemental 106 Phase 2 project which focused on a baseline of the existing water quality and coral condition with intent to observe the changes to both from run-off events. There was focus on heavily impacted water bodies in the USVI. Those areas were the Charlotte Amalie & Crown Bay areas in St. Thomas and the Christiansted area in St. Croix. Like the previous study, water quality monitoring and coral condition assessment was done concurrently, using the same methods. This was done over a shorter period covering 26 monitoring stations from around Sept. 2013 to July of the following year.

Data generated during the above described studies is being used to derive TN criterion and reevaluate already existing TP criterion to protect marine waters around the USVI, with a specific focus on protection of coral reefs and their critical habitats. Because the coral reef ecosystems are very sensitive to environmental impacts and require very high water quality, the DPNR believes that criteria protective of corals will also be as protective (if not more) of other marine life.

Data generated during the above described projects was statistically analyzed through the NSTEPS (Nutrient Scientific Technical Exchange Partnership and Support) Program in support of the numeric nutrient criteria development process for US Virgin Islands estuarine/marine waters. This effort included compiling existing data, developing a conceptual model, summarizing current nutrient criteria assessment methods, supporting the US Virgin Islands (DPNR) Division of Environmental Protection (DEP) in the development of sound nutrient database, and conducting analyses of nutrient data.

The data were also analyzed for significant correlations among stressor and response variables provided. Oliver et al. (2011) successfully related the LDI (landscape development intensity) index to marine ecosystems, specifically coral reefs off St. Croix in the US Virgin Islands. The LDI showed negative correlations with stony corals colony density, taxa richness, colony size, and total coral cover. Percent impervious surface in the watershed was also negatively correlated with total coral cover. Correlations included relationships between LDI and stressors (nutrients), LDI and endpoints (Chl, DO, biotic metrics), and between stressors and endpoints

Distributional statistics for water quality parameters were calculated (please refer to the Technical Report included in Appendix D below for details). This analysis was performed for the entire dataset, the subset of data within 500 m of shore, and the subset of data within 500 m of shore and closest to a catchment with LDI under 2. This last category was presumed to represent least impacted coastal waters of the USVI. The 90th percentiled concentrations of TP and TN in these waters were 14.8 uM TN and 1.8 uM TP (207 ug/L and 56 ug/L respectively). The USVI existing TP standard is very close to this value from least disturbed waters, the comparable TN value, therefore, may also represent an appropriate line of evidence.

(1) <u>Statistical analysis of Local data – reference approach used to derive proposed criterion</u>

How does the proposed TN criterion compare to historical water quality data?

To further evaluate how proposed TN criterion will reflect the existing ambient conditions, the USVI DPNR reviewed all available TN data taken during the water quality monitoring events from 2004 to Present from sample points within the USVI Territory. Of 495 data point points, 79 (approximately 16%) were found to exceed the proposed standard of 207 ug/L, spread randomly across 31 different sites that cover a wide range of areas and waterbody types, in a total of 17 different Assessment Units (AUs) throughout the Territory. However, no single AU had data covering 3 years, so analysis of whether any of these AUs would be impaired is impossible, so the data is just noted as showing distribution of sample results individually. However, it is expected that in future assessments of compliance with the TN criterion, there will be sufficient and consistent data to draw from.

(2) Proposed criteria compared to criteria adopted by other states with coral reef ecosystems ecosystems (FL, HI)

How does the proposed TN criterion compare to criteria adopted by other states?

FLORIDA:
Criteria derived based on reference condition approach. Biological endpoints (obtained from unimpaired sites): DO (conctr or % saturation). Chlorophyll a (20 ug/l NTE > 10% of time protective of algae) and seagrass indicators (20 % of surface light at the bottom of the water column, colonization depth, water clarity, coverage and extend).

		TP (ug/L)	TN (ug/L)	Chll a (ug/L)
Florida:				
Annual average values not to be exceeded more than once every three years.	Florida Keys	7-11	170-250	0.2-0.7

FL Keys segment is considered to be most relevant to the USVI. FL adopted nutrient (TP and TN) and chlorophyll criteria for 7 individual estuaries within Florida Keys, expressed as an annual geometric mean not to be exceeded more than once in three-year period (geometric means are used to attenuate the short-term variability reflected by skewed data to provide a more reliable long-term estimate of the nutrient status).

- **TN: range from 170 to 250 ug/L**

Approach - Most estuary standards are based on distributional statistics applied to data for reference conditions. However, TMDLs were submitted as site-specific standards for those segments that were currently or had been previously identified on the state's 303(d) impaired waters list as impaired for nutrients or DO. The reference condition approach was applied to some segments which had been previously been identified as impaired by nutrients or DO, but had since attained designated uses. In these cases, data from years when the segment was impaired or from specific impaired portions of the segment were excluded from the calculation. The FDEP determined reference conditions using biological endpoint data from currently unimpaired segments or data from the period of time when a segment was unimpaired.

Endpoints - Florida's NNC are intended to prevent eutrophication and promote healthy conditions. The indicators representing healthy conditions include seagrass metrics, Chl-a concentrations, and DO regime.

- Seagrass metrics are used as indicators of estuary health worldwide, with declines in spatial extent, density and biomass integrating the influences of multiple stressors
- Chl-a is a useful indicator of plankton growth which integrates nutrient loading in an aquatic system. Overstimulation of photosynthesis and increased algal growth by excess nutrients elevates Chl-a levels above natural conditions
- The DO regime of a system is also an indicator of eutrophic conditions, with photosynthesizing algal biomass elevating DO to supersaturated levels in daylight and the oxygen consumption processes of respiration and algal decomposition and decay depleting DO at night time.

Endpoints used in this determination included DO concentration and/or percent saturation, chlorophyll-a concentration, and the seagrass indicators: colonization depth, water clarity, coverage, and extent. Specifically, achievement of 20 percent of the surface light at the bottom of the water column is considered to be protective of seagrass communities and a chlorophyll-a concentration of 20 µg/L, not to be exceeded more than 10 percent of the time, is considered to be indicative of balanced algal populations. Taken with spatial attributes of seagrass indicators and Florida's DO criteria previously approved by EPA and applied by FDEP, EPA concluded that these endpoints are expected to indicate the health of the system as a whole and, at reference levels, represent conditions that protect aquatic life and recreation uses.

The NNC for TP and TN are nutrient levels at which Florida expects these indicators will meet thresholds reflecting the reference conditions expected to protect aquatic life.

Methodology - Florida's NNC standards for Florida Keys were arrived at using a "Maintain Healthy Conditions" methodology. The objective of this methodology is to maintain current nutrient regimes considered to be biologically healthy from the standpoint of nutrient enrichment. Monitoring data representing "healthy conditions" were used to calculate standards as annual geometric means not to be exceeded more than once in three years. The process specifically shields against identifying a healthy system as impaired by selecting thresholds at the 90th percentile. This approach may appear counter-intuitive because the goal of an environmental indicator is often to detect and mitigate environmental problems, and the failure to detect a problem may result in a failure to recover an impaired system. However, the FDEP strategy for developing protective standards was to avoid identification of a problem, for example unhealthy conditions, when none actually exists, to avoid the loss of information about healthy conditions for that particular system.

Coastal Waters

Due to lack of sufficient TP and TN data, only criteria for chlorophyll have been adopted for coastal waters of FL, based on satellite remote sensing data for chlorophyll-a collected between 1998 and 2009, which were validated using available field observations. Coastal waters adjacent to impaired estuaries and data obtained during harmful algal bloom events were excluded from the dataset prior to calculating standards.

Standards for each segment are the 90th percentiles of the annual geometric means of chlorophyll-a levels in the reference dataset. Sufficient field monitoring data for TP and TN were not available, so only chlorophyll-a standards were established for coastal waters.

In summary, the FL Keys criteria falls in a similar range as the USVI's proposed TN criterion, but to provide appropriate context, it should be emphasized that there are different endpoints and methodology in each criterion derivation. FL criteria is based on analysis of seagrass metrics, Chl-a concentrations, and DO regime in relatively unimpaired waterbodies, while USVI's approach involved LDI and endpoints that include Chl-a, DO, and biotic metrics.

HAWAII:

Embayments:

(ug/L)	Geom mean		NTE more than 10% of the time		NTE more than 2% of the time	
	wet	dry	wet	dry	wet	dry
TN	200	150	350	250	500	350
TP	25	20	50	40	75	60

NTE – not to exceed

- Wet – criteria apply when the average freshwater inflow from land equals or exceeds 1% of embayment volume per day
- Dry - criteria apply when the average freshwater inflow from land is less than 1% of embayment volume per day

Open coastal waters:

(ug/L)	Geom mean		NTE more than 10% of the time		NTE more than 2% of the time	
	wet	dry	wet	dry	wet	dry
TN	150	110	250	180	350	250
TP	20	16	40	30	60	45

NTE – not to exceed

- Wet – criteria apply when the open coastal waters receive more than 3 million gallons per day of freshwater discharge per shoreline mile
- Dry - criteria apply when the open coastal waters receive less than 3 million gallons per day of freshwater discharge per shoreline mile

Although the overall range of TN criteria adopted by HI is consistent with the TN criterion being proposed by the USVI, these values are not directly comparable due to different water conditions and coral species present at both locations. As a result, HI criteria are provided above just for the informational purposes rather than comparability. FL criteria are much more relative for our analysis.

Nutrient Criteria Work Group

In order to evaluate the proposed TN criterion, the VIDPNR set up the WG panel consisting of various agencies including USEPA, NOAA, FWS, NPS, USGS, and UVI. Over a period of few months, panel evaluated available data and provided recommendations to the VIDPNR. Several workgroup calls were hosted to discuss the proposed nutrient criteria development, as well as changes to the VIWQSR in general.

Discussions via phone conferences were conducted over a 6-month period, consisting of 5 separate calls that included representatives from USVI-DPNR, USEPA, NPS, FWS, UVI, NOAA, NRCS and USDA.

Topics discussed relating to TN include an interest in focus on runoff through watershed ghuts and inland areas. The Workgroup had interest in seeing a focus on areas with potential for large influxes of nutrients as well as other pollutants. A desire to see the differences in water quality in various water body types, and classes was also brought up.

Suggestions and data on what other states or territories have established and how they were established was put forth and discussed, as well as concern that the proposed limits would not be appropriate based on what other studies have shown.

Further calls presented concerns regarding statistical analysis of data, model use and insufficient datasets.

Most of these concerns were discussed in full, and were addressed through this document's content and its justification for the TN criterion derivation.

(3) Proposed criteria compared to the Literature

Results of literature review evaluating the impacts of proposed TN criterion on coral protection

Literature review identified approximately 150 publications focusing on the impacts of nutrients on coral condition. Only a small subset of this literature discussed specifically corals around the US Virgin Islands. Due to the fact that the similar water conditions and coral ecosystems are found around the Puerto Rico and Florida (including the same coral species listed as threatened under the ESA), this review also considered publications related to these two additional locations. In addition, in some cases information/data generated at other locations have been considered, if reviewers determined that information/data may be helpful or relevant to locations around the USVI.
Majority of the literature reviewed included very general observations stating that eutrophication resulting from the anthropogenic activities has a potential to seriously degrade or modify coral reef ecosystems and that nutrients have a negative effect on coral survival, growth, and/or reproduction. Only a subset of the studies reviewed had quantified the amount of nutrients the corals were exposed to. The review was focused on publications describing specific coral responses to specific nutrient concentrations. Unfortunately, all of the identified publications focused on concentrations of specific

nitrogen components as opposed to total nitrogen concentrations. The comparison of the USVI's proposed TN criterion of 207 ug/L to values published in the literature is thus complicated by the different ways/units nitrogen data are expressed in the published literature (e.g., nitrate, ammonia, or the aggregate metric for dissolved inorganic nitrogen (DIN= nitrate+ nitrite+ ammonia). Since these other measures of nitrogen only represent a fraction of TN, effects occurring at concentrations at or above the USVI's TN criterion were considered evidence that criterion is protective. On the other hand, the relative contribution of measured constituents to TN were considered when evaluating exposure data was reported to be below the USVI's TN criterion.

Water overlaying coral reefs typically contain low concentrations of dissolved nutrients (Bythell 1990). Zooxanthellae associated with reef-building corals are believed to at least partially overcome this nutrient deficiency by utilizing excretory ammonium produced by host, thus conserving assimilated nitrogen and returning it to the host in the form of amino acids. In addition to the internal recycling of ammonium, algae have been shown to be capable of removing net quantities of ammonium and nitrate from seawater. Due to the host excretion, algae are expected to be exposed to elevated tissue ammonium concentrations, which is reported to suppress nitrate uptake by algae in the presence of significant amounts of ammonium. Bythell (1990) conducted 15 months of study in a backreef environment in Teague Bay, St. Croix, USVI to assess the net exchange of dissolved inorganic nitrogen (DIN) by the *Acropora palmata* under near –natural environmental conditions. He demonstrated the capability of dual uptake of both, nitrate and ammonium by algae. His research suggests however that under the natural conditions, nitrate rather than ammonium provides the primary external sources of DIN in this species. Bythell also points out, that although ammonium may be a sole nitrogen source involved in the internal recycling between algae and coral, nitrate must be regarded as an important (if not major), external source of nitrogen for the symbiosis.

Monitoring data from the Florida Keys National Marine Sanctuary Water Quality Protection Program indicated that, in the Keys, organic nitrogen makes up the vast majority of TN, with other forms of nitrogen (DIN, ammonia, nitrate, and nitrite) together comprising 1-10% or less of measures of total nitrogen (Briceno and Boyer, 2013). The same is reported for waters of southern Florida. Another study of estuarine waters of Broward County, Florida also indicates that organic nitrogen dominates TN, with inorganic forms generally comprising less than 20% of TN (Broward County, 2001). Because the data identifying various forms of nitrogen in marine waters around the USVI was not identified, data generated for the areas of Florida Keys is believed to be relevant to the USVI and thus was used in this analysis.

Within the study specific discussions below, the literature nutrient values are compared to the proposed criterion for TN. In most of cases, the precise comparison cannot be made as the literature values reflect different nitrogen components. To convert the proposed TN criterion of 0. 207 mg/L into the components evaluated in the studies, a Nitrogen (N) equivalent was calculated (10% of the proposed TN criterion), which was more representative of the values contained in the literature.

Given the information provided above, an "N equivalent value" of 0.02 mg/L (1.48 uM), representing 10% (median value between 1 and 20%) of the proposed TN criterion will be used to compare it to the literature data reported in DIN.

Brief summary of published research

Dissolved Inorganic Nitrogen (DIN)

(1) Fabricius (2005) published a detailed summary of experiments/studies evaluating the direct effects of DIN on coral calcification, tissue growth and zooxanthellae. A brief summary of results published by Dubinsky and Stambler (1996) and Szmant (2002) are presented in Table below. In general, studies show that most experiments were conducted at environmentally unrealistically high levels, and that significant inconsistencies exist across studies that are as yet unresolved. Many studies found that high levels of DIN reduce calcification up to 50%, while other studies found no change in growth rates, or reported slightly increased rates of calcification and linear extension but reduced skeletal densities. Effects of DIN on tissue growth and composition vary across studies, with some reporting reduced lipids (Koop et al., 2001), and others finding enhanced zooxanthellae protein but unaltered host protein (Marubini, 1996). Increased DIP appears to have little effect on tissue growth. Most studies found that increased DIN increases zooxanthellae density, increases the contents of nitrogen and chlorophyll *a* per zooxanthellae, and increases photosynthetic rates. It was also noted that the stimulated growth of the zooxanthellae can lead to a reduction in calcification rates and promote coral bleaching (Marubini and Davies 1996; Nagelkerken 2006; Wooldridge and Done 2009). In experimental studies, colony survival was generally unaffected by DIN, while coral mortality increased, for unknown reasons, in one species after a 1-year field exposure to high daily pulses of DIN (Koop et al., 2001); however, such high and frequent nutrient pulses are unlikely to be encountered in nature for sustained periods except near sewage outfall sites.

Zooxanthellae densities increase in response to enhanced DIN availability because this nutrient is preferentially used for zooxanthellae growth rather than the growth of host tissue. Reduced calcification at elevated DIN has been explained as follows: zooxanthellae populations increase after release of N limitation, these cells have preferential access to the available CO_2 which they use for photosynthesis, hence less CO_2 is available for calcification and CO_2 becomes a limiting factor. In the field, DIN is quickly taken up by phytoplankton and bacteria and benthic food webs; the soluble inorganic nutrients are taken up rapidly by the algae, hence their concentrations will generally be quite low (Laws and Redalje 1979) and elevated nutrients are available in their dissolved inorganic form only for short periods of time over relatively limited areas. Severe direct effects of dissolved inorganic nutrients on corals appear restricted to heavily polluted, poorly-flushed locations such as semi-enclosed lagoons and bays, where they are linked to reduced reef calcification, coral cover and biodiversity.

In summary, the available information suggests that short-term exposure to high levels of unprocessed DIN does not kill or greatly harm individual coral colonies, however chronically increased levels of dissolved inorganic nutrients may alter reef metabolism and reef calcification sufficiently to cause noticeable changes in coral communities. Also, The addition of nutrients to such a complex system will tend to breakdown the necessity for the natural symbiotic/recycling processes and in doing so will lead to a reduction in the stability of the system and leave it poised in a relatively unstable state of equilibrium and thus prone to sudden phase shifts. Typical triggers for such phase shifts would be events that lead to the physical damage of the coral matrix, e.g., hurricanes/cyclones, bleaching events, and attacks by corallivores (e.g., COTS). If the system were not eutrophic the corals and the associated complex structure would most likely re-establish over time, i.e., the hermatypic corals in such a system could be considered robust (Bell and Elmetri 1995; Bell et al. 2007). However, if the system is eutrophic there will be increased competition for space by other organisms (e.g., algae and filter feeders such as soft corals, sponges, and bivalves) that will now flourish in the more fertile/productive waters. Under such a situation it is less likely that the hermatypic corals will re-establish to dominate the benthos as before. Hence in eutrophic conditions the hermatypic corals should be considered as being fragile and the system overall would exhibit a low resilience (Bell et al. 2007; Littler et al. 2009).

Existing data indicate that: (a) there is strong evidence that zooxanthellae numbers, chlorophyll per unit surface area, and photosynthetic rates increase with increasing DIN, affecting the transfer of energy, CO_2 and nutrients between zooxanthellae and host; (b) there is little evidence that dissolved inorganic nutrients alter tissue thickness, lipids or coral protein per unit surface area; and (c) while some studies found increased or unaltered skeletal growth (measured as linear skeletal extension, skeletal density and/or calcification), many controlled experimental studies found a reduction in growth at elevated levels of DIN.

In most cases where terrestrial runoff causes reef degradation, disturbances other than eutrophication were the proximate causes of coral mortality, and runoff effects only became obvious when hard corals failed to reestablish after such disturbances or phase shifts, as noted above. The limited available experimental data suggest that the three main pre-settlement stages of coral reproduction (gamete production, egg fertilization, and larval development and survival), as well as larval settlement rates, are sensitive to dissolved inorganic nutrients. In acroporid corals, fecundity, egg sizes, egg fertilization rates and embryo development are all reduced, and the occurrence of irregular embryos increased, at slightly elevated levels of dissolved inorganic nutrients (from 1 μM NH_4 and 0.1 μM PO_4, i.e., at <10% of concentrations that detrimentally affect adult corals; Ward and Harrison, 2000; Harrison and Ward, 2001). Furthermore, spat densities were reduced at elevated levels of nitrogen (Ward and Harrison, 1997). Other observed effects include failed planulation in the brooding coral *Pocillopora damicornis*, and reduced egg sizes in *Montipora* that releases zooxanthellate eggs, after four months of exposure to elevated ammonium levels (Cox and Ward, 2002).

(2) Wiedenmann et al (2013) investigated responses to controlled additions of DIN and/or phosphorus on an offshore One Tree Island reef at the southern end of the Great Barrier Reef, Australia (The ENCORE experiment). This study considered three combinations of nutrients: a low treatment (low 0.07 µM DIN; 0.009 mg/L and low phosphate), a replete treatment (high 6.5 µM DIN; 0.81 mg/L and high phosphate), and an imbalanced treatment (high >3 µM DIN; 0.38 mg/L and ambient phosphate) that was used for comparison against the USVI's proposed TN criterion. Results are summarized in Table 1 below. Wiedenmann et al (2013) determined that DIN in combination with limited phosphate concentrations resulted in an increased susceptibility of corals to temperature and light-induced bleaching. The study also suggested the most severe impact on coral health might not actually arise from the over-enrichment with one group of nutrients but from the resulting relative depletion of other groups.

<u>DIN in uM to mg/L Calculations:</u>
DIN molecular weight: 125.0408 g/mol

- *Molarity Conversion for Low treatment: 0.07 uM*

*0.07 uM solution/1 L * 1 mol/1,000,000 umol = 7x10(-8) mol/L*
*7x10(-8) mol/L * 125.0408 g/mol * 1000mg/1g = 0.009 mg/L*

- *Molarity Conversion for Imbalanced treatment: 3 uM*

*3 uM solution/1 L * 1 mol/1,000,000 umol = 3x10(-6) mol/L*
*3x10(-6) mol/L * 125.0408 g/mol * 1000mg/1g = 0.38 mg/L*

- *Molarity Conversion for Replete treatment: 6.5 uM*

*6.5 uM solution/1 L * 1 mol/1,000,000 umol = 6.5x10(-6) mol/L*
*6.5x10(-6) mol/L * 125.0408 g/mol * 1000mg/1g = 0.81 mg/L*

Based on the conversion of the treatment values into concentrations of mg/L, and a direct comparison of DIN values to TN criterion, the low treatment of the Wiedenmann et al study for DIN resulted in value lower than the proposed TN criterion. However, the levels of DIN for the imbalanced and replete treatments were higher than the TN criterion. When comparing the low treatment literature values to the calculated equivalent the comparison result did not change; however, for the imbalanced and replete treatments the literature values were greater than the equivalent concentration in all cases.

The interpretation of responses reported in the Weidenmann et al study is further complicated by the fact that bleaching (paling) of corals observed during the study was light–induced. As a result, the influence of high light exposure and resulting interaction between the coral and algae played a significant role in response to increased DIN levels in combination with limited phosphate concentrations.

Additionally, the Weidenmann et al publication acknowledges that other researchers have shown corals can thrive in high nutrient waters or, as noted in the paper itself, that the depletion of one parameter (for example, phosphate) can negatively influence coral health, particularly when other stressors are present. Based on the likelihood that the paling of the corals in the low treatment is due to insufficient nutrient addition, the comparison to the low treatment was not considered to indicate a negative conclusion with regard to the proposed TN criterion and protection of the corals. Proposed TN criterion is greater than an insufficient level of nutrient concentrations but lower than concentrations when combined with high light levels that negatively affect coral health, particularly when considered the comparison using N equivalent.

(3) Thurber et al (2014) evaluated a link between chronic nutrient enrichment and coral disease/bleaching. Over 3 years, researchers continuously exposed areas of coral reefs to elevated levels of N and P. Levels of DIN in the control plots were within the range of concentrations for offshore reefs, as measured in a 15-year water monitoring program in the Florida Keys (Boyer & Briceño, 2010). DIN was approximately three times higher in the enrichment plots (3.91 ± 1.34 µM) than control plots (1.15 ± 0.05 µM). Over 1200 corals were monitored for signs of disease and bleaching. Corals within enrichment plots had a twofold increase in both the prevalence and severity of disease compared with corals in unenriched control plots. In addition, elevated nutrient loading increased coral bleaching; Agaricia spp. of corals exposed to nutrients suffered a 3.5-fold increase in bleaching frequency relative to control corals. However, 1 year later, after nutrient enrichment had been terminated for 10 months, there were no differences in coral disease or coral bleaching prevalence between the previously enriched and control treatments.

Ammonium

(4) Harrison and Ward (2001) studied the effects of nutrient (ammonium and/or phosphate) enrichment on ferritization success and early embryo development of *Acropora longicyathus* collected at One Tree Island and *Goniastrea aspera* collected at Magnetic Island, Great Barrier Reef, Australia. Gametes of *A. longicyatlms* were treated with 1, 10, and 100 µM ammonium, phosphate, and combined ammonium and phosphate and gametes of *G. aspera* with 0.5, 1, 5, and 50 µM of the same nutrients. For this study, ammonium level of 0.65 +/- 0.69 uM was used as a background level. Harrison and Ward concluded that phosphate had a more pronounced negative effect on fertilization rates; however, in one fertilization cross, the rates actually improved as the concentrations got higher. For ammonium values of 1 µM and higher, mean fertilization rates and success were reduced for *A. longicyathus* for one cross, but not the other cross, which remained comparable to the sea water control, except at the 100 µM level. For *G. aspera* the rates were only significantly reduced at the combined nutrients level of 50 µM.

Ammonium in uM to mg/L Calculations:
Ammonium (NH4) molecular weight: 18.03846 g/mol

- _0.5 uM_

0.5 uM solution/1 L * 1 mol/1,000,000 umol = 5x10(-7) mol/L
5x10(-7) mol/L * 18,03846 g/mol * 1000mg/1g = 0.009 mg/L

- _1 uM_

1 uM solution/1 L * 1 mol/1,000,000 umol = 1x10(-6) mol/L
1x10(-6) mol/L * 18,03846 g/mol * 1000mg/1g = 0.018 mg/L

- _5 uM = 0.09 mg/L_

When comparing the proposed TN criterion against the values in the 2001 study, the ammonium and combined nutrient values associated with the *A. longicyatlms* (1 µM) and *G. aspera* (0.5 µM) were both lower than the proposed TN concentration. As noted above, based on the varied responses depending on the cross analyzed, the study also found some responses were improved or not affected as concentrations of the individual nutrients were increased. In general, the effects noted in this study were more adverse to corals than in the Wiedenmann et al study. Important differences are that the Harrison and Ward study used higher phosphate concentrations and instead of DIN (represented as primarily nitrate+nitrite) used ammonium, which is generally considered toxic at lower levels than nitrate or nitrite.
As with the previous study, the literature values of this study were also compared against the calculated N equivalent. For the ammonium dosings of *A. longicyatlms* (1 µM) and *G. aspera* (0.5 µM), the literature values were lower than N equivalent.

(5) Koop et al (2001) reported the results of the ENCORE project performed on the Great Barrier Reef. An important aspect of this field experiment is that nutrients enrichment was done under pristine conditions (with no increased sedimentation, high organic loadings or overfishing). A total of twelve microatolls were divided into 4 treatments (control, N, P and N+P) and dosed with nutrients daily over 2 years. The ambient concentration of 0.65 +/- 0.69 uM ammonium was used in the study as a control concentration. Loading was increased from 10 uM ammonium to 20 uM ammonium in the second year because of lack of clear effects by the original level of enrichment on most systems studies. Due to the fact that concentration of nutrients used in this study was significantly higher than the concentration of TN proposed as criterion, the decision was made not to evaluate this particular study as a part of our analysis.

(6) O'Neil, in his final thesis (O’Neil 2015), includes general recommended ranges for water quality parameters commonly measured in coral aquaculture, from published sources, including recommended ranges for *A. cervicornis*. For *A. cervicomis*, the recommended values for total ammonia, nitrite and nitrate were <0.03 mg/L, <0.03 mg/L and <1.0 mg/L, respectively. The primary objective of

the study was to assess if *A. cervicomis* fragments can be produced in aquarium conditions at comparable rates to offshore nurseries and whether corals out planted from land-based nurseries vs offshore nurseries experience different rates of survival success.

When all of the nitrogen components are combined the result is <l.06 mg/L which is greater than proposed TN criterion. Therefore, despite not specifically addressing specific health impacts or endpoints, the results of this paper regarding what water quality would support an environment to grow corals was an important indicator of the necessary levels of nutrients to support the growth and culture of *A. cervicomis*. The work was based on existing published information on coral aquaculture.

(7) In a Review Paper by Bell (1992), a study by Tomascik and Sander (1985), which attempted to address the apparent degradation of reefs in Barbados, suggests that mild amounts of eutrophication may have a positive effect on coral growth but beyond the threshold limit adverse impacts occur. A comparison of the 1981-1982 results from the least polluted stations indicates that measurable changes for decreased coral growth rate occur for annual mean suspended particulate matter concentration (SPM) greater than 4-5 mg/I with a corresponding annual mean chlorophyll a level above 0.4 mg/m 3. The corresponding annual mean nutrient levels were: dissolved (total i.e. NH3-N + NO3- + NO2-N) inorganic nitrogen (DIN) concentration of around 1 uM and P-PO, levels of 0.06-0.08 uM. Historical data presented by Tomascik and Sander (1985) for Barbados indicated that the onset of eutrophication may be characterized by even lower levels of the above water quality parameters. For the Kaneohe Bay studies, chlorophyll a levels were also highly correlated with the degree of eutrophication (Smith et al., 1981; Laws and Redalje, 1979). The results of Smith et al. (1981) for the least polluted section of Kaneohe Bay before diversion, yet still considered eutrophic by Laws and Redalje, show an annual mean chlorophyll a level of 0.68 mg/m3 and annual mean nutrient levels of 0.23 uM P-PO4 and 1.1 uM DIN. Laws and Redalje (1979) report annual mean values for the same region of 0.61 mg/m3 chlorophyll a, 0.20 uM P-PO, and 1.97 uM DIN. As noted above, after diversion of the sewage considerable improvement in the coral community structure occurred, the annual mean chlorophyll a level in the least polluted region was now 0.55 mg/m3 with corresponding levels for P-PO4 of 0.11 uM and for DIN of 0.78 uM (Smith et al., 1981).

(8) Lapointe (1997), performed water and macroalgal sampling starting in 1987 in Discovery Bay, Jamaica. Concurrent surveys along transects in 1987, 1992, and 1996 showed eutrophic indicator species expansion along this timeline. Additionally, studies on the Floridian reefs were performed during summer 1994 and 1995 at four sites between West Palm Beach and Hobe Sound (Princess Anne, North Colonel’s Ledge, Jupiter Ledge and Hobe Sound), -2-3 km offshore.

Water samples were collected by SCUBA divers from the surface and bottom of the water column (0.5-3-m depth in Jamaica; 0.5-28-m depth in Florida) into clean, high-density polyethylene Nalgene bottles. The water samples were immediately passed through combus Apical portions were excised and exposed to overnight (12 h) nutrient pulses in polycarbonate aquaria. During the overnight pulses, the macroalgae were gently aerated to provide mixing to facilitate nutrient uptake. The factorial enrichment treatments included NO3- (160 uM), and an unenriched control. These NO3 - concentrations are high compared to ambient concentrations on most coral reefs but are only 2X the NO3 concentrations in undiluted spring water at Discovery Bay (D'Elia et al. 1981). The Florida experiments utilized lower nutrient concentrations and NH4 + rather than NO3 - as the DIN source, because preliminary sampling indicated higher concentrations of NH4+ compared to NO, and generally lower DIN concentrations on these reefs; accordingly, treatments included NH4+ (110 uM), and an unenriched control.

Results of this study support the hypothesis that nutrient enrichment and bottom-up control were major factors causing increased productivity and standing crops of macroalgae on reefs in Jamaica and southeastern Florida. The DIN and SRP concentrations on both reef systems were at or above 1.0 uM, and 0.1 uM, respectively-threshold concentrations noted for macroalgal overgrowth of seagrass and coral reef communities along natural nutrient gradients around seabird mangrove rookeries on the Belize Barrier Reef (Lapointe et al. 1993). The DIN and SRP concentrations on corals reefs impacted by macroalgal blooms globally all seem to be above these thresholds, suggesting that increased nutrient concentrations associated with coastal eutrophication exert a significant kinetic control leading to the development of macroalgal blooms.

Table 1. Summary of the results found in the most relevant literature.

