NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7
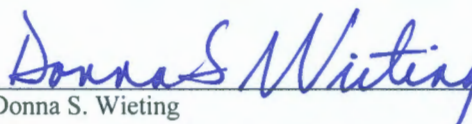# BIOLOGICAL OPINION

Title: Biological Opinion on the Environmental Protection Agency's Approval of Florida's Proposed Water Quality Criteria for 4-Nonyphenol

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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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

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 (a not likely to adversely affect determination) and NMFS concurs with that determination for species under NMFS jurisdiction, consultation concludes informally (50 C.F.R. §402.14(b)).

When consultation is not concluded informally, Section 7(b)(3) of the ESA requires that at the conclusion of consultation, NMFS provide an opinion stating whether the Federal agency's action is likely to *jeopardize the continued existence* of an ESA-listed species or result in *destruction or adverse modification* of designated critical habitat. Specifically:

"Jeopardize the continued existence" 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 CFR 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 CFR 402.02).

If NMFS determines that the action is likely to jeopardize the continued existence of listed species or result in the destruction or adverse modification of designated 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 incidental take is expected, ESA section 7(b)(4) requires NMFS to provide an incidental take statement 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.

The action agency for this consultation is the United States Environmental Protection Agency (EPA). The EPA proposes to approve water quality criteria for 4-nonylphenol proposed by the Florida Department of Environmental Protection (FDEP), pursuant to Section 303(c) of the Clean Water Act, 33 U.S.C. §1313(c).

This biological opinion, and incidental take statement, was prepared by NMFS Office of Protected Resources Endangered Species Act Interagency Cooperation Division ("We") in accordance with section 7(a)(2) of the statute (16 U.S.C. 1536(a)(2)), associated implementing regulations (50 C.F.R. Part 402), and agency policy and guidance.

During consultation we considered the effects of EPA's approval of FDEP's revision of ambient water quality criteria for 4-nonylphenol on the following ESA-listed species and designated critical habitat; green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles, giant manta ray, smalltooth sawfish, shortnose and Atlantic sturgeon, Nassau grouper, elkhorn, staghorn, rough cactus, pillar, lobed star, mountainous star, and boulder star coral, Johnson's seagrass, and the designated critical habitat for elkhorn and staghorn coral, loggerhead sea turtle, smalltooth sawfish, and Johnson's seagrass.

NMFS concluded that the proposed action is not likely to adversely affect for green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtle, smalltooth sawfish, Gulf sturgeon, shortnose or Atlantic sturgeon, Nassau grouper, giant manta ray, oceanic white tip shark, and Johnson's seagrass. While the action is not expected to jeopardize the continued existence of ESA-listed coral species or adversely modify critical habitat designated for elkhorn and staghorn coral, incidental take is anticipated. We include an Incidental Take Statement (ITS) that specifies the impact of the take, Reasonable and Prudent Measures (RPMs) to minimize the impact of the take, and Terms and Conditions to implement the RPMs. A complete record of this consultation is on file at the NMFS Office of Protected Resources in Silver Spring, Maryland.

1.1 Background

EPA's authorities regarding water quality criteria are contained in sections 303(c) and 304(a) of the Clean Water Act. Water quality standards consist of three components: (1) the designated uses of waters, which can include use for public water supplies, propagation of fish and wildlife, recreational, agricultural, industrial and other uses; (2) water quality criteria, expressed in numeric or narrative form, reflecting the condition of the water body that is necessary to protect its designated use, and (3) an antidegradation policy that protects existing uses and provides a mechanism for maintaining high water quality.

Under section 303(c) of the Clean Water Act, the development of water quality standards is primarily the responsibility of States and Tribes, with EPA exercising an oversight role. States and Tribes are required to review their standards every three years and any revisions or new standards must be submitted to EPA for approval. EPA approvals of these standards are federal actions subject to consultation under section 7 of the ESA. Section 7 consultation is required if EPA determines that its approval of any of the standards may affect ESA-listed species or designated critical habitat.

Under section 304(a) of the Clean Water Act, EPA has published recommended water quality guidelines that serve as scientific guidance for use by States or Tribes in establishing and revising water quality criteria. These guidelines are not enforceable requirements, but are

recommended pollutant concentration limits that States or Tribes may adopt as part of their legally enforceable water quality criteria. States or Tribes may propose to adopt alternative scientifically defensible criteria instead of EPA's recommended water quality guidelines (see 40 CFR 131.11(b)). The state must obtain approval of these alternative criteria from EPA before they can be adopted for use.

Section 303(c) of the Clean Water Act contains time frames for EPA to review and either approve or disapprove water quality criteria submitted by a State or Tribe. Once EPA receives the criteria proposed by a state, the agency is required, under the Clean Water Act, to review and approve the criteria within 60 days or disapprove them within 90 days. If disapproved, EPA is required to promulgate the water quality criteria developed under 304(a) to supersede the disapproved State or Tribal criteria. In addition, section 303(c) authorizes EPA to promulgate Federal criteria whenever the Administrator determines that such criteria are necessary to meet the requirements of the Clean Water Act. Regulations implementing section 303(c) are codified at 40 CFR part 131.

1.2 Consultation History

On January 26, 2017, the NMFS Southeast Regional Office received a letter from EPA Region 4 requesting informal consultation and concurrence with its not likely to adversely affect determination for approval of FDEP's proposed 4-nonylphenol criteria. Additional information was requested on January 30, 2017, and again on March 22, 2017. We received a final response on March 22, 2017, and initiated informal consultation that day. On June 1, 2017, EPA Region 4 notified us in writing that the January 26, 2017, letter to initiate consultation mentioned a revision of the FDEP water quality criterion for alkalinity, but that EPA determined revisions to the alkalinity criteria would have no effect for any NMFS species. On August 7, 2017, we provided a letter to EPA explaining that we cannot concur with EPA's determination that approval of FDEP's 4-nonylphenol ambient water quality criteria is not likely to adversely affect ESA-listed species under NMFS's jurisdiction. On August 31, 2017, NMFS Southeast Regional Office received a letter from EPA Region 4 requesting formal consultation on its approval of FDEP's proposed 4-nonylphenol criteria. On January 4, 2018, EPA Region 4 agreed to extend the deadline for delivery of the opinion to February 28, 2018, to allow time to incorporate anticipated data for 4-nonylphenol toxicity to coral that would otherwise potentially trigger reinitiation.

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. Accordingly, the analysis in this opinion evaluates whether the proposed action would directly, or indirectly, adversely affect individual survival or fitness such that the continued existence of ESA-listed populations or species would be jeopardized, or that designated critical habitat necessary for the

conservation of ESA-listed species would be adversely modified or destroyed. Risk hypotheses are statements that organize this analysis by describing the relationships among stressor, exposure, and these environmental values to be protected (assessment endpoints) by placing information on stressors in context of potential responses of species and designated critical habitat. The following risk hypotheses were used to evaluate whether ESA-listed species are likely to respond to 4-nonylphenol at the proposed criteria:

- Exposure to 4-nonylphenol at the proposed criteria will directly affect the survival and fitness of individuals of ESA-listed species under NMFS jurisdiction through:
  - Increased mortality
  - Impaired growth or development
  - Impaired reproduction
- The proposed 4-nonylphenol criteria will indirectly affect the survival and fitness of individuals of ESA-listed species under NMFS jurisdiction through:
  - Consuming prey that have accumulated toxic levels of 4-nonylphenol
  - Reduced quality or extent of habitat due to effects on biological habitat features (e.g., Sargassum, mangroves)
  - Reduced availability and quality of forage due to population-level effects on survival, growth, reproduction of forage species.

This opinion is structured as follows:

*Description of the Proposed Action (*Section 3): We describe the proposed action and those aspects (or stressors) of the proposed action that may have direct or indirect effects on the physical, chemical, and biotic environment. This includes subsections on the *Stressor of the Action* (3.1), *Mitigation to Minimize or Avoid Exposure* (Section 3.2) and *Interrelated and Interdependent Actions (*Section 3.3). *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 use, apart from the action under consideration.

*Action Area (*Section 4): We describe the action area with the spatial extent of those stressors that may have direct or indirect effects on the physical, chemical, and biotic environment.

Species and Designated Critical Habitat that are Not Likely to be Adversely Affected (Section 6): We use two criteria to identify the ESA-listed species or designated critical habitat that are not likely to be adversely affected by the proposed action: exposure to stressors of the action and the probability of response given an exposure. If we conclude that an ESA-listed species or designated critical habitat is not likely to be exposed to the stressors of the action, or, if exposed, is not likely to respond, we must also conclude that the species or designated critical habitat is not likely to be adversely affected by those activities.

Subsections identify *Species that are Not Likely to be Exposed to Waters under Florida's Jurisdiction* (Section 6.1), *Species that are Not Likely to Respond to Exposures at the Proposed*

Criteria (Section 6.2), and *Designated Critical Habitat Not Likely to be Modified by the Proposed Criteria* (Section 6.3). Sections 6.2 and 6.3 apply the risk hypotheses described above.

*Effects of the Action on Species (*Section 7): Applies the risk hypotheses to evaluate the adverse effects of the action on ESA-listed species and designated critical habitat under NMFS jurisdiction that are likely to respond to 4-nonylphenol at the criteria. If adverse effects are indicated for individuals or the essential features, we evaluate whether those responses would affect populations or subpopulations of species or the designated critical habitat (Risk Analysis, Section 9.1).

Status of Species and Designated Critical Habitat Addressed *(*Section 7): We identify the ESA-listed species that are likely to co-occur with and respond to those stressors in space and time and evaluate the status of those species and habitat.

Environmental Baseline (Section 8): We describe the environmental baseline in the action area where potentially adversely affected species occur. The environmental baseline includes 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, impacts of state or private actions that are contemporaneous with the consultation in process.

*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 CFR 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 Effects of the Action, Status of the Species, and the Environmental Baseline to summarize the consequences to ESA-listed species and designated critical habitat under NMFS jurisdiction.

*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 Alternatives (RPAs) to

the action, if any, or indicate that to the best of our knowledge there are no RPAs. See 50 C.F.R. § 402.14.

In addition, we include an ITS (Section 13) that specifies the impact of the take, RPMs to minimize the impact of the take, and Terms and Conditions to implement the RPMs. ESA Section 7(b)(4); 50 CFR 402.14 (i). We also provide discretionary *Conservation Recommendations* (Section 13) that may be implemented by EPA. 50 CFR 402.14 (j). Finally, in the *Reinitiation Notice* (Section 15) we identify the circumstances in which reinitiation of consultation is required. 50 CFR 402.16.

To comply with our obligation to use the best scientific and commercial data available, we collected information identified through searches of Web of Science, scientific publisher databases (e.g., Elsevier), government databases (e.g., EPA's National Service Center for Environmental Publications), and literature cited sections of peer reviewed articles, 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:

- EPA's biological evaluations for FDEP's proposed 4-nonylphenol criteria

- fact sheets and technical support documents

- data from EPA's STOrage and RETrieval database (STORET)

- status reviews, recovery plans, and listing notices for ESA-listed species and designated critical habitat

- reports on the status and trends of water quality

- Data collected under Organization for Economic Co-operation and Development testing guidelines, and

- the best available commercial and scientific information, including peer reviewed research.

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.