Fabricius (2005)		
NH4, NH4 + PO4	Increased zooxanthellae density, increased protein synthesis by zooxanthellae	Muscatine et al. (1989)
NH_4 (15 µM)	After 8 weeks, increased zooxanthellae density, increased chlorophyll and N per zooxanthella	Snidvongs and Kinzie (1994)
NO_3 (0, 1, 2, 5, 20 µM)	Calcification decreases with increasing NO_3 to 50% of controls, effects significant at $\geqslant$ 1 µM. After 30–	Marubini (1996)

	40 days: at ⩾ 1 µM, increased N per zooxanthellae, increased zooxanthellae density. At ⩾ 5 µM NO_3, increased zooxanthellae size, chlorophyll per zooxanthellae, photosynthesis, increased coral protein through greater zooxanthellae biomass. At 20 µM NO_3, 30% increased chlorophyll and zooxanthellae density, reduced respiration per unit protein	
NH_4 (10 µM and 20 µM)	After 9 weeks: unaltered buoyant weight gain at 10 µM, reduced buoyant weight gain (−60%) at 20 µM	Ferrier-Pages et al. (2000)
NO_3 (2 µM)	No change in zooxanthellae density or rate of photosynthesis. Reduced buoyant weight gain (−34%) after 3 weeks	Ferrier-Pages et al. (2001)
NH_4 (10 or 20 µM)	Inconsistent effects on linear extension and buoyant weight after 1 year: 10–20% reduction, or no effect, or slight increase. Reduced lipids	Koop et al. (2001)
NH4	increased zooxanthellae density, chlorophyll concentration. Decreased linear extension	Stambler et al. (1991)
NO_3 (15 µM)	After 2 weeks, reduced primary production, unaltered zooxanthellae density and chlorophyll concentrations. Temperature effects enhanced by presence of nitrate	Nordemar et al. (2003)
⩾ 1 µM NH_4 and/or ⩾ 1 µM P	Reduced egg fertilisation rates in *Acropora*, increased rate of abnormally formed embryos	Harrison and Ward (2001)
NH_4 (11–36 µM m^{-3}) and/or PO_4 (2–5 µM m^{-3})	Reduced spat densities on tiles in NH_4 enriched, but not in PO_4 enriched treatments	Ward and Harrison (1997)
NH_4 (11–36 µM m^{-3}) and/or PO_4 (2–5 µM m^{-3})	Smaller and fewer eggs per polyp, reduced egg fertilization, increased proportion of irregular embryos	Ward and Harrison (2000)
20 µM NH_4 for 4 months	Failed planulation in *Pocillopora damicornis*. Reduced egg size, but no difference in fecundity and fertilisation in *Montipora* with zooxanthellate eggs	Cox and Ward (2002)

Thurber et al (2014)		
DIN enrichment plots (3.91 ± 1.34 µM)	twofold increase in prevalence and severity of disease compared to controls;	Value greater than TN criterion
	3.5-fold increase in bleaching frequency relative to control	

Weidenmann et al (2013)			
DIN: 0.009 mg/L	Low treatment	Effects: Light-induced paling of corals observed over a period of I 2 weeks potentially due to insufficient nutrient addition.	Value below N equivalent and TN criterion

DIN: 0.38 mg/L	Imbalanced Treatment	After 4 weeks, no visible signs of bleaching. With doubled light iNTEnsity exposure, bleached colonies partially or completely died. Two weeks later with light levels doubled, the corals lost 50% of zooxanthellae	Value greater than TN criterion
DIN: 0.81 mg/L	Replete Treatment	After 4 weeks, no visible signs of bleaching. With doubled light iNTEnsity exposure, various degrees of negative effects observed	Value greater than TN criterion
Harrison and Ward (2001)			
Ammonium: 1uM (0.018 mg/L) 10uM (0.18 mg/L) 100 uM (1.8 mg/L)	*A. longicyathus*	For cross 1m mean fertilization rates and percentage of embryos that developed were comparable to control (94%/93%) at 1 and 10 uM doses. At 100 uM dose, rates were significantly reduced (75%/68%). In cross 2, fertilization rates were reduced (75-15%) at all doses. Mean fertilization success was reduced in treatments with phosphate only.	Effects observed above N equivalent and above TN criterion.
Ammonium: 0.5 uM (0.009 mg/L) 1 uM (0.018 mg/L) 5uM (0.09 mg/L) 50 uM (0.9 mg/L)	*G. aspera*	No significant differences in fertilization rates among 4 treatments and the controls. Fertilization rates and success were only significanty reduced (by 8%) at combined ammonium and phosphate levels of 50 uM. In both crosses, the mean percentage of regular embryos was significantly higher in controls, 0.5 and 1 uM treatments compared to 5 and 50 uM treatments.	Effects observed above N equivalent and TN criterion.
O'Neil (2015)			
Ammonia <0.03 mg/L Nitrate <1 mg/L Nitrite <0.03 mg/L		Presumed beneficial effects – provided values reflect general recommended ranges for water quality parameters commonly measured in coral aquaculture, from published sources. Total nitrogen components <1.06 mg/L. *Also, the recommendation of 0.02-to 0.05 mg/L is consistent with existing TP criterion.*	Greater than TN criterion when combining ammonia, nitrite, and nitrate. Individual components greater than N quivalent.

Bell (1992)			
DIN 1uM/L (0.125 mg/L) P-PO4 0.06-0.08 uM/L (0.007-0.08 mg/L)		In Barbados the coral species Montastrea anmdaris was found to exhibit much slower growth rates in the more eutrophic regions.	Value below N equivalent and TN criterion
Lapointe (1997)			
DIN 1uM/L (0.125 mg/L)		Increased productivity and standing crops of macroalgae on reefs in Jamaica and southeastern Florida.	Value below N equivalent and TN criterion

Summary

Table 1 above compiles all of the concentration based values from the published literature, along with a comparison against the proposed TN criterion and calculated N equivalent. In many cases, the differences between the values reported in the literature and proposed TN criterion changed when the study values were compared to N equivalents, improving the ability to support the comparison between the study values and the USVI's proposed TN value. It is important to mention that the studies done in ambient waters may provide a significantly different result than studies completed in the laboratory setting, which makes the above analysis even more challenging. This introduces the additional uncertainties, since numerous additional factors (not discussed in this evaluation) can further influence corals response to nutrients in the ambient environment (e.g, temperature, light and disease). Additionally, water quality concentrations can attenuate through biological or physical processes before reaching the offshore locations around the USVI. Therefore, many variables complicate a direct comparison or definitive decision as to the amount of nutrients that may cause adverse effects.

Conclusions

Although there are numerous factors that impact corals (increased water temperature, reduced water clarity, disease, predation, etc.), it is clear that excess of nutrients are also a contributing factor to corals degradation (Zhou et al 2017). As it is clearly indicated in the literature, tropical reef-building corals commonly flourish in nutrient-poor oligotrophic environments (Radecker et al 2015, Zhou et al 2017). The availability of nitrogen sources in coral reefs underlies strong seasonal and diel variations, and can be significantly affected by anthropogenic activities. In general, low internal nutrient availability, specifically of nitrogen, appears to be crucial for maintaining high primary production, while simultaneously controlling algae growth.
At the same time, a healthy balance is needed for corals to prosper. Zhou et al (2017), for example, reported that elevated ammonium concentrations could reduce the negative effect of heat stress on the coral *P. damicornis*. The elevated concentrations of ammonium have been confirmed to significantly raise the population density of symbiotic zooxanthellae in stony corals, which may partially neutralize the loss of algae under the heat stress. The resulting moderate algae would provide corals with enough organic nutrition and oxygen and repress the heat stress response of corals to a certain

extent. Zhou et al concluded that the availability of inorganic nitrogen was important to determine the resilience of coral to thermal stress and help to reduce a negative effects of heat stress response and maintain metabolism balance in stony corals.

The science describing impacts of nutrient (nitrogen and phosphorus) pollution on coral reefs is complex. Excessive nutrients are only a small component on the long list of coral degradation causes including thermal stress, ocean acidification, increased frequency and magnitude of tropical storms, increased pollutants from coastal development, dredging and runoff (sediment, metals and pathogens), sewage discharge, over fishing, coral diseases and physical damage. Based on the research results published by various scientists, it is clear that isolation of negative (or in some cases positive) nutrient impacts on corals from impacts related to other anthropogenic or natural stressors is extremely difficult (Zhou et al 2017). Because nitrogen/phosphorus pollution is often considered as one of the factors affecting coral health (even though it is difficult to directly quantify it) in general, it is clear that any reduction in anthropogenic nutrient load will be beneficial for corals and their critical habitats.

Through the discussion of qualitative and quantitative analysis comparing the “nitrogen protective values” published in the peer-reviewed literature with the proposed by the USVI TN criterion and the associated effects on coral health, it is believed that the adoption of TN criterion to protect coral reefs and their critical habitats is a step in the right direction.

In addition, it is important to point out that the USVI already adopted TP criterion of 0.05 mg/L into its Water Quality Standard Regulations to protect all of the USVI’s marine waters. The adoption of dual nutrient criteria is also consistent with EPA's recommendations most recently highlighted in EPA's Fact Sheet "Preventing Eutrophication: Scientific Support for Dual Nutrient Criteria (February 2015)."

Based on the best available data, the USVI considers the adoption of TN criterion, and its subsequent implementation into permitting and assessment programs, to be protective for corals as well as other threatened and endangered species. If in the future, the additional data becomes available, the USVI will reevaluate the applicable TP and TN criteria and revise them, if necessary.

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APPENDIX D
NSTEPS (Tetra-Tech) Report

Analysis of Existing Water Quality Data from the United States Virgin Islands under Nutrient Scientific Technical Exchange Partnership Support (N-STEPS)

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February 1, 2018 DRAFT 3.0

TETRA TECH, INC.

Introduction

The USVI DPNR sought support from the USEPA Nutrient Scientific Technical Exchange Partnership Support (NSTEPS) program with help developing nutrient criteria for their estuarine/marine waters. Since 2003, USVI DPNR has completed multiple projects in relation to nutrients and coral health. DPNR's overall goal is to use this data to adopt TN criteria for all estuarine/marine waters and revise their existing TP criterion, if needed.

The goal of this NSTEPS analysis was to conduct analyses in support of this numeric nutrient criteria development process for US Virgin Islands estuarine/marine waters. This included compiling existing data, developing a conceptual model, summarizing current nutrient criteria assessment methods, supporting the US Virgin Islands Department of Planning and Natural Resources (DPNR) Division of Environmental Protection (DEP) in the development of sound nutrient database, and conducting analyses of nutrient data.

The conceptual modeling and data summary exercises were conducted as separate tasks. This report summarizes the data collected and provides a summary of the analyses conducted and their results. The hope is that these analyses will support USVI DPNR's other analyses and ongoing efforts for developing numeric criteria.

Background

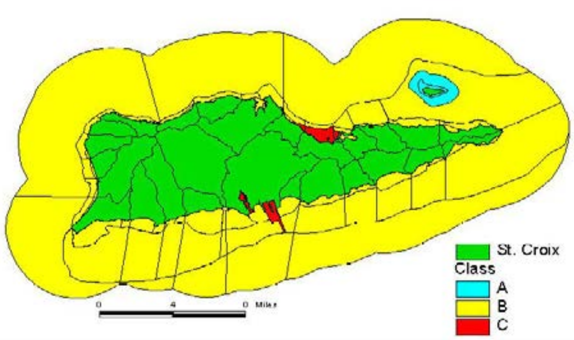
The following excerpts from various documents provide background information related to available data used to conduct the analytical methods that follow. The first is relevant to much of the underlying data, the second to existing P criteria, and the third outlines the Landscape Development Index for USVI, which was used for estimating reference based conditions.

From: US Virgin Islands Nutrient Standards Plan: October 13, 2010

This document describes the US Virgin Island's (USVI) present (as of 2010) and future approaches to addressing eutrophication in estuarine and marine waters. As it is stated in the Plan, the most recent USVI Water Quality Standards Regulations (WQSR), adopted in 2010, include a numeric criterion for phosphorus of 50 ug/L. The USVI does not currently have a numeric criterion for nitrogen. Numeric TN criterion needs to be developed and TP criterion reevaluated for the >600 mi^2 of territorial marine waters in the USVI, according to the following class structure (Table 1, Figure 1):

	Class A	Class B	Class C
Best Usage of Waters	Preservation of natural phenomena requiring special conditions, such as the Natural Barrier Reef at Buck Island, St. Croix and the Under Water Trail at Trunk Bay, St. John.	For maintenance and propagation of desirable species of aquatic life (including threatened and endangered species listed pursuant to section 4 of the federal Endangered Species Act) and for primary contact recreation (swimming, water skiing, etc.).	For maintenance and propagation of desirable species of aquatic life (including threatened and endangered species listed pursuant to section 4 of the federal Endangered Species Act) and for primary contact recreation (swimming, water skiing, etc.).
Legal Limits (physical location)	St. Croix: within 0.5 miles of the boundaries of Buck Island's Natural Barrier Reef. St. John: Trunk Bay.	Any coastal waterbody not classified as Class A or Class C is considered a B waterbody.	St. Thomas Harbor beginning at Rupert Rock and extending to Haulover Cut; Crown Bay enclosed by a line from Hassel Island at Haulover Cut to Regis Point at West Gregerie Channel; and Krum Bay. St. Croix: Christiansted Harbor from Fort Louise Augusta to Golden Rock; Frederiksted Harbor from La Grange to Fisher Street and seaward to the end of the Frederiksted Pier; Hess Oil Virgin Islands Harbor; and Martin-Marietta Alumina Harbor. St. John: Enighed Pond Bay.

Table 1. Water quality classification in the USVI.



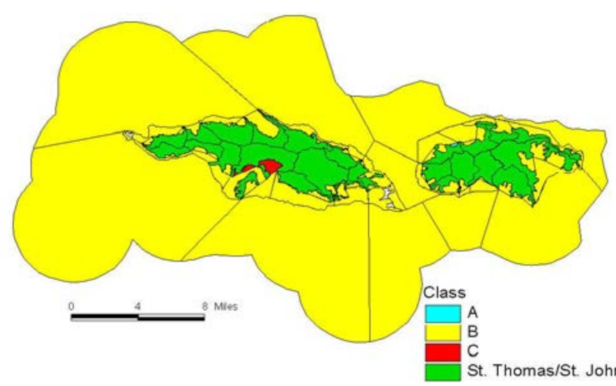


Figure 1. Classes of territorial waters in the USVI.

Coastal waters of USVI are divided into four different assessment units (Figure 2):

1. Embayments – waters surrounded primarily by land; restricted mixing and circulation; most susceptible to water quality problems.
2. Nearshore – at least some portion of water unit in contact with land; mixing may not be restricted; longshore currents help to disperse pollutants; may be impacted by point and non-point sources from land.
3. Offshore – water units bounded on all sides by water; generally thought to be well mixed; any inputs from land are likely to be highly dispersed prior to reaching these waters.
4. Requiring TMDL – waters which do not meet Virgin Islands Water Quality Standards and are therefore considered impaired.

For the process of nutrient criteria development, priority is given to embayments and nearshore waterbodies, based on proximity to land and pollution sources, as well as monitoring priorities of the 303(d) list.
EPA has recommended that states (and territories) establish criteria for nitrogen (N) and phosphorus (P), along with response variables (chlorophyll a and/or water clarity. According to the 2010 USVI WQSR and the EPA N/P Criteria Progress Map (http://cfpub.epa.gov/wqsits/nnc-development), UVSI currently has statewide P criteria for its estuaries and marine waters. TP limits for all marine waters (class A, B, and C) ≤ 50 μg/L. The USVI is in the process of developing TN limits as well. Data have been collected and analyzed, but the proposal of criteria will not be formally submitted until Spring of 2016. According to the most recent schedule, the VIDPNR plans to public notice proposed criteria in Summer of 2016 and adopt revisions in 2017/2018.

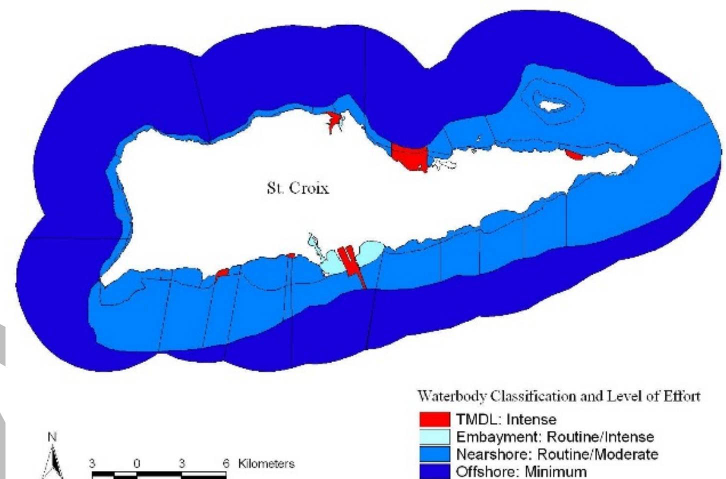


Figure 2. St. Croix example of waterbody classification and level of effort.

The USVI initiated coastal/estuarine nutrient sampling efforts in 2000 with a goal of developing a sufficient database to be used for nutrient criteria development, with special emphasis on examining the relationship of the level of nutrients in the water column and the effect of algal growth on coral reefs. Since then, the following steps/studies have been completed:

- **2001** DPNR received funds from EPA to support a Nutrient Criteria Development Program. These funds supported the procurement of equipment for nutrient analysis.

- **2002** A testing facility was established at the University of the Virgin Islands' MacLean Marine Science Center.

- **2003** USVI Department of Planning and Natural Resources (DPNR) Division of Environmental Protection (DEP) began collecting water samples for N and P around St. Croix.
  - Variety of known/suspected point sources including municipal sewage discharge and a rum distillery outfall, as well as seasonal stormwater runoff.

- **2003-2005** Opportunistic collection of water samples during regularly scheduled ambient monitoring surveys at 43 stations on St. Croix.
  - 137 samples collected and analyzed for TKN
  - 118 samples collected and analyzed for TP

- **2006** Preliminary data analysis on DEP's nutrient dataset (Toller and Villanueva-Mayor 2006)
  - Low levels of nutrients observed raised concerns that analytical detection limits were not sufficiently sensitive. Were the most appropriate lab measure being used?
  - Lack of stratified sampling design precluded statistical tests and compromised interpretation of nutrient datasets.

- **2007-2008:** Nutrient Work Group (local and federal scientists) provided recommendations on strategic decisions regarding nutrient studies
  - Continue nutrient data collection using existing methods but attempt to improve data reliability through evaluation of method detection limits (MDLs) and independent verification of lab results
  - Include a stratified sampling design in future studies to distinguish various waterbody types based on the degree of hydrographic restriction and distance from shore (i.e., estuaries, embayments, open coasts, pelagic waters)

- **2008-2009:** DPNR received the CWA 106 Supplemental funding from EPA to continue work on nutrient criteria development w/ the University of USVI.
  - MDL was used for TKN and TP, but for consistency reasons, DPNR considered replacing TKN with TN, nitrate and nitrite
  - Quantitation Limit (QL) was determined for TKN and TP. Based on confirmed methods to be used for future nutrient studies, DPNR may develop QLs for TN as well.

Ongoing and future efforts (as of 2010) include:

- **Continuation of CWA 106 Supplemental Funding:** Determine nutrient concentrations in different types of waterbodies. These data are necessary to interpret data from future comparative studies of impacted vs. un-impacted waters, which may be obtained from different waterbody types. At the time of this publication (*N.B. this is a 2010 work plan*), the sampling breakdown was 20 samples/year in estuaries & bays, semi-enclosed bays, nearshore open/coastal waters, and pelagic waters. Samples were analyzed for TKN and TP at the UVI MacLean Marine Science Center.
- **RARE Project:** Relate nutrient concentration to coral health (N-STEPS_Template-RARE.xls). Bioassessment surveys were completed by USEPA Office of Research and Development (ORD) for St. Croix from November-December 2007, and for St. John's and St. Thomas in February-March 2009. St. Croix study included nutrients (TP, TN,

nitrate, nitrite, and ammonia). St. John and Thomas study included nutrients, sediment, and biology (coral reefs and reef communities).

USVI has chosen to measure TKN and TP as their two core indicators of water quality. They hope that the studies described above will help them know: 1) What is the operational detection limit for TKN and TP using existing analytical procedures? and 2) Are those detection limits appropriate for nutrient monitoring in the previously listed USVI waterbodies?

The USVI Code's narrative describes the designated best use of waters as "for maintenance and propagation of desirable species of aquatic life (including threatened and endangered species…) and for primary contact recreation (swimming, water skiing, etc.)". This definition is currently interpreted to mean that water body impairment occurs whenever either one of the two usages is compromised. Multiple studies have shown that nutrients and eutrophication can reduce both the aesthetic value and biological quality of a water body (LaPointe 1997).

USVI is interested in understanding how their waterways "experience" excessive nutrient inputs. They have (and continue to) conducted assessment and monitoring studies of nutrients in USVI coastal marine waters in an attempt to understand the full extent of anthropogenic eutrophication. The hope is for this nutrient data to form the foundation for future Total Maximum Daily Load (TMDL) development. For example, nutrient monitoring data collected by the National Park Service was utilized to create a TMDL for Great Cruz Bay, St. John (Tetra Tech, 2005). That analysis indicated that elevated P inputs were primarily responsible for phytoplankton blooms, elevated pH, and reduced water clarity in the Bay.

US EPA Nutrient Policy Data – USVI State Development of Numeric Criteria
According to the US EPA Nutrient Policy Database, the US Virgin Islands has statewide numeric P criteria for all of the VI's marine waters. There are no numeric criteria on the books for nitrogen (N) or chlorophyll-a. The TP criterion for marine waters covers class A, B, and C waters at: **TP ≤ 50 µg/L.**

Landscape Development and Coral Reef Condition
Oliver et al. (2011) successfully related the LDI (landscape development intensity) index to marine ecosystems, specifically coral reefs off St. Croix in the US Virgin Islands. The LDI showed negative correlations with stony corals colony density, taxa richness, colony size, and total coral cover. Percent impervious surface in the watershed was also negatively correlated with total coral cover. This study verifies multiple other studies on the adverse effects that terrestrial anthropogenic activities have on coral reefs in the near coastal zone (Loya 1976, Hubbard 1986, Richmond et al. 2007, Miller and Cruise 1995, Burke and Maidens 2004, Wolandski et al. 2004, Fabricius 2005, Warne et al. 2005). Oliver et al. 2011 also revealed the majority of watersheds with low LDI values and higher proportions of undeveloped land were associated with relatively good coral condition. The authors conclude that future application of the study could be improved by using a survey design that balances the number of sites representing each watershed;

incorporating landscape weighting factors for runoff potential, and better estimates coastal transport runoff – particularly near watershed boundaries. Once refined, they hope the tool can advise coastal management decisions that may have otherwise been made solely on an economic basis.

Methods

Data

Data provided by USVI included the EPA ORD RARE/National Coastal Assessment (NCA) data from 2007-2011, USVI 106 Phase 1 and 2 data (2012-2014), and a Stony Coral dataset for USVI (2007-2009) that was matched to the RARE data.

Data – RARE study (2007, 2009, and 2011)

Station Attributes: Latitude Longitude locations for Station IDs
Chemistry data for 106 stations; including TN, TP (available for all sample dates), NO3, HPO4, NH4, NO2, Urea, NO3+NO2, and lesser samples of SiO2 and HSiO3 (available for some sample dates). Approximately 1-2 samples per site.

Data – 106 Phase 1 (2012, 2103)

Station Attributes: Latitude Longitude locations for 42 Station IDs on Saint John's. There are 19 additional stations on Saint John's, and 22 on Saint Croix with chemical data that do not have latitude/longitude information.

Chemistry data for 83 stations; including depth, salinity, chlorophyll a, LDO, pH, temperature, NO3+NO2, NO2, NO3, ortho-PO4. Approximately 12 samples per site and a limited number of NH4 samples per site.

Data – 106 Phase 2 (2012, 2103)

Station Attributes: Latitude Longitude locations for 22 Station IDs on St. Thomas. Latitude and longitude for 7 additional Stations, located around St. Croix, are provided in separate spreadsheets, now merged.

Chemistry data for 28 stations; including depth, salinity, TN, TP, chlorophyll a, LDO, pH, temperature, TDS, pH, turbidity, TSS, NO3+NO2, NO2, NO3, PO4, NH4, and Secchi depth. Approximately 5-6 samples per site.

Data – Stony Coral Data (2007-2009)

We received 2 sets of Stony Coral Data for USVI for 2007 (50 stations) and 2009 (52 stations). These contain raw data on coral taxa, percent live, height, diameter, and health conditions (i.e. bleached, diseased, clionid presence). Sampling stations have now been associated to those from the RARE data to join with the corresponding water chemistry provided for those sites.

For 22 of the Data – 106 Phase2, we have paired water chemistry and coral data. For 101 sites in the RARE/NCA dataset, we have paired water chemistry and coral information.

Data Preparation – Cleaning and Averaging

Some data records were found to contain negative and zero values for various nutrient measurements. These values were assumed to be errors and were removed. Measurements from different dates were averaged for each station after log-transformation (if needed) to generate site averages for some analyses.

Geospatial Analysis

The station coordinates were used to analyze the spatial relationships between the measurements and the islands. The data was combined with the Land Use/Land Cover and Landscape Development Index (LDI) values for catchments of the USVI, from EPA ORD Gulf Ecosystem Division (Oliver et al. 2011). For each chemistry station, the distance from shore was calculated along with the LDI value of the closest catchment. A number of data points had coordinates located within the USVI catchment polygons, and thus had a distance of 0. These distances were manually changed to 1 m so that the distance variable could be $\log_{10}$ transformed for analysis.

Exploratory Statistical Analysis

For each of the three datasets (full chemistry dataset, Phase 2 coral dataset with matched chemistry data, and RARE/NCA Coral dataset), exploratory statistical analyses were performed for each measurement. Each variable was plotted, assessed for normality, and $\log_{10}$ transformed where necessary. The variables considered were:

Full chemistry dataset: Depth, latitude, longitude, (log10) distance from shore, LDI, (log10) salinity, pH, temperature, (log10) TN, (log10) TP, (log10) NO2, (log10) NO3, (log10) NO2 + NO3, (log10) NH4, (log10) PO4, (log10) DO, and (log10) Chlorophyll.

Phase 2 coral dataset: Depth, latitude, longitude, (log10) distance from shore, LDI, (log10) salinity, pH, (log10) TN, (log10) TP, (log10) NO2, (log10) NO3, (log10) NO2 + NO3, (log10) NH4, (log10) PO4, (log10) DO, (log10) Chlorophyll, total dissolved solids, total suspended solids, and turbidity, as well as the following biotic metrics: Coral, Gorgonian, Sponge, Macroalgae, and DCA.

RARE/NCA coral dataset: Latitude, longitude, (log10) distance from shore, LDI, (log10) TN, (log10) TP, (log10) NO3, (log10) NO2 + NO3, (log10) NH4, (log10) HPO4, (log10) SiO2, (log10) Urea, as well as the coral metrics of average percent alive, average height, and average max diameter.

Distributional Statistics

Distributional statistics (mean, median, standard deviation, and 10th, 25th, 75th, and 90th percentiles) were calculated for the water chemistry data. This was performed for the entire data set, the subset of data within 500 m of shore, and the subset of data within 500 m of shore and with adjacent watershed LDI < 2. This final category was assumed to represent conditions least-disturbed by human activity related runoff, accepting that the boundary condition around the USVI is large and dilute and mixing likely affects this signal as distance increases (thus the 500m criterion).

Stressor-Response Analysis

Stressor-response relationships were analyzed using linear regression models. The correlation coefficient was calculated for each pair of variables in each dataset. Each pair of variables was plotted, along with a linear regression model if significant ($p < 0.05$). Relationships between LDI and nutrients were of particular interest, as were relationships between nutrients and coral metrics.

Results

Geospatial Analysis

Geospatial analyses for the three datasets are shown below. The sampling points are colored based on their distance from shore. The subset of data within the two closest classifications (within 500 m from shore) were used in further analyses to study the effects of LDI. The catchments are colored based on their LDI value. Each sampling point is assigned the LDI value of the closest catchment.

The sampling density, especially for points within 500 m of shore, is somewhat lower for St. Croix than for the two northern islands. This is true for the full chemistry dataset and the RARE/NCA coral dataset. The Phase 2 coral dataset is quite small and the sampling points are clustered within three catchments, all with relatively high LDI. There are no Phase 2 data points around St. John. Hence, this dataset may have limited statistical power.

Note that data points without coordinates were not visualized, but were included in the analyses that did not require locations (e.g. used for nutrient vs. chlorophyll, but not LDI vs nutrient). There were also a few data points with coordinates that were far from the US Virgin Islands. Any data point farther than 10 km from USVI shoreline was eliminated completely.

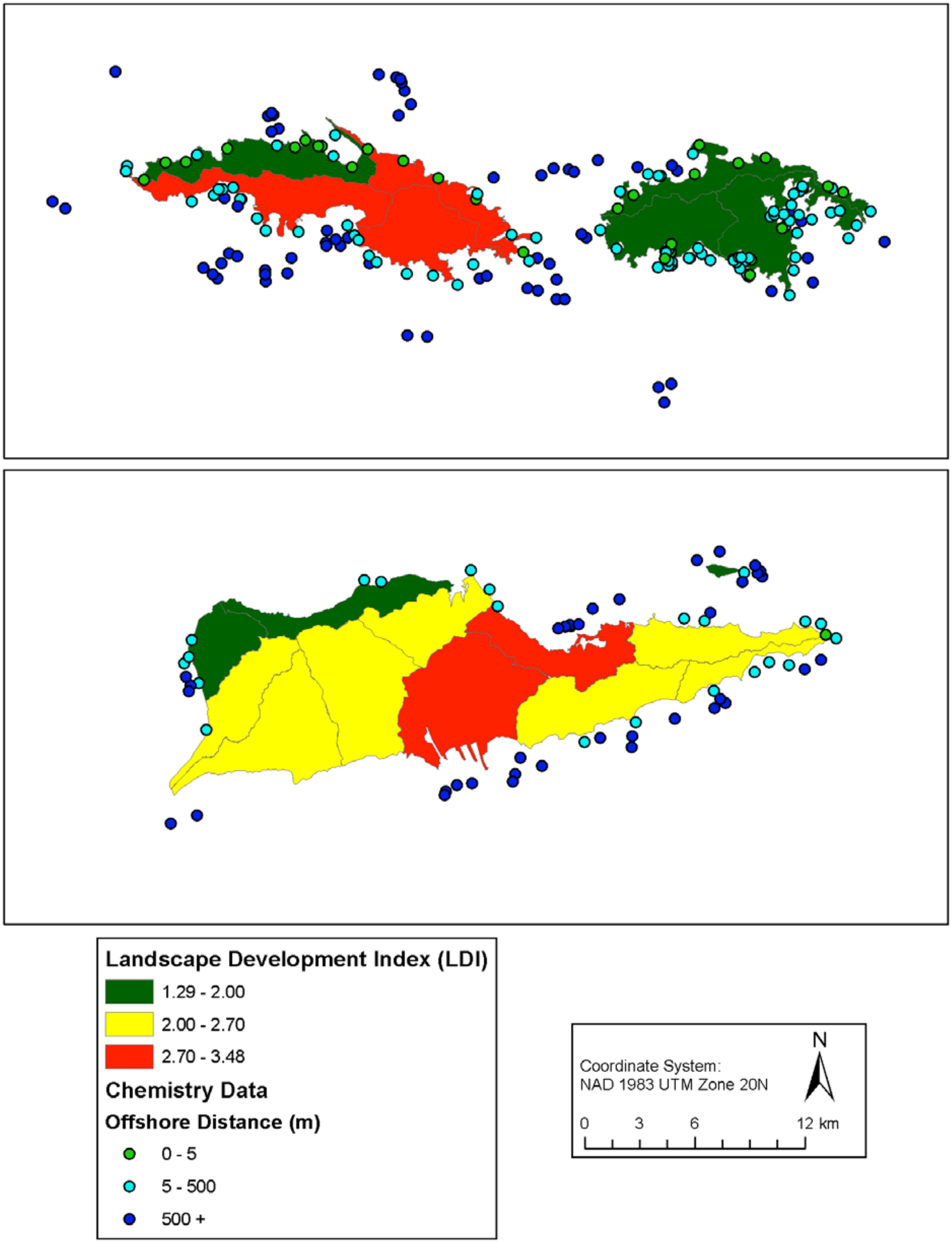

Landscape Development Index (LDI)
1.29 - 2.00
2.00 - 2.70
2.70 - 3.48
Chemistry Data
Offshore Distance (m)
0 - 5
5 - 500
500 +
Coordinate System:
NAD 1983 UTM Zone 20N
N
0 3 6 12 km

Figure 3. Locations of sampling points for the full chemistry dataset, along with the LDI values for catchments within the USVI.

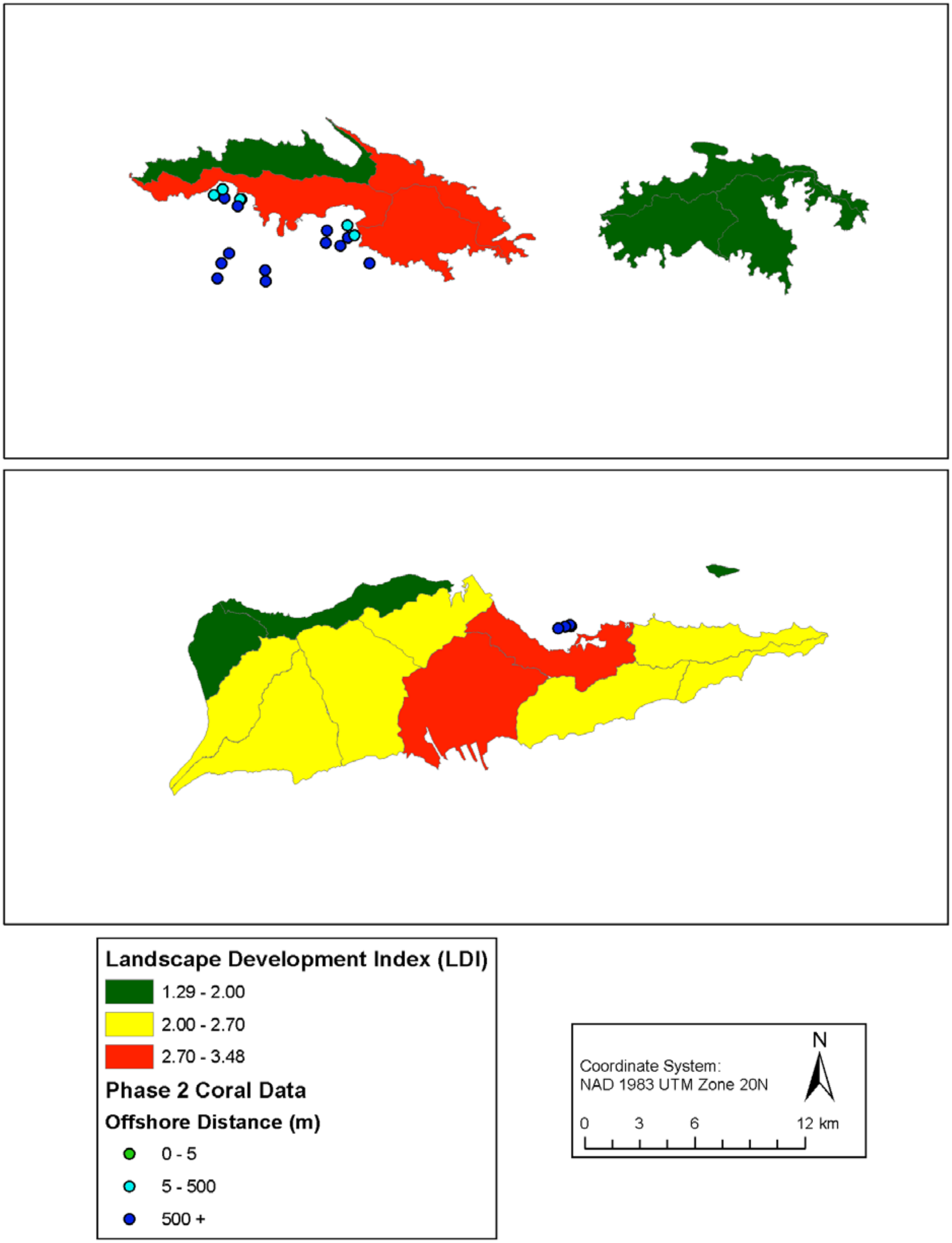

Landscape Development Index (LDI)
1.29 - 2.00
2.00 - 2.70
2.70 - 3.48
Phase 2 Coral Data
Offshore Distance (m)
0 - 5
5 - 500
500 +
Coordinate System:
NAD 1983 UTM Zone 20N
N
0 3 6 12 km

Figure 4. Locations of sampling points for the Phase 2 coral dataset, along with the LDI values for catchments within the US Virgin Islands.