3 DESCRIPTION OF THE PROPOSED ACTION

The FDEP's proposed 4-nonylphenol criteria are specific to the aquatic use designation for each water body. Class I waters are designated for use as potable water supplies. Class II waters are suitable for shellfish propagation or harvesting. Class III waters support fish consumption; recreation, propagation and maintenance of a healthy, well balanced population of fish and wildlife. Class III-limited waters support fish consumption; recreation or limited recreation;

and/or propagation and maintenance of a limited population of fish and wildlife. Class III-Limited waters are either:(a) Wholly artificial waterbodies that were created by excavation; or(b) Altered waterbodies that were dredged or filled prior to November 28, 1975 (62-302.400 Florida Administrative Code, F.A.C.). Class IV waters are agricultural water supplies. These are generally located in agriculture areas around Lake Okeechobee. Class V waters are designated for navigation, utility, and industrial use. Currently, there are not any designated Class V bodies of water. FDEP has not proposed 4-nonylphenol criteria for class IV or V waters. EPA proposes to approve FDEP's adoption of the following water quality criteria for 4-nonylphenol:

- Not to exceed 6.6 micrograms 4-nonylphenol per liter in Class I freshwaters
- Not to exceed 1.7 micrograms 4-nonylphenol per liter in Class II freshwaters
- Not to exceed 6.6 micrograms 4-nonylphenol per liter in Class III and Class III-limited predominantly fresh waters
- Not to exceed 1.7 micrograms 4-nonylphenol per liter in Class III and Class III-limited predominantly marine waters

The criteria above are maximum never-to-be-exceeded at any time except within established mixing zones or in accordance with site-specific effluent limitations developed pursuant to Rule 62-620.620 F.A.C. The criteria are based on EPA's 2005 recommended chronic guideline values, which are intended to be implemented as four-day average concentrations not-to-be-exceeded more than once every three years. FDEP's proposed adoption of the chronic criteria as never-to-be-exceeded concentrations is more protective than adopting the criteria as recommended by EPA.

To arrive at the guideline values, EPA screened and evaluated chronic 4-nonylphenol toxicity test results measuring the effects of 4-nonylphenol in ambient water on growth, reproduction, and long term-survival. The freshwater chronic guideline values for 4-nonylphenol were derived using data from seven independent studies. These studies included data for two fish, rainbow trout and fathead minnow and for three invertebrate species, two species of water flea, *Ceriodaphnia dubia* and *Daphnia magna*, and the midge, *Chironomous tentans*. Three of these seven studies were *Daphnia magna* exposures. Only two saltwater chronic toxicity tests were used for the 4-nonylphenol guideline for predominantly marine waters, and both these tests were for the mysid shrimp, *Americamysis bahia*.

3.1 Stressor of the Proposed Action

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. Discharges of 4-nonylphenol from publically owned treatment works are not expected because 4-nonylphenol and 4-nonylphenol ethoxylates have been phased out of household products. Other uses of

nonylphenol ethoxylate surfactants, such as dust-control agents and deicers, lead to direct release to the environment. 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 (USEPA 2017). In the environment, 4-nonylphenol is persistent and accumulates in sediment to concentration several orders of magnitude greater than concentrations in water. Bottom-feeding fish can be significantly exposed to these persistent and toxic compounds (Brooke 1993a, USEPA 2010). Half-life in water and sediment is determined by ambient conditions. Nonylphenol accumulates in sediment. Half-lives have been reported to range from 1.1 to 99 days in sediment (Reviewed by Mao et al. 2012) and from 28 to 104 days (Maguire 1999) both reports indicated that persistence was reduced by increased light intensity and the presence of microorganisms (Reviewed by Mao et al. 2012).

Most toxicants in ambient water, 4-nonylphenol among them, are not readily absorbed through the skin of fish, marine mammals, reptiles, or invertebrates, but can be absorbed through the skin of amphibians. The pathway for exposure to pollutants in aquatic toxicity tests is uptake via the gills in freshwater and saltwater fish and aquatic invertebrates. For saltwater fish, exposure to toxicants in water also occurs through ingestion. Most marine fish need to maintain a lower concentration of solutes in their body fluids than saltwater and they must therefore "osmoregulate" by drinking water and excreting solute. Most marine invertebrates have the same internal concentration of solutes as the water they live in and do not osmoregulate (Larsen et al. 2014).

Accumulation rates vary, depending on exposure duration, concentration, species, and lipid content (Hecht 2002, Hu et al. 2005). Dietary exposures result in accumulation of 4-nonylphenol, but trophodynamic studies indicate that 4-nonylphenol is metabolized and does not biomagnify (i.e., increase in concentration from prey to predator) in the food web (Hu et al. 2005, Diehl et al. 2012, Korsman et al. 2015). The EPAs 2005 water quality criteria document reported bioconcentration factors ranging from 4.7 to 344 (Ward and Boeri 1991b, Brooke 1994 After EPA 2005) in freshwater and 78.5 to 2,168 in salt water (Ekelund et al. 1990). Accumulated 4-nonylphenol may be transferred to offspring (Thibaut et al. 2002) with concentrations in eggs increased over maternal levels 30-100 fold (Ishibashi et al. 2006). Persistence and global distribution is indicated by the presence of 4-nonylphenol in organisms living among marine debris. Plastic marine debris contains 4-nonylphenol, but also absorbs 4-nonylphenol from ambient water. The presence of marine debris can result in enhanced exposures through the creation of 4-nonylphenol-concentrated microhabitats (e.g., poorly flushed areas, relatively sheltered areas of reefs and rocky substrates) or incidental ingestion (Gassel et al. 2013, Guerranti et al. 2014, Hamlin et al. 2015, Staniszewska et al. 2016). While the proposed criteria are intended to limit exposure of aquatic organisms to harmful levels of 4-nonylphenol, the presence of any 4-nonylphenol in water and the dynamic flux between ambient water, sediment, and debris may result in fluctuating microhabitat exposures to concentrations above the proposed criteria in otherwise 4-nonylphenol-compliant waters.
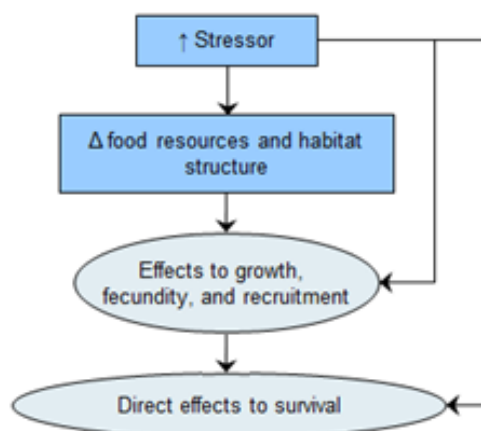
Figure 1. Generalized pathway from stressor to effects and the relationship of effects on species.

In the environment, 4-nonylphenol is a proximate stressor. A proximate stressor is the actual toxicant, physiological impact, or resource limitation most directly linked to a biological response. Physiological responses that result in effects expressed in individuals translate to effects on the interdependence of species (generalized in Figure 1). Proximate stressors that directly affect organism survival and lifecycle processes (i.e., effects to growth, fecundity, and recruitment) can also affect the survival and lifecycle processes of prey species upon which they rely.

Toxicity tests show that 4-nonylphenol disrupts endocrine systems by mimicking the female hormone 17β-estradiol. Exposure of aquatic animals resulted in abnormal gonad development, changes in reproductive behavior, altered sex ratio of offspring, and the production of yolk proteins (vitellogenin) by immature male fish. Vitellogenin induction in fish by 4-nonylphenol at ambient fresh and salt water occurred concentrations ranging from 5-100 micrograms per liter (Hemmer et al. 2002, Zhang et al. 2005a, Ishibashi et al. 2006, Arukwe and Roe 2008) and resulted in altered sex ratios after dietary exposures as low as 1 milligrams per kilogram feed (Demska-Zakes and Zakes 2006). Vitellogenin is an egg yolk protein produced by mature females in response to 17-β estradiol.

Vitellogenin is a robust biomarker of 4-nonylphenol exposure and potential to affect fitness, but without concurrent indicators of effects on fitness, a linkage between the intensity of the response and consequences to the survival and fecundity of individuals is difficult. Ishibashi et al. (2006) reported vitellogenin induction and reduced egg production and fertility after exposure of medaka to 100 micrograms 4-nonylphenol per liter for 21 days. Tilapia gonad development and, sperm abnormalities, and intersex (the presence of oocytes in the testes) after two months of exposure to the same concentration (Ali et al. 2014). A retrospective analysis of an Atlantic salmon population crash implicated 4-nonylphenol, applied as an adjuvant in a series of pesticide applications in Canada, as the causal agent (Fairchild et al. 1999, Brown and Fairchild 2003). Additionally, processes involved in sea water adaptation of salmonid smolts are impaired by 4-

nonylphenol (Madsen et al. 2004, Jardine et al. 2005, Luo et al. 2005, McCormick et al. 2005, Lerner et al. 2007a, Lerner et al. 2007b).

While these data are not for vertebrate species that are present in Florida, they establish 4-nonylphenol as a persistent pollutant with endocrine disrupting properties, providing a plausible mechanism for fitness effects and survival in the wild, while providing a broad sense of its potency in causing such effects. Data specific to ESA-listed species found in Florida or closely related surrogates are rare (e.g., data for one grouper species to infer effects on another species of grouper).

The importance of estrogen and vitellogenin, and therefore implications of 4-nonylphenol effects, is well established for fish and reptiles. Our understanding of role of estrogens in coral species is a growing area (Tarrant 2005). The hormone 17β-estradiol occurs 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 of rice coral resulting in 29 percent fewer egg-sperm bundles and growth rates of finger coral fragments reduced 13 to 24 percent (Tarrant et al. 2004). Further evidence for streroidal 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 for that an estrogen mimic such as 4-nonylphenol would have reproductive effects on coral, mechanism (s) for such effects have not been established.

Similarly, while studies have indicated 4-nonylphenol toxicity to plants (Bokern and Harms 1997, Bokern et al. 1998, Domene et al. 2009, de Bruin et al. 2016, Esteban et al. 2016, Zhang et al. 2016), the mechanisms for these responses are not fully understood. Esteban et al. (2016) proposed that the allelopathic properties of the active hydroxyl groups of phenolic compounds like 4-nonylphenol potentially caused germination impairment by mimicking natural allelochemicals and/or phytohormones.

Since we are interested in whether the proposed criteria may result in adverse effects to ESA-listed species under NMFS jurisdiction in Florida, we would want to know whether adverse effects have been reported at concentrations below the proposed criteria. Ideally, the available toxicity data would report, or allow us to determine, the threshold exposure concentration at which a response would not occur or would be insignificant in ESA-listed species under NMFS jurisdiction. However, toxicity tests are not performed on ESA-listed species, so species commonly used in laboratories must serve as surrogates representing the species of interest. Further, most toxicity studies do not identify response threshold concentrations.

Most of the available data on the effects of 4-nonylphenol are summarized in EPA's Ecotoxicology database (ECOTOX). In evaluating the data, studies reporting study interferences(Schoenfuss et al. 2008) or where test organisms were obtained from a contaminated environment (Chaube, et al. 2013) were eliminated. The database identifies the highest exposure concentration that did not differ significantly from controls, the no observed

effects concentration (NOEC), and the lowest concentration that differed significantly from controls, the lowest observed effect concentration (LOEC). The NOECs and LOECs are not ideal measures of effects because they are influenced by study design (e.g., distribution and number of concentrations tested). Depending on exposures tested and underlying variability in responses, the LOEC may actually result in a 30 percent difference in response from controls. The distribution of NOECs and LOECs showing the variability among species and response types among ECOTOX data are presented in Table 1. Exposure duration and life stages used in the individual toxicity studies contributes to the variability in the data presented in this table.