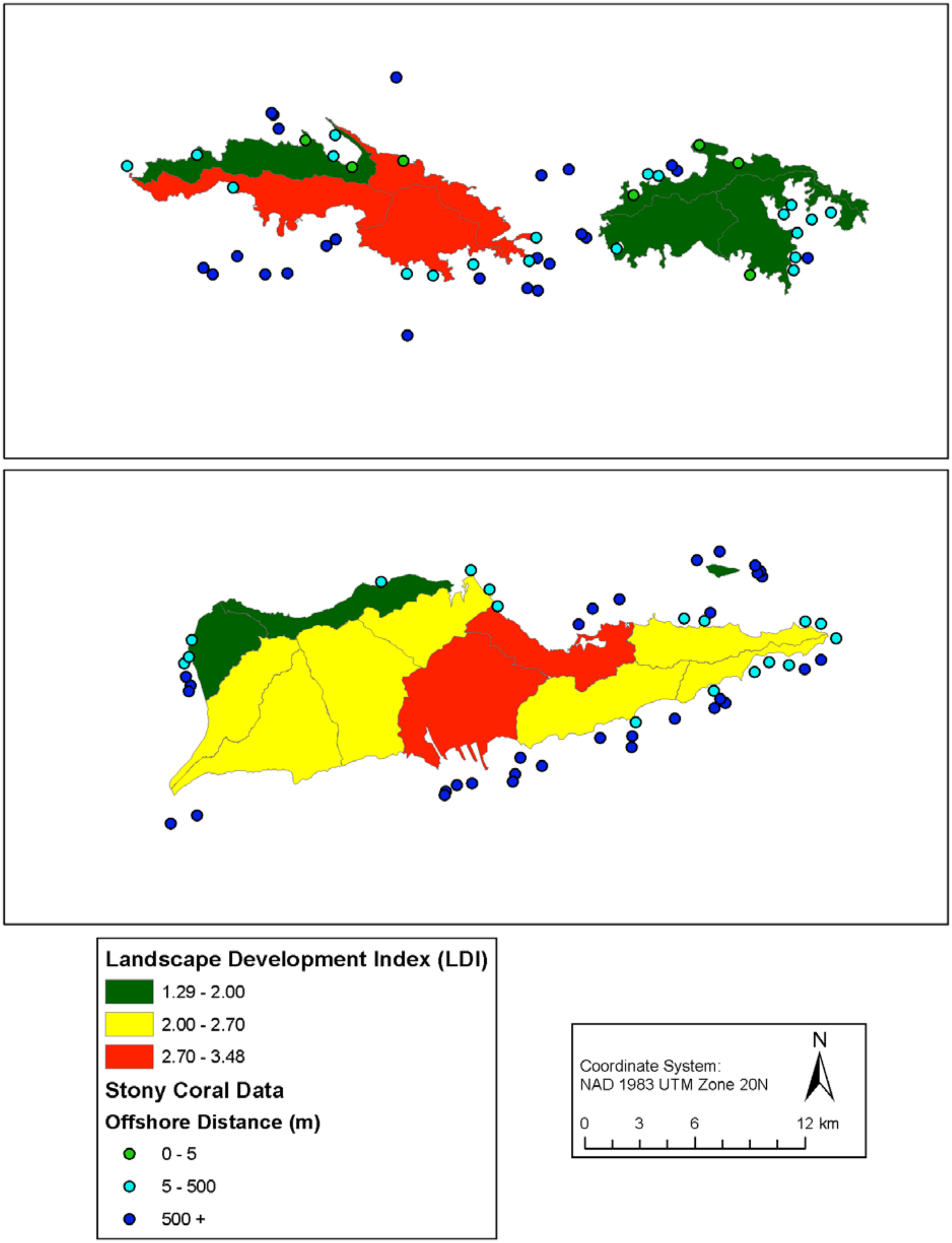

Landscape Development Index (LDI)
1.29 - 2.00
2.00 - 2.70
2.70 - 3.48
Stony Coral Data
Offshore Distance (m)
0 - 5
5 - 500
500 +
N
Coordinate System:
NAD 1983 UTM Zone 20N
0
3
6
12 km

Figure 5. Locations of sampling points for the RARE/NCA coral dataset, along with the LDI values for catchments within the US Virgin Islands.

Exploratory Statistical Analysis – Full Chemistry Dataset

Exploratory analysis of the full chemistry dataset is shown below. Note that some variables have been $\log_{10}$ transformed in order to better meet assumptions of normality. However, even after the transformation, some variables do not exhibit clean normal behavior (especially chlorophyll and total phosphorus). Outliers have not been removed, as they may be important in understanding the distribution of water quality parameters. Additional unused variables can be found in The Addendum.

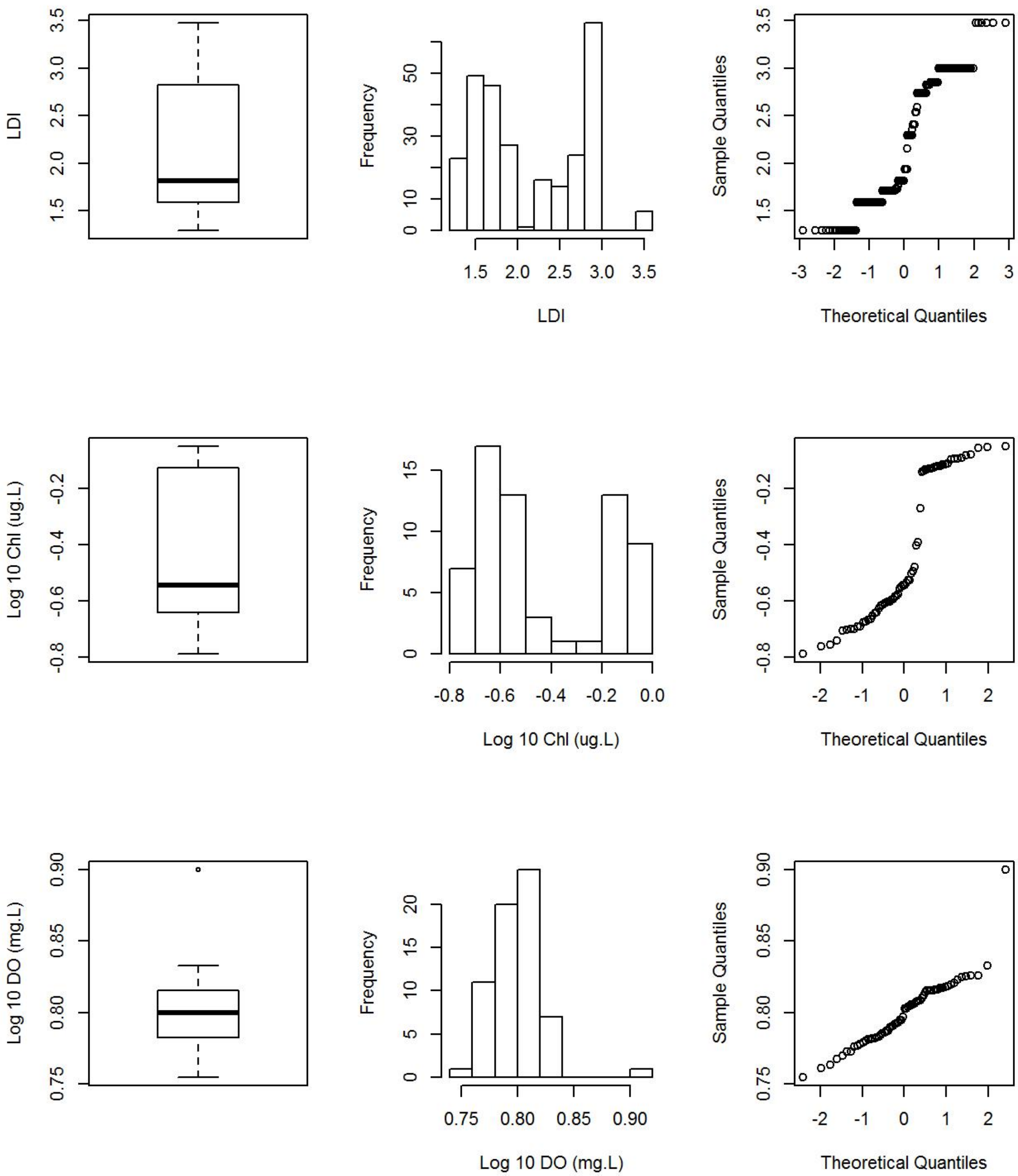


Figure 6-8. Distributions of Landscape Development Index, Log10 Chlorophyll (ug/L), and Dissolved Oxygen (mg/L) values in full chemistry dataset.

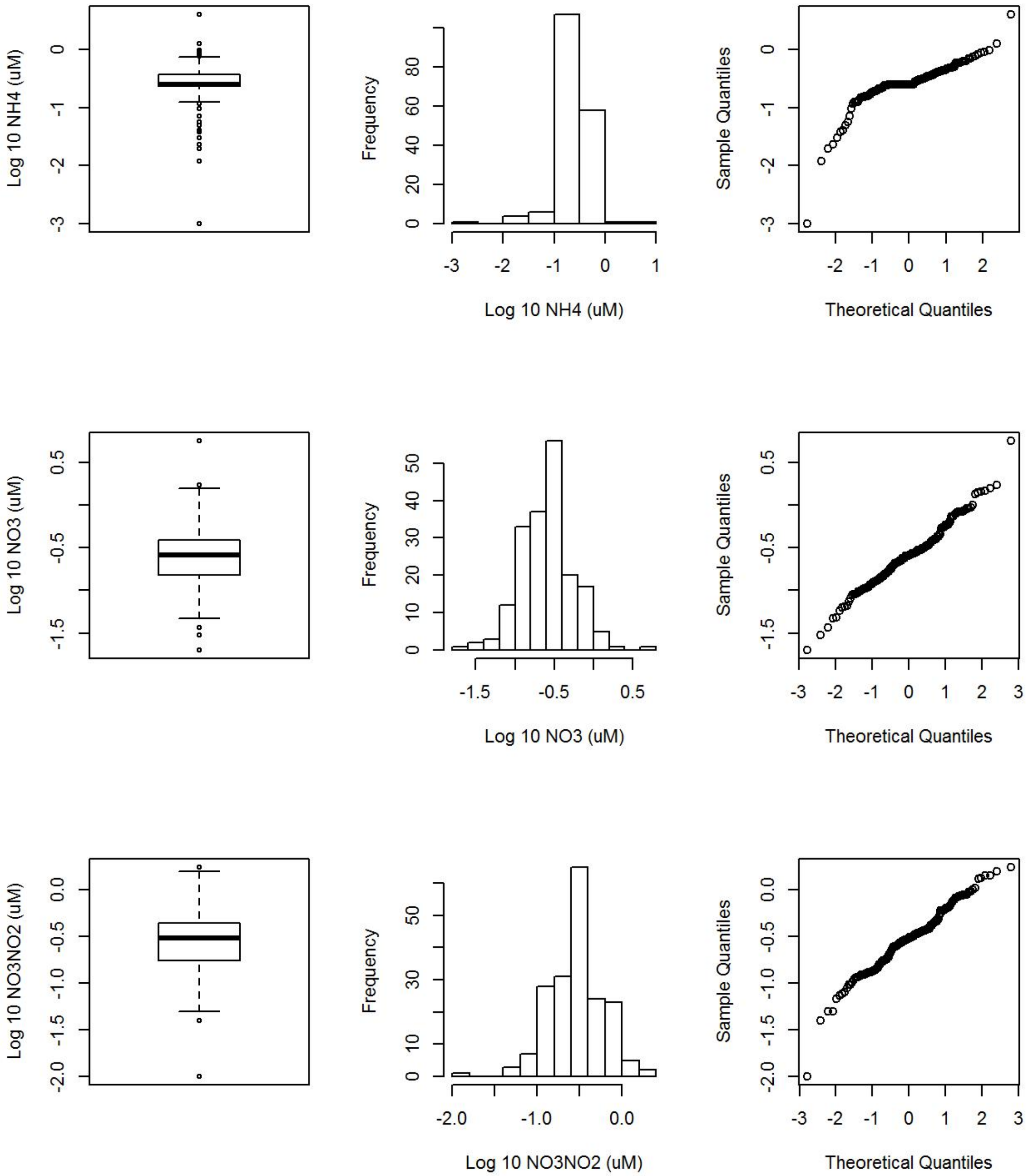


Figure 9-11. Distributions of Ammonium, Nitrate, and Nitrate + Nitrite values (uM) in full chemistry dataset. All Log10 transformed.

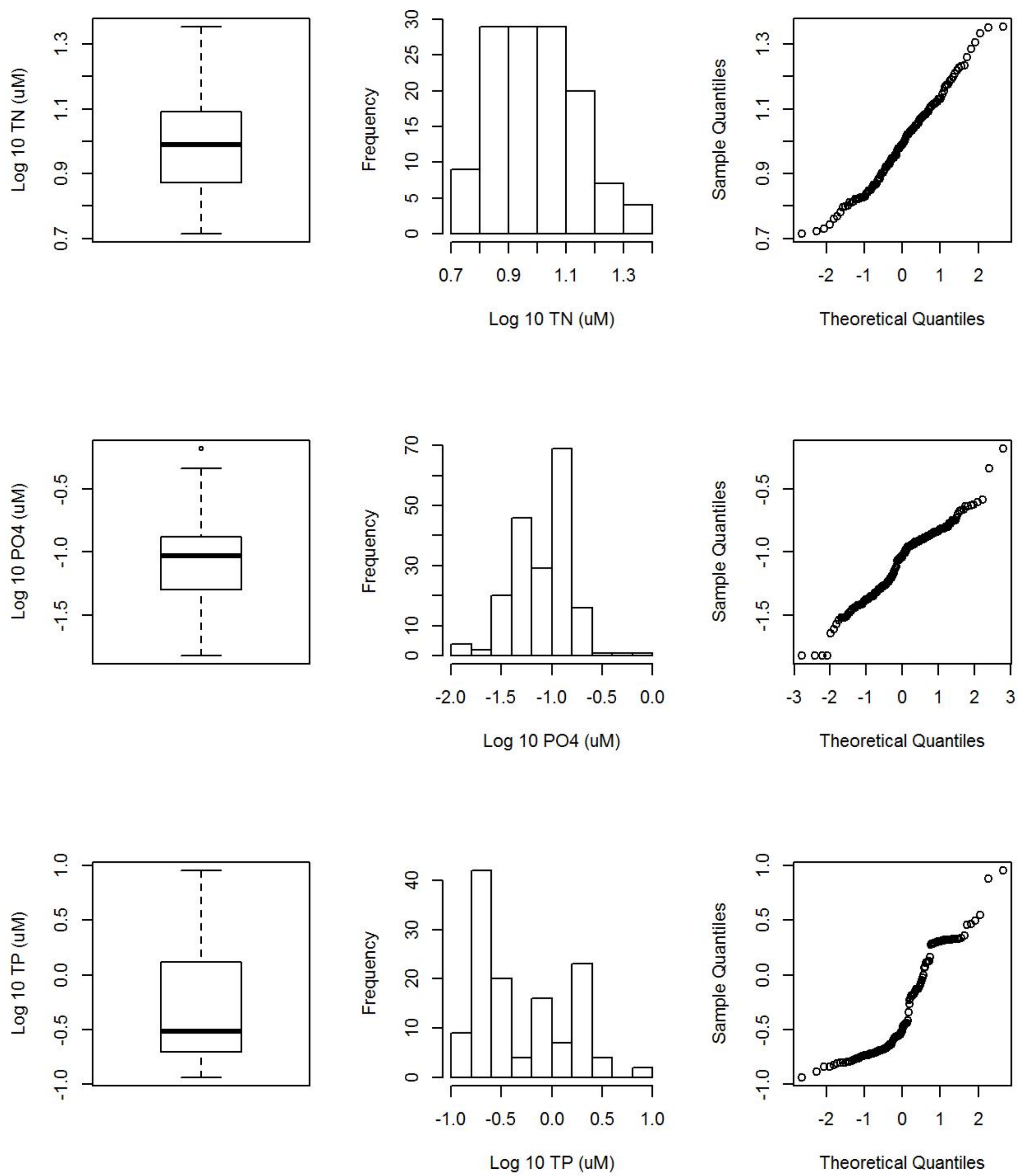


Figure 12-14. Distributions of Total Nitrogen, Phosphate, and Total Phosphorus values (uM) in full chemistry dataset. All Log10 transformed.

Exploratory Statistical Analysis – Phase 2 Coral Dataset

Exploratory analysis for the Phase 2 coral dataset is shown below. Note that some variables have been $\log_{10}$ transformed in order to better meet assumptions of normality. Even after the transformation, some variables do not exhibit clean normal behavior (although they are generally cleaner than those in the full chemistry dataset). Outliers have not been removed, as they may be important in understanding the distribution of water quality parameters. Additional unused variables can be found in The Addendum.

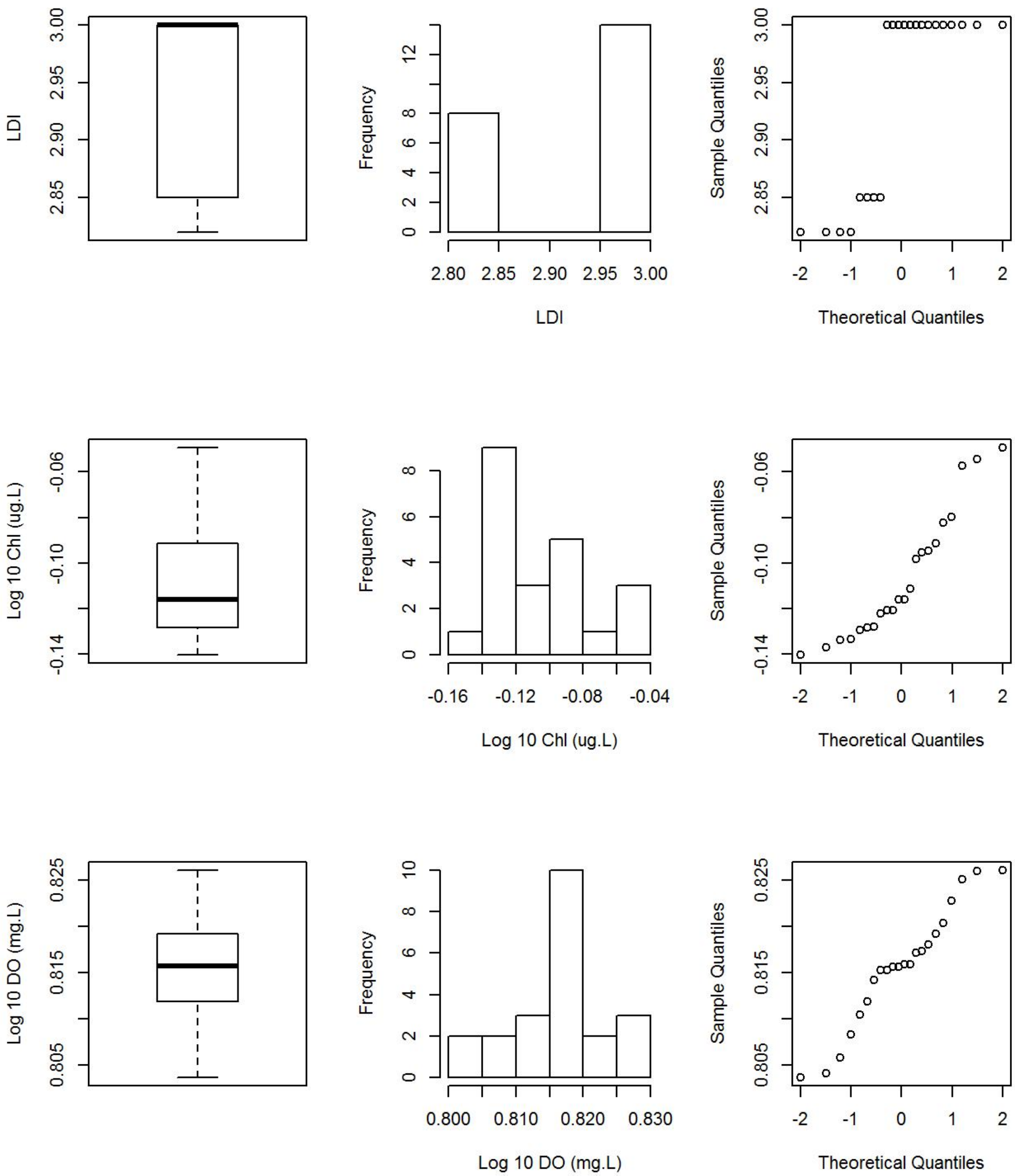


Figure 15-17. Distributions of Landscape Development Index, Log10 Chlorophyll (ug/L), and Dissolved Oxygen (mg/L) values (uM) in Phase 2 coral dataset.

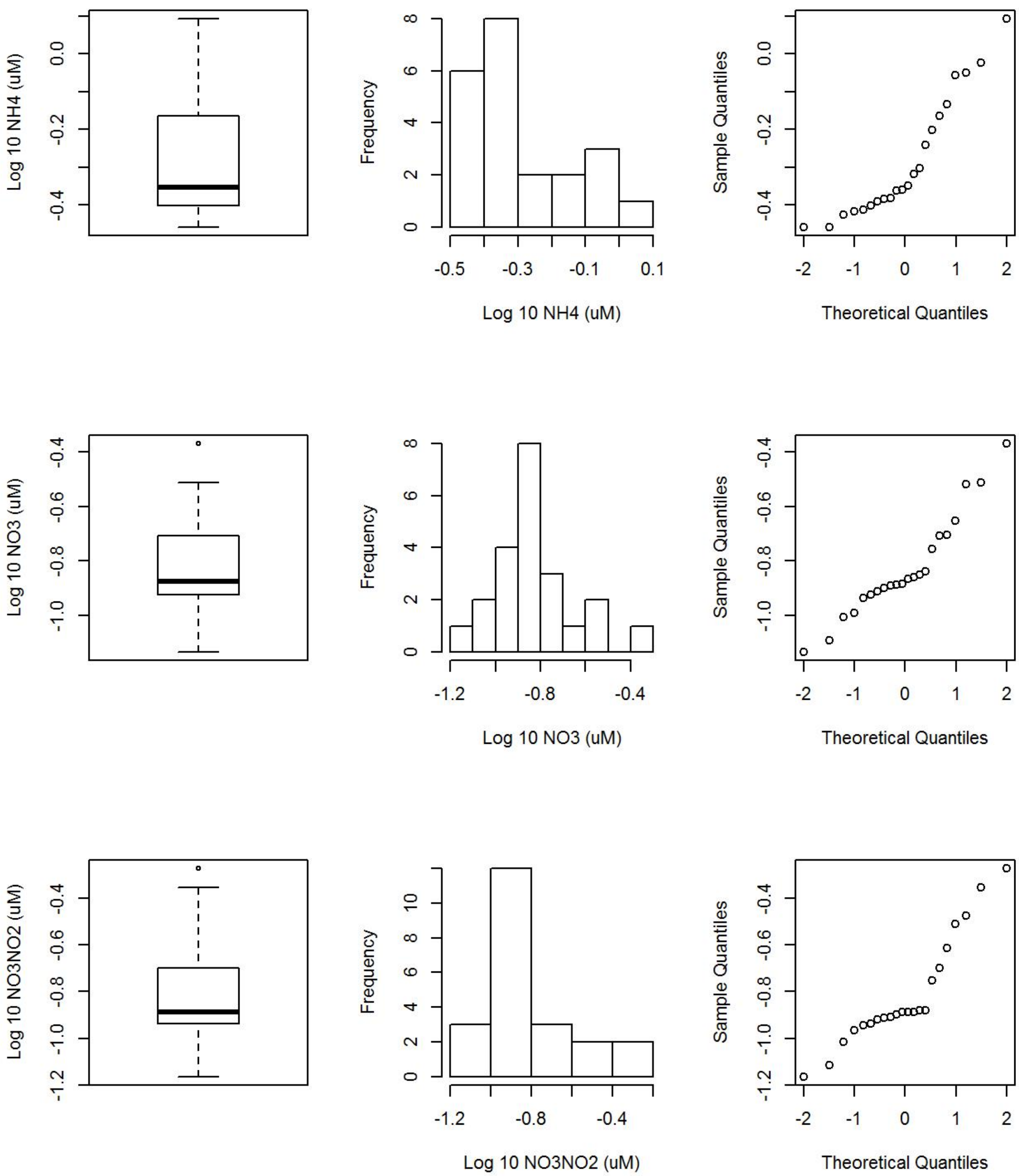


Figure 18-20. Distributions of Ammonium, Nitrate, and Nitrate + Nitrite values (uM) in Phase 2 coral dataset. All Log10 transformed.

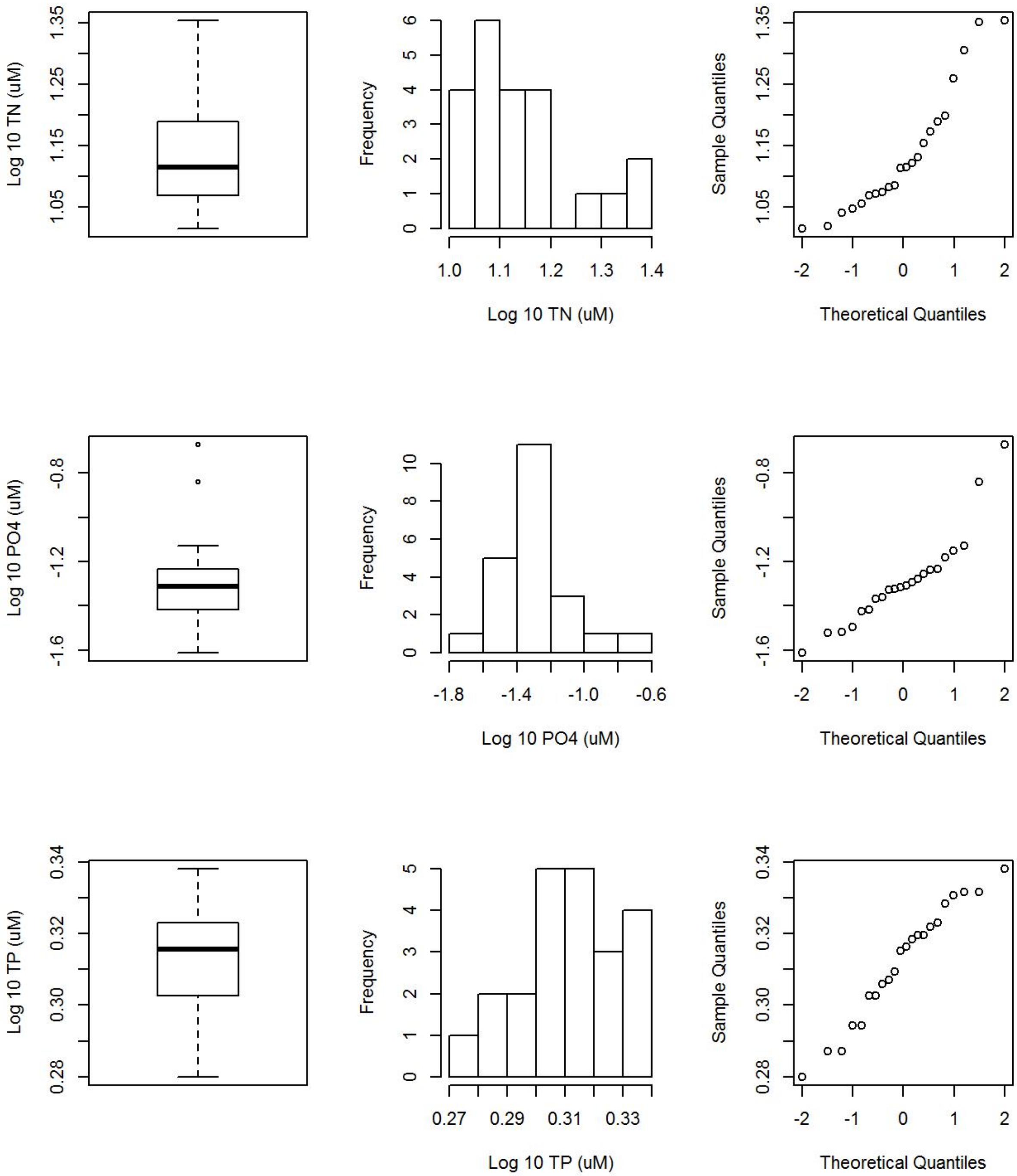


Figure 21-23. Distributions of Total Nitrogen, Phosphate, and Total Phosphorus values (uM) in Phase 2 coral dataset. All Log10 transformed.

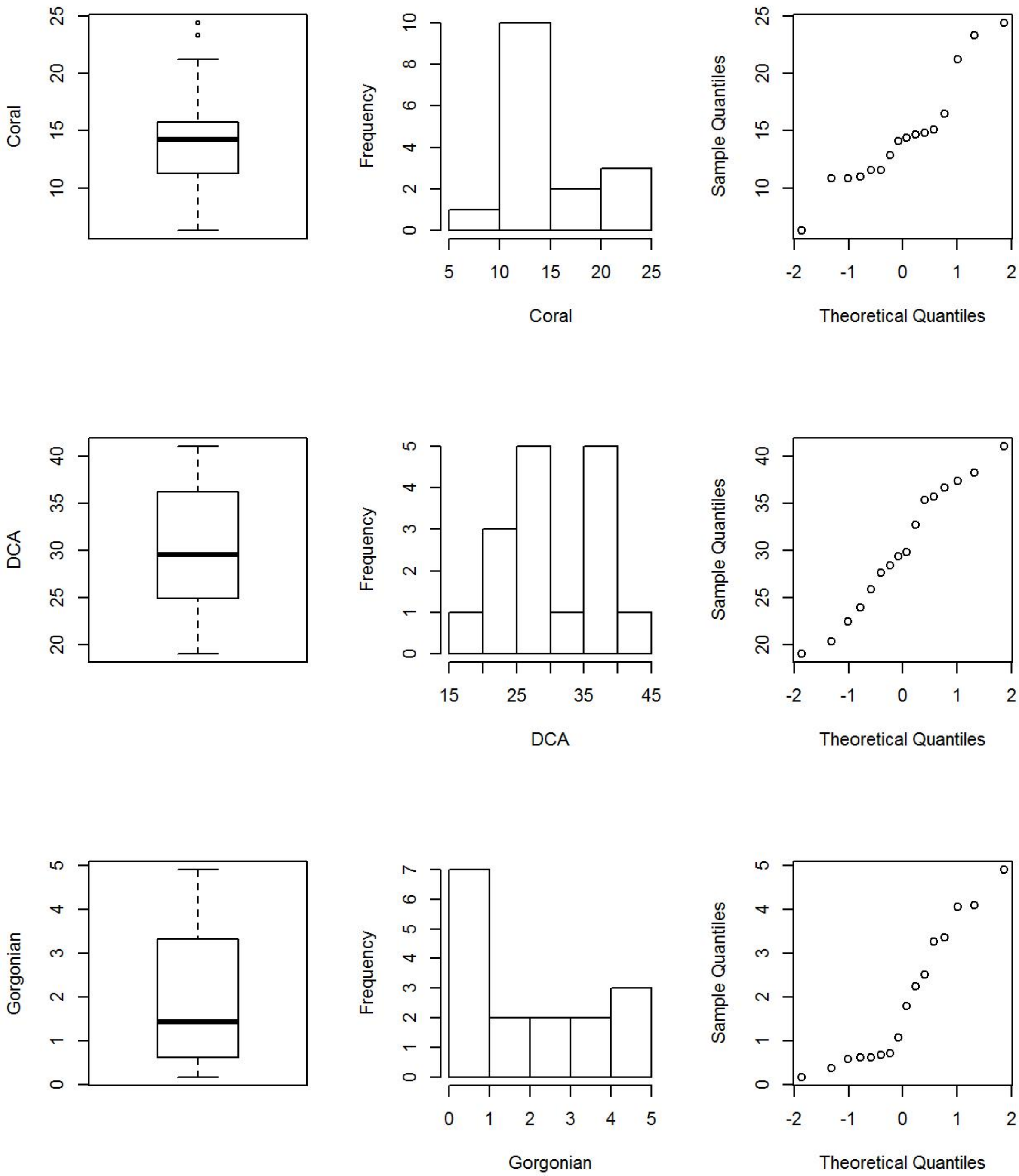


Figure 24-26. Distributions of Coral, DCA, and Gorgonian metric values in Phase 2 coral dataset.

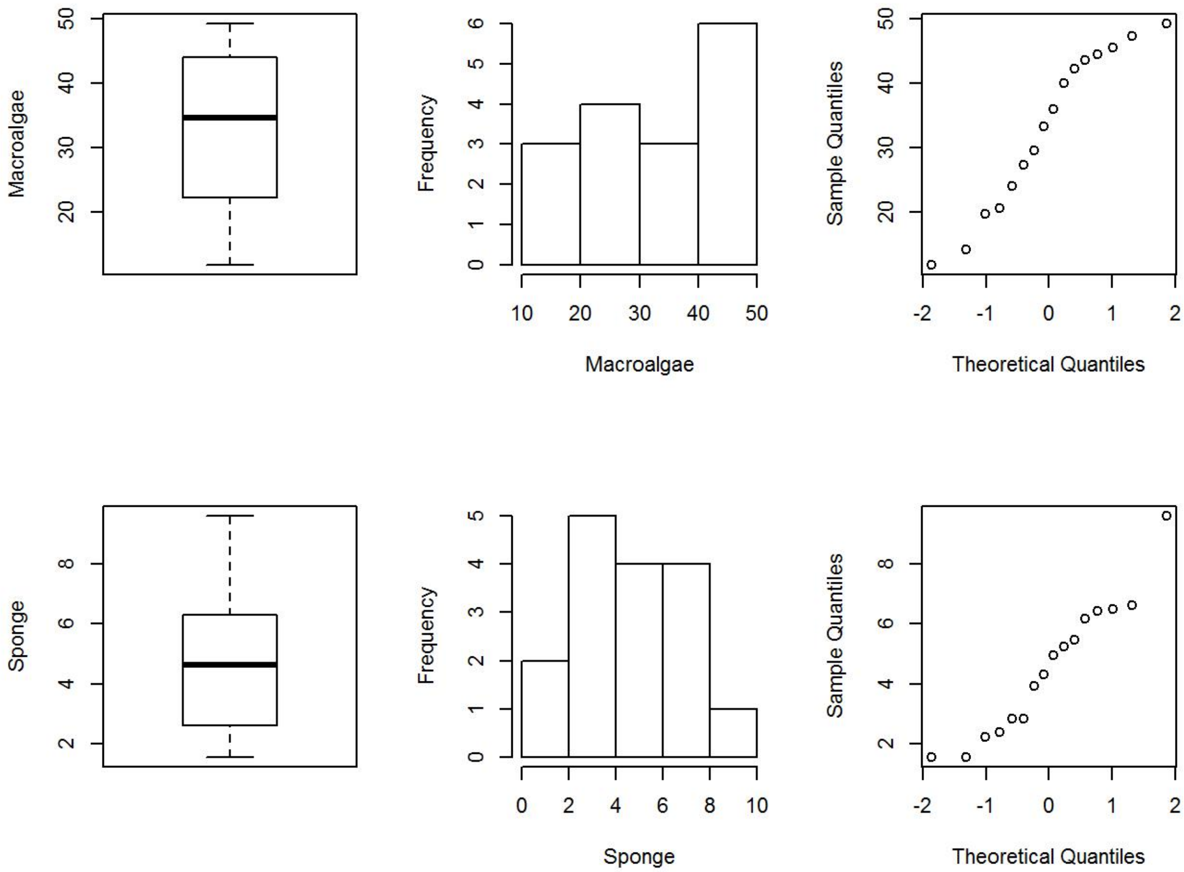


Figure 27-29. Distributions of Macroalgae and Sponge metric values in Phase 2 coral dataset.

Exploratory Statistical Analysis – RARE/NCA Coral Dataset

Exploratory analysis for the full chemistry dataset is shown below. Note that some variables have been log_{10} transformed in order to better meet assumptions of normality. Even after the transformation, some variables do not exhibit clean normal behavior (especially NH4 and TP). Outliers have not been removed, as they may be important in understanding the distribution of water quality parameters. Additional unused variables can be found in The Addendum.

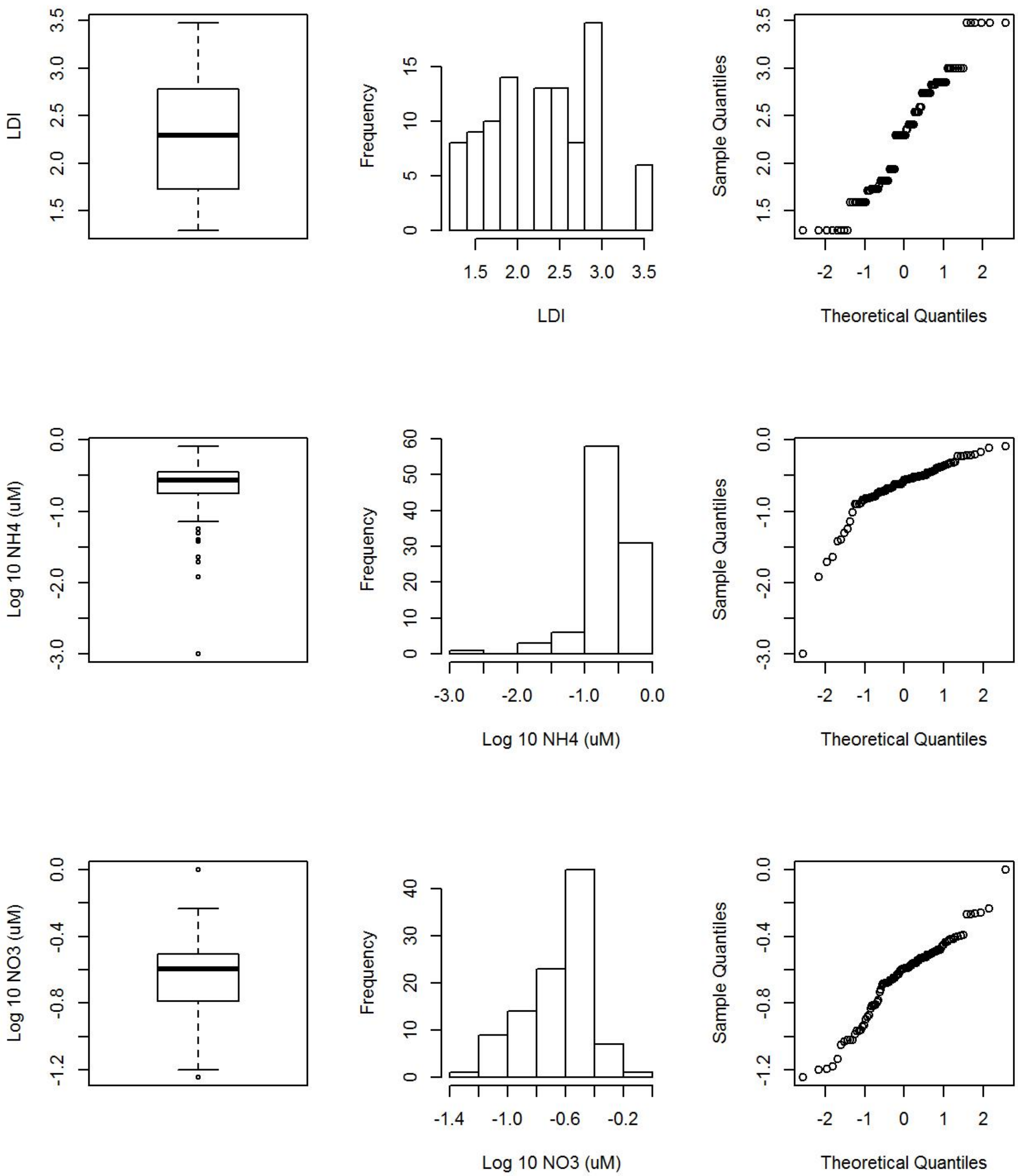


Figure 30-32. Distributions of Landscape Development Index, Ammonium and Nitrate, and Nitrate + Nitrite values (uM) in RARE/NCA coral dataset. Nutrient values Log10 transformed.

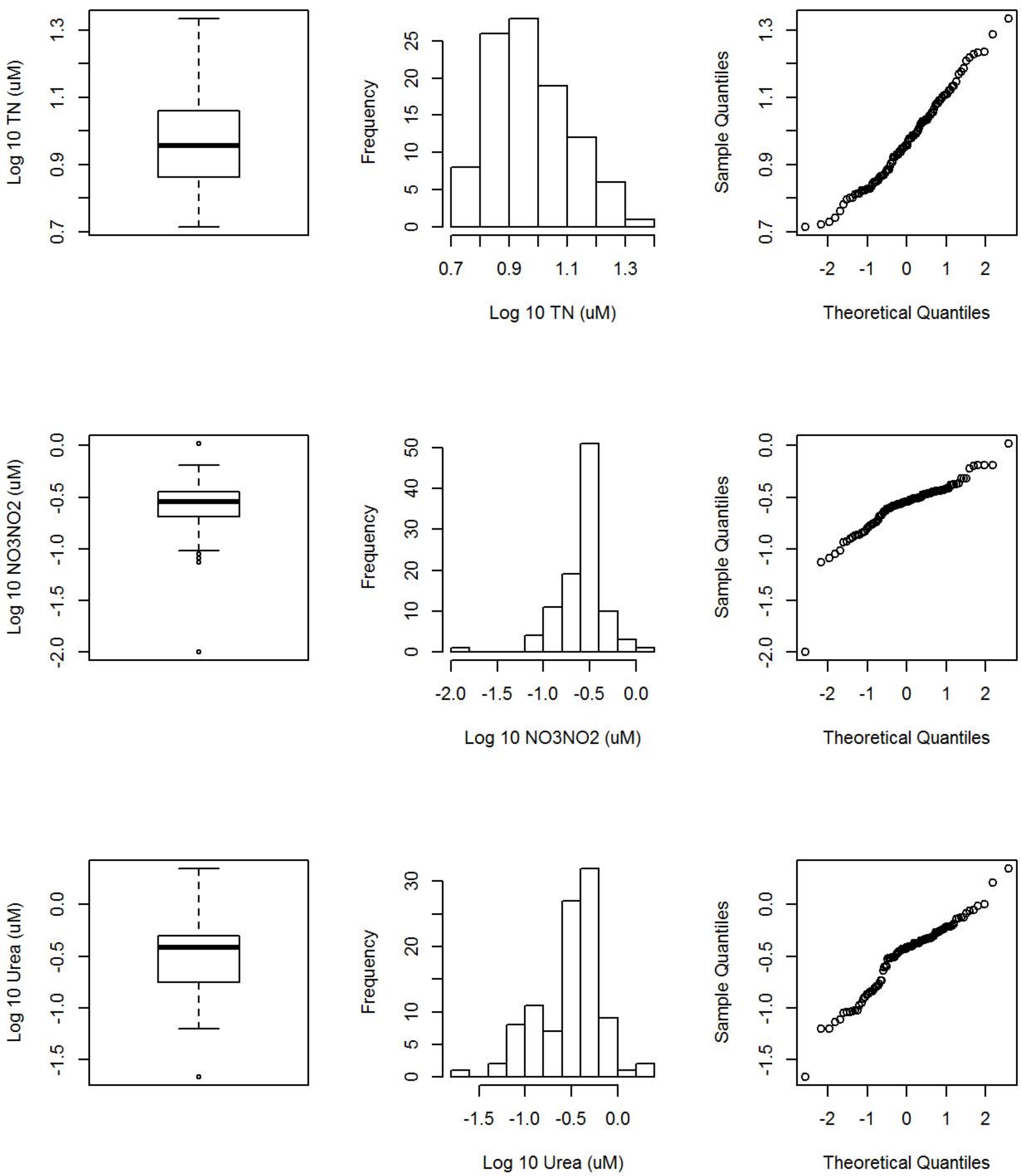


Figure 33-35. Distributions of Nitrate + Nitrite, Total Nitrogen, and Urea values (uM) in RARE/NCA coral dataset. All Log10 transformed.

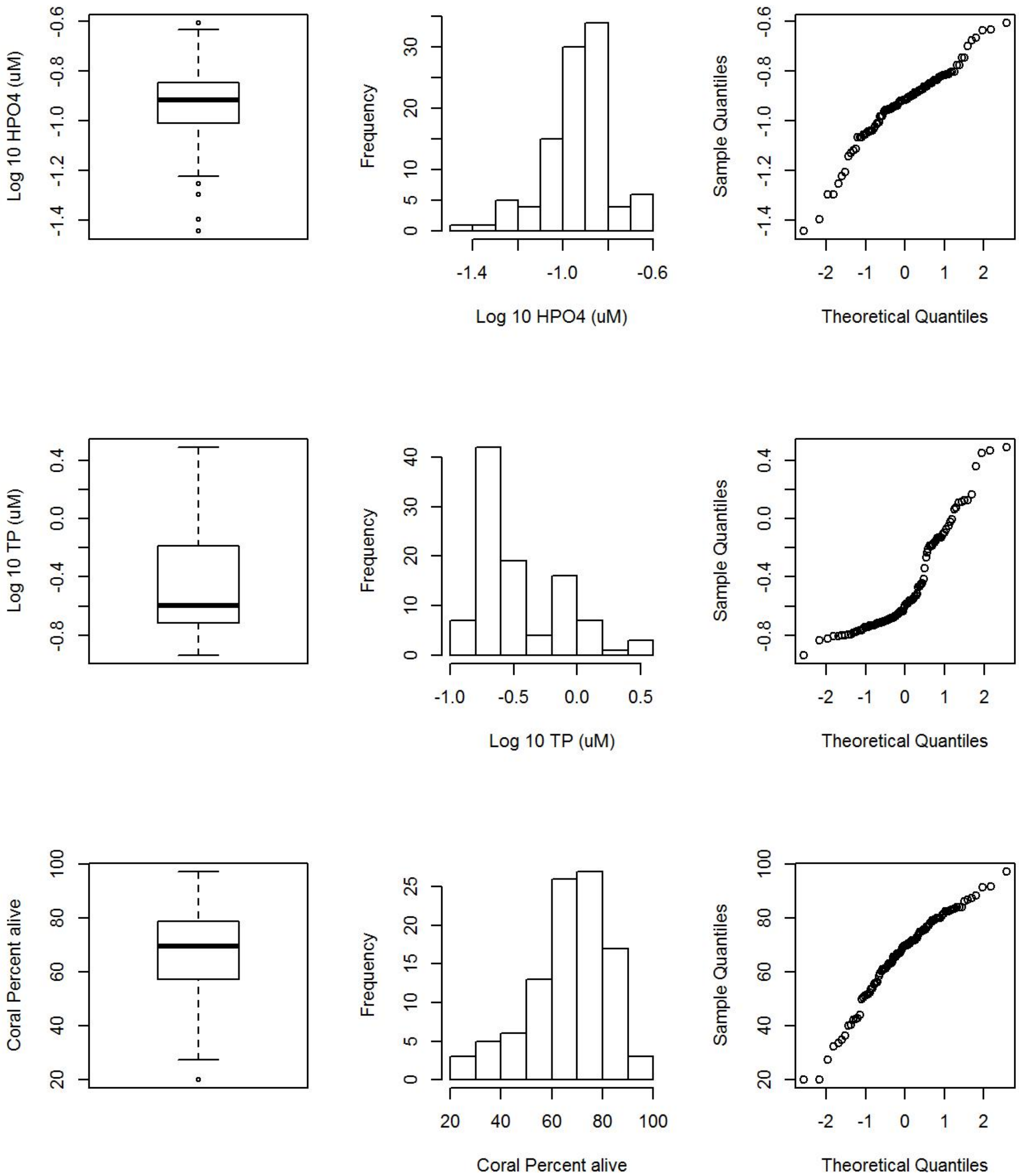


Figure 36-38. Distributions of Phosphate and Total Phosphorus (uM) and Percent of Coral Alive in RARE/NCA coral dataset. Nutrient values Log10 transformed.

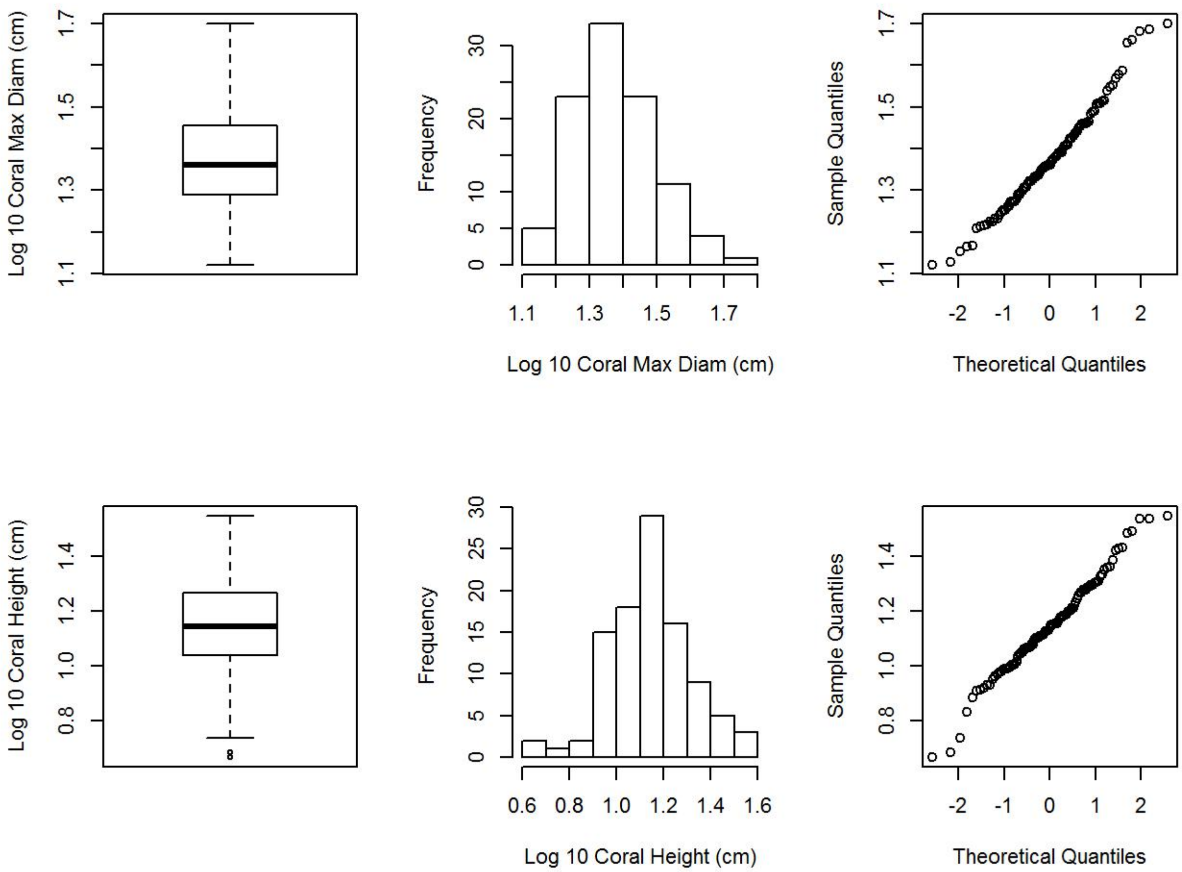


Figure 39-40. Distributions of coral Max Diameter and Height (cm) values in RARE/NCA coral dataset. All Log10 transformed.

Distributional Statistics

Distributional statistics (number of observations, geometric mean, median, and 10th, 25th, 75th, and 90th percentiles) for important parameters in the full chemistry dataset are shown below. Parameters included are the landscape development index, total phosphorus (TP, uM), total nitrogen (TN, uM), ammonium (NH4, uM), nitrate + nitrite (NO3/NO2, uM), phosphate (PO4, uM), chlorophyll (Chl, ug/L), and dissolved oxygen (DO, mg/L).

Statistics were calculated for the entire data set (Table 2), the subset of data within 500 m of shore (Table 3), and the subset of data within 500 m of shore and with LDI < 2 (Table 4). Again, this final category was used as an estimate of conditions least impacted by human associated runoff.

	N	Geometric Mean	10th percentile	25th percentile	Median	75th percentile	90th percentile
LDI	262	2.054	1.590	1.590	1.810	2.740	3.000
TP (uM)	127	0.494	0.165	0.197	0.303	1.339	2.109
TN (uM)	127	9.874	6.513	7.404	9.777	12.360	15.537
NH4 (uM)	178	0.259	0.145	0.235	0.250	0.376	0.584
NO3/NO2 (uM)	189	0.291	0.122	0.174	0.302	0.439	0.821
PO4 (uM)	189	0.082	0.036	0.050	0.094	0.132	0.167
Chl (ug/L)	64	0.369	0.198	0.227	0.286	0.745	0.807
DO (mg/L)	64	6.299	5.921	6.053	6.305	6.541	6.664

Table 2. Distributional statistics for all measurements in the full chemistry dataset.

	N	Geometric Mean	10th percentile	25th percentile	Median	75th percentile	90th percentile
LDI	154	1.880	1.440	1.590	1.710	2.350	2.850
TP (uM)	54	0.412	0.153	0.173	0.229	1.037	2.144
TN (uM)	54	9.366	5.948	7.118	9.411	12.268	14.913
NH4 (uM)	99	0.267	0.155	0.250	0.250	0.340	0.480
NO3/NO2 (uM)	108	0.349	0.131	0.248	0.341	0.604	0.878
PO4 (uM)	108	0.077	0.035	0.045	0.069	0.130	0.159
Chl (ug/L)	43	0.298	0.199	0.216	0.255	0.321	0.766
DO (mg/L)	43	6.252	5.896	6.039	6.176	6.422	6.683

Table 3. Distributional statistics for all measurements within 500 m of shore in the full chemistry dataset.

	N	Geometric Mean	10th percentile	25th percentile	Median	75th percentile	90th percentile
LDI	108	1.612	1.290	1.590	1.590	1.710	1.810

TP (uM)	26	0.366	0.145	0.160	0.223	0.795	1.817
TN (uM)	26	9.071	5.474	6.663	9.194	12.184	14.773
NH4 (uM)	71	0.249	0.156	0.250	0.250	0.250	0.468
NO3/NO2 (uM)	80	0.419	0.174	0.300	0.412	0.699	0.945
PO4 (uM)	80	0.070	0.033	0.042	0.057	0.125	0.171
Chl (ug/L)	37	0.255	0.199	0.214	0.248	0.289	0.344
DO (mg/L)	37	6.193	5.863	6.016	6.122	6.348	6.595

Table 4: Distributional statistics for all measurements within 500 m of shore and closest to catchments with LDI less than 2, in the full chemistry dataset. These are assumed to be minimally impacted sites.

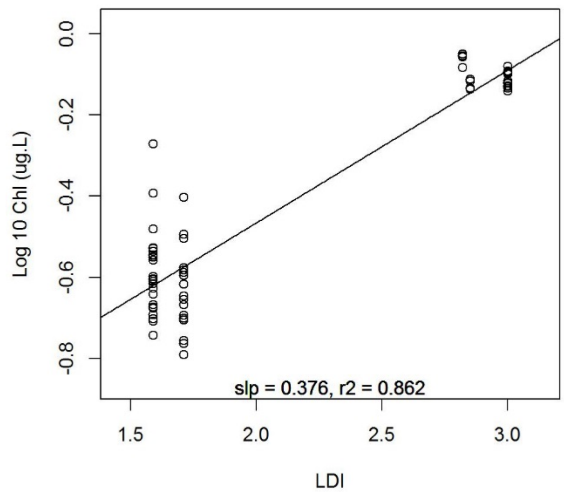
Assuming the near-shore coastal areas draining watersheds with LDI < 2 represent minimally disturbed conditions, the 90th percentiles of TP and TN were 1.82 and 14.8 μM respectively (56.4 and 207 μg/L respectively) or an N:P molar ratio of 8.1 (range 0.7 to 97).

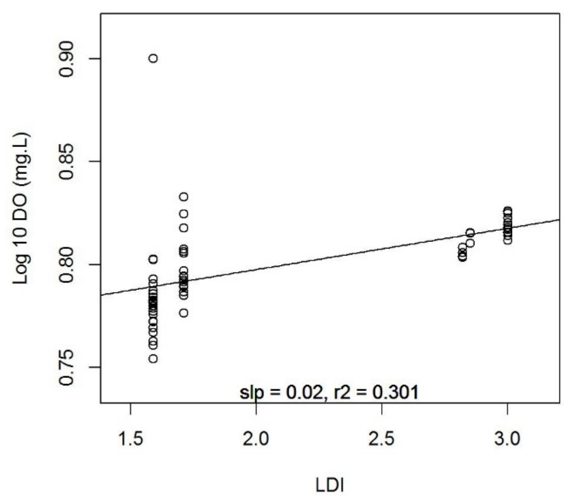
Stressor-response Analysis – Full Chemistry Dataset

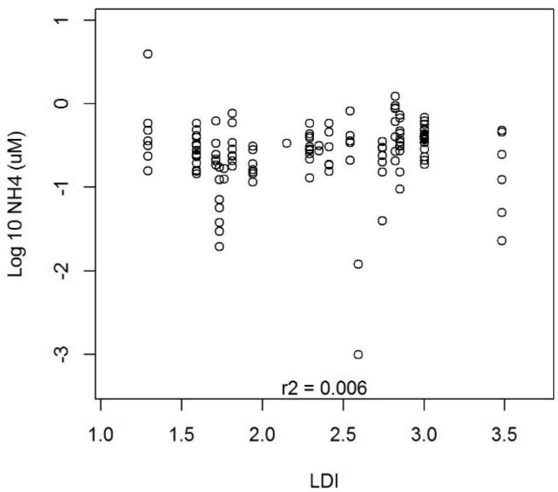
Correlations and linear regression analyses for stressor-response pairs in the full chemistry dataset are shown below. Only statistically significant regression models are displayed ($p < 0.05$), along with the slope of the relationship. The R-squared value is displayed for all relationships. All nutrient concentrations have been $\log_{10}$ transformed.

Statistically significant relationships were as follows. LDI was found to be positively correlated with Chl, DO, TN and TP, and negatively correlated with nitrite (NO2) and NO3/NO2. NH4 and TN were found to be positively correlated with Chl, while nitrate (NO3) and NO3/NO2 were negatively correlated with Chl. NH4 and TP were found to be positively correlated with DO, while NO3, NO3/NO2, and TN were negatively correlated with DO. Chl was positively correlated with DO. The significant correlations were of varying strength, with R-squared values ranging from 0.044 (LDI and TN) to 0.862 (LDI and Chl).

The final figure in this dataset explores the relationship between TN and Chl for sites within 500m of shore. No significant relationship was identified for this plot using long-term average TN and Chl; however, a significant negative relationship was identified using individual grab pairs, strongly influenced by samples at the detection limit for TN. Without those samples, the relationship is non-significant.


Log 10 Chl (ug.L)
0.0
-0.2
-0.4
-0.6
-0.8
slp = 0.376, r2 = 0.862
1.5
2.0
2.5
3.0
LDI


Log 10 DO (mg.L)
0.90
0.85
0.80
0.75
slp = 0.02, r2 = 0.301
1.5
2.0
2.5
3.0
LDI



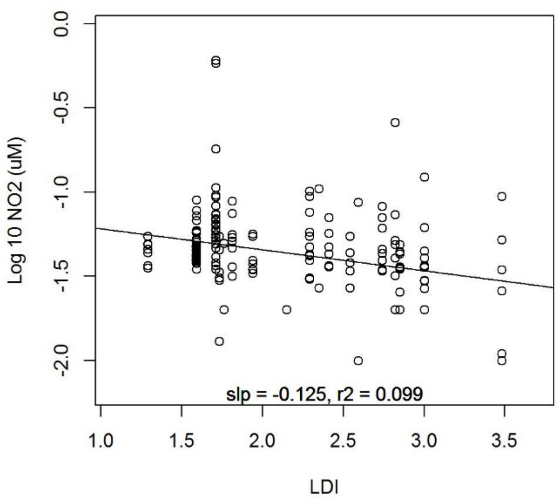
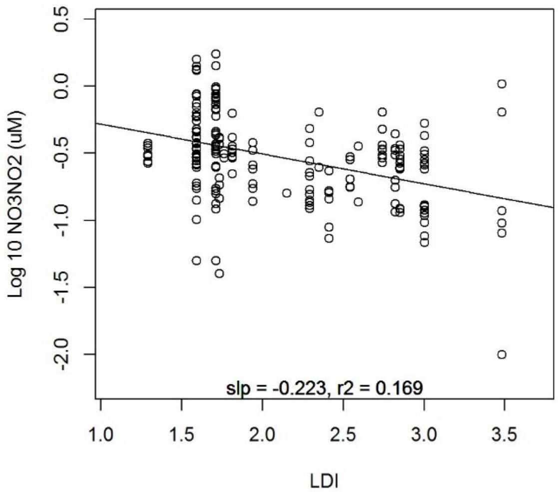
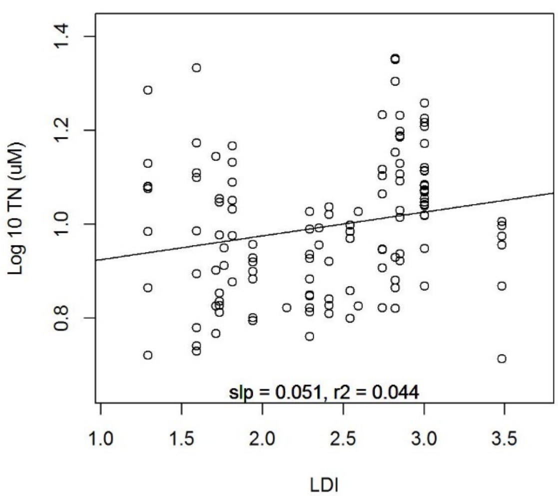
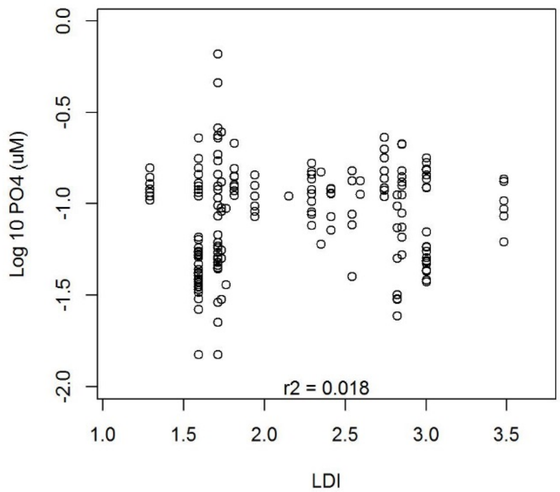


Figure 41-44. Relationships between Landscape Development Index and Chlorophyll (ug/L), Dissolved Oxygen (mg/L), Ammonium (uM), and Nitrite (uM) in the full chemistry dataset. All nutrient values Log 10 transformed.


Log 10 NO3NO2 (uM)
LDI
slp = -0.223, r2 = 0.169


Log 10 TN (uM)
LDI
slp = 0.051, r2 = 0.044



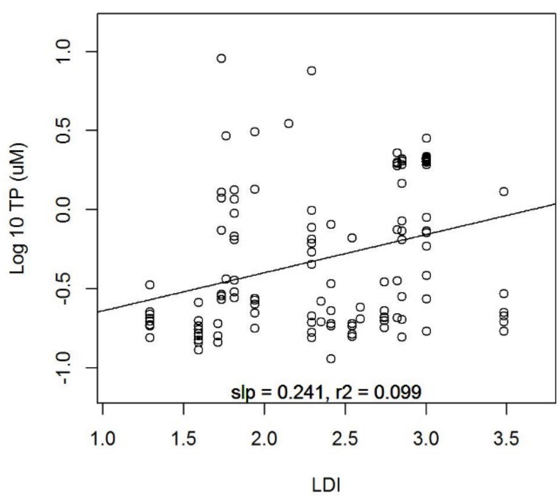
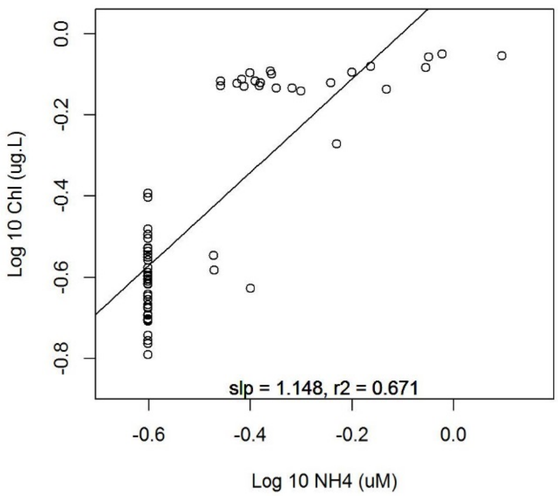
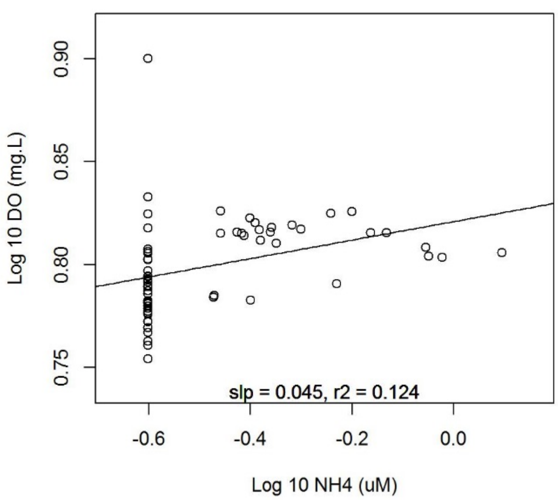
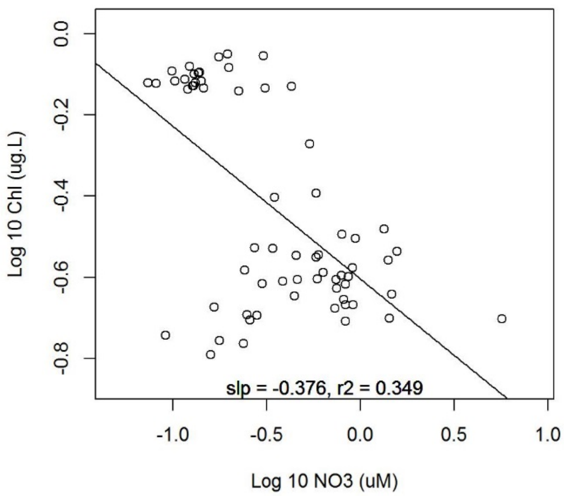


Figure 45-48. Relationships between Landscape Development Index and Nitrate + Nitrite, Total N, Phosphate, and Total P (uM) in the full chemistry dataset. All nutrient values Log 10 transformed.