We have data for species that do not occur in the United States among the data used to determine whether ESA-listed species or biological features of designated critical habitat are likely to respond to 4-nonylphenol exposures at the proposed criteria. Use of nonresident species is consistent with the Stephen et al. (1985) *Guidelines for Deriving Numerical National Water Quality Criteria.* The guidelines indicate that data obtained with non-resident species may be used to provide auxiliary information but should not be used to develop a criterion. In our analysis, the data are used to evaluate whether surrogate species are likely to respond to exposure to 4-nonylphenol at or below FDEP's proposed 4-nonylphenol criteria. The guidelines used to develop the criteria are intended to protect most aquatic ecosystems under most but not all circumstances. Stephen et al. (1985) states:

> "*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. If acceptable data are available for a large number of appropriate taxa from an appropriate variety of taxonomic and functional groups, 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.*"

EPA's water quality guidelines, and state water quality criteria based on those guidelines, therefore cannot be assumed to be exposure concentrations that would not adversely affect threatened and endangered species. EPA worked with USGS to conduct studies comparing the sensitivity of threatened and endangered species relative to laboratory species commonly used in toxicity tests (Sappington et al. 2001, Besser et al. 2005, Dwyer et al. 2005a, Dwyer et al. 2005b). These studies suggest applying generic adjustment factors of about 0.5 to EPA water quality guidelines for the protection of ESA-listed species. However, the proposed adjustment factors are based on studies using fish. Differences in pollutant sensitivities could vary widely among taxonomic families within some species groups. For example, the LOECs for coral species calicoblast cells[1] exposed to 2,4,6-trinitrotoluene (TNT) were 500 micrograms per liter for *Porites divaricate,* a hardy species of the taxonomic family Poritidae (C. Woodley, NOAA National Center for Coastal Science, pers. comm. to P. Shaw-Allen, NMFS OPR, November 2,

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[1] Calicoblast cells secrete the calcium carbonate matrix skeleton of stony corals.

2017), and 0.5 micrograms per liter for a more sensitive species from the Pocilloporidae family, *Pocillopora damicornis* (Woodley and Downs 2014). The concentrations at which half of exposed cells died, or the LC50, were 716 and 16 micrograms TNT per liter, respectively.

In many cases, data for 4-nonylphenol effects on the species of interest or suitable surrogate species (i.e., within the same taxonomic family) are not available in ECOTOX or in data found through the literature search efforts conducted for this assessment. In addition to these sources of toxicity information, EPA's Web-based Interspecies Correlation Estimation application predicts acute toxicity for a species of interest based on what is known about the sensitivity of that species, or the species' genus or family, relative to other more commonly tested species. For example, if we have toxicity data on the responses of both species to several toxicants, we can determine the extent to which Atlantic sturgeon are consistently more (or less) sensitive to those toxicants than rainbow trout. These models are only available for the three ESA-listed species of sturgeon found in Florida waters. This information can be used to extrapolate the expected sensitivity of Atlantic sturgeon to an untested toxicant when we only have data for rainbow trout responses to that toxicant. In this way, EPA's model uses existing toxicity data for rainbow trout response to 4-nonylphenol to predict the response of Atlantic sturgeon to 4-nonylphenol. Unfortunately, the application has limited use for this consultation. There are no prediction models for reptiles, giant manta ray, Nassau grouper, corals, or plants.

Table 1. No observed effect data and lowest observed effect data for aquatic organisms.

Species	Response	NOEC	LOEC	SOURCE[a]
Fresh Water Fish				
Atlantic Salmon	Growth & Morphology	73.9		(Lerner et al. 2007a)*
Bluegill	Mortality	74 (n=3, 59.5-86.5)	193 (n=3, 126-243)	(Brooke 1993a, Liber et al. 1999)
Buffalo Springs Tilapia	Growth	30		(Bin-Dohaish 2012)
	Morphology	3.5	15	
Rare Minnow	Growth	26.17 (n=6, 18.53-30)		(Zha et al. 2007, Zha et al. 2008)
	Morphology	18.77 (n=12, 3-30)	10.436 (n=5, 4.52-18.53)	
	Mortality and Reproduction	18.53		
Duckweed	Growth	901	2080	(Brooke 1993a)
European Physa	Mortality	75 (n=2, 50-100)	175 (n=2, 100-250)	(Hong and Li 2007)
Fathead Minnow	Growth	23	23	(Ward and Boeri 1991b)
	Mortality	42.8 (n=5, 7.4-83.1)	75.57 (n=7, 14-230)	(Ward and Boeri 1991b, Brooke 1993a)
Japanese Medaka	Development	8.84 (n=2, 6.08-11.6)	17.55 (n=2, 11.6-23.5)	(Seki et al. 2003)
	Growth	43.74 (n=5, 11.6-61.2)	34.1 (n=2, 23.5-44.7)	(Seki et al. 2003, Ishibashi et al. 2006)
	Morphology	79.62 (n=10, 2.9-184)	59.67 (n=7, 8.7-184)	(Kang et al. 2003, Balch and Metcalfe 2006, Ishibashi et al. 2006)
	Mortality	44.7		(Seki et al. 2003)
	Reproduction	65.15 (n=22, 16.5-184)	97.37 (n=15, 61.2-200)	(Kang et al. 2003, Ishibashi et al. 2006)
Pearl Mullet	Reproduction	200		(Kaptaner and Unal 2011)
Rainbow Trout	Growth	8.85 (n=6, 6-23.1)	17.42 (n=6, 10.3-53)	(Brooke 1993a, Spehar et al. 2010)
	Morphology	10.15 (n=2, 2.3-18)	18	(Shelley et al. 2012a)
	Mortality	62.15 (n=4, 10.3-114)	23.1	(Brooke 1993a, Spehar et al. 2010)
Southern Platyfish	Morphology	1280		(Kinnberg et al. 2000)
Zambezi Barbel	Morphology		50	(Sayed et al. 2012)
Zebra Danio	Development	16.67 (n=3, 10-30)	100	(Hill and Janz 2003, Lin and Janz 2006)
	Growth & Morphology	100		(Hill and Janz 2003, Lin and Janz 2006)
	Mortality	86 (n=5, 30-100)	100	(Hill and Janz 2003, Lin and Janz 2006, Xu et al. 2013)
	Reproduction	82.5 (n=4, 30-100)	100	(Hill and Janz 2003, Weber et al. 2003, Lin and Janz 2006)

Species	Response	NOEC	LOEC	SOURCE[a]
Fresh Water Invertebrates				
Freshwater Prawn	Mortality	100	250	(Hong and Li 2007)
Midge	Growth	80 (n=3, 21-143)	147 (n=3, 39-252)	(England and Bussard 1993)
	Mortality	83.33 (n=6, 39-143)	143 (n=4, 81-252)	(England and Bussard 1993, Kahl et al. 1997)(Martínez-Paz et al. 2014)
Scud	Development		0.05	(Geffard et al. 2010)
	Reproduction	5		
Shrimp	Mortality	250	350	(Hong and Li 2007)
Water Flea	Development	57.37 (n=10, 12.9-100)	100	(Baer and Owens 1999, Gibble and Baer 2003, Zhang et al. 2003, Campos et al. 2012)
	Growth	55.84 (n=17, 10-116)	69 (n=10, 10-215)	(Brooke 1993a, Comber et al. 1993, Gibble and Baer 2003, Spehar et al. 2010, Campos et al. 2012)
	Morphology		100	(Gibble and Baer 2003)
	Mortality	87.19 (n=14, 10-215)	140.07 (n=11, 10-500)	(Brooke 1993a, Comber et al. 1993, Hong and Li 2007, Spehar et al. 2010, Campos et al. 2012)
	Reproduction	59.62 (n=34, 1-215)	161 (n=14, 10-500)	(Brooke 1993a, Baer and Owens 1999)(Comber et al. 1993, Kopf 1997, Gibble and Baer 2003, Zhang et al. 2003, Spehar et al. 2010, Campos et al. 2012)
Zebra Mussel	Growth	500		(Quinn et al. 2006)
	Mortality	1000	5000	
Salt Water Fish				
Atlantic Cod	Growth & Morphology	29		(Martin-Skilton et al. 2006)
Sheepshead Minnow	Mortality	240	420	(Ward and Boeri 1990a)
Turbot	Growth & Morphology	29 (condition, weight)	29 (length)	(Martin-Skilton et al. 2006)
Salt Water Invertebrates				
Australian Barnacle	Development	10 (renewal, larvae)	0.6 (static, nauplii)	(Billinghurst et al. 2001)
Harpacticoid Copepod	Development	0.3 (n=4, 0.01-1)	3 (n=4, 0.1-10)	(Marcial et al. 2003)
	Mortality	50 (n=3, 10-130)		(Marcial et al. 2003, Lee et al. 2007)
	Reproduction	10	27.88 (n=8, 20-41)	(Bechmann 1999, Marcial et al. 2003)

Species	Response	NOEC	LOEC	SOURCE[a]
Opossum Shrimp	Development	23 (n=17, 3-30)	26 (n=5, 10-30)	
	Growth	4.7 (n=3, **0.3**-10)	12.57 (n=3, **1**-30)	
	Morphology	3	10	(Ward and Boeri 1991a, Hirano et al. 2009)
	Mortality	18.35 (n=2, 6.7-30)	9.1	
	Reproduction	6.7	9.1	

[a]Data from EPA's Ecotoxicology database.
[b]Values within criteria, i.e., less than 1.7 µg/L for marine waters and less than 6.6 for freshwaters, are in bold.

3.2 Conservation Measures to Avoid Exposure

The EPA's action is an approval of Florida's proposed water quality criteria for 4-nonylphenol to protect aquatic life from adverse effects due to exposure to this chemical. Once approved, as a state with delegated authority under the Clean Water Act, Florida will implement the criteria in establishing effluent limits for discharge permits and in identifying impaired waters. The only actions within EPA's authority that would minimize or avoid exposure are the disapproval of the 4-nonylphenol criteria with either the requirement that Florida revise its criteria or EPA promulgation of alternative 4-nonylphenol criteria.

3.3 Interrelated and Interdependent Actions

During consultation we did not identify any additional interdependent or interrelated actions for EPA's approval of FDEP's proposed 4-nonylphenol criteria.

4 ACTION AREA

The action area is defined by regulation as "all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action" (50 CFR 402.02). The proposed 4-nonylphenol criteria apply to all freshwater in the state of Florida and all of Florida's territorial marine waters (Figure 2 and 3). As a land sourced pollutant, waters extending beyond Florida's jurisdiction are not expected to exceed concentrations observed in the territorial waters of Florida. The action area considered consultation was therefore the territorial waters of Florida.

The Florida Gulf of Mexico continental shelf is a wide, low energy area extending a distance of approximately 900 kilometers (about 486 nautical miles) in length from the western Panhandle to the extreme Southwest margin off the Florida Keys. The shelf is a carbonate platform that is approximately 100 kilometers (about 54 nautical miles) in width off St. Augustine tapering to less than two kilometers (about one nautical mile) by West Palm Beach to the south. Florida state waters are defined as extending from shore to three nautical miles (about 5.5 kilometers) on the Atlantic coast and from shore to nine nautical miles (about 16.7 kilometers) on the Gulf coast (Figure 3, Adapted from Robbins et al. 2007). The depth of state waters is important in identifying which ESA-listed species under NMFS jurisdiction may be exposed to the proposed water quality criteria. Florida state waters in the Gulf of Mexico off the panhandle and the Dry Tortugas reach a depth to 30 meters and are at ten to 20 meters deep along the Gulf coasts of the peninsula and Florida Keys. State waters are deepest, to 190 meters, in the Atlantic coastal plain off the southern coasts of the Florida Keys. Moving northward along the Atlantic coast of the peninsula to West Palm Beach, the continental slope of the Atlantic coastal plain is relatively close to shore and state waters reach depths to about 170 meters. State waters north of West Palm Beach along the peninsula are shallow, at depths of ten to 20 meters.