Log 10 Chl (ug.L)
0.0
-0.2
-0.4
-0.6
-0.8
slp = 1.148, r2 = 0.671
-0.6
-0.4
-0.2
0.0
Log 10 NH4 (uM)


Log 10 DO (mg.L)
0.90
0.85
0.80
0.75
slp = 0.045, r2 = 0.124
-0.6
-0.4
-0.2
0.0
Log 10 NH4 (uM)



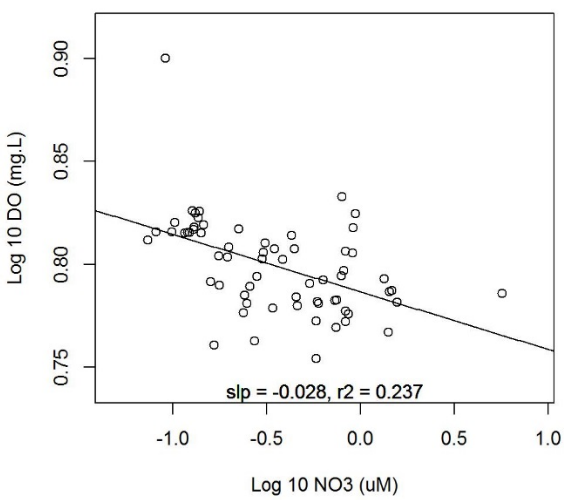
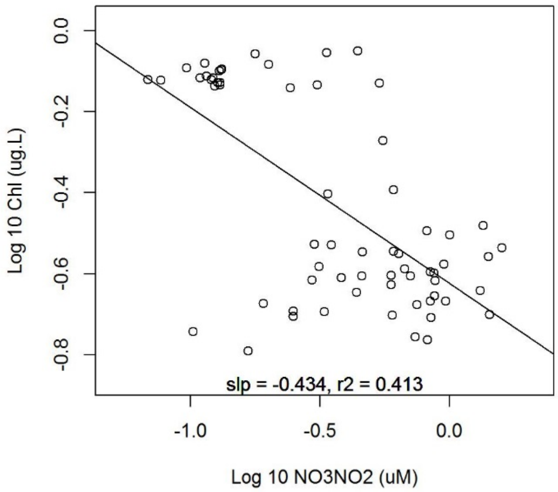
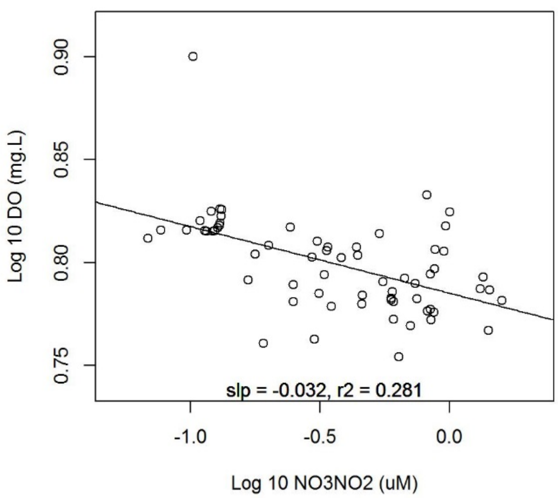
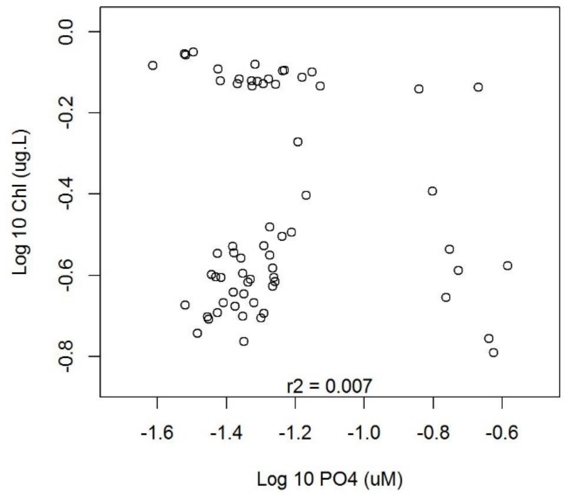


Figure 49-52. Relationships between Ammonium and Nitrite (uM) and Chlorophyll (ug/L) and Dissolved Oxygen (mg/L) in the full chemistry dataset. All nutrient values Log 10 transformed.


Log 10 Chl (ug.L)
0.0
-0.2
-0.4
-0.6
-0.8
slp = -0.434, r2 = 0.413
-1.0
-0.5
0.0
Log 10 NO3NO2 (uM)


Log 10 DO (mg.L)
0.90
0.85
0.80
0.75
slp = -0.032, r2 = 0.281
-1.0
-0.5
0.0
Log 10 NO3NO2 (uM)



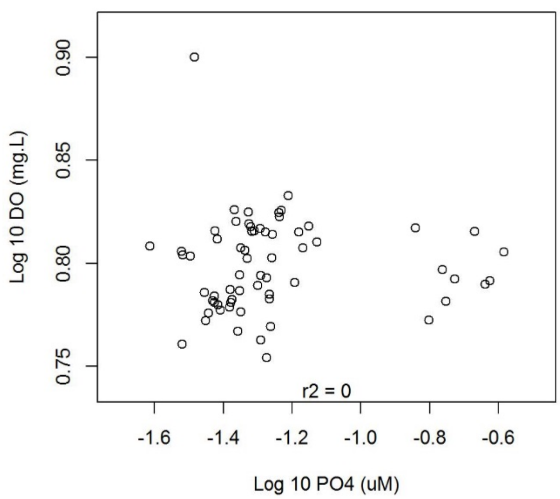
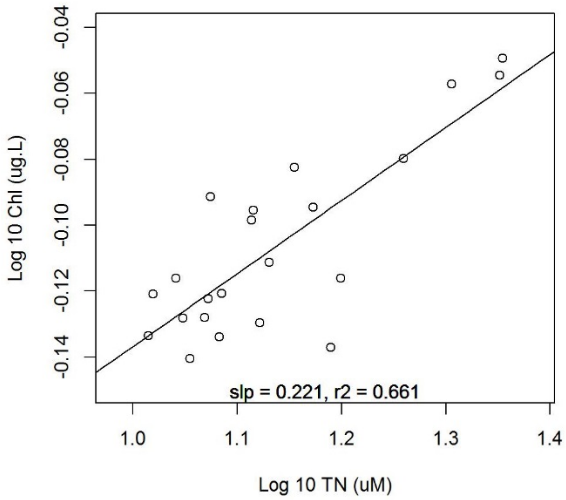
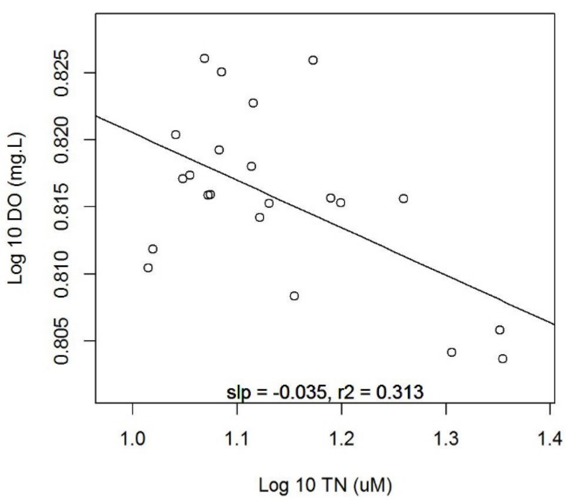


Figure 53-56. Relationships between Nitrate + Nitrite and Phosphate (uM) and Chlorophyll (ug/L) and Dissolved Oxygen (mg/L) in the full chemistry dataset. All nutrient values Log 10 transformed.


Log 10 Chl (ug.L)
-0.04
-0.06
-0.08
-0.10
-0.12
-0.14
slp = 0.221, r2 = 0.661
1.0
1.1
1.2
1.3
1.4
Log 10 TN (uM)


Log 10 DO (mg.L)
0.825
0.820
0.815
0.810
0.805
slp = -0.035, r2 = 0.313
1.0
1.1
1.2
1.3
1.4
Log 10 TN (uM)

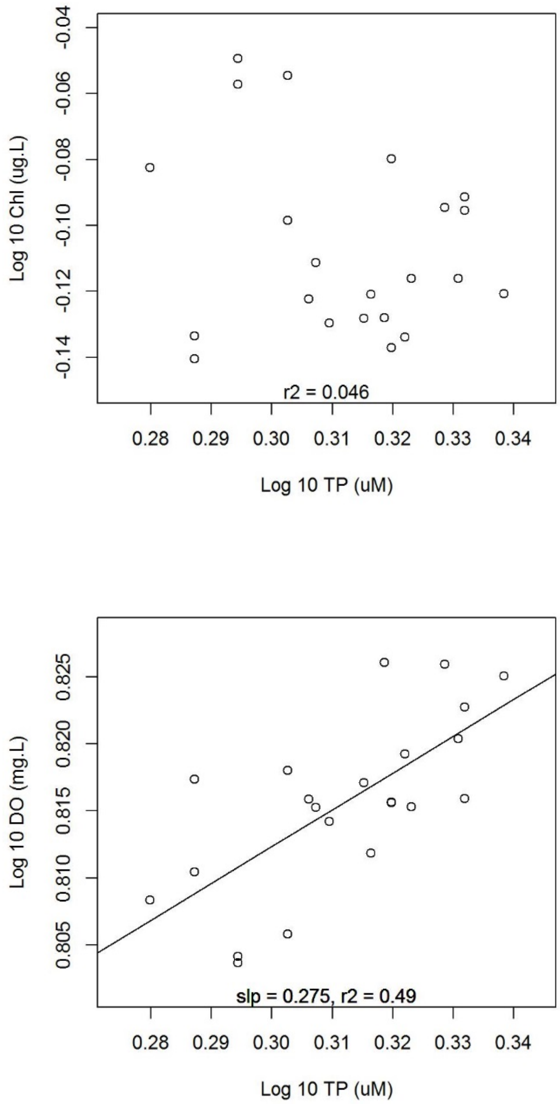


Figure 57-60. Relationships between Total N and Total P (uM) and Chlorophyll (ug/L) and Dissolved Oxygen (mg/L) in the full chemistry dataset. All nutrient values Log 10 transformed.

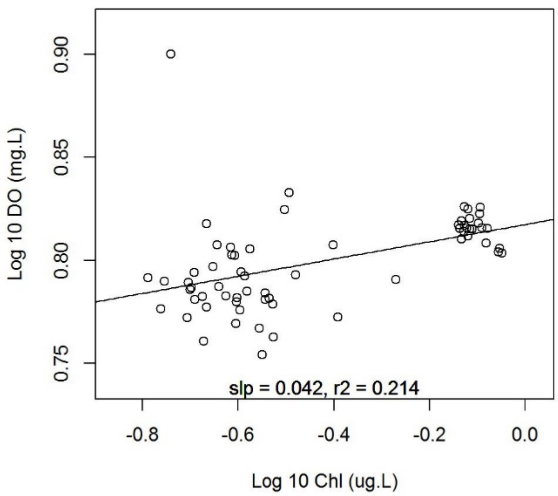


Figure 61. Relationship Chlorophyll (ug/L) and Dissolved Oxygen (mg/L) in the full chemistry dataset. All nutrient values Log 10 transformed.

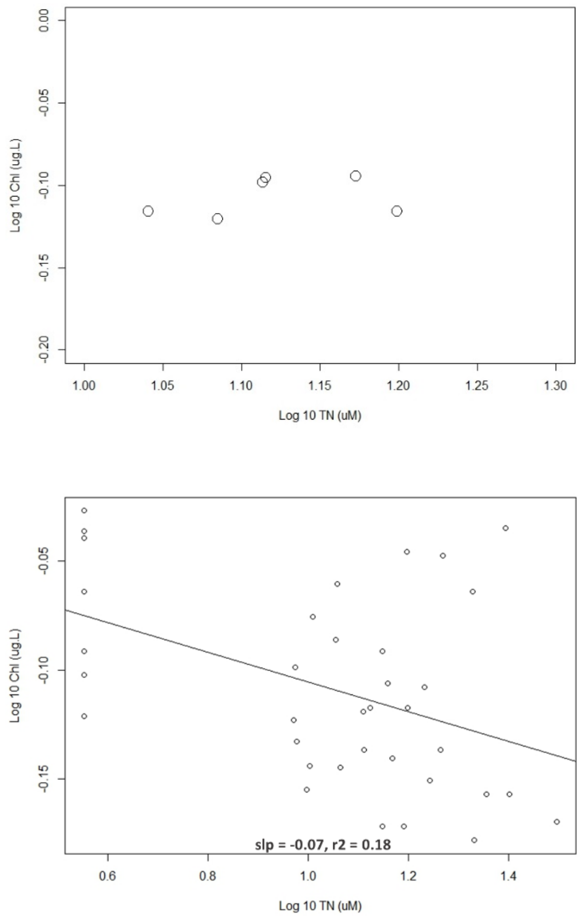
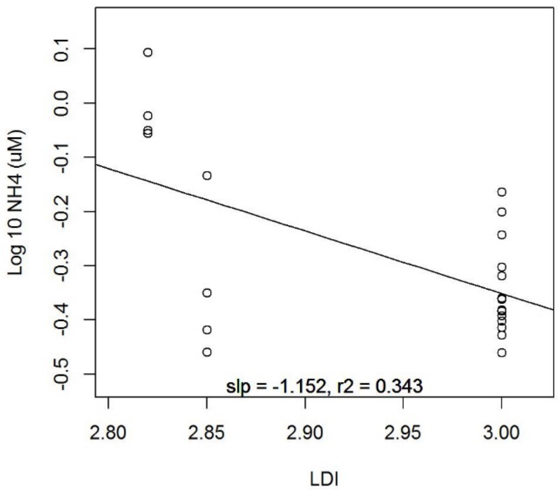


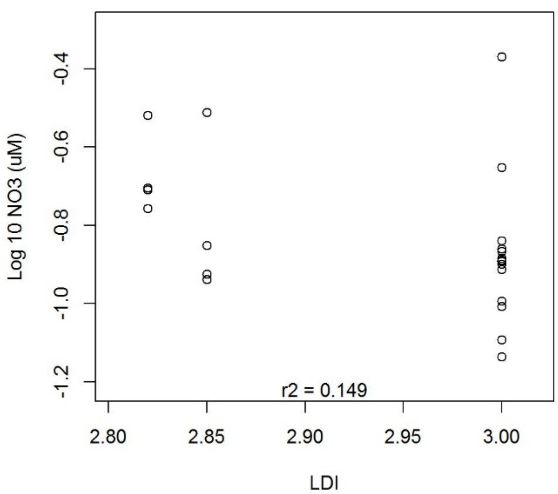
Figure 62. Relationship between Total N (uM) and Chlorophyll (ug/L) in the full chemistry dataset, using only sites within 500m. Figure on the left is long-term average and figure on the right is for individual grab samples.

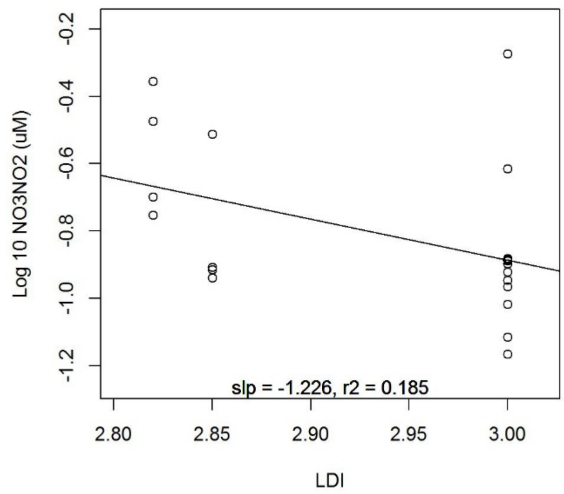
Stressor-response Analysis – Phase 2 Coral Dataset

Correlations and linear regression analyses for stressor-response pairs in the Phase 2 Coral dataset are shown below. Only statistically significant regression models are displayed ($p < 0.05$), along with the slope of the relationship. The R-squared value is displayed for all relationships. All nutrient concentrations have been Log 10 transformed.

Statistically significant relationships were as follows. LDI was found to be positively correlated with TP and DO, and negatively correlated with NH4, NO3/NO2, TN, and Chl. LDI was not significantly related to any of the 5 biotic metrics. DO was found to be negatively correlated with Chl. NH4 was positively correlated with Chl and Gorgonian, and negatively correlated with DO. NO3 was negatively correlated with coral. NO3/NO2 was negatively correlated with DO. PO4 was positively correlated with Coral and negatively correlated with Chl and Gorgonian. TN was positively correlated with Chl, DO, and Gorgonian. TP was positively correlated with DO and Coral. The significant correlations were of varying strength, with R-squared values ranging from 0.212 (LDI and Chl) to 0.682 (Chl and TN). The significant correlations with biotic metrics were generally weaker, ranging from 0.260 (TP and Coral) to 0.374 (NH4 and Gorgonian).







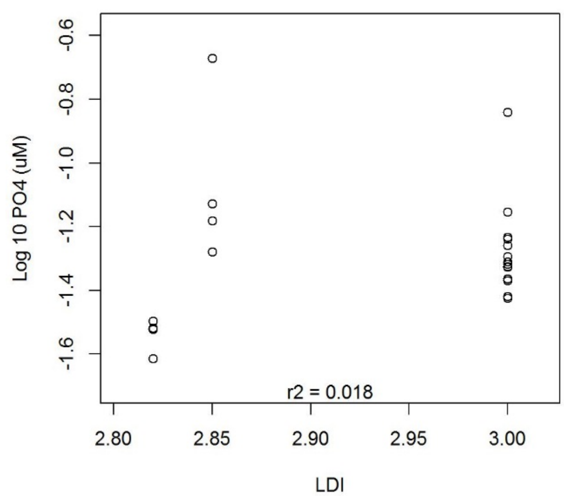
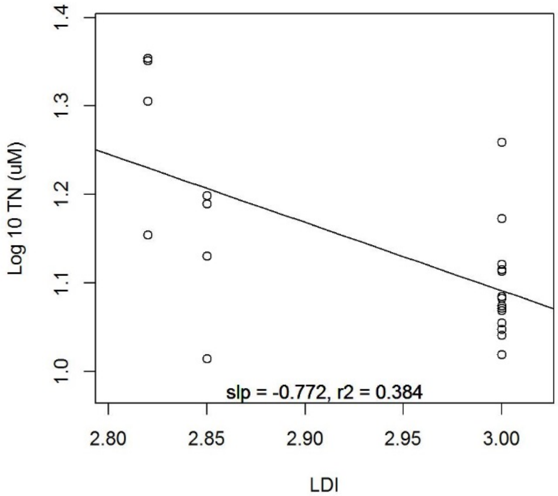
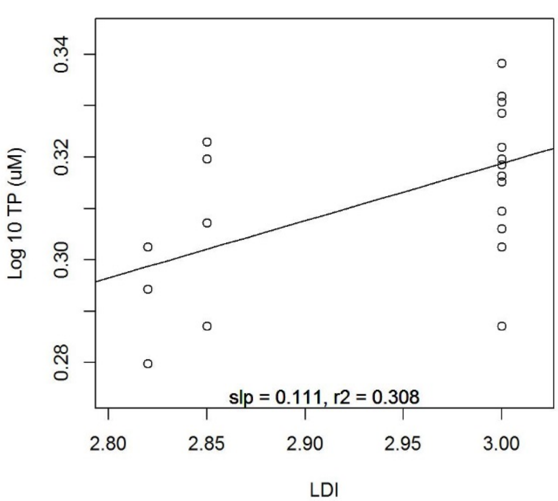
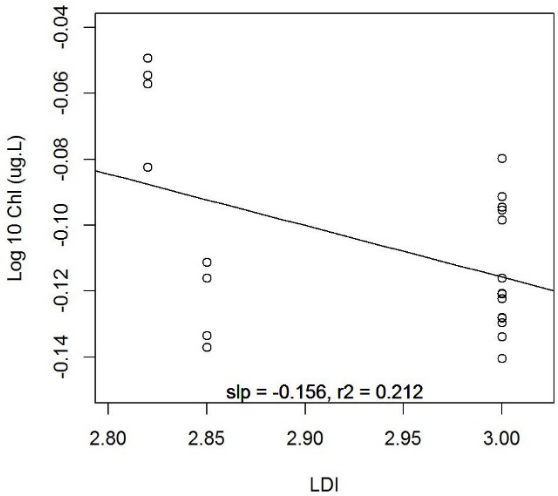


Figure 63-66. Relationship between Landscape Development Index and Ammonium, Nitrate, Nitrate + Nitrite, and Phosphate (uM) in the Phase 2 coral dataset.


Log 10 TN (uM)
1.0
1.1
1.2
1.3
1.4
slp = -0.772, r2 = 0.384
2.80
2.85
2.90
2.95
3.00
LDI


Log 10 TP (uM)
0.28
0.30
0.32
0.34
slp = 0.111, r2 = 0.308
2.80
2.85
2.90
2.95
3.00
LDI



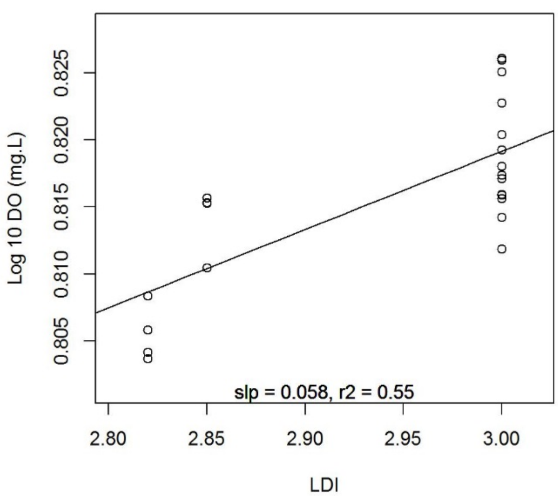
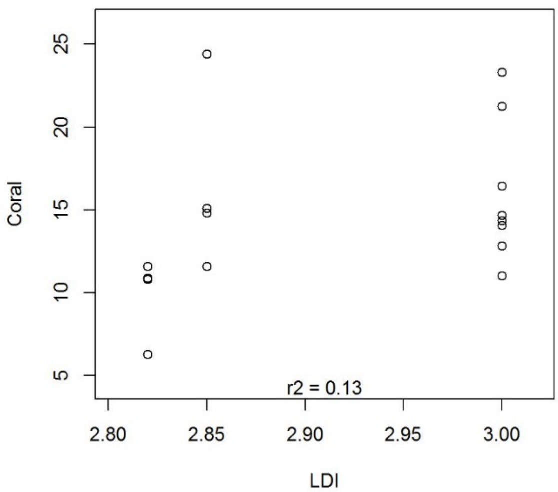
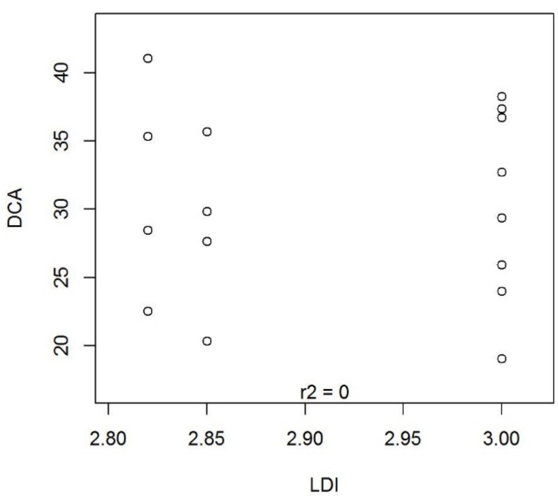
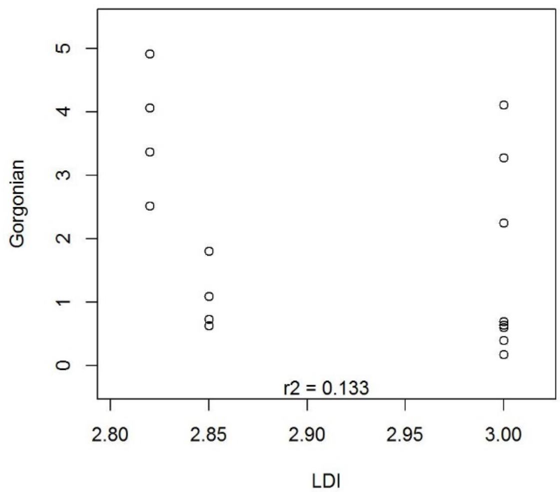


Figure 67-70. Relationship between Landscape Development Index and Total N and Total P (uM), Chlorophyll (ug/L), and Dissolved Oxygen (mg/L) in the Phase 2 coral dataset.


Coral
25
20
15
10
5
r2 = 0.13
2.80
2.85
2.90
2.95
3.00
LDI


DCA
40
35
30
25
20
r2 = 0
2.80
2.85
2.90
2.95
3.00
LDI



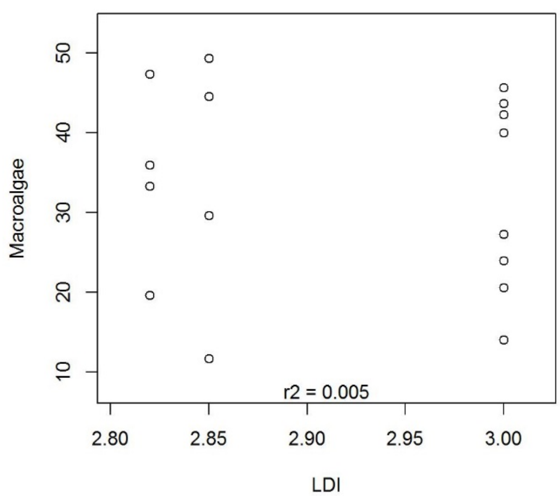
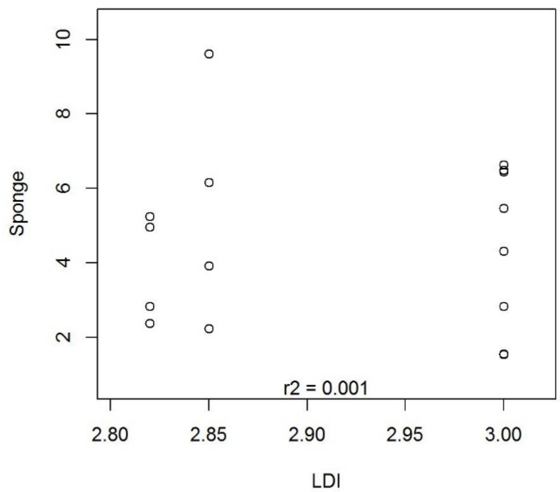


Figure 71-74. Relationship between Landscape Development Index and the Coral, DCA, Gorgonian, and Macroalgae metrics in the Phase 2 coral dataset.



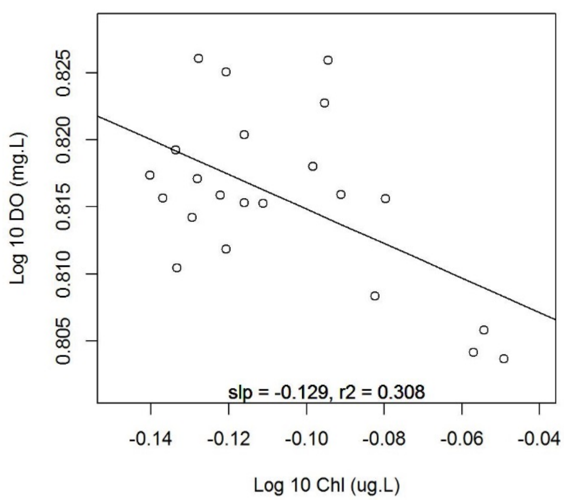
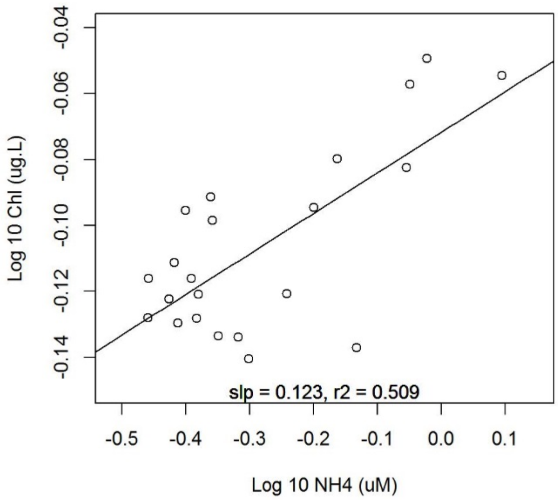
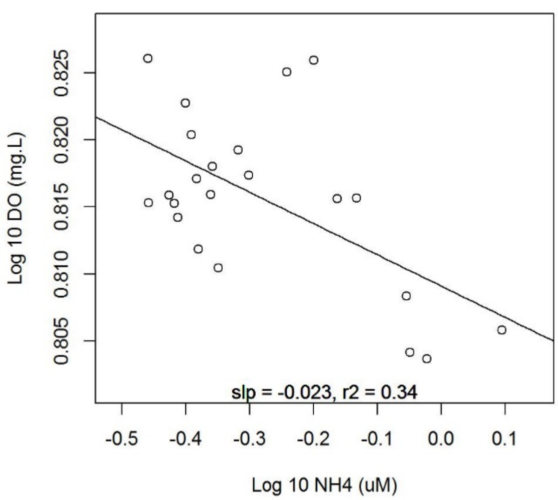
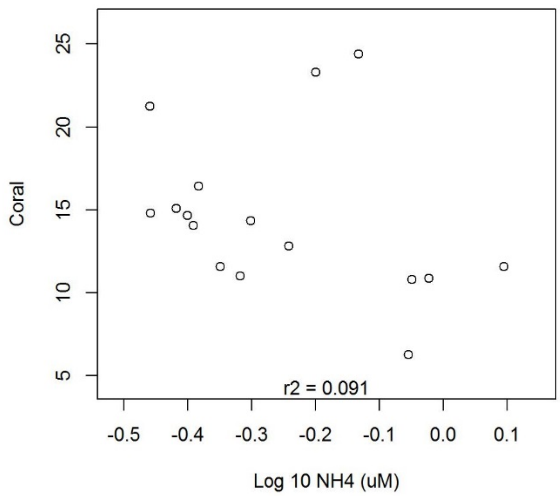


Figure 75-76. Relationship between Landscape Development Index and the Sponge metric, and between Chlorophyll (ug/L) and Dissolved Oxygen (mg/L) in the Phase 2 coral dataset.


Log 10 Chl (ug.L)
-0.04
-0.06
-0.08
-0.10
-0.12
-0.14
slp = 0.123, r2 = 0.509
-0.5
-0.4
-0.3
-0.2
-0.1
0.0
0.1
Log 10 NH4 (uM)


Log 10 DO (mg.L)
0.825
0.820
0.815
0.810
0.805
slp = -0.023, r2 = 0.34
-0.5
-0.4
-0.3
-0.2
-0.1
0.0
0.1
Log 10 NH4 (uM)



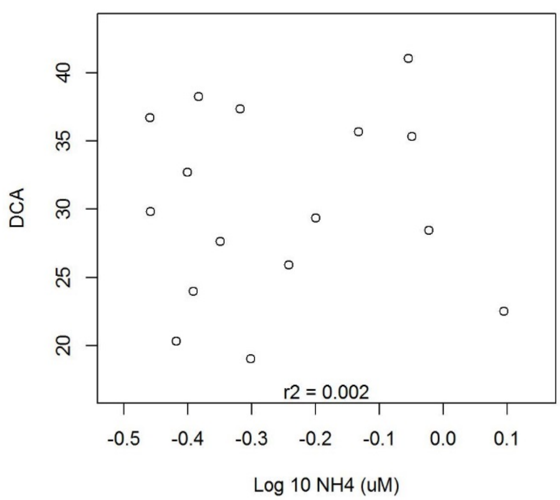
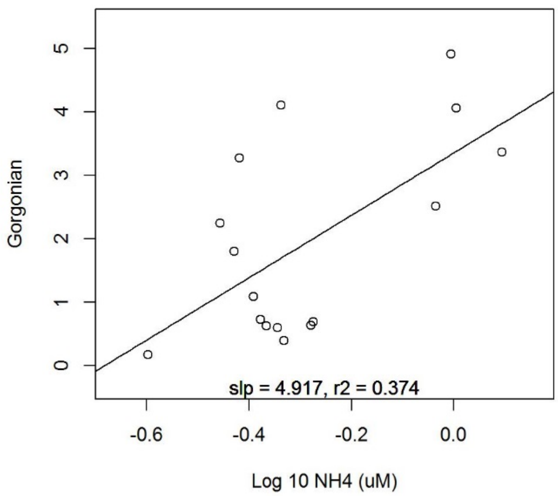
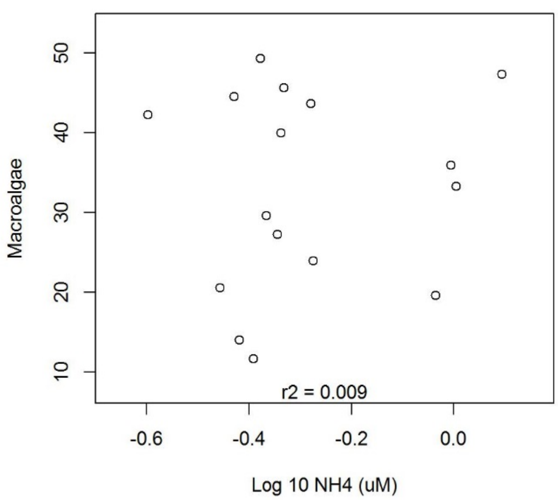


Figure 77-80. Relationship between Ammonium (uM) and Chlorophyll (ug/L), Dissolved Oxygen (mg/L), and the Coral and DCA in the Phase 2 coral dataset.