Figure 2. Major surface waters in Florida[2].

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[2] Adapted from http://www.thepalmbeachtimes.com/TravelNavigator/FloridaMileage.php. Added north arrow, labeled Perdido River, Ten Thousand Islands and Indian River, recolored and moved scale bar for legibility.
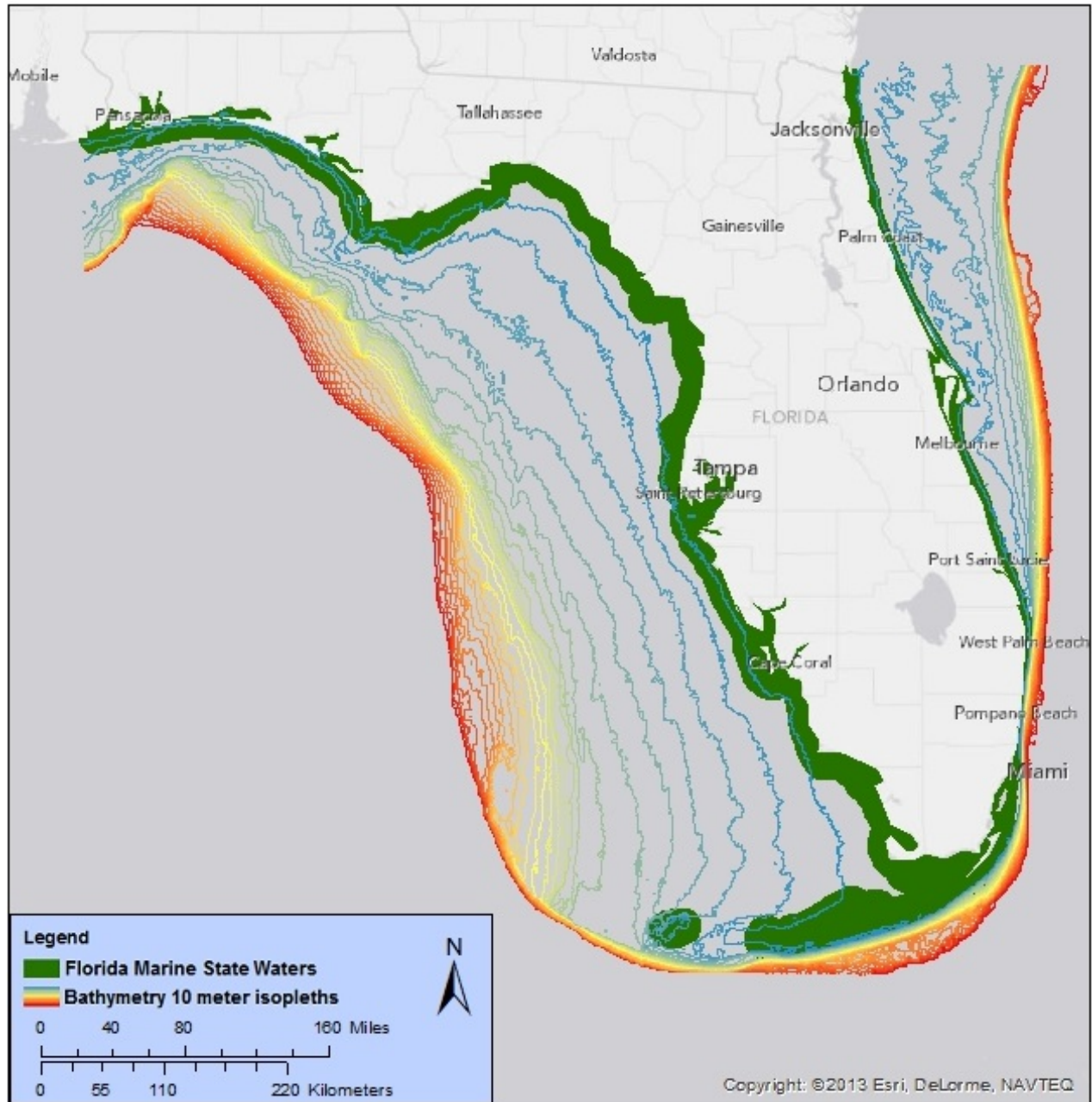
Figure 3. Florida state waters and depth of waters along the coast of Florida (ten meter isopleths).

5 SPECIES WITH RANGES THAT OVERLAP THE ACTION AREA

Table 2 identifies the ESA-protected species and designated critical habitat under NMFS jurisdiction that have ranges overlapping with waters under Florida's jurisdiction.

Table 2. Species protected under the ESA with ranges that overlap with waters under Florida's jurisdiction.

Species	ESA Status	Designated Critical Habitat	Recovery Plan
Cetaceans			
Blue Whale (Balaenoptera musculus)	E – 35 FR 18319	--	1998
Fin Whale (Balaenoptera physalus)	E – 35 FR 18319	--	75 FR 47538
North Atlantic Right Whale (Eubalaena glacialis)	E – 35 FR 18319 & 73 FR 12024	81 FR 4837	70 FR 32293
Sei Whale (Balaenoptera borealis)	E – 35 FR 18319	--	2011
Sperm Whale (Physeter macrocephalus)	E – 35 FR 18319	--	75 FR 81584
Sea Turtles			
Green sea turtle (Chelonia mydas)	E – 43 FR 32800	63 FR 46693	63 FR 28359
Hawksbill sea turtle (Eretmochelys imbricata)	E – 35 FR 8491	63 FR 46693	57 FR 38818
Kemp's Ridley sea turtle (Lepidochelys kempii)	E – 35 FR 18319	--	75 FR 12496
Leatherback sea turtle (Dermochelys coriacea)	E – 61 FR 17	44 FR 17710	63 FR 28359
Loggerhead sea turtle (Caretta caretta) – Northwest Atlantic Distinct Population Segment	E – 76 FR 58868	78 FR 39856	63 FR 28359
Fish			
Smalltooth Sawfish (Pristis pectinata)	E – 68 FR 15674	74 FR 45353	74 FR 3566
Giant Manta Ray (Manta birostris)	T – 83 FR 2916	--	--
Oceanic Whitetip Shark (Carcharhinus longimanus)	T – 83 FR 4153	--	--
Shortnose Sturgeon (Acipenser brevirostrum)	E – 32 FR4001	--	63 FR 69613
Atlantic Sturgeon (Acipenser oxyrinchus oxyrinchus) South Atlantic Distinct Population Segment	E – 77 FR 5914	82 FR 39160	--
Gulf Sturgeon (Acipenser oxyrinchus desotoi)	T – 56 FR 49653	68 FR 13370	1995
Nassau Grouper (Epinephelus striatus)	T – 81 FR 42268	--	report
Corals			
Elkhorn Coral (Acropora palmata) and Staghorn Coral (Acropora cervicornis)	T – 71 FR 26852	73 FR 72210	--
Rough Cactus Coral (Mycetophyllia ferox) Pillar Coral (Dendrogyra cylindrus) Lobed Star Coral (Orbicella annularis) Mountainous Star Coral (Orbicella faveolata) Boulder Star Coral (Orbicella franksi)	T – 79 FR 54122	--	--
Marine Plant			
Johnson's Seagrass (Halophila johnsonii)	T – 63 FR 49035	65 FR 17786	2002

6 SPECIES NOT LIKELY TO BE ADVERSELY AFFECTED

We use two criteria to identify the ESA-listed species or designated critical habitat under that are likely to be adversely affected by a proposed action, or by the effects of activities that are interrelated to or interdependent with a 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 under NMFS jurisdiction is not likely to be exposed to the proposed activities, we must also conclude that the species or designated critical habitat is not likely to be adversely affected by those activities.

The second criterion is the probability of a response given exposure. The ESA-listed species or designated critical habitat under NMFS jurisdiction that are exposed to a potential stressor but are likely to be unaffected by the exposure are also not likely to be adversely affected by the proposed action.

We applied these criteria to the risk hypotheses species ESA-listed in Section 2 and we summarize our results below.

In addition, 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. That means the ESA-listed species may be expected to be affected, but not harmed or harassed.

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 ESA-listed species), but it is very unlikely to occur.

6.1 Species Not Likely to be Exposed to Waters Under Florida's Jurisdiction

Species with ranges that surround Florida, but do not frequent waters under Florida's jurisdiction, are not expected to be affected by EPA's approval of the proposed 4-nonylphenol criteria. While blue and Sei whales may be found along the continental slope, they are extremely rare in Florida waters. Fin whales are found in deep, offshore waters of all major oceans, primarily in temperate to polar latitudes, and less commonly in the tropics. Sperm whales tend to inhabit areas with a water depth of 1968 feet (600 meters) or more, and are uncommon in waters

less than 984 feet (300 meters) deep. Oceanic white tip sharks occur in the open ocean, well offshore along the southeastern coast of the United States. The essential fish habitat for this species, that is to say: *those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity*, is adjacent to, but does not overlap with Florida state waters. Exposures of blue, fin, sei, and sperm whales, and oceanic white tip shark to waters under Florida's regulatory jurisdiction is expected to be extremely unlikely, therefore exposure of these species to waters meeting FDEP's proposed 4-nonylphenol criterion for predominantly marine waters is considered discountable. Therefore, NMFS concluded that EPA's approval of FDEP's proposed 4-nonylphenol criterion for predominantly marine waters are not likely to adversely affect blue, fin, sei, and sperm whales and the oceanic white tip shark. These species are not discussed further in this opinion.

As an aquatic toxicant, 4-nonylphenol is not readily absorbed through mammalian skin, so exposure would require direct uptake from the water column through membranes that are in contact with ambient water, ingesting water, or indirect uptake through ingesting organisms that have accumulated 4-nonylphenol. Understanding that whales do not drink seawater[3] and that the North Atlantic right whale birth its calves, but does not feed in Florida waters, their exposures would be insignificant. Therefore, NMFS concluded that EPA's approval of FDEP's proposed 4-nonylphenol criterion for predominantly marine waters are not likely to adversely affect the North Atlantic right whale. The North Atlantic right whale will not be discussed further in this opinion.

6.2 Species Not Likely to Respond to 4-Nonylphenol Exposures at the Proposed Criteria

The ESA-listed species under NMFS jurisdiction that are exposed to waters where the 4-nonylphenol criteria will be applied will not necessarily be adversely affected, directly or indirectly, to these exposures. Toxicity data from the literature or summarized from ECOTOX in Table 2 are applied to the risk hypotheses identified in Section 2 to determine whether or not species are likely to respond to exposures at the criteria.

6.2.1 Sea Turtles

Sea turtles breathe air. Since they do not have gills, their only exposures to 4-nonylphenol in sea-water are through drinking the sea-water and limited absorption through mucous membranes. Indirect exposures occur through ingestion of food that has accumulated pollutants. The ECOTOX does not include data on reptiles exposed to 4-nonylphenol, so studies from the open literature were used in this assessment.