Gorgonian
slp = 4.917, r2 = 0.374
Log 10 NH4 (uM)


Macroalgae
r2 = 0.009
Log 10 NH4 (uM)

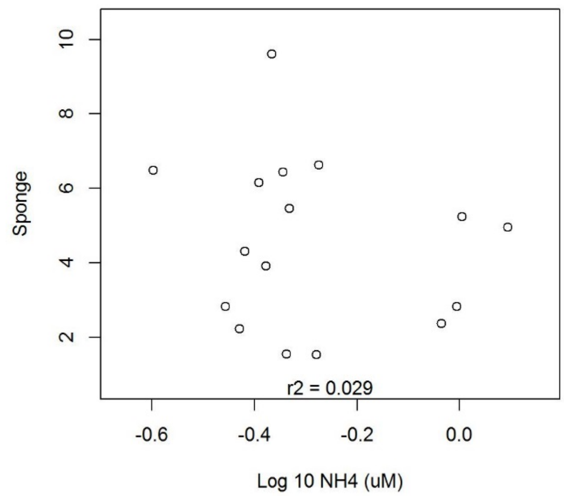
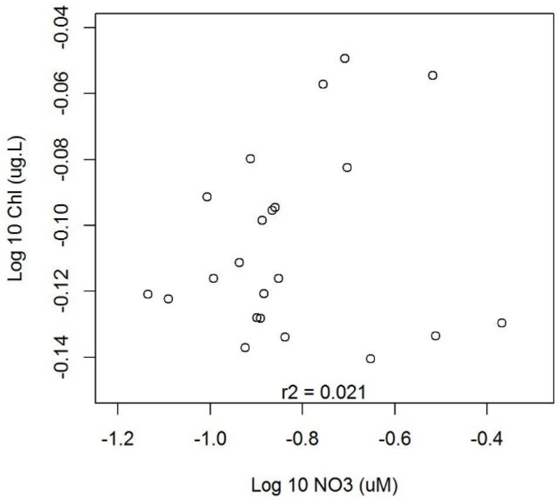
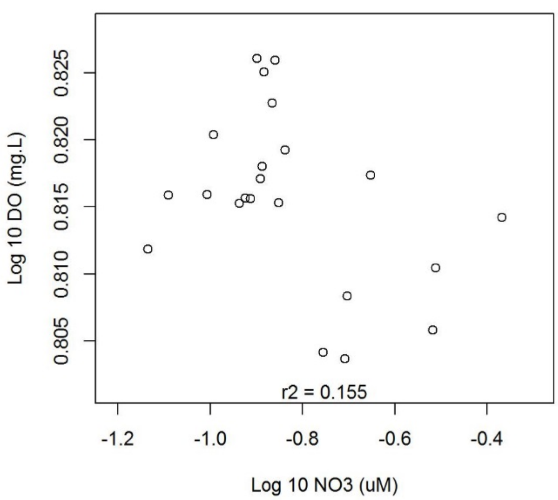
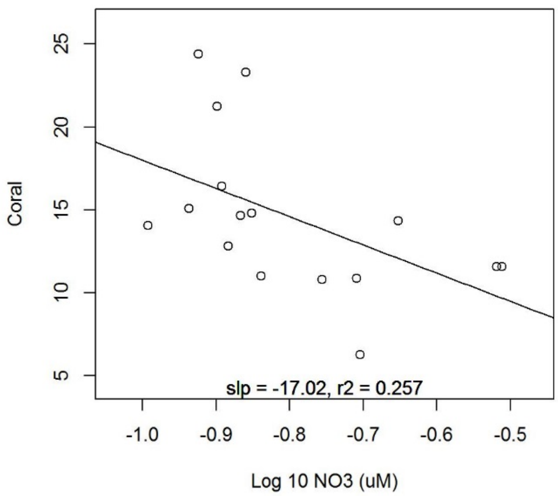


Figure 81-83. Relationship between Ammonium (uM) and the Gorgonian, Macroalgae, and Sponge metrics in the Phase 2 coral dataset.


Log 10 Chl (ug.L)
-0.04
-0.06
-0.08
-0.10
-0.12
-0.14
r2 = 0.021
-1.2
-1.0
-0.8
-0.6
-0.4
Log 10 NO3 (uM)


Log 10 DO (mg.L)
0.825
0.820
0.815
0.810
0.805
r2 = 0.155
-1.2
-1.0
-0.8
-0.6
-0.4
Log 10 NO3 (uM)



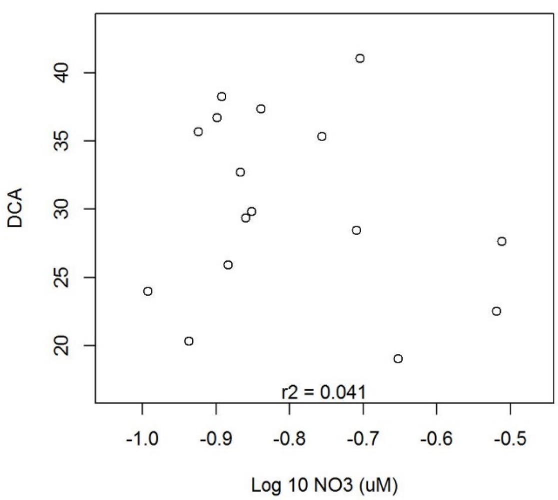
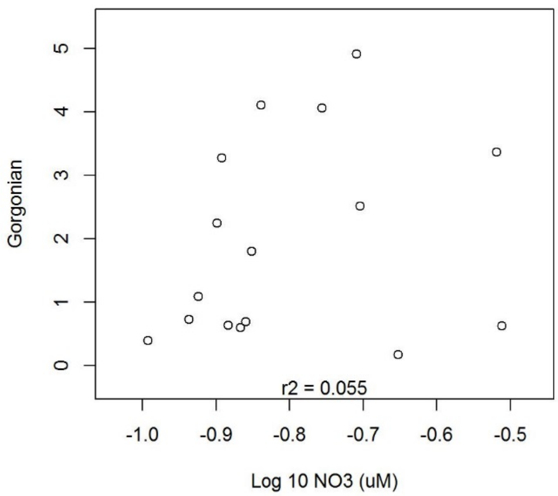
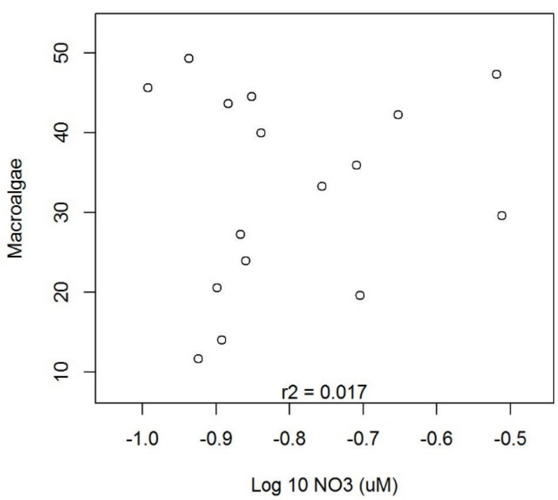


Figure 84-87. Relationship between Nitrate (uM) and Chlorophyll (ug/L), Dissolved Oxygen (mg/L), and the Coral and DCA in the Phase 2 coral dataset.


Gorgonian
5
4
3
2
1
0
r2 = 0.055
-1.0
-0.9
-0.8
-0.7
-0.6
-0.5
Log 10 NO3 (uM)


Macroalgae
50
40
30
20
10
r2 = 0.017
-1.0
-0.9
-0.8
-0.7
-0.6
-0.5
Log 10 NO3 (uM)

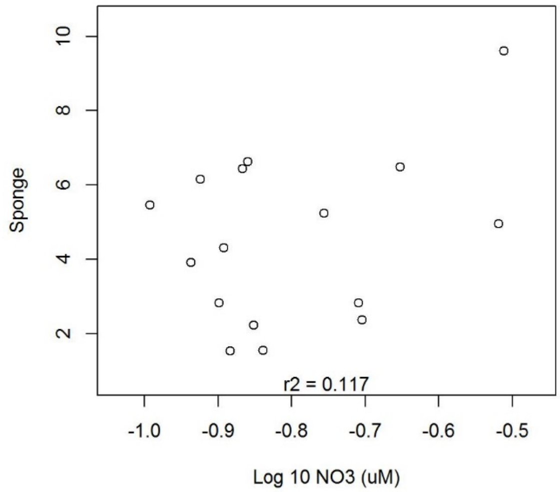
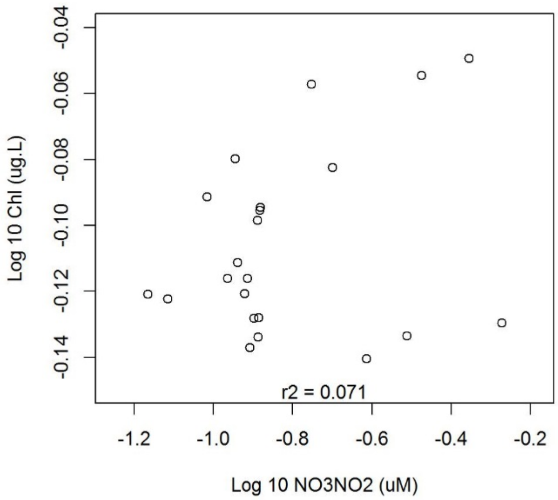
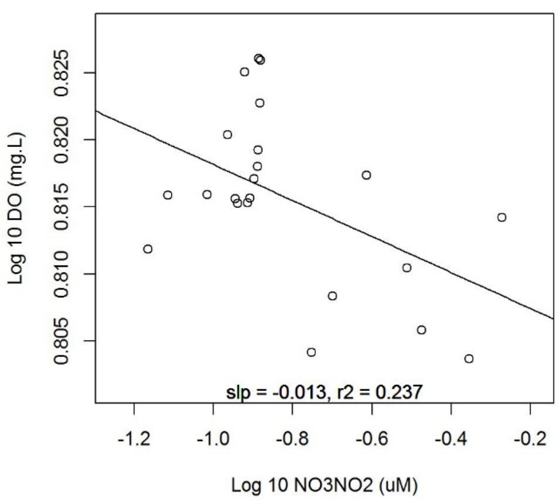
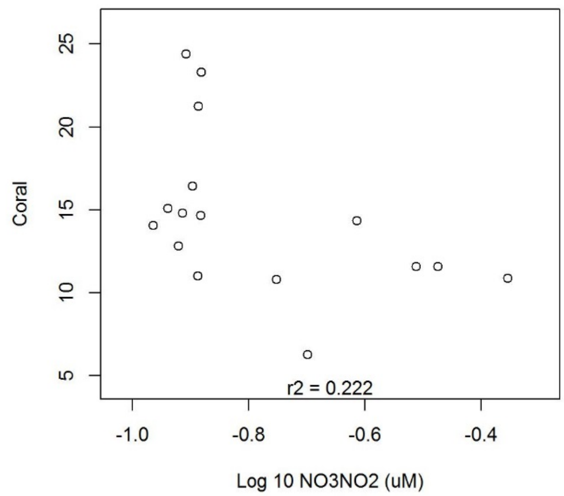


Figure 88-90. Relationship between Nitrate (uM) and the Gorgonian, Macroalgae, and Sponge metrics in the Phase 2 coral dataset.


Log 10 Chl (ug.L)
-0.04
-0.06
-0.08
-0.10
-0.12
-0.14
r2 = 0.071
-1.2
-1.0
-0.8
-0.6
-0.4
-0.2
Log 10 NO3NO2 (uM)


Log 10 DO (mg.L)
0.825
0.820
0.815
0.810
0.805
slp = -0.013, r2 = 0.237
-1.2
-1.0
-0.8
-0.6
-0.4
-0.2
Log 10 NO3NO2 (uM)



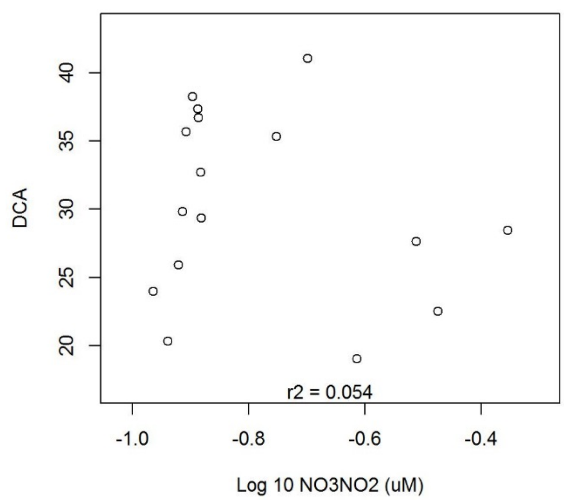
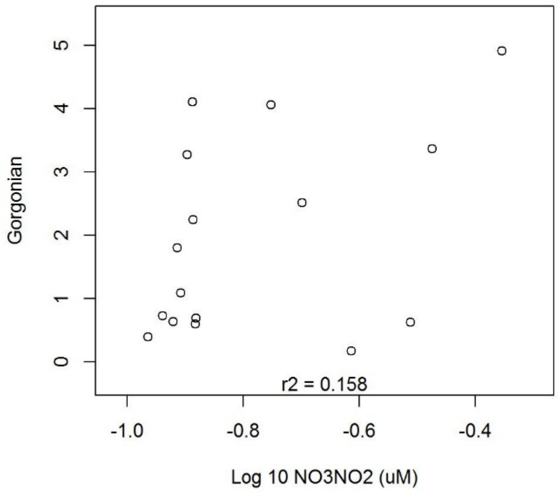
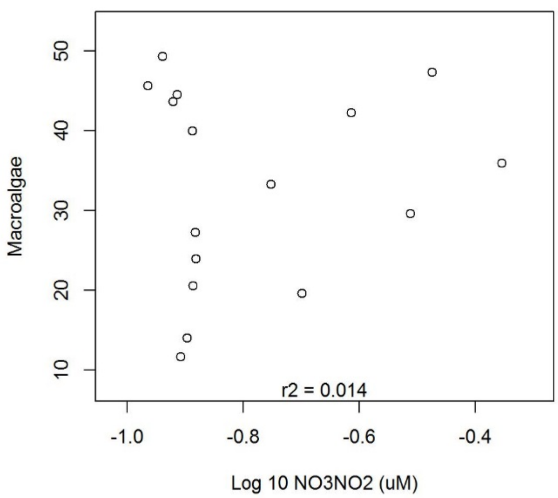


Figure 91-94. Relationship between Nitrate + Nitrite (uM) and Chlorophyll (ug/L), Dissolved Oxygen (mg/L), and the Coral and DCA in the Phase 2 coral dataset.


Gorgonian
r2 = 0.158
Log 10 NO3NO2 (uM)


Macroalgae
r2 = 0.014
Log 10 NO3NO2 (uM)

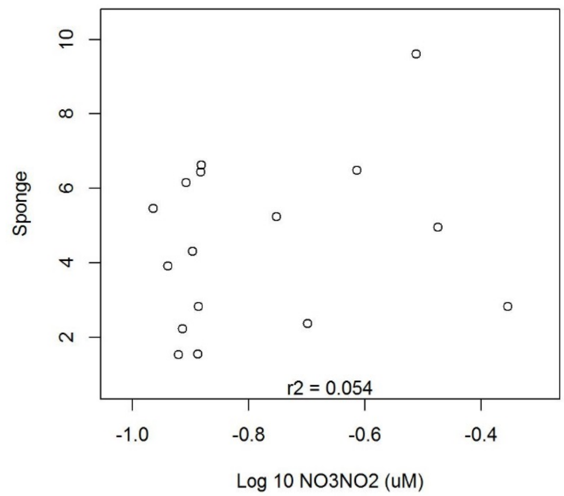
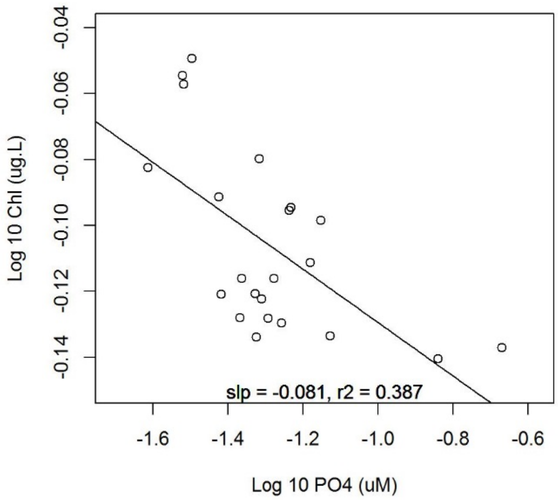
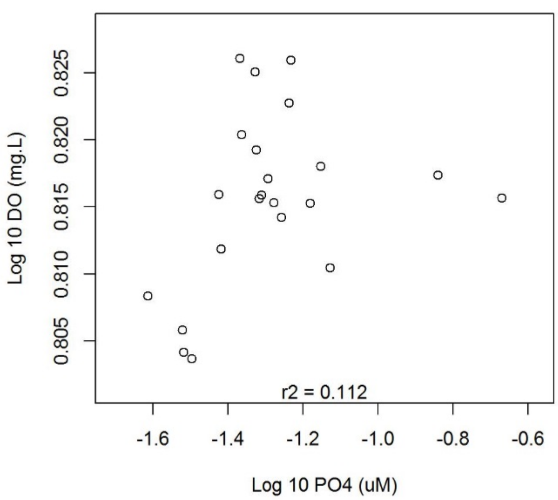
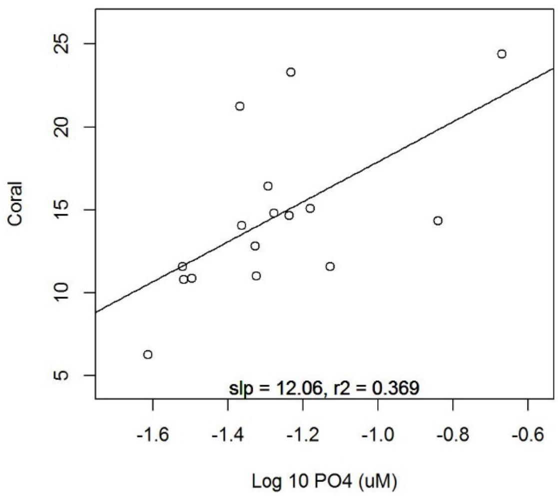


Figure 95-97. Relationship between Nitrate + Nitrite (uM) and the Gorgonian, Macroalgae, and Sponge metrics in the Phase 2 coral dataset.


Log 10 Chl (ug.L)
-0.04
-0.06
-0.08
-0.10
-0.12
-0.14
slp = -0.081, r2 = 0.387
-1.6
-1.4
-1.2
-1.0
-0.8
-0.6
Log 10 PO4 (uM)


Log 10 DO (mg.L)
0.825
0.820
0.815
0.810
0.805
r2 = 0.112
-1.6
-1.4
-1.2
-1.0
-0.8
-0.6
Log 10 PO4 (uM)



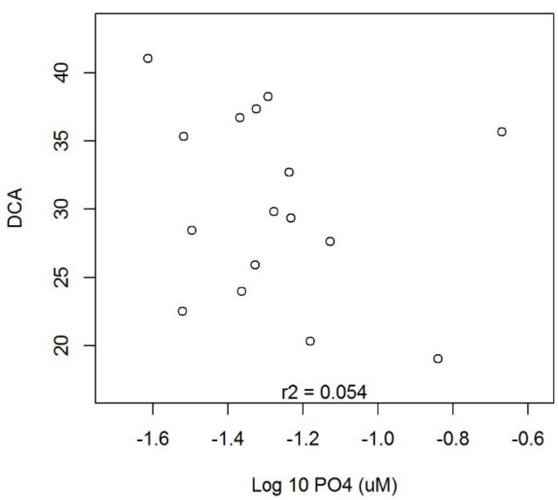
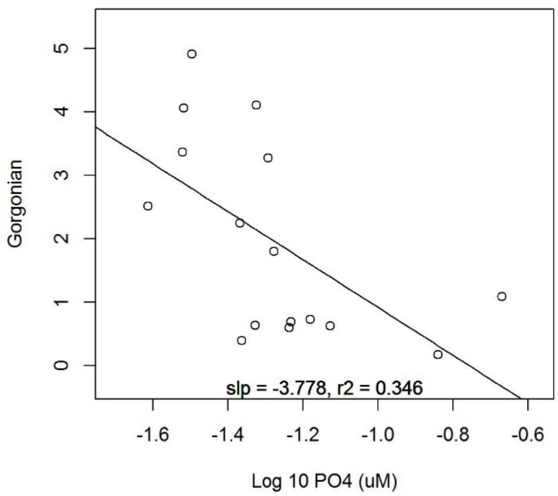
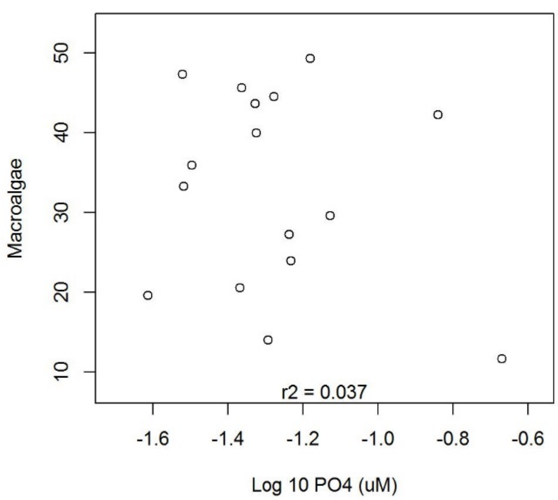


Figure 98-101. Relationship between Phosphate (uM) and Chlorophyll (ug/L), Dissolved Oxygen (mg/L), and the Coral and DCA in the Phase 2 coral dataset.


Gorgonian
slp = -3.778, r2 = 0.346
Log 10 PO4 (uM)


Macroalgae
r2 = 0.037
Log 10 PO4 (uM)

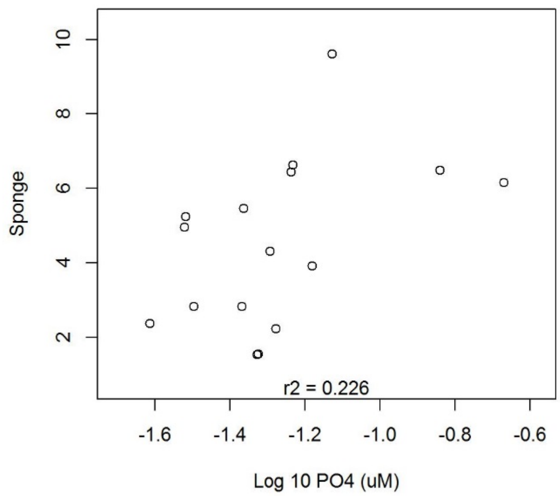
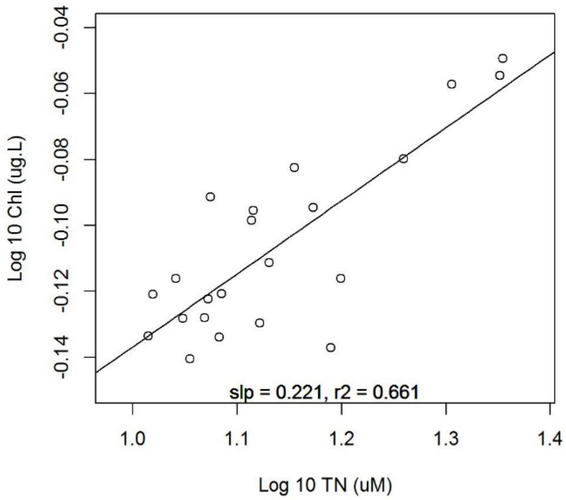
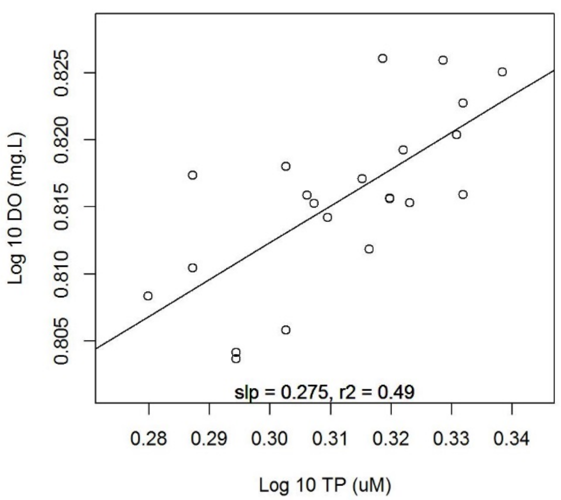
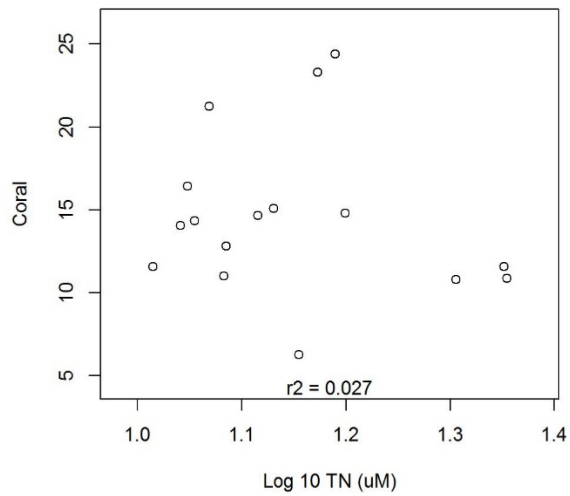


Figure 102-104. Relationship between Phosphate (uM) and the Gorgonian, Macroalgae, and Sponge metrics in the Phase 2 coral dataset.


-0.04
-0.06
-0.08
-0.10
-0.12
-0.14
Log 10 Chl (ug.L)
slp = 0.221, r2 = 0.661
1.0
1.1
1.2
1.3
1.4
Log 10 TN (uM)


0.825
0.820
0.815
0.810
0.805
Log 10 DO (mg.L)
slp = 0.275, r2 = 0.49
0.28
0.29
0.30
0.31
0.32
0.33
0.34
Log 10 TP (uM)



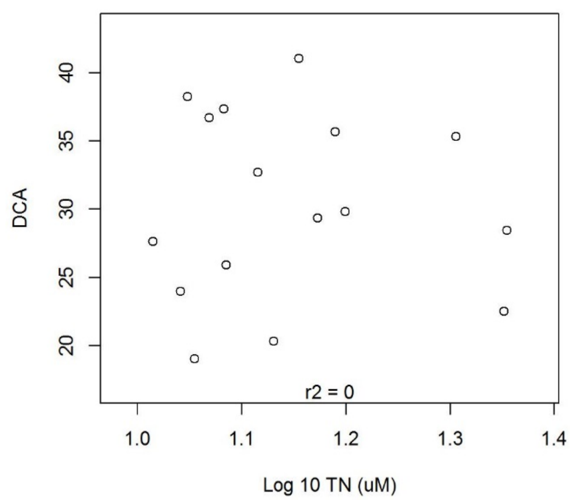
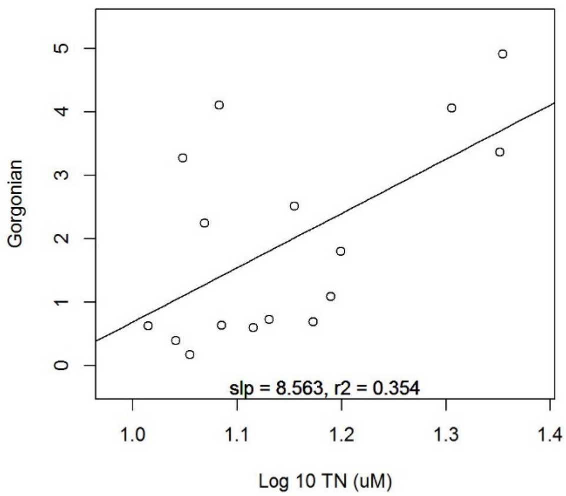
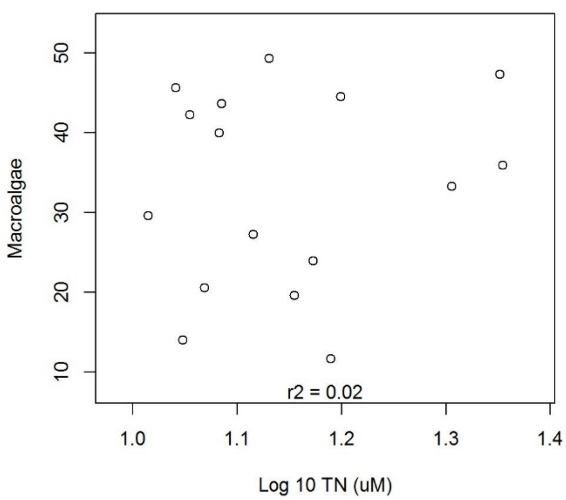


Figure 105-108. Relationship between Total N (uM) and Chlorophyll (ug/L), Dissolved Oxygen (mg/L), and the Coral and DCA in the Phase 2 coral dataset.


Gorgonian
0
1
2
3
4
5
slp = 8.563, r2 = 0.354
1.0
1.1
1.2
1.3
1.4
Log 10 TN (uM)


Macroalgae
10
20
30
40
50
r2 = 0.02
1.0
1.1
1.2
1.3
1.4
Log 10 TN (uM)

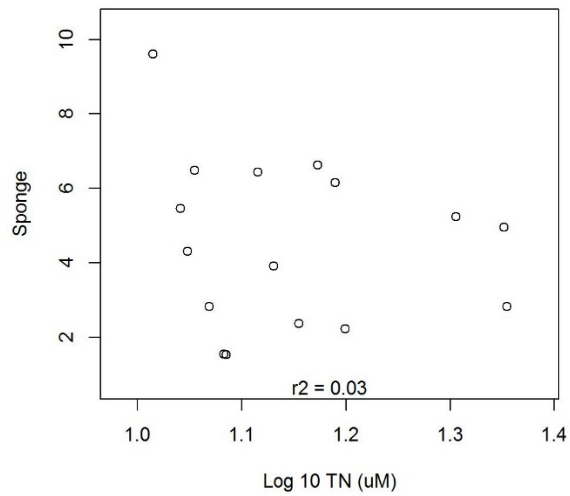
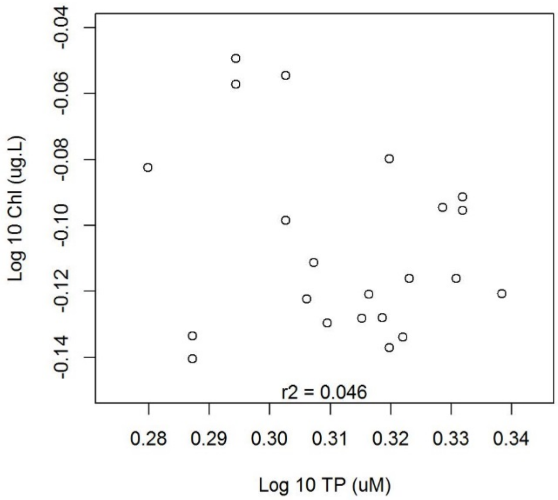
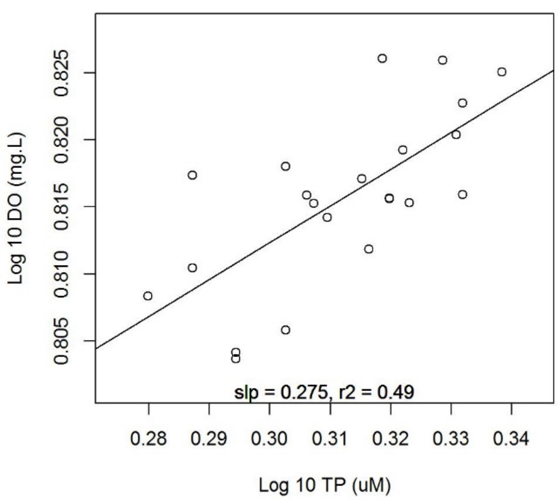
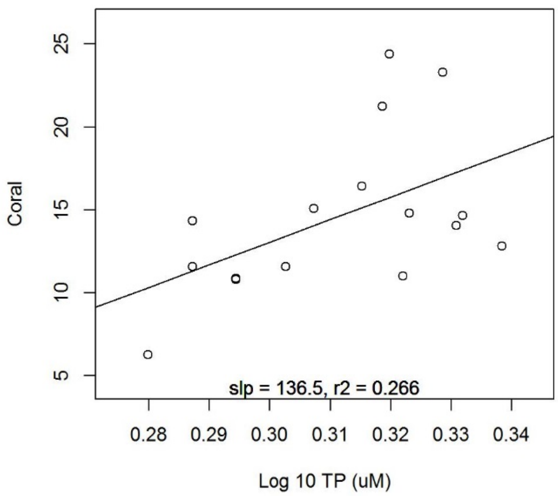


Figure 109-111. Relationship between Total N (uM) and the Gorgonian, Macroalgae, and Sponge metrics in the Phase 2 coral dataset.


Log 10 Chl (ug.L)
-0.04
-0.06
-0.08
-0.10
-0.12
-0.14
r2 = 0.046
0.28
0.29
0.30
0.31
0.32
0.33
0.34
Log 10 TP (uM)


Log 10 DO (mg.L)
0.805
0.810
0.815
0.820
0.825
slp = 0.275, r2 = 0.49
0.28
0.29
0.30
0.31
0.32
0.33
0.34
Log 10 TP (uM)



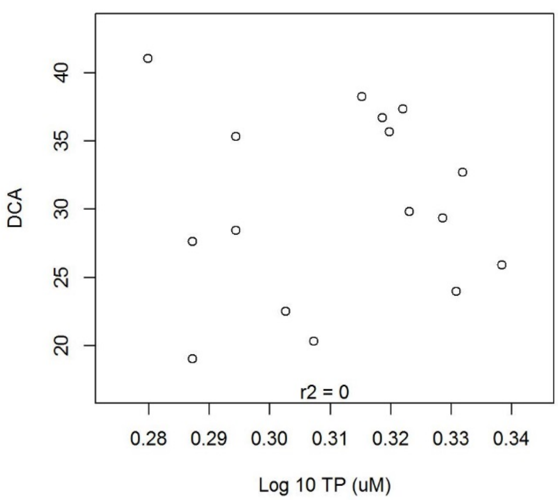
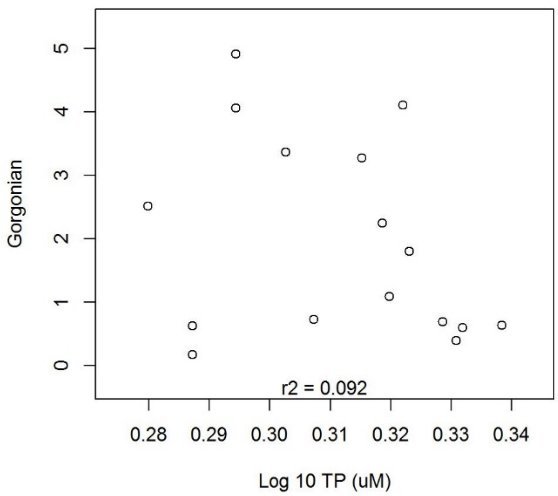
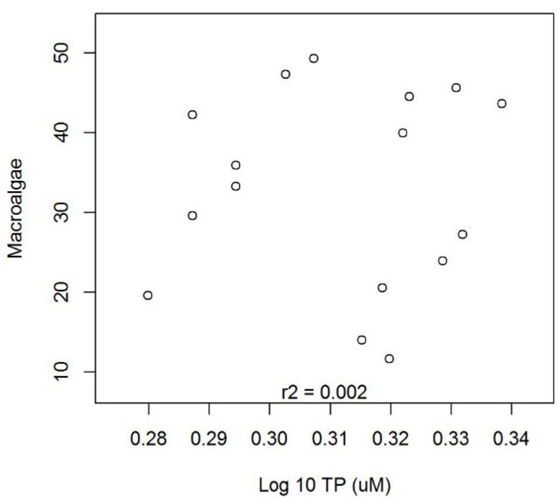


Figure 112-115. Relationship between Total P (uM) and Chlorophyll (ug/L), Dissolved Oxygen (mg/L), and the Coral and DCA in the Phase 2 coral dataset.


Gorgonian
0
1
2
3
4
5
r2 = 0.092
0.28
0.29
0.30
0.31
0.32
0.33
0.34
Log 10 TP (uM)


Macroalgae
10
20
30
40
50
r2 = 0.002
0.28
0.29
0.30
0.31
0.32
0.33
0.34
Log 10 TP (uM)

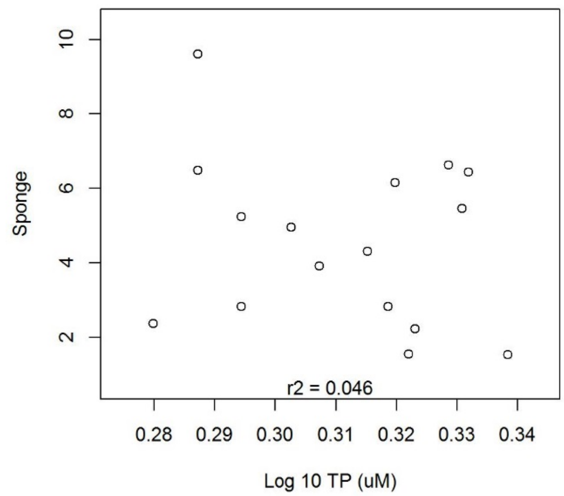
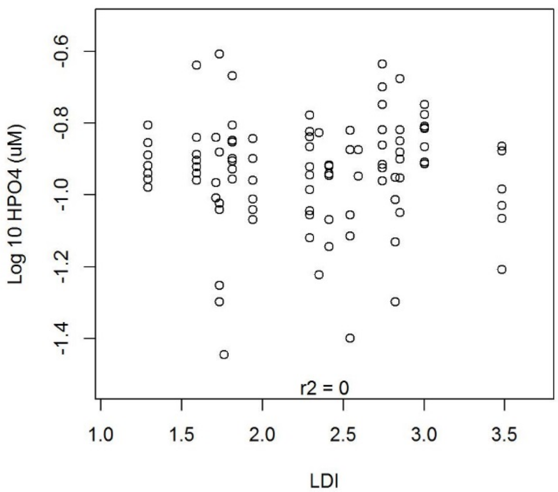


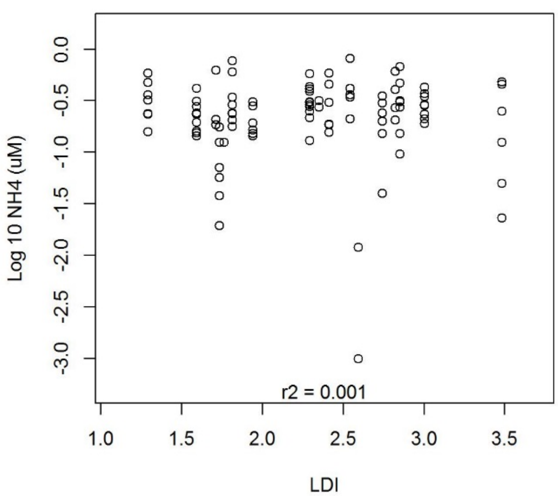
Figure 116-118. Relationship between Total P (uM) and the Gorgonian, Macroalgae, and Sponge metrics in the Phase 2 coral dataset.

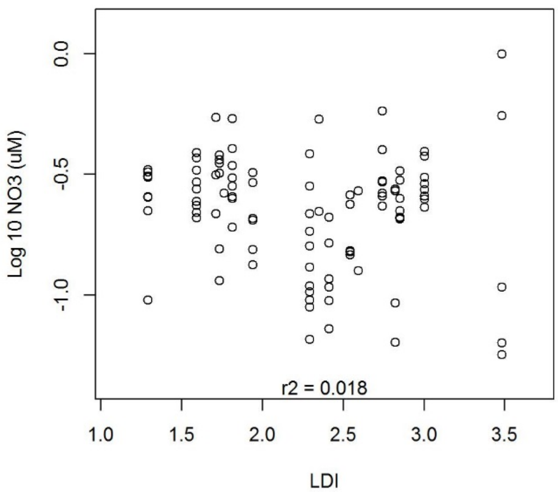
Stressor-response Analysis – RARE/NCA Coral Dataset

Correlations and linear regression analyses for stressor-response pairs in the RARE/NCA coral dataset are shown below. Only statistically significant regression models are displayed ($p < 0.05$), along with the slope of the relationship. The R-squared value is displayed for all relationships. All nutrient concentrations and coral heights and diameters have been Log 10 transformed.

Statistically significant relationships were as follows. LDI was found to be negatively correlated with NO3 + NO2, and not significantly related to any of the 3 coral metrics. NH4, TN, and Urea were found to be positively correlated with the percent of coral alive. All correlations were weak, with R-squared < 0.10. HPO4, NO3, NO3 + NO2, and TP were not found to be significantly correlated with any of the coral metrics.


Log 10 HPO4 (uM)
-0.6
-0.8
-1.0
-1.2
-1.4
r2 = 0
1.0
1.5
2.0
2.5
3.0
3.5
LDI


Log 10 NH4 (uM)
0.0
-0.5
-1.0
-1.5
-2.0
-2.5
-3.0
r2 = 0.001
1.0
1.5
2.0
2.5
3.0
3.5
LDI



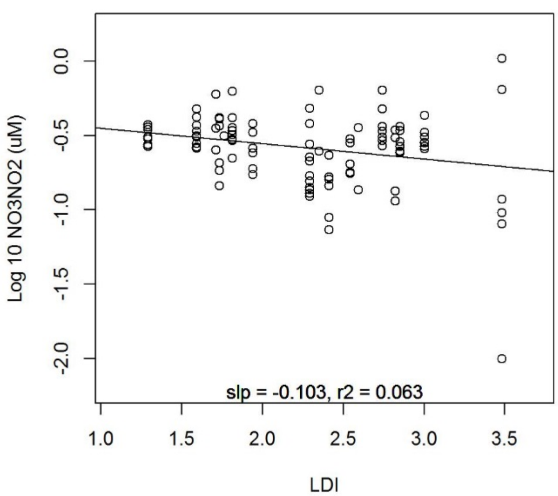
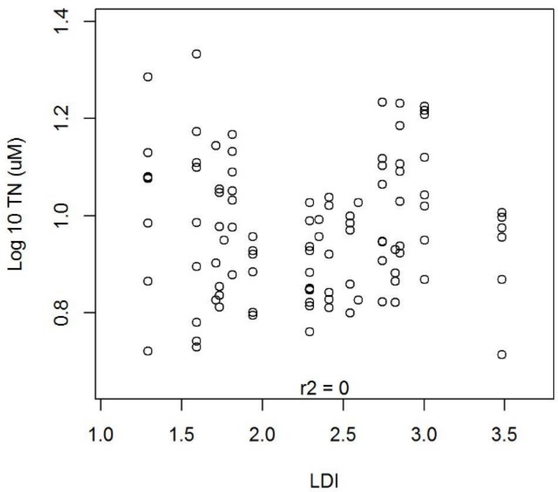
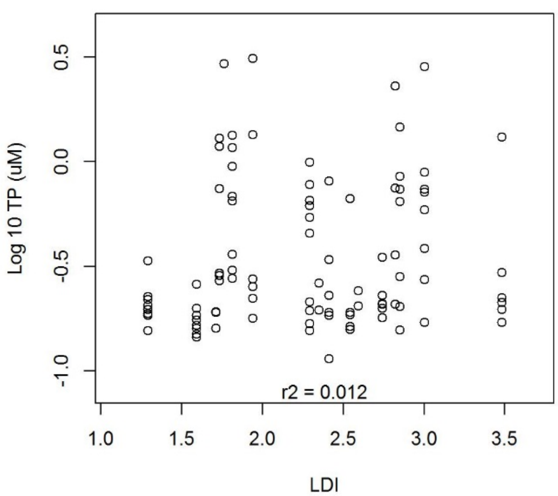


Figure 119-122. Relationship between Landscape Development Index and Phosphate, Ammonium, Nitrate, and Nitrate + Nitrite (uM) in the RARE/NCA coral dataset.


Log 10 TN (uM)
1.4
1.2
1.0
0.8
r2 = 0
1.0
1.5
2.0
2.5
3.0
3.5
LDI


Log 10 TP (uM)
0.5
0.0
-0.5
-1.0
r2 = 0.012
1.0
1.5
2.0
2.5
3.0
3.5
LDI

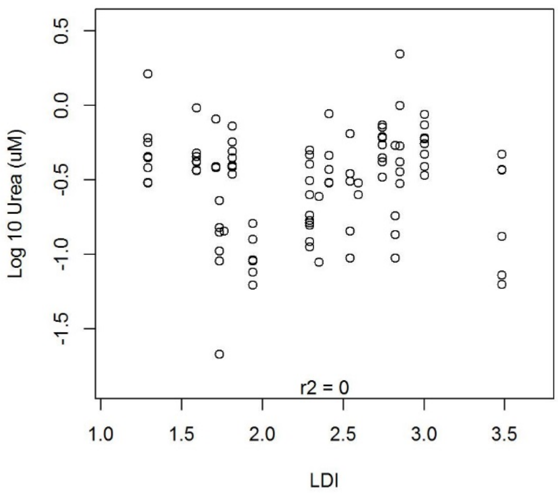
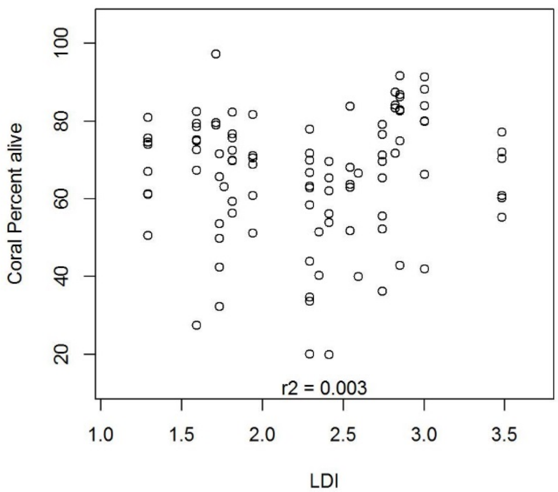
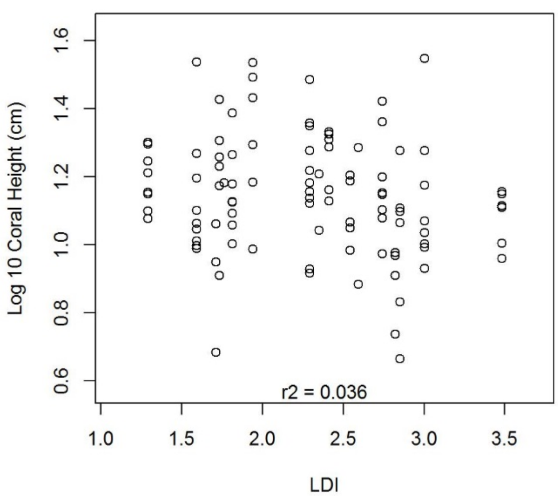


Figure 123-125. Relationship between Landscape Development Index and Total N, Total P, and Urea (uM) in the RARE/NCA coral dataset.


Coral Percent alive
100
80
60
40
20
r2 = 0.003
1.0
1.5
2.0
2.5
3.0
3.5
LDI


Log 10 Coral Height (cm)
1.6
1.4
1.2
1.0
0.8
0.6
r2 = 0.036
1.0
1.5
2.0
2.5
3.0
3.5
LDI

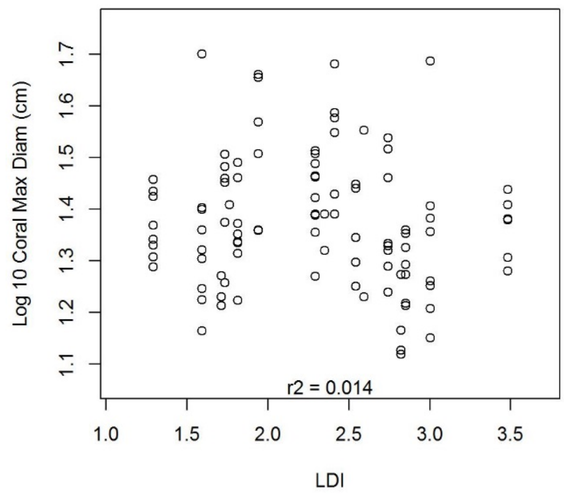
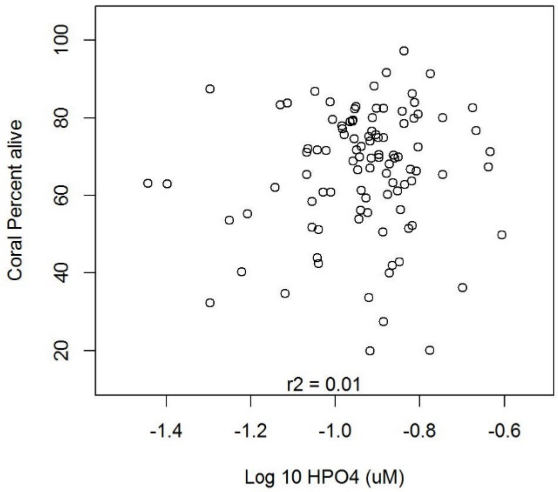
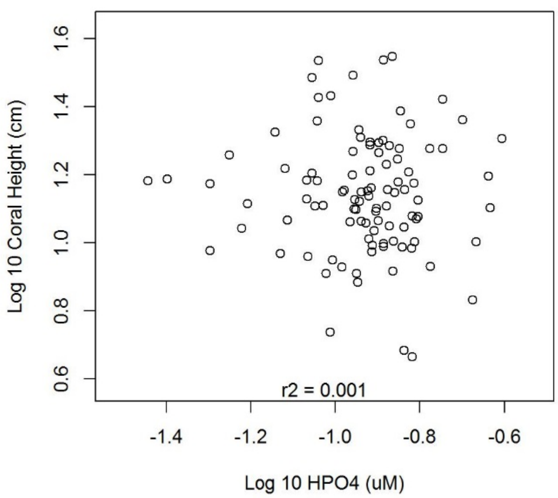


Figure 126-128. Relationship between Landscape Development Index and the Percent of Coral Alive, Coral Height, and Coral Max Diameter (cm).


Coral Percent alive
100
80
60
40
20
r2 = 0.01
-1.4
-1.2
-1.0
-0.8
-0.6
Log 10 HPO4 (uM)


Log 10 Coral Height (cm)
1.6
1.4
1.2
1.0
0.8
0.6
r2 = 0.001
-1.4
-1.2
-1.0
-0.8
-0.6
Log 10 HPO4 (uM)

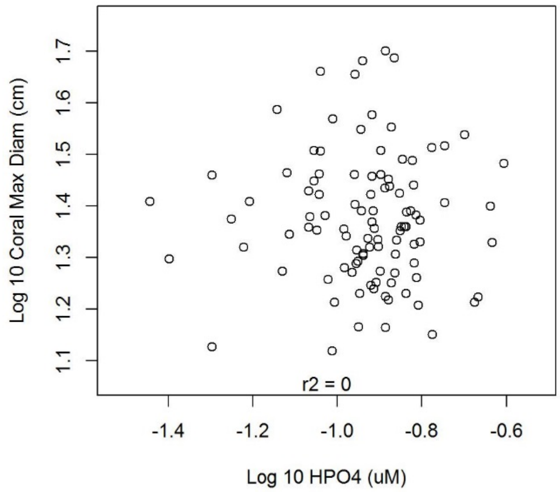
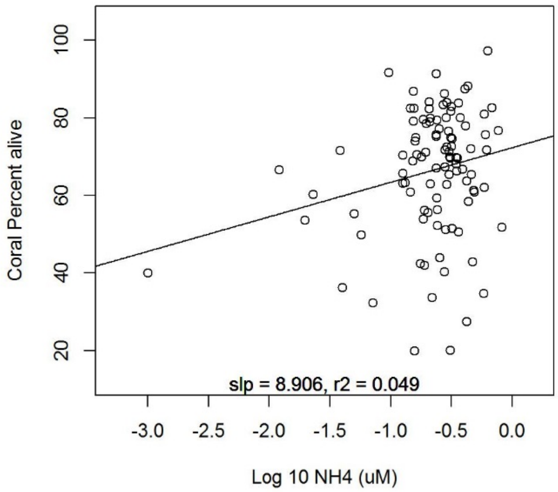
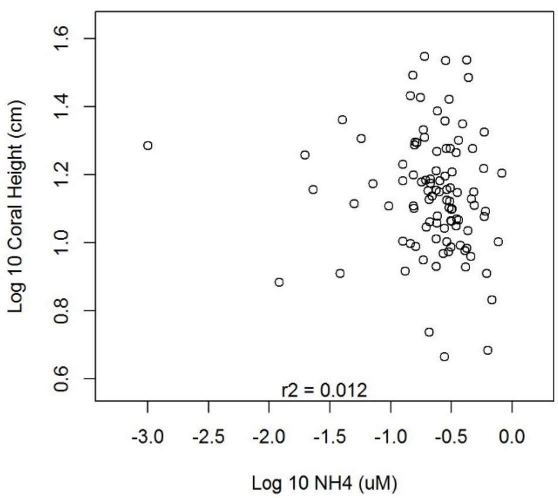


Figure 129-131. Relationship between Phosphate (uM) and the Percent of Coral Alive, Coral Height, and Coral Max Diameter (cm).


Coral Percent alive
100
80
60
40
20
slp = 8.906, r2 = 0.049
-3.0 -2.5 -2.0 -1.5 -1.0 -0.5 0.0
Log 10 NH4 (uM)


Log 10 Coral Height (cm)
1.6
1.4
1.2
1.0
0.8
0.6
r2 = 0.012
-3.0 -2.5 -2.0 -1.5 -1.0 -0.5 0.0
Log 10 NH4 (uM)

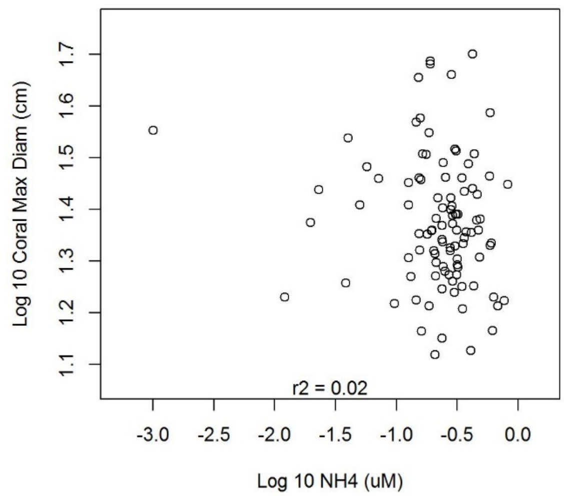
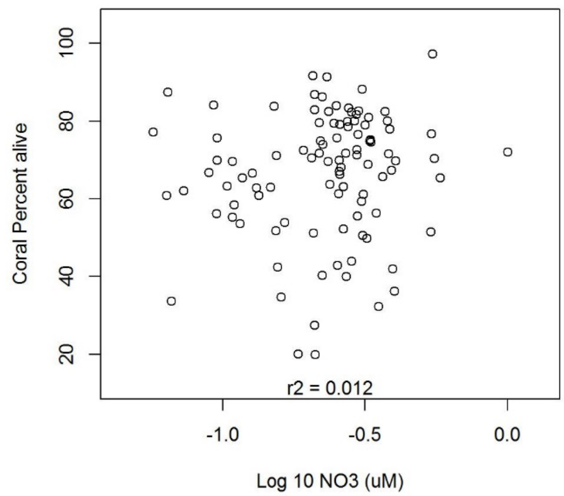
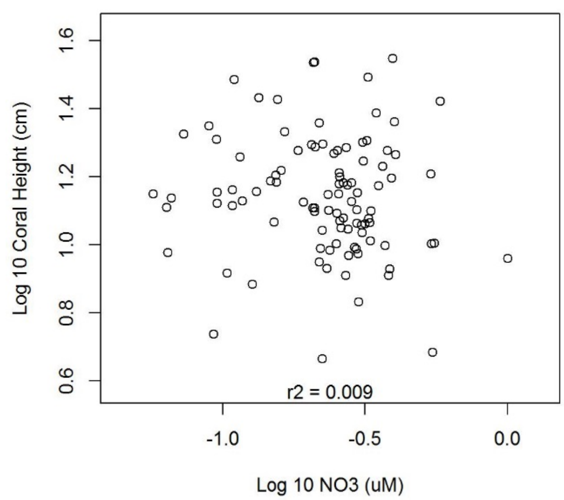


Figure 132-134. Relationship between Ammonium (uM) and the Percent of Coral Alive, Coral Height, and Coral Max Diameter (cm).


Coral Percent alive
100
80
60
40
20
r2 = 0.012
-1.0
-0.5
0.0
Log 10 NO3 (uM)


Log 10 Coral Height (cm)
1.6
1.4
1.2
1.0
0.8
0.6
r2 = 0.009
-1.0
-0.5
0.0
Log 10 NO3 (uM)

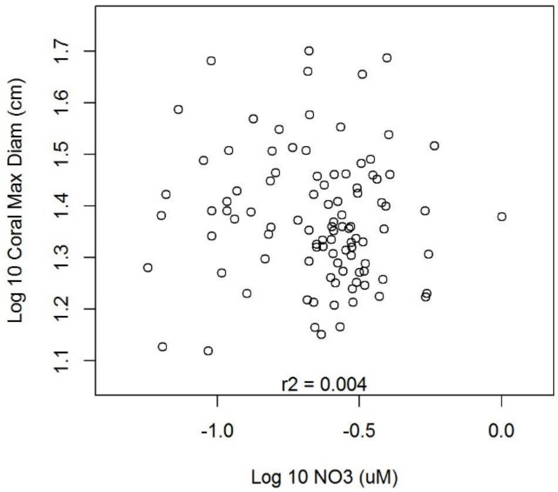
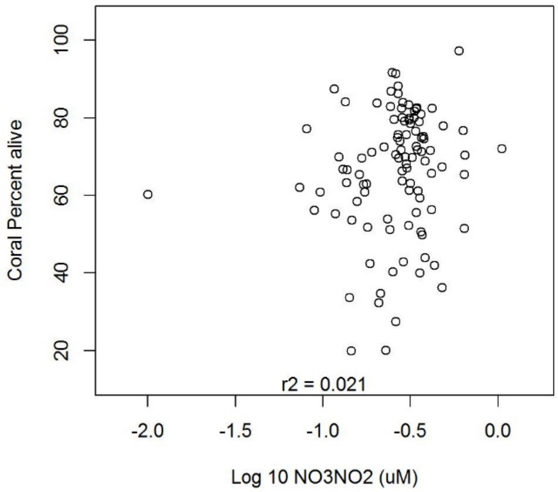
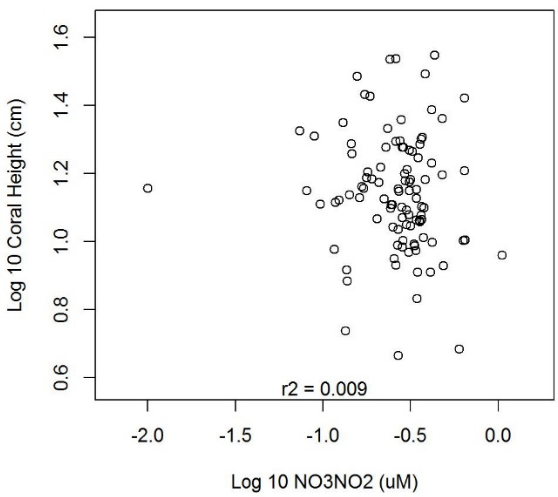


Figure 135-137. Relationship between Nitrate (uM) and the Percent of Coral Alive, Coral Height, and Coral Max Diameter (cm).


Coral Percent alive
100
80
60
40
20
r2 = 0.021
-2.0
-1.5
-1.0
-0.5
0.0
Log 10 NO3NO2 (uM)


Log 10 Coral Height (cm)
1.6
1.4
1.2
1.0
0.8
0.6
r2 = 0.009
-2.0
-1.5
-1.0
-0.5
0.0
Log 10 NO3NO2 (uM)

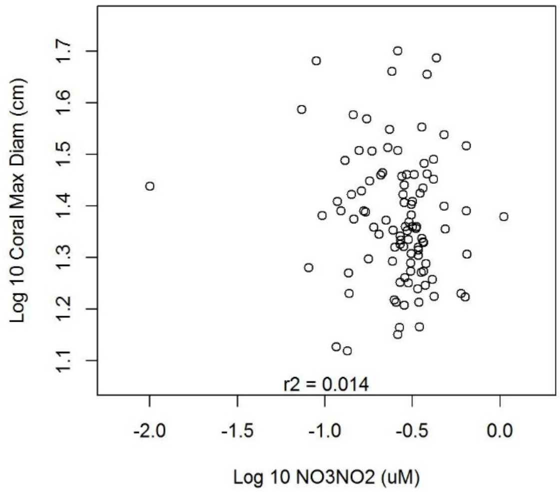
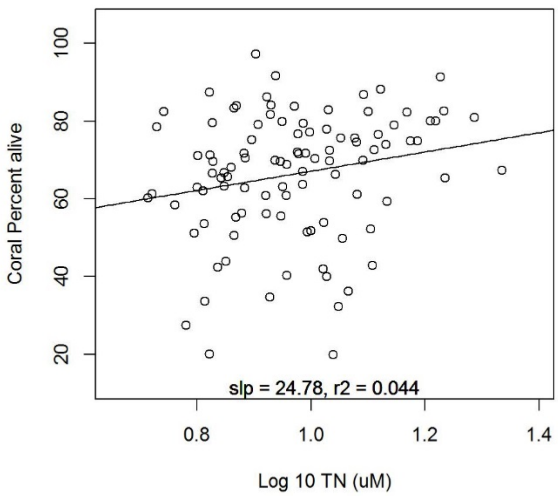
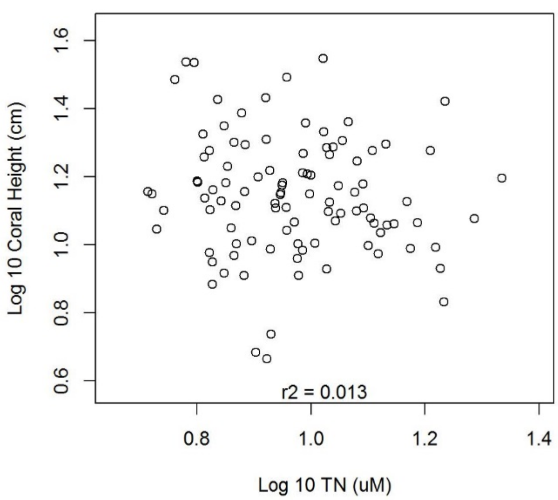


Figure 138-140. Relationship between Nitrate + Nitrite (uM) and the Percent of Coral Alive, Coral Height, and Coral Max Diameter (cm).


Coral Percent alive
100
80
60
40
20
slp = 24.78, r2 = 0.044
0.8
1.0
1.2
1.4
Log 10 TN (uM)


Log 10 Coral Height (cm)
1.6
1.4
1.2
1.0
0.8
0.6
r2 = 0.013
0.8
1.0
1.2
1.4
Log 10 TN (uM)

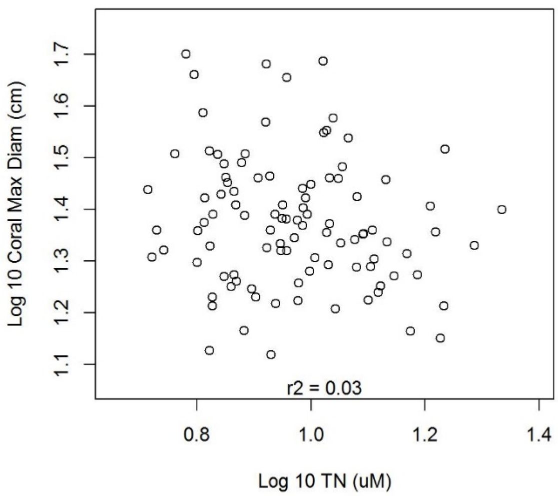
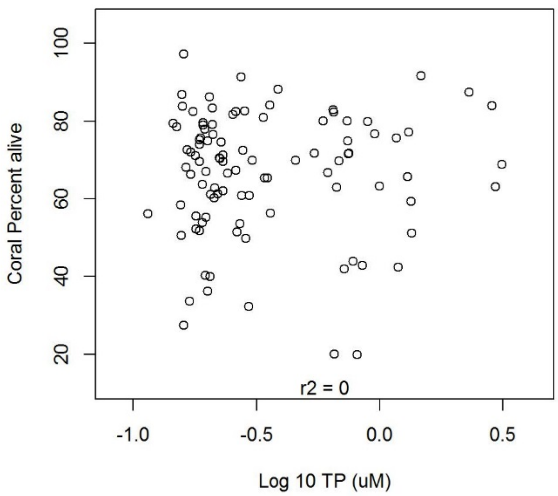
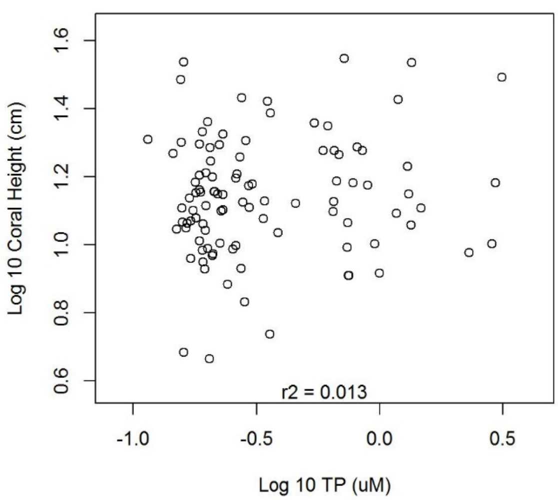


Figure 141-143. Relationship between Total N (uM) and the Percent of Coral Alive, Coral Height, and Coral Max Diameter (cm).