Direct Toxicity

Toxicity data are not available for exposure of green, hawksbill, Kemp's ridley, leatherback, or loggerhead sea turtles to 4-nonylphenol in ambient water. As air breathing species, direct toxicity

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[3] Whale osmoregulation employs genetic and allometric adaptations such as increased filtration rates, urine volume, and kidney size along with high solute levels in urine and plasma (Kjeld 2003, Birukawa et al. 2005).

under the 4-nonylphenol criterion for predominantly marine waters would have to occur as a result of ingesting seawater. Both the aquatic and terrestrial data in ECOTOX did not contain NOECs for ingestion of 4-nonylphenol in water by reptiles. One terrestrial study found in the open literature did evaluate effects in a reptile species resulting from exposure to 4-nonylphenol in drinking water and diet. While the study did not evaluate mortality or impaired growth or development, it suggests such effects are unlikely.

Combined diet and drinking water exposures leading to induction of vitellogenin, impairment of spermatogenesis, and gonad abnormalities were reported in the Italian wall lizard (Verderame et al. 2011, Verderame and Limatola 2015). Histopathological changes in gonads, including visual evidence of impaired spermatogenesis, were not quantified. Vitellogenin in 4-nonylphenol-treated males reached about half that of reproductive females, while vitellogenin was undetectable in untreated males. Exposures to food and drinking water were ad libitum. The dietary 4-nonylphenol exposure was intended to simulate surface concentrations on invertebrate prey sprayed with 4-nonylphenol at a concentration of 2,500,000 micrograms per liter. Drinking water was dosed with 500,000 micrograms 4-nonylphenol per liter, about 294 thousand times higher than the proposed 4-nonylphenol marine criterion.

In contrast, data for drinking water 4-nonylphenol exposures in birds, which are commonly used as a sensitive surrogate species group for reptiles in risk assessments, suggests dose dependent impairment of reproductive ability at concentrations closer to the proposed 4-nonylphenol standards. Fertilization rates in Japanese quail were significantly reduced relative to controls at concentrations as low as 0.1 micrograms 4-nonylphenol per liter, 14 day survival rates were reduced at 1 microgram per liter, and reduced hatchability occurred at 10 micrograms per liter drinking water (Cheng et al. 2017).

The exposures in the Verderame studies were extremely high, but did not result in mortality. While estrogenic effects were reported, these cannot easily be placed in context of potential effects at the proposed 4-nonylphenol marine criterion. These data only tell us that reproductive responses may occur in reptiles as a result of 4-nonylphenol exposures at intensities resembling terrestrial pesticide applications. Meanwhile the data for Japanese quail indicate effects on reproduction resulting from drinking water exposures at concentrations below or near the proposed marine criterion.

The physiological differences and similarities between birds and reptiles as well as some characteristics of sea turtles must be taken into consideration when interpreting these data. The most obvious physiological difference among these species is that reptiles are cold blooded while birds are warm-blooded and therefore differ in metabolic rates. A high physiological metabolic rate increases relative food and water intake (and thereby dietary exposure) as well as toxicant metabolism. When differences in size are accounted for, warm-blooded mammals have higher metabolic rates than reptiles (Krogh 1916, Else and Hulbert 1981). The field metabolic rate for non-passerine birds is reported to be comparable to mammals, with both species groups having energetic costs nearly 17 fold that of a desert reptile (Nagy 1987). It has been argued that sea

turtles have a more active lifestyle relative to other reptiles, and that this higher activity would result in higher metabolic rates. For example, a green turtle's pectoral muscle may be as much as 7°C above water temperature when the animal is vigorously swimming (Standora et al. 1982). However, a comparative analysis accounting for size, ambient temperature, and relative activity status indicated that the metabolic rates of sea turtles do not differ from other reptiles and that birds and mammals have a metabolic rate that is five to ten times higher than reptiles (Willard 2013). In addition to differences in adult metabolic rates, reptile embryos grow more slowly than bird embryos and also have slower metabolic rates (Vleck and Hoyt 1991, Ricklefs and Starck 1998).

NMFS takes these data to indicate that reptiles can be affected by 4-nonylphenol, but are expected to be much less sensitive to the effects of 4-nonylphenol than the avian surrogate species (i.e., Japanese quail). Data for gilled species (i.e., fish, aquatic invertebrates, mollusks) and the expected exposure intensities in sea turtles relative to gilled species are also taken into consideration when evaluating the proposed 4-nonylphenol criterion for predominantly marine waters. When gilled species continuously pass water over their gills to oxygenate their blood and regulate ion balance, contaminants like 4-nonylphenol in the ambient water partition into the bloodstream. In contrast, while sea turtles are immersed in seawater, contaminants like 4-nonylphenol do not readily pass through their shell and skin into the body. Unlike gilled species, sea turtle exposures are not continuous because they do not drink continuously.

In summary:

- Mortality did not occur in a reptile species exposed to 4-nonylphenol in drinking water at concentrations many orders of magnitude higher than the proposed criterion,

- The mean NOECs for marine fish species are one or more orders of magnitude larger than the proposed criterion, and

- The net exposure of sea turtles to 4-nonylphenol in water at the proposed criterion is expected to be much lower than that of marine fish.

Taken together, direct mortality due to exposure to ambient concentrations of 4-nonylphenol at the proposed criterion for predominantly marine waters is extremely unlikely such that any effects are discountable.

While uncertainties remain regarding whether reptile exposures would result in significant sublethal reproductive responses affecting growth, development, and reproduction (fitness) at or below the marine criterion, NMFS believes such responses would be extremely unlikely such that these effects are discountable. Therefore, NMFS concluded that EPA's approval of the proposed criterion is not likely to adversely affect for the survival of ESA-listed green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles.

Indirect Effects

Consuming prey that have accumulated toxic levels of 4-nonylphenol. Indirect toxic effects of the proposed 4-nonylphenol marine criterion to sea turtles would result from ingestion of forage species that have accumulated 4-nonylphenol from ambient water. The combined dietary and drinking water exposure studies of Verderame et al. (2011) and Verderame and Limatola (2015) described above inform this risk hypothesis for sea turtles. Data for dietary exposures of accumulated 4-nonylphenol through aquatic forage species were not found. A single study implicated dietary exposure to 4-nonylphenol in reduced clutch sizes of tree sparrows (Dods et al. 2005). However, this was a field study at a wastewater treatment facility and we cannot eliminate other factors that may have influenced clutch size, such as the exposure to unmeasured toxicants or the additive or synergistic interaction of toxicants that were detected.

The hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles eat animal prey, so the Verderame et al. (2011) and Verderame and Limatola (2015) were used to assess their dietary risk. Meanwhile green sea turtles eat seagrass and algae, and only data for 4-nonylphenol uptake by algae, marine diatoms, were found. Sargassum is a marine brown algae that serves as important nursery habitat for sea turtles. Neonate loggerheads are omnivorous and potentially consume sargassum. While adult loggerhead are not expected to consume sargassum, sargassum may serve as an 4-nonylphenol sink for ambient 4-nonylphenol and as a source of 4-nonylphenol in prey species (sensu Correa-Reyes et al. 2007; Staniszewska et al. 2015). The bioconcentration factors[4] for 4-nonylphenol in diatoms exposed from one to 80 micrograms 4-nonylphenol per liter ranged from 750 to 2250 (Liu et al. 2010), suggesting that algae exposed to ambient 4-nonylphenol concentrations of 1.7 micrograms 4-nonylphenol per liter would have a bioconcentration factor of closer to 750. Taking the Verderame et al. (2011) and Verderame and Limatola (2015) studies into consideration, accumulation would not likely amount to toxic concentrations.

As discussed above, only sublethal effects occurred in a reptile species exposed to 4-nonylphenol at dietary levels that would result from ambient water concentrations many orders of magnitude higher than the proposed criterion. This indicates that EPA's approval of FDEP's proposed 4-nonylphenol criterion for predominantly marine waters is discountable because it is extremely unlikely to result in toxic dietary exposures to sea turtles. Therefore, NMFS concluded that EPA's approval of the proposed 4-nonylphenol criterion is not likely to adversely affect ESA-listed green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles through dietary exposures.

Reduced extent of habitat and effects on refugia. Sargassum is a marine brown algae that serves as important nursery habitat for sea turtles. As stated previously, the only data on 4-nonylphenol effects on marine algal species in ECOTOX are for 4-nonylphenol effects on the

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[4] A bioconcentration factor is the ratio of a substance's concentration in the tissue of an aquatic organism to its concentration in the ambient water.

abundance of diatoms and 4-nonylphenol bioconcentration in diatoms. Diatom cell numbers declined by ten percent at 12.5 micrograms 4-nonylphenol per liter and by 50 percent at 32 micrograms 4-nonylphenol per liter (Ward and Boeri 1990b). These values are well above the proposed 4-nonylphenol criterion for predominantly marine waters of 1.7 micrograms 4-nonylphenol per liter.

Since the available toxicity data do not suggest that exposure to 4-nonylphenol at, or below, the proposed criterion for predominantly marine waters is expected to result in adverse effects to algae or vascular plants, adverse effects are extremely unlikely to occur such that they are discountable. Therefore, NMFS concluded that EPA's approval of the proposed 4-nonylphenol criterion is not likely to adversely affect the extent of habitat or refugia for ESA-listed green, hawksbill, Kemp's ridley, leatherback, and loggerhead turtles.

Reduced availability and quality of forage due to effects on forage species populations. The hawksbill, Kemp's ridley, leatherback, and loggerheads eat various species of animal prey while green sea turtle adults eat sea grasses and algae. Limited data are available for the effects of 4-nonylphenol on marine life (Table 1). For fish, sublethal effects growth and development were reported with NOECs at 29 micrograms per liter (Martin-Skilton et al. 2006) and a NOEC for mortality was reported as 240 micrograms per liter (Ward and Boeri 1990a). The sublethal NOECs for invertebrates ranges from 0.01 to 30 micrograms per liter, with five out of 28 NOECs reported at concentrations below the proposed 4-nonylphenol criterion of 1.7 micrograms per liter for predominantly marine waters. The five NOECs for mortality ranged from 6.7 to 130 micrograms per liter. Taken together, population level effects of 4-nonylphenol on fish and invertebrate forages species are not expected.

Seagrass is an important component of the diet of adult green sea turtles. The only NOEC and LOEC data available for the effects of 4-nonylphenol on plant life in ECOTOX is for a single study of growth in the freshwater species duckweed. The NOEC is reported at 901 micrograms 4-nonylphenol per liter and LOEC at 2080 micrograms 4-nonylphenol per liter (a 62 percent decline relative to control, Brooke 1993a). Growth is a sublethal effect, and these values are orders of magnitude greater than the proposed criteria, effects of 4-nonylphenol on seagrass forage availability and quality are not expected.

In summary, the available toxicity data do not suggest that exposures to 4-nonylphenol at or below the proposed criterion would significantly affect marine fish, invertebrate, or seagrass communities. This indicates that ambient concentrations of 4-nonylphenol at the proposed criterion for predominantly marine waters is extremely unlikely to reduce the availability or quality of forage species consumed by sea turtles such that it is discountable. Therefore, NMFS concluded that EPA's approval of the proposed criterion is not likely to adversely affect for ESA-listed green, hawksbill, Kemp's ridley, leatherback, and loggerhead turtles.