Coral Percent alive
100
80
60
40
20
r2 = 0
-1.0
-0.5
0.0
0.5
Log 10 TP (uM)


Log 10 Coral Height (cm)
1.6
1.4
1.2
1.0
0.8
0.6
r2 = 0.013
-1.0
-0.5
0.0
0.5
Log 10 TP (uM)

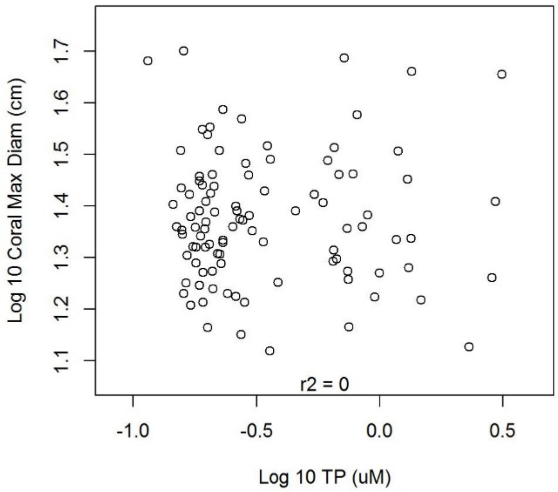
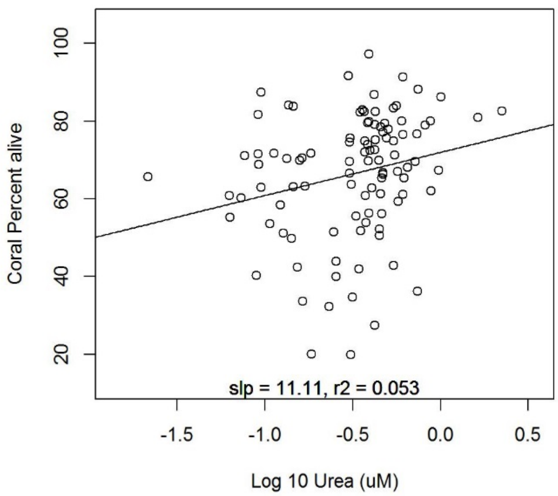
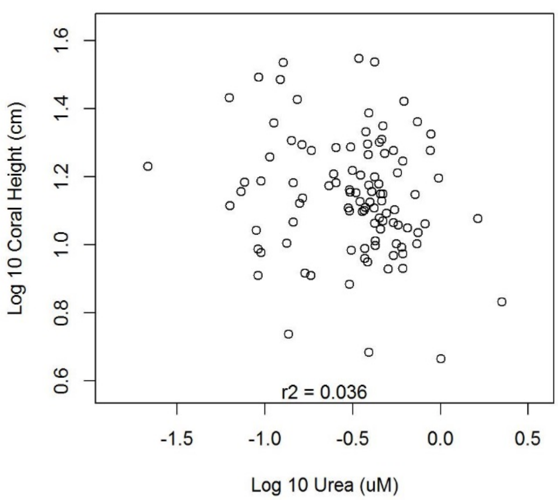


Figure 144-146. Relationship between Total P (uM) and the Percent of Coral Alive, Coral Height, and Coral Max Diameter (cm).


Coral Percent alive
Log 10 Urea (uM)
slp = 11.11, r2 = 0.053


Log 10 Coral Height (cm)
Log 10 Urea (uM)
r2 = 0.036

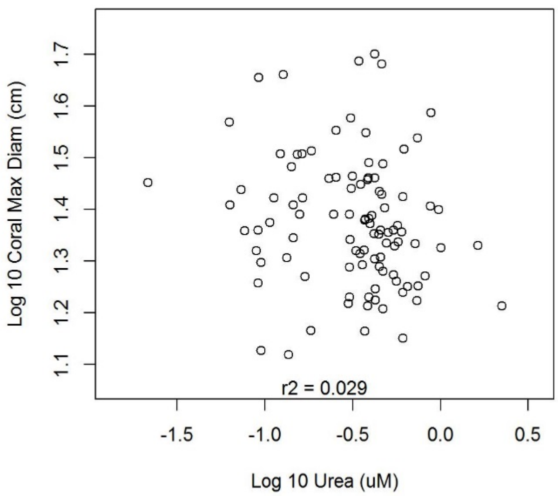


Figure 147-149. Relationship between Urea (uM) and the Percent of Coral Alive, Coral Height, and Coral Max Diameter (cm).

Summary

The fullest nutrient gradient is provided by the full combined chemistry dataset. The individual datasets only provide part of the nutrient or LDI gradient, therefore the results can be misleading. Looking at the combined dataset, however, the stressor-response analyses provide indication that changes in LDI relate to increases in nutrients (TN and TP) and chlorophyll, however, the forms of nitrate differ. Nitrate/nitrite decreases with LDI, which may be due to a preponderance of relatively more uptake of these oxidized forms in these coastal waters or greater relative production of these forms by more natural watersheds. Because TN is increasing with LDI, but NO3/NO2 decreasing, it suggests that organic N forms are predominant under increased LDI, consistent with both higher chlorophyll as well as potentially untransformed human N sources. Chlorophyll exhibited its strongest relationship with ammonium and TN, which is consistent with the N content of phytoplankton as well as hypotheses about relative N limitation. The least disturbed concentrations of N and P discussed above under the distributional statistics with regards to LDI certainly suggest these waters are naturally more N limited. The average N:P molar ratio there was essentially 8:1, consistent with relative N limitation in these waters.

Unfortunately, the gradient of TN associated with existing paired chlorophyll and TN data is insufficient to resolve TN concentrations associated with reference chlorophyll conditions (for LDI<2 within 500m of the coast, the 90th percentile Chl a is 0.344 μg/L, Table 4). There are actually no paired chlorophyll and TN and TP data for least disturbed sites within 500m of shore. There would need to be a greater effort to collect paired TN, TP, and chlorophyll data) in these coastal waters, especially within least disturbed waters, to resolve this relationship better and to more confidently derive TN and TP values from the stressor-response line of evidence. The existing dataset only had approximately 22 samples with paired TN, TP and Chl.

However, if we use the upper percentiles of chlorophyll a from the full dataset (Table 2) as avoidance concentrations of chlorophyll (75th = 0.745 μg/L, 90th percentile = 0.807 μg/L), using the full dataset TN vs. Chl regression model [$Log_{10}Chl = 0.22142(log_{10}TN) - 0.3582$] yields TN concentrations of approximately 11.2 μM (157 μg/L) and 15.8 μM (222 μg/L) respectively. Such concentrations bracket the distributional value for TN found for sites with LDI<2 within 500m of the coast (14.8 μM or 207 μg/L).

It was unclear which coral measures or target levels are desired by USVI, and in case the relationships were highly variable and were not taken along as full a nutrient gradient as represented in the full dataset. USVI is encouraged to consider these plots, however, and determine if the relationships provide promising direction for future sampling effort or analysis. Certainly, the threat of algae or macroalgae overgrowing corals exists, but the lack of an appropriate response measure for this or substantial data on it precludes a recommendation.

Conclusions

In this report, water quality and coral data analysis from the coastal waters of the US Virgin Islands have been summarized, in order to inform the development of numeric nutrient criteria in addition to the existing TP criterion.

Distributional statistics for water quality parameters were calculated. This analysis was performed for the entire dataset, the subset of data within 500 m of shore, and the subset of data within 500 m of shore and closest to a catchment with LDI under 2. This last category was presumed to represent least impacted coastal waters of the USVI. The 90th percentile concentrations of TP and TN in these waters were 14.8 μM TN and 1.8 μM TP (207 μg/L and 56 μg/L respectively). The USVI existing TP standard is very close to this value from least disturbed waters, the comparable TN value, therefore, may also represent an appropriate line of evidence.

The data were also analyzed for significant correlations among stressor and response variables provided. This included relationships between LDI and stressors (nutrients), LDI and endpoints (Chl, DO, biotic metrics), and between stressors and endpoints.

Most correlations were insignificant or significant but weak and there was limited resolution of temporal trends, being limited by available data. Using the full dataset as representative of the greatest gradients in stressor and response conditions, significant models between TN and Chl were identified. Using Chl target values associated with the upper percentiles of the full dataset as representative of avoidance concentrations yielded TN concentrations between 157 and 222 μg/L, consistent with the least disturbed reference condition estimate. Note that such chlorophyll a concentrations are not linked to a particular harm to use, they are merely an upper concentration associated with waters draining watersheds with higher LDI values. Lacking particular chlorophyll targets, this was the best that could be achieved.

The state may wish to pursue identification of specific chlorophyll targets from the literature or other nearby countries or territories and calculate TN concentrations associated with those concentrations.

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Addendum

Additional Variables from Exploratory Statistical Analysis – Full Chemistry Dataset

Additional variables from the exploratory analysis for the full chemistry dataset are shown below. Note that some variables have been $\log_{10}$ transformed in order to better meet assumptions of normality.

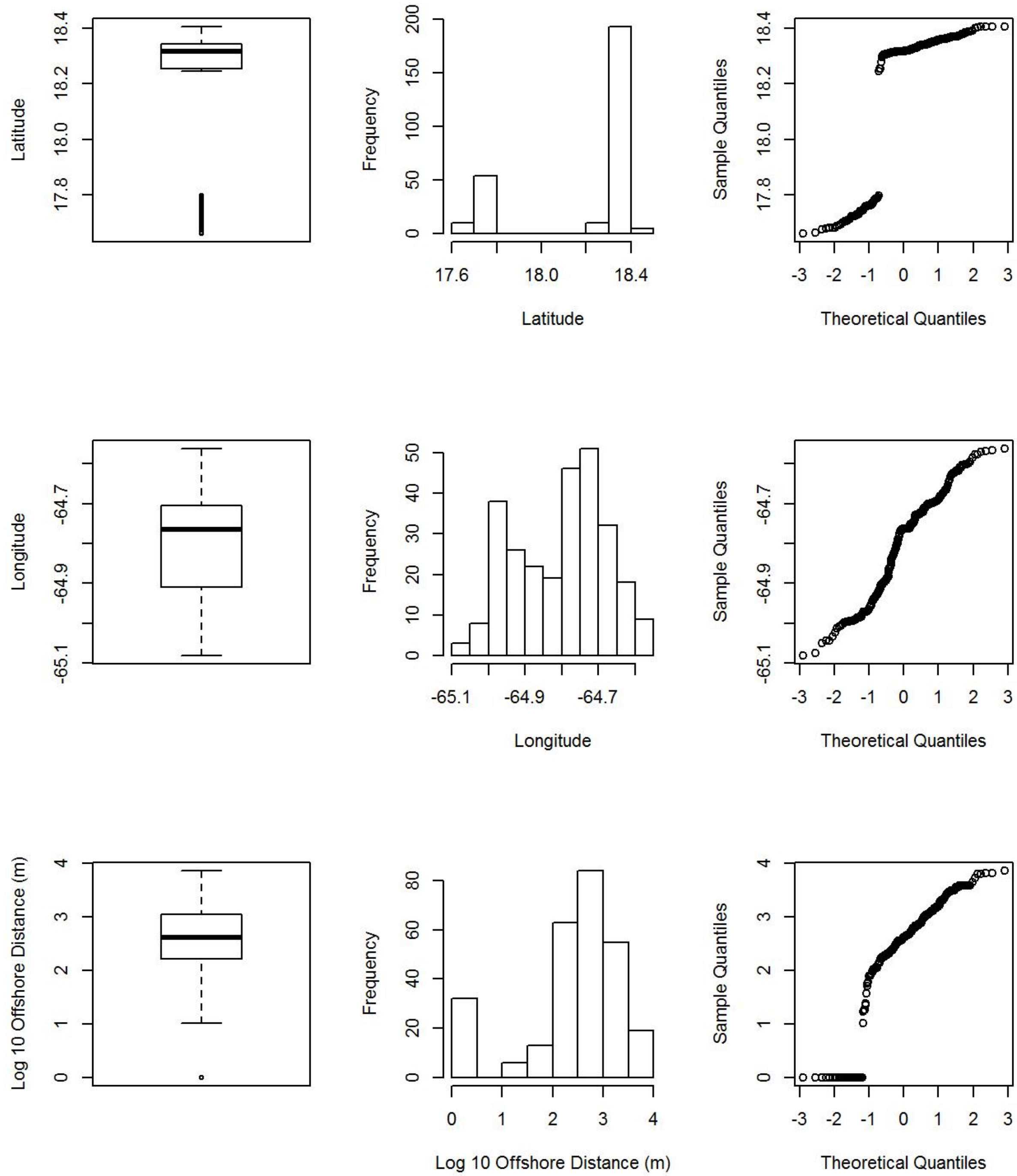

Figure A1-A3. Distributions of measurement latitude, longitude, and distance from shore (m) in full chemistry dataset.

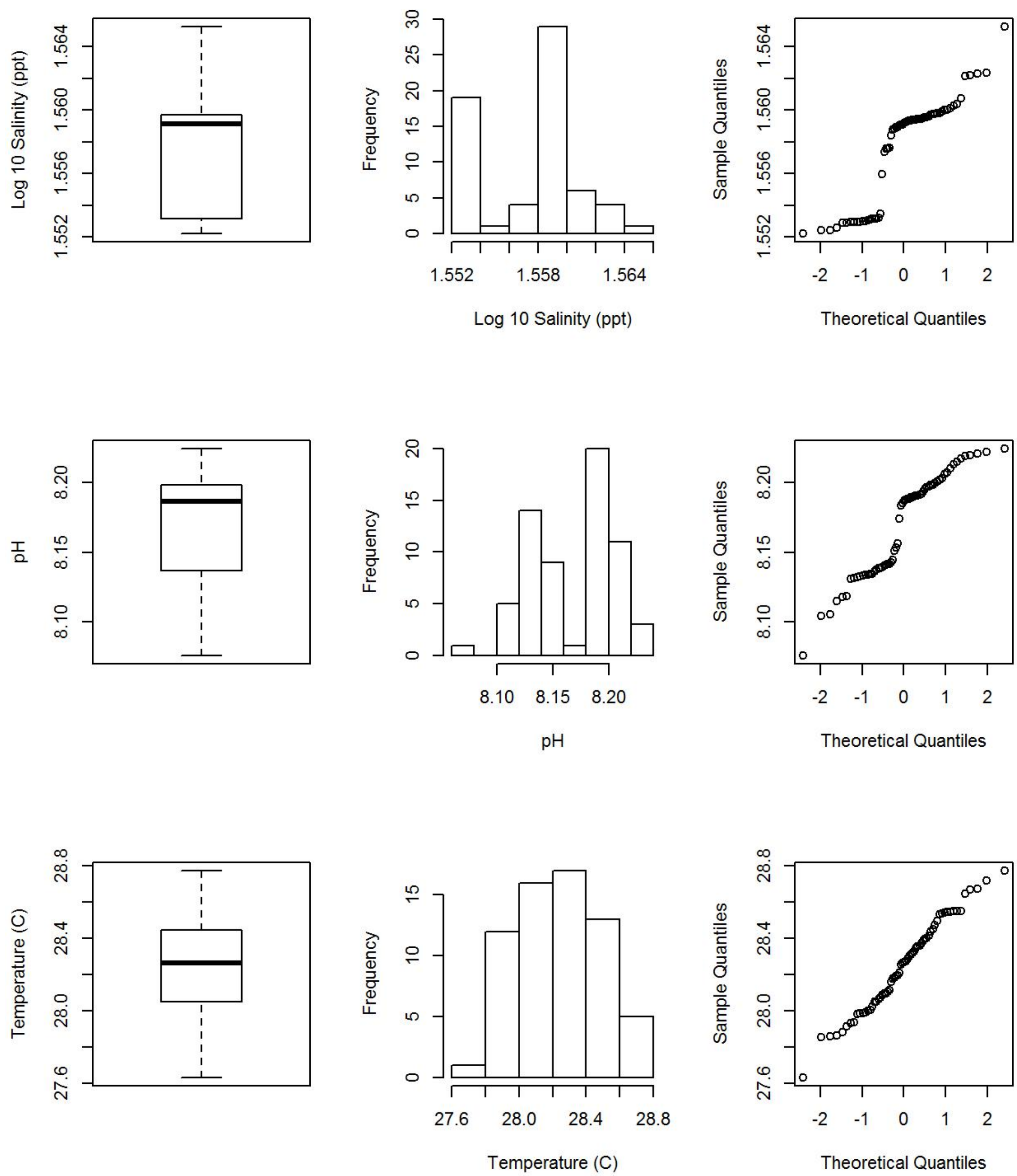

Figure A4-A6. Distributions of Log 10 Salinity, pH, and Temperature (C) values in full chemistry dataset.

Additional Variables from Exploratory Statistical Analysis – Phase 2 Coral Dataset

Additional variables from the exploratory analysis for the Phase 2 coral dataset are shown below. Note that some variables have been $\log_{10}$ transformed in order to better meet assumptions of normality.

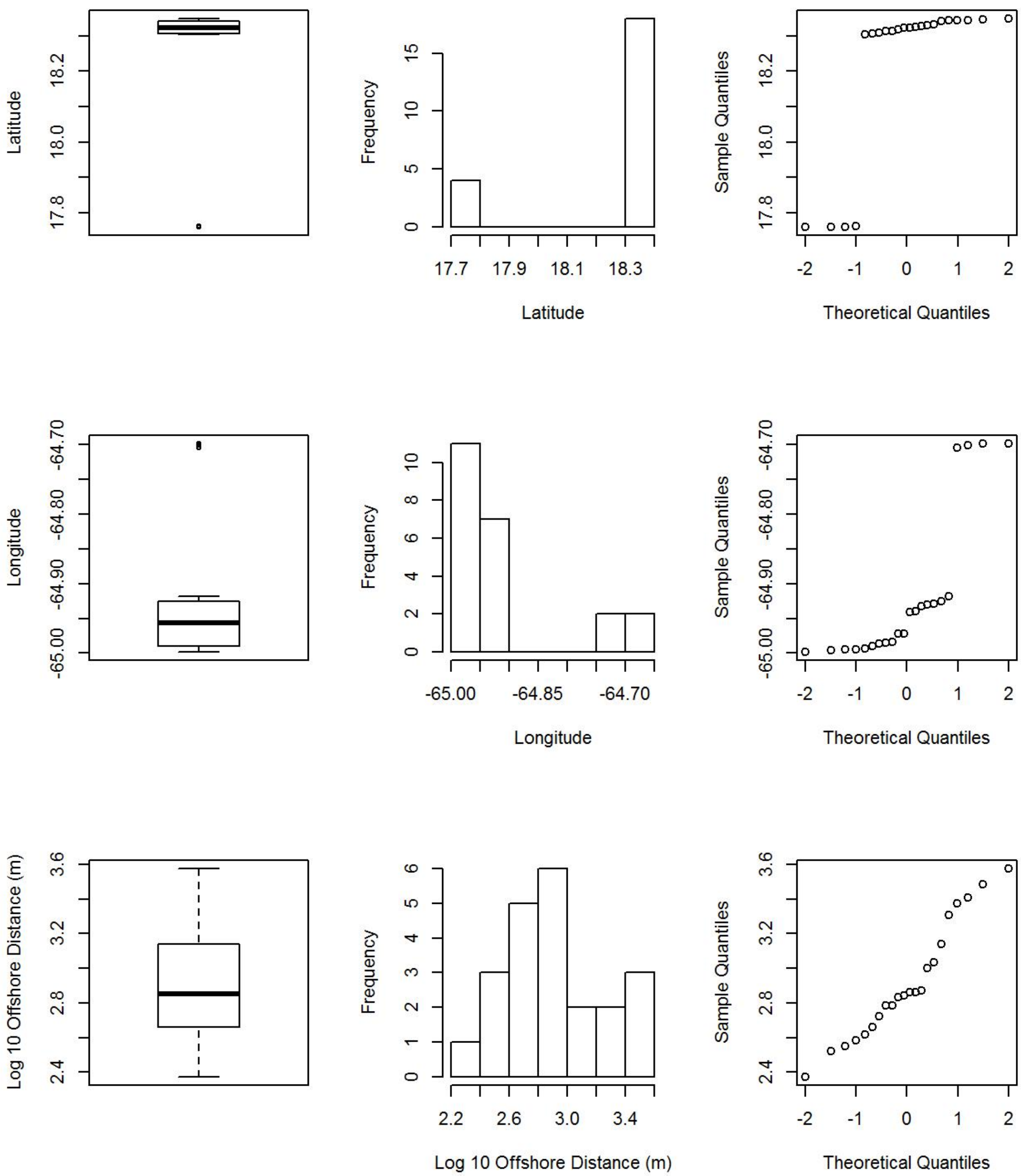

Figure A7-A9. Distributions of measurement latitude, longitude, and distance from shore (m) in Phase 2 coral dataset.

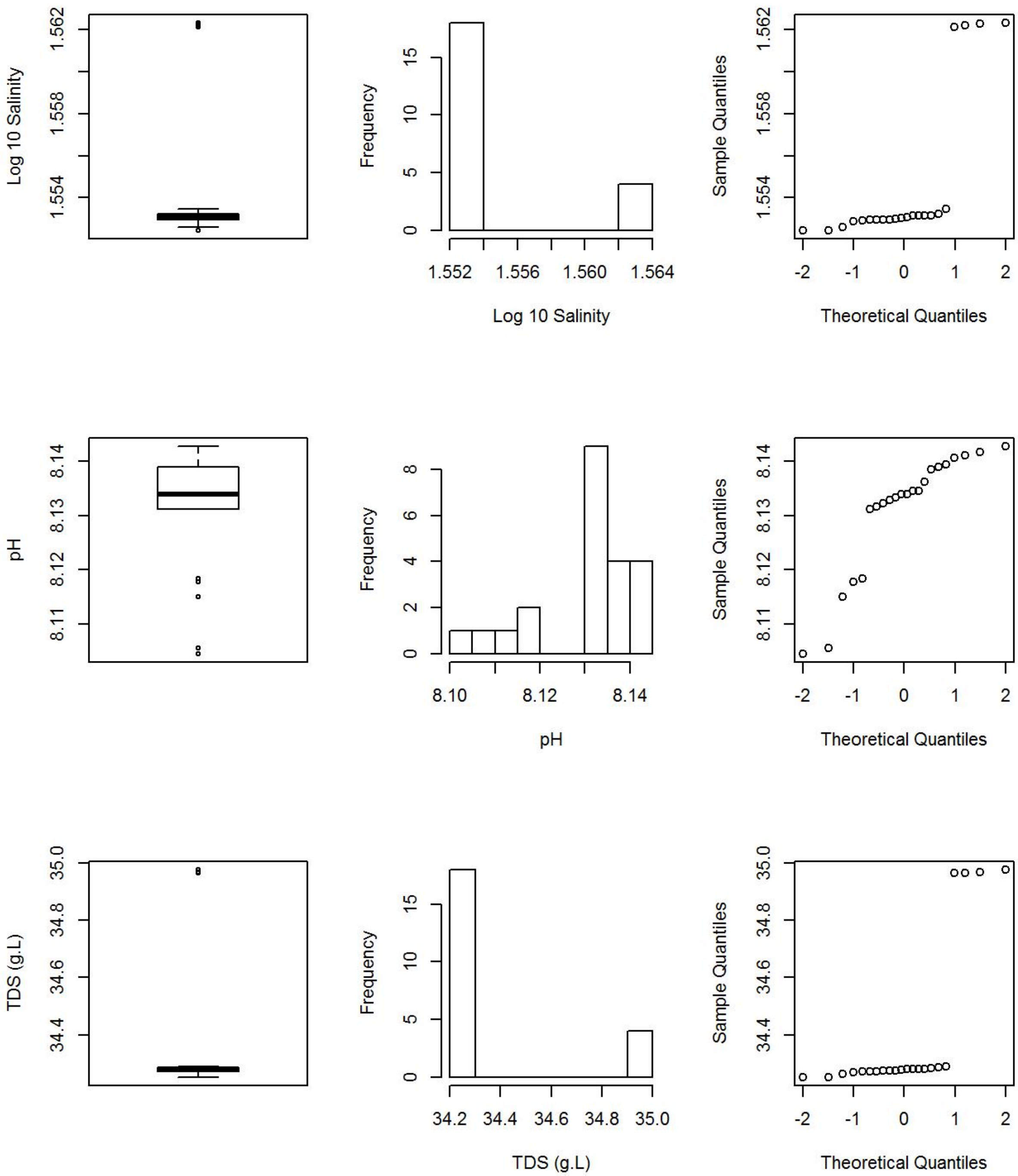

Figure A10-A12. Distributions of Log 10 Salinity, pH, and Total Dissolved Solids (g/L) values in Phase 2 coral dataset.

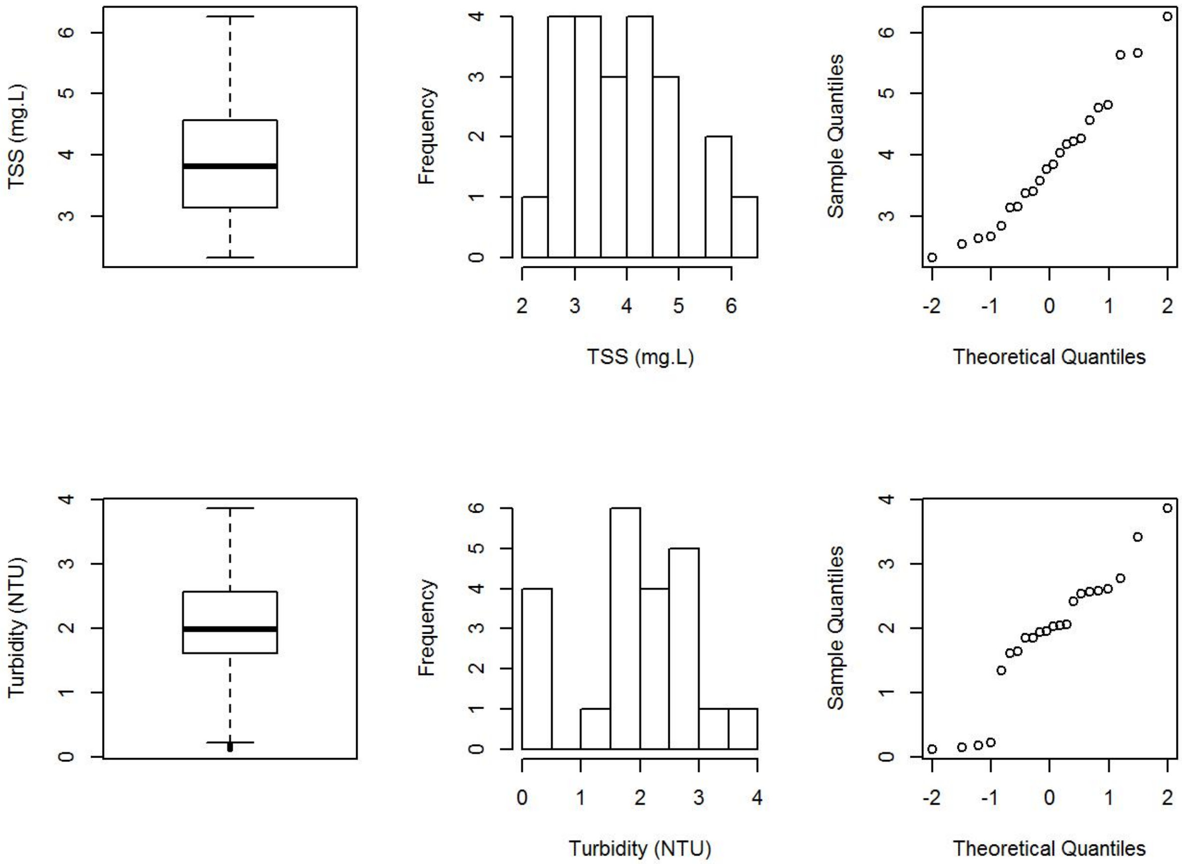

Figure A13-A15. Distributions of Total Suspended Solids (mg/L) and Turbidity (NTU) values in full chemistry dataset.

Additional Variables from Exploratory Statistical Analysis – RARE/NCA Coral Dataset

Additional variables from the exploratory analysis for the RARE/NCA coral dataset are shown below. Note that some variables have been Log10 transformed in order to better meet assumptions of normality.

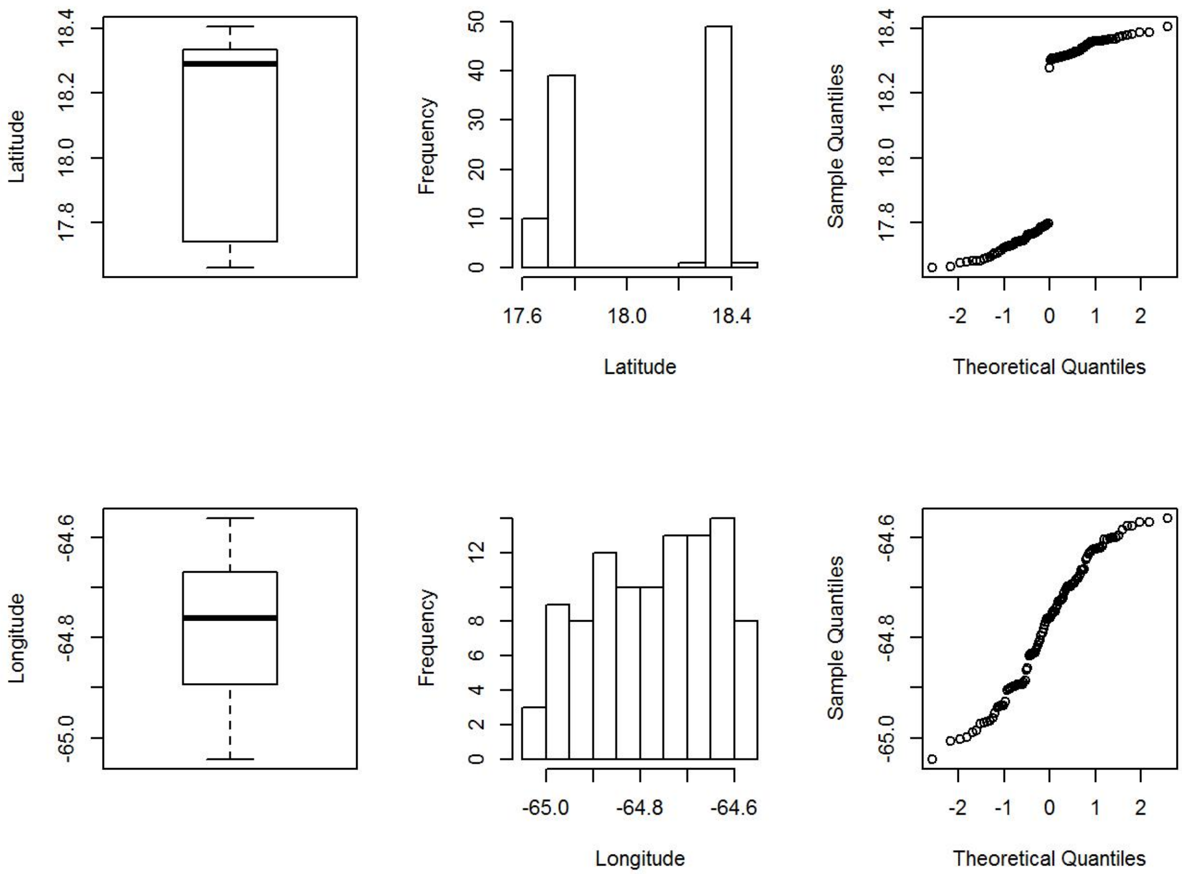

Figure A16-A18. Distributions of measurement latitude and longitude in RARE/NCA coral dataset.

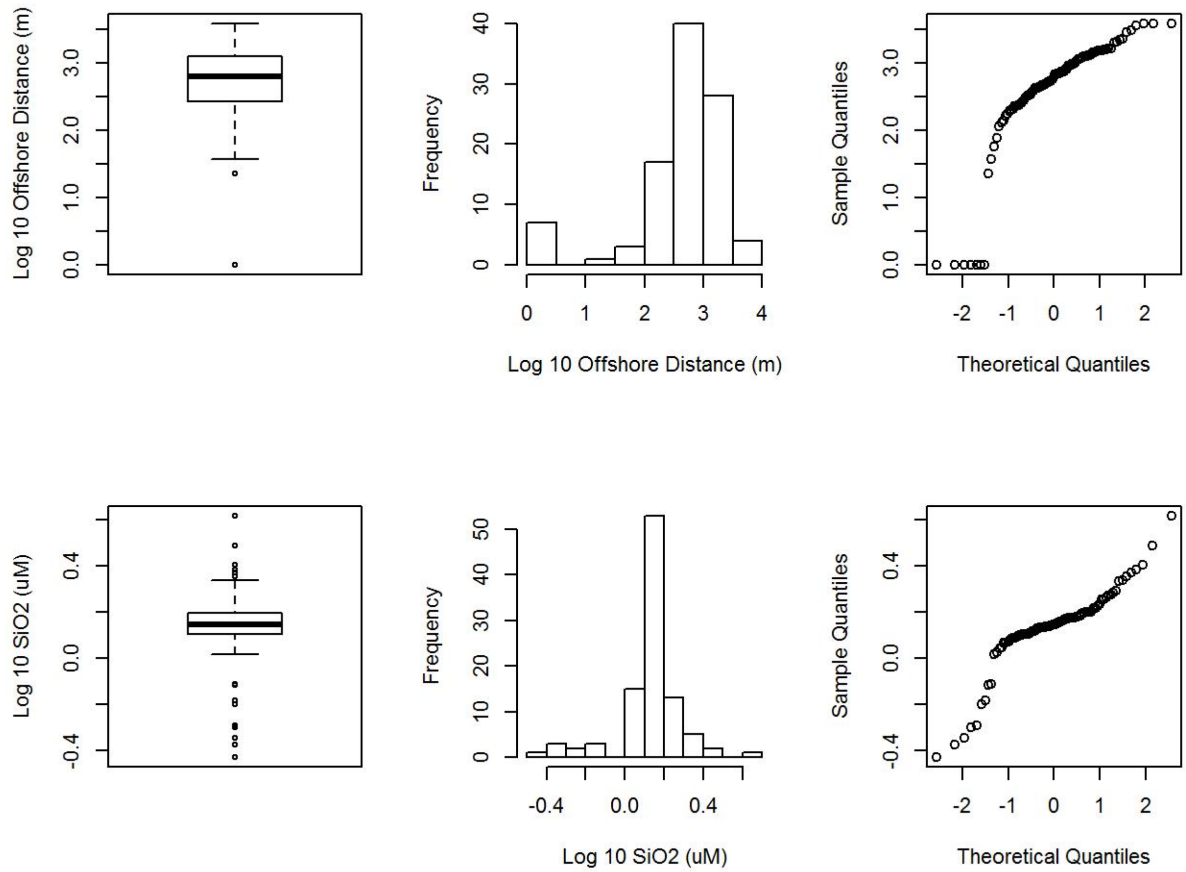

Figure A19-A21. Distributions of measurement distance offshore (m) and silica (uM) values in RARE/NCA coral dataset.