Conclusion for ESA-Listed Sea Turtles: Our analyses of both direct and indirect effects for each pathway of effect concluded that it is extremely unlikely that ambient concentrations of 4-nonylphenol at the proposed criterion for predominantly marine waters would affect survival of

sea turtles, toxicity of their prey, quality of their habitat, or the abundance and quality of forage species. Such effects were found to be discountable. Therefore, NMFS concluded that EPA's approval of the proposed nonylphenol criterion for predominantly marine waters is not likely to adversely affect ESA-listed green, hawksbill, Kemp's ridley, leatherback, and loggerhead turtles. The ESA-listed sea turtles will not be discussed further in this opinion.

6.2.2 Fish

Marine and anadromous fish are more vulnerable to aquatic pollutants than sea turtles because their gill epithelium is necessarily in direct contact with ambient water in order to meet the physiological needs of oxygen uptake, water, and ion balance, and excretion of nitrogen wastes. Gills are an arrangement of highly vascularized filaments that resemble stacked feathers, creating large surface areas through which water is sieved and oxygen, ions, and waste exchange occur. Toxicants present in ambient water may be drawn or diffuse into the bloodstream at the gill, bind to ion exchange components of the gill epithelia, or otherwise irritate or damage tissue, resulting in morphological abnormalities and impaired function (Evans 1987, Evans et al. 2005).

Survival. Dwyer et al. (2005b) reported the 96-hour LC50 of 4-nonylphenol to Atlantic and shortnose sturgeon to be 50 and 80 micrograms 4-nonylphenol per liter, respectively. However, a response threshold of 50 percent mortality is not an acceptable outcome for these ESA-listed species. Data reported in ECOTOX include fish mortality NOECs ranging from 77 percent survival at 7.4 micrograms 4-nonylphenol per liter (Ward and Boeri, 1991b) to 100 percent survival at 240 micrograms 4-nonylphenol per liter (Ward and Boeri 1990a). The LOECs range from 67 percent survival at 14 micrograms 4-nonylphenol per liter (Ward and Boeri 1991b) to 10 percent survival at 420 micrograms 4-nonylphenol per liter (Ward and Boeri 1990a).

The sole marine fish data set for mortality is for a single study of effects to sheepshead minnow (Ward and Boeri 1990a). The mortality NOEC (95 percent survival at 240 micrograms 4-nonylphenol per liter) and LOEC (zero percent survival 420 micrograms 4-nonylphenol per liter) from this study exceed those observed for freshwater fish. This same study also reported 100 percent survival at concentrations of 150 micrograms 4-nonlyphenol and below. Given the absence of mortality data for ESA-listed species under NMFS jurisdiction or closely related surrogates, the available data are taken together to infer whether 4-nonylphenol exposures would result in mortality of anadromous sturgeon, smalltooth sawfish, giant manta ray, or Nassau grouper.

Studies reporting mortality LOECs and NOECs for exposures of fish embryos and larvae are one or more orders of magnitude higher[5] than the criteria EPA proposed to approve (Ward and Boeri 1990a, Brooke 1993a, Hill and Janz 2003, Demska-Zakes and Zakes 2006, Lin and Janz 2006, Zha et al. 2007, Xu et al. 2013). Mortality in adults would be expected to occur at even higher concentrations. Taking taxonomic relatedness into consideration, the available data represent a

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[5] As determined from ECOTOX and original sources

diverse group of fish orders (i.e., Acipenseriformes, Beloniformes, Cypriniformes, Cyprinodontiformes, Perciformes, and Salmoniformes). Meanwhile, the fish species of interest in this work are in the orders Acipenseriformes (Atlantic, Gulf, and shortnose sturgeon), Perciformes (Nassau grouper), and super order Batoidea (giant manta ray and smalltooth sawfish). Interpretation of the available data for implications for the Batoidea includes a great deal of uncertainty. The endocrine physiology of elasmobranchs is poorly studied, but those studies that are available indicate similarities with bony fish species (Pankhurst 2011, Awruch 2013). These similarities, taken with evidence that fish species from a broad range of taxonomic orders are not likely to respond to 4-nonylphenol exposures at the proposed criteria, leads NMFS to expect that the ESA-listed smalltooth sawfish and giant manta would likewise not respond to such exposures.

In summary, toxicity data for other fish species suggest that exposures to 4-nonylphenol at, or below, the proposed criteria are extremely unlikely to result in mortality such that it is discountable. Therefore, NMFS concluded that EPA's approval of FDEP's proposed 4-nonylphenol criteria is not likely to result in the death of ESA-listed Atlantic, Gulf, and shortnose sturgeon, Nassau grouper, giant manta ray, and smalltooth sawfish.

Fitness: We begin the discussion on 4-nonylphenol effects on fitness with an evaluation of on growth and development. This includes effects on size (e.g., mass, length), development (e.g., maturation, metamorphosis) and morphology (e.g. development outcomes such as deformity). NOECs for these responses ranged from 2.3 micrograms 4-nonylphenol per liter (liver mass, Shelley et al. 2012) to 1280 micrograms 4-nonylphenol per liter (testes mass, Kinnberg et al. 2000) and LOECs ranged from 11.6 micrograms 4-nonylphenol per liter (testis-ova induction, Seki, et al., 2003) to 2270 micrograms 4-nonylphenol per liter (ten percent decrease in ovarian weight, Van den Belt et al. 2004).

The lowest NOEC is above the proposed criterion for predominantly marine waters, but below the criterion for freshwater. Most of the NOECs that are below the freshwater 4-nonylphenol criterion of 6.6 micrograms 4-nonylphenol per liter were for growth: organism size or organ mass relative to body mass (Brooke 1993b, a, Zha et al. 2007, Zha et al. 2008, Spehar et al. 2010, Bin-Dohaish 2012, Shelley et al. 2012a, Shelley et al. 2012b). The significance of growth impairment to survival and reproduction is attributed to smaller fish being at greater risk of predation, having a smaller diversity of available prey, competing poorly for mates or nests, and having poor reproductive output. Extrapolating size and body metric responses from taxonomically distant laboratory species to implications for the survival and reproduction of ESA-listed anadromous and marine fish species is tenuous.

The ability to reproduce is also a factor in an individual's fitness. Some of the sexual development NOECs were below the proposed criterion for Class III freshwaters (Seki et al. 2003, Balch and Metcalfe 2006). In the (Balch and Metcalfe 2006) study, 12 percent of exposed fish had mixed secondary sex characteristics at the NOEC of 2.9 micrograms 4-nonylphenol per liter while 20 percent exhibited this effect at a LOEC 8.7 micrograms 4-nonylphenol per liter.

This study illustrates a shortcoming in using NOECs and LOECs, they are based on statistical significance, rather than biological significance, relative to a control (i.e., a 12 percent response might be biologically important). The baseline level of abnormalities in the organisms tested needs to be considered when interpreting the biological significance of reported NOECs. A meta-analysis by Grim et al. (2007) showed that 15 out of 41 studies had control groups exhibiting intersex rates (gonads with both male and female characteristics). Intersex ranged from 0.2 to 24 percent in these studies. The rate of intersex among controls appeared to be related to which strain was used by each study facility. A rate of 12 percent effect over baseline would likely have negative implications for a population already at risk of endangerment or extinction. However, the medaka strain used in the Balch and Metcalfe (2006) study had a baseline mixed secondary sex characteristic rate of eight percent.

In the final sexual development study, the Seki et al. (2003), reported a NOEC for medaka testis tissue abnormalities at 6.08 micrograms 4-nonylphenol per liter (no fish with intersex) with a LOEC of 11.6 micrograms 4-nonylphenol per liter (20 percent fish with intersex). Interpolating the degree of response between the NOEC and LOEC reported by Seki et al (2003) suggests about three percent of exposed organisms would exhibit testis-ova at the criterion of 6.6 micrograms 4-nonylphenol per liter. Even with these few observed NOECs below the proposed freshwater/predominantly freshwater 4-nonylphenol standard of 6.6 micrograms 4-nonylphenol per liter, the overall mean growth and development NOEC values among studies for each species tested exceed 6.6 micrograms 4-nonylphenol per liter.

Duffy et al. (2013) examined the suitability of standard test methods to detect endocrine disruption in shortnose sturgeon and Atlantic sturgeon exposed to 4-nonylphenol in ambient water. Using vitellogenin as the response indicator, the study demonstrated that routine laboratory test methods are insufficient for determining impacts from endocrine disruptors to these listed species. Vitellogenin mRNA in early life stage (less than 1 year old) shortnose and Atlantic sturgeon was not induced by 4-nonylphenol or more potent vitellogenin inducing substances: α-ethinylestradiol, or 17-β estradiol. The 4-nonylphenol exposures used were as high as 88 micrograms per liter.

The report stated that applying the precautionary principle indicated that brook trout, which mature by age two, would be a suitable surrogate species for endocrine disruption in sturgeon despite differences in lie history. The exposure resulting in responses for this species was 44 micrograms 4-nonylphenol per liter over 21 days exposure, well above FDEP's proposed criteria.

Data for endocrine disruption and responses more readily associated with fitness in Nassau grouper, smalltooth sawfish, or giant manta taxonomic families were not found among data in ECOTOX or searches of published literature. The ECOTOX database (see Table 1) did not contain NOECs or LOECs for 4-nonylphenol effects on other measures of fitness in marine fish species. It is therefore necessary to rely on data for freshwater fish for our analysis of 4-nonylphenol effects on fitness in these species.

The lowest fecundity NOEC for freshwater fish NOEC is 16.5 micrograms 4-nonylphenol per liter (97 percent hatch success in medaka, Ishibashi et al., 2006) and the lowest LOEC was 61.2 micrograms 4-nonylphenol per liter (83 percent hatch success in medaka, Ishibashi et al., 2006).

Because the available toxicity data indicate that ambient exposure to 4-nonylphenol at the proposed criteria is extremely unlikely to change the growth, development, or reproductive success of Atlantic, Gulf, and shortnose sturgeon, Nassau grouper, giant manta ray, and smalltooth sawfish, such effects are discountable. Therefore NMFS concludes that EPA's approval of FDEP's proposed 4-nonylphenol criteria is not likely to adversely affect the fitness of these ESA-listed species.

Indirect Effects

Consuming prey that have accumulated toxic levels of 4-nonylphenol. Data were not found on the dietary toxicity of 4-nonylphenol to fish species that could be related to ambient water concentrations. Regarding sublethal effects, Pickford et al. (2003) hypothesized that a hydrophobic chemical such as 4-nonylphenol, entering via the gills, may be more estrogenic than one entering through the diet, because dietary exposures undergo metabolism in the small intestine and liver before entering the bloodstream. In this way, metabolism may reduce or eliminate the estrogenic potential of 4-nonylphenol before it reaches target organs such as the gonads or liver. The result showed a 10-fold greater sensitivity for 4-nonylphenol in fish exposed via the water compared with exposure via the oral route. Results obtained from this study indicate that a chemical such as 4-nonylphenol has a higher estrogenic potential when it enters the bloodstream via the gills/skin of a fish compared with exposure through the diet

As described above, direct exposures via gill uptake has been demonstrated to be more likely than dietary exposures to result in adverse effects. In addition, NMFS previously concluded that direct exposures at the proposed 4-nonylphenol criteria are not likely to adversely affect survival or fitness. For this reason, dietary exposures under the proposed 4-nonylphenol criteria are extremely unlikely to affect the fitness of ESA-listed Atlantic, Gulf, and shortnose sturgeon, Nassau grouper, giant manta ray, and smalltooth sawfish such that it is discountable.

Reduced availability and quality of forage due to effects on forage species. NMFS' prior conclusion that direct exposures at or below the 4-nonylphenol criteria are not likely to result in adverse effects to individual ESA-listed fish extends to the populations of fish and invertebrates they prey upon. While marine invertebrate data include NOECs and LOECS below the marine criterion of 1.7 micrograms per liter, significant changes in the overall abundance and distribution of prey species from the invertebrate community are not expected because the criterion was derived to generically protect aquatic life, or "most of the species most of the time." NMFS does not expect water concentrations at or below the proposed criteria would result in significant changes in prey availability and quality.

Since NMFS previously concluded that adverse effects to individual fish are extremely unlikely such that they are discountable and that effects to the invertebrate prey community are expected

to be insignificant, NMFS concludes that EPA's approval of the proposed 4-nonylphenol criteria is not likely to adversely affect due to reduction in the availability or quality of prey affecting the fitness of ESA-listed Atlantic, Gulf, and shortnose sturgeon, Nassau grouper, giant manta ray, and smalltooth sawfish.

Conclusion for ESA-Listed Fish Species Under NMFS jurisdiction: Our analyses of both direct and indirect effects each concluded that it is extremely unlikely that ambient concentrations of 4-nonylphenol at the proposed criteria for fresh and predominantly marine waters would affect survival of ESA-listed fish species, toxicity of their prey, quality of their habitat, or the abundance and quality of forage species. Such effects were found to be discountable. Therefore, NMFS concluded that EPA's approval of the proposed nonylphenol criteria is not likely to adversely affect for ESA-listed Atlantic, Gulf, and shortnose sturgeon, Nassau grouper, giant manta ray, and smalltooth sawfish.

6.2.3   Seagrass

Plant species, such as Johnson's seagrass, would not be expected to respond to an exogenous estrogen mimic like 4-nonylphenol. The only NOEC and LOEC data available for the effects of 4-nonylphenol on plant life in ECOTOX is for a single study of growth in the freshwater species duckweed. The NOEC is reported at 901 micrograms 4-nonylphenol per liter and LOEC at 2080 micrograms 4-nonylphenol per liter (a 62 percent decline relative to control, Brooke 1993a). Growth is a sublethal effect, and these values are orders of magnitude greater than the proposed criteria, suggesting that other sublethal effects would also occur at concentrations greater than the proposed criteria and mortality could occur at yet higher concentrations. Indirect effects to Johnson's seagrass are not expected because they do not prey on other organisms and do not require biological habitat features (e.g., refugia,cover).

The available toxicity data indicate direct toxic effects to Johnson's seagrass are extremely unlikely to occur through exposures at or below the proposed criterion for predominantly marine waters, and are thus discountable. Therefore, NMFS concluded that EPA's approval of FDEP's this criterion is not likely to adversely affect Johnson's seagrass.

6.3 Designated Critical Habitat that is Not Likely to be Adversely Modified by 4-nonylphenol Exposures at the Proposed Criteria

Because 4-nonylphenol is a toxicant, its effects to designated critical habitat would be responses of the *biological* essential features of designated critical habitat to 4-nonylphenol exposure at the criteria concentrations. The designated critical habitats for Atlantic sturgeon, staghorn and elkhorn coral, and Johnson's seagrass do not include biological essential features and therefore will not be discussed further in this opinion. While some of the essential features of designated critical habitat for the North Atlantic Right whale are biological (i.e., forage), this species does not feed in Florida waters, so the action is not expected to affect designated critical habitat for this species. Designated critical habitat for the North Atlantic Right whale will not be further discussed in this opinion.

The biological features of the designated critical habitat for loggerhead turtle include sargassum to support prey, abundance of prey, and cover. Biological features of the designated critical habitat for smalltooth sawfish include red mangroves and the nursery area functions they provide (e.g., cover, prey).

Modification of Sargassum Essential Element of Designated Critical Habitat for Loggerhead Sea Turtle: Sargassum is a marine brown algae but the only data on 4-nonylphenol effects on marine algal species in ECOTOX are for 4-nonylphenol effects on the abundance of diatoms and 4-nonylphenol bioconcentration in diatoms. Diatom cell numbers declined by ten percent at 12.5 micrograms 4-nonylphenol per liter and by 50 percent at 32 micrograms 4-nonylphenol per liter. These values are well above the proposed 4-nonylphenol standard for predominantly marine waters of 1.7 micrograms 4-nonylphenol per liter. The bioconcentration factors[6] for 4-nonylphenol in diatoms exposed from one to 80 micrograms 4-nonylphenol per liter ranged from 750 to 2250 L/g (Liu et al. 2011). Neonate loggerheads are omnivorous and potentially consume 4-nonylphenol laden sargassum. While adult loggerhead are not expected to consume sargassum, sargassum may serve as a 4-nonylphenol sink for ambient 4-nonylphenol and as a source of 4-nonylphenol in prey species (sensu Correa-Reyes et al. 2007; Staniszewska et al. 2015). Data on flux of 4-nonylphenol in marine algal species was looked for, but not found, so it is uncertain whether 4-nonylphenol accumulated in sargassum may be metabolized or released back into ambient water. However, data for freshwater algal species suggests that algae may metabolize 4-nonylphenol (Sun et al. 2014, Otto et al. 2015, He et al. 2016).

In summary, the available toxicity data indicate that adverse effects are extremely unlikely to occur in plant species at, or below, the proposed criterion for predominantly marine waters and is discountable. Therefore, NMFS concluded that EPA's approval of FDEP's proposed criterion is not likely to adversely affect the sargassum essential element of designated habitat for the endangered North West Atlantic distinct population segment of the loggerhead turtle. EPA's approval of the proposed criterion is therefore not expected to result in the destruction or modification of critical habitat designated for this species.

Modification of Mangrove Essential Element of Designated Critical Habitat for Smalltooth Sawfish: Mangrove exposure to 4-nonylphenol in water would occur through uptake by roots in the water and saturated sediment. NMFS looked for and did not find data on 4-nonylphenol uptake by mangroves or 4-nonylphenol effects on marine plants. A 20 percent decline in frond production was reported for duckweed exposed to 2,080 micrograms 4-nonylphenol per liter for four days (Spehar et al. 2010). This is several orders of magnitude greater than the proposed standard for predominantly marine waters.

As stated for sargassum, the available toxicity data indicate that adverse effects are extremely unlikely to occur in plant species at or below the proposed criterion for predominantly marine

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[6] The ratio of a substance's concentration in the tissue of an aquatic organism to its concentration in the ambient water

waters and is discountable. Therefore, NMFS concluded that EPA's approval of FDEP's proposed criterion is not likely to adversely affect the mangrove habitat essential element of designated critical habitat for the endangered smalltooth sawfish. EPA's approval of the proposed criterion is therefore not expected to result in the destruction or modification of critical habitat designated for this species.

Modification of Prey Abundance and Availability Essential Elements of Designated Critical Habitats for Loggerhead Sea Turtle and Smalltooth Sawfish: The designated critical habitat for both loggerhead sea turtles and smalltooth sawfish include the biological essential element of prey availability and abundance. This work previously determined that EPA's approval of FDEP's proposed 4-nonylphenol criteria is not likely to result in increased mortality, fecundity, or growth and development effects in ESA-listed species under NMFS jurisdiction. This conclusion logically extends to the fish prey species. However, crustaceans are an important component of endangered loggerhead sea turtles and smalltooth sawfish. Data collected from ECOTOX indicate that the 4-nonylphenol concentrations affecting the survival, reproduction, and population size of marine crustacean species are higher than the proposed 4-nonylphenol standard for predominantly marine waters.

As discussed previously, the presence of 4-nonylphenol in aquatic habitats results in food web transfer, but bioaccumulation and biomagnification studies indicate that the metabolism can attenuate accumulation (Hu et al. 2005, Korsman et al. 2015). EPA's biological evaluation discussed bioconcentration, but did not attempt to interpret the proposed criteria in context of the available data. Bioaccumulation and biomagnification in an ecosystem is affected by the length and structure of the food web, lipid content of food web components, and metabolic capacity of the species present, system-flushing rate, and seasonal movements/presence of the species that use the habitats. NMFS acknowledges the importance of dietary exposures resulting from pollutants in ambient water, but the location and season-specific data for each of the factors needed to extrapolate FDEP's proposed 4-nonylphenol criteria to trophic-scale outcomes are not available and, given environmental stochasticity, would be intractable to model.

NMFS previously concluded that adverse effects to individual fish are extremely unlikely such that they are discountable and that effects to the invertebrate prey community are expected to be insignificant. For this reason, NMFS concluded that EPA's approval of the proposed 4-nonylphenol criterion for predominantly marine waters is not likely to adversely affect the forage species essential element of designated critical habitat for the endangered smalltooth sawfish and Northwest Atlantic Distinct Population Segment of loggerhead sea turtle. EPA's approval of the proposed criterion is therefore not expected to result in the destruction or modification of critical habitat designated for this species.

7 STATUS OF SPECIES ADDRESSED IN THIS OPINION

Thus far, we determined that ESA-listed cetacean, sea turtle, and fish species within the action area are not likely to be adversely affected by the proposed action. Below we discuss our analysis of species likely to be adversely affected. These are the ESA-listed coral species of the Caribbean (Table 2). The status includes the existing level of risk that the ESA-listed species face, based on parameters considered in documents such as recovery plans, status reviews, and listing decisions. Each species status section helps to inform the description of the species' current "reproduction, numbers, or distribution," which is part of the jeopardy determination 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 sites.

This consultation applied the most recent recovery plans and status reports available at the time it was conducted. While the following discussions focus on the use of Florida waters by these species, consideration of the status of populations outside of the action area is also important in our evaluating how the risk to affected population (s) influences the status of the species as a whole.

Factors that are common to all of the ESA-listed Caribbean coral species are discussed first before describing the status for each species of coral that may be affected by EPA's approval of the 4-nonylphenol criteria, with particular emphasis on aspects that may be influenced by the criteria. As explained in Section 6.3, designated critical habitat for elkhorn and staghorn coral does not include biological features that would respond to exposures to 4-nonylphenol.

7.1 ESA-listed Coral Species of the Florida Reef Tract

Among the corals protected under the ESA, seven species occur on the shallow reefs (i.e., to 30 meters in depth) of south Florida, Puerto Rico, and the U.S. Virgin Islands. These include four genera: the *Acropora*, elkhorn and staghorn coral, the *Orbicella*, boulder, lobed, and mountainous star corals; a *Dendrogyra* species, pillar coral; and a *Mycetophyllia* species, rough cactus coral. The distribution of coral species within a reef is influenced by depth, distance from shore, and wave exposure (Goreau 1959, Bak 1977). Elkhorn, staghorn, and the star corals lend their name to classical reef zonation descriptions due to their historic importance in reef structure (Figure 4). Star corals now dominate these areas.
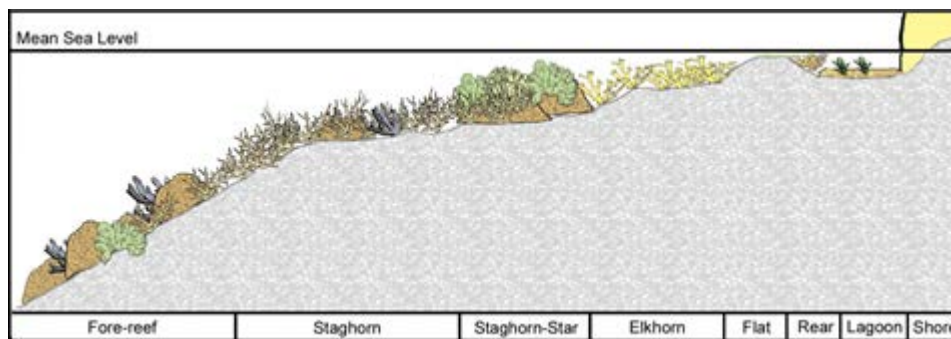
Figure 4. Reef zonation schematic example modified from several reef zonation-descriptive studies (Bak 1977; Goreau 1959).

All seven species belong to the taxonomic Order Scleractinia; stony corals that secrete calcium carbonate to form hard exoskeletons. Stony corals are polyp body forms typically aggregated into dense colonies. Each polyp is an individual organism with a digestive, nervous, respiratory, and reproductive system (Figure 5). Using nematocyst-equipped[7] tentacles, polyps are capable of catching food. In addition, the tissues of most coral polyps contain zooxanthellae, symbiotic algae that support the coral's energy budget and calcium carbonate secretion (Figure 6, Credit: Gini Kennedy, NOAA). Zooxanthellae give coral its color. Under adverse conditions, particularly temperature extremes (Kemp et al. 2016, Hughes et al. 2017), the zooxanthellae can be expelled, resulting in coral bleaching (Goreau et al. 1979, Brainard et al. 2011).
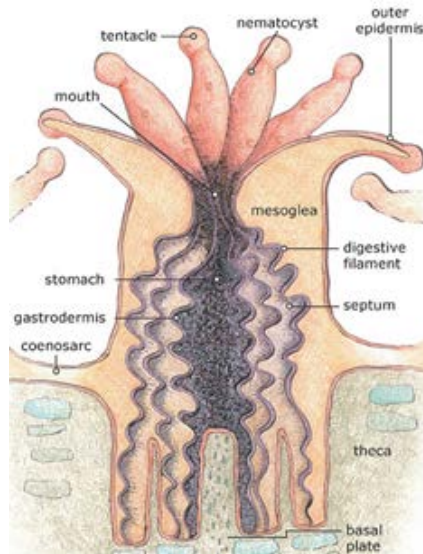
Figure 5. General anatomy of a coral polyp.

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[7] A nematocyst is a specialized cell characteristic of cnidarian organisms, such as jellyfish, coral, and anemones, that discharges a barbed thread, often containing toxin, used to ward off predators or to stun and capture prey.
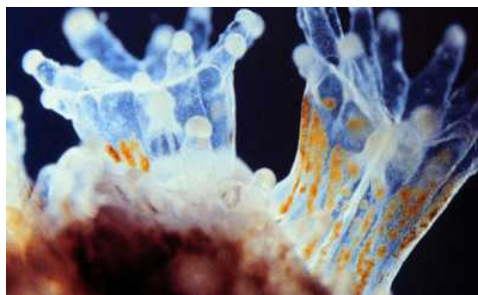
Figure 6. Close up image of translucent coral polyps containing photosynthetic algae (zooxanthellae).

7.1.1 Reproduction in Threatened Species of Stony Corals with Populations in Florida Waters

With the exception of pillar coral, which has separate sexes, the ESA-listed coral species are hermaphroditic. Rough cactus coral fertilize and brood larvae within the colony before release while elkhorn, staghorn, boulder star, lobed star, mountainous star, and pillar corals are broadcast spawners (Brainard et al. 2001, Figure 7, Photo: Maricopa Community College). Broadcast spawners release sperm and egg cells in 'bundles' such that fertilization occurs. The larvae, called planula, develop externally in the open water and drift as plankton until finding suitable substrate upon which to settle and metamorphose into polyps (Harrison et al. 1984). Goreau et al. (1981) working with planula of the coral species Porites porites, report that mortality rates prior to settling is likely greater than 90 percent due to predation, larval defects, and environmental stress. The planula of the ESA-listed species are expected to experience similar mortality rates.
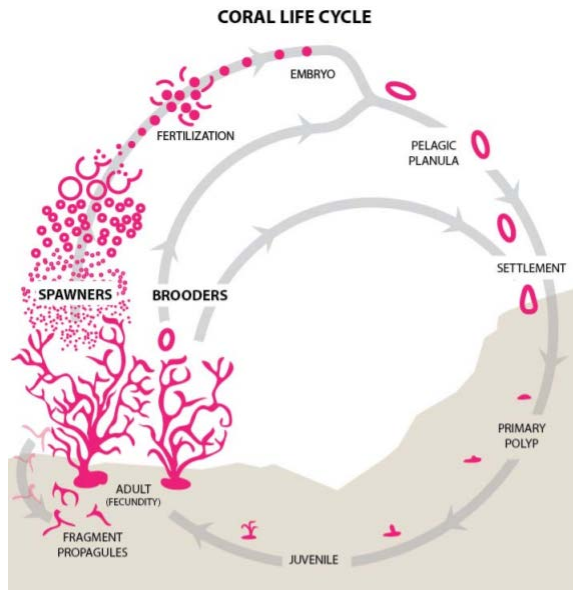
Figure 7. General depiction of coral life cycles.

Successful recruitment of larvae into sexually reproducing adults is the only means by which genetically unique individuals enter a population, thereby maintaining or increasing genotypic diversity. Genotypic diversity is important for stress resilience and disease causing organisms (Foret et al. 2007, Baskett et al. 2010, Schopmeyer et al. 2012). Planula are also the only life cycle phase that disperse over long distances, genetically linking populations and providing potential to re-populate depleted areas (Hughes and Tanner 2000, Jackson et al. 2014, NMFS 2015b).

Asexual reproduction in stony corals occurs through budding, with new polyps forming from a parent polyp, or through fragmentation, with broken pieces attaching to suitable substrate, forming a new, but genetically identical, colony (Brainard et al. 2011). The size and weight of fragments can limit their dispersal range. Fragmentation is important in maintaining local populations when sexual recruitment is limited. In contrast with sexually produced planula, fragments are larger, are less likely to be smothered by overgrowth by other organisms (Jackson 1977), and have higher survivorship. Contrasted with sexual reproduction, fragmentation can occur year round (Szmant 1986). However, potential consequences of the dominance of genetically identical colonies include poor to no reproductive success, because these species do not self-fertilize and increased susceptibility to stress events for which that clone is not adapted. Additionally, severe fragmentation, as commonly observed after storms, may limit future sexual reproduction by reducing the biomass of colonies and shifting energy allocation from reproduction to regeneration (Jackson 1986).

Biological and physical factors affect spatial and temporal patterns of sexual recruitment. These include substrate availability and community structure, grazing pressure, fecundity, mode and timing of reproduction, behavior of larvae, hurricane disturbance, physical oceanography, the structure of established coral assemblages, and chemical cues (Lewis 1974, Birkeland 1977, Goreau et al. 1981, Rogers et al. 1984, Baggett and Bright 1985, Harriott 1985, Hughes and Jackson 1985, Sammarco 1985, Morse et al. 1988, Fisk and Harriott 1990, Richmond and Hunter 1990).

7.1.2 Threats to Stony Coral Species with Populations in Florida Waters

The coral species living off the coast of Florida are vulnerable to the same anthropogenic stressors that threaten corals worldwide: climate change, fishing impacts, recreation impacts, and pollution. The following discussion was adapted from the NOAA Coral Reef Conservation Program threat summaries (NOAA 2015), the NOAA Fisheries Southeast Region Recovery Outline for five species of corals in the region listed in September 2014 (NMFS 2015a), the NMFS 2014 listing determination for these corals (see Table 2), and the Recovery Plan for Elkhorn and Staghorn Coral (NMFS 2015b).

Increased water temperatures and ocean acidification resulting from climate change increases coral vulnerability to bleaching and disease and impairs the construction and maintenance of calcium carbonate-based skeletal frameworks. Mass coral bleaching, which results from the expulsion of the symbiotic zooxanthellae algae, is linked to excursions in ocean temperatures outside of coral physiological tolerances (Hughes et al. 2017). Warm water bleaching events typically co-occur with high subsurface light levels and are associated with major El Niño-Southern Oscillation events (e.g., 1982–83, Glynn and D'croz, 1990; 1997–98, Wilkinson et al 2000; and 2002, Berkelmans et al 2004). Laboratory experiments have confirmed this association (Coles and Jokiel 1978, Glynn and D'croz 1990). Increased coral mortality due to the stress from bleaching and subsequent disease outbreaks alters reef habitats, structures, and biodiversity (Eakin 2001, Graham et al. 2006). The most severe and extensive Caribbean mass warm water

bleaching event occurred in 2005. Only localized warm water bleaching was observed in the years between 2006 and 2013 (Manzello 2015) and a cold water bleaching event occurred in the Florida Keys over the winter of 2009-2010. In 2005, wide-scale bleaching occurred throughout the Caribbean with wide-scale mortality, with some areas reaching 95 percent of coral colonies affected (Wilkinson and Souter 2008). Puerto Rico and Florida also experienced disease rates of 50 percent of coral colonies or greater. Following the 2005 bleaching event, monitoring data indicate that total coral cover is now less than 12 percent on many reefs (Rogers et al. 2008). Coral mortality due to the 2005 bleaching event was more severe than at any time in the last 40 years of monitoring in U.S. Virgin Islands (Woody et al. 2008). Bleaching events can lead to increased thermal tolerance in affected reefs, meaning that subsequent bleaching events are not as severe (Maynard et al. 2008).

Ocean acidification is caused by increased dissolved CO_2 in ocean water. This changes the solubility and form of seawater minerals in even slightly more acidic seawater. Most critically, acidification reduces seawater saturation with aragonite, the form of calcium carbonate used by corals and other marine species to construct protective shells and skeletal frameworks. Without sufficient aragonite, formation of calcified parts is impaired (Anthony et al. 2008, De'ath et al. 2009, Wei et al. 2009, Crawley et al. 2010). Acidification also reduces thermal tolerance of corals, meaning that bleaching can occur at lower temperatures (Anthony et al. 2008).

Taken together, disease and ocean warming are major threats affecting the potential for coral recovery in the southeast U.S. because they are severe, ongoing, synergistic, and have increased in the recent past. Mortality rates after disease and bleaching events have not been compensated for through recruitment or growth. Sea-surface temperature is expected to continue to rise over time and exacerbate disease impacts. Climate change effects, such as sea level rise, altered ocean circulation, and changes in the frequency, intensity, and distribution of tropical storms, will affect corals. These changes may increase physical damage to coral reefs (Madin et al. 2012, Teixidó et al. 2013) or harm corals by severely reducing salinity with large influxes of stormwater runoff (Berkelmans et al. 2012, Lough et al. 2015). Hurricanes fueled by warmer waters can cause wide-scale inhibition of recruitment in years following storm passage as well as physical damage to coral colonies themselves (Mallela and Crabbe 2009).

A record number of hurricanes in 2005 caused extensive damage to coral reefs; the prevalence of hurricanes and subsequent coral reef damage has been linked to climate change (Wilkinson and Souter 2008).

Fishing and recreation impacts on coral reefs include direct harvests of coral, cascading effects due to the removal or reduction of important functional species from coral reef communities, and physical damage by certain fishing gears and fishing methods that can directly contact coral reefs such as anchoring, accidental grounding, and prop wash of vessels. Cascading effects resulting from altered trophic structure of the reef community degrades coral condition and habitat and increases synergistic stress effects (e.g., bleaching, disease).

Coastal development contributes localized threats through run-off of land-based pollutants, including excess nutrients and sediment, and through physical damage from activities such as dredging, cable and pipeline deployment, construction, and beach nourishment. Stressors may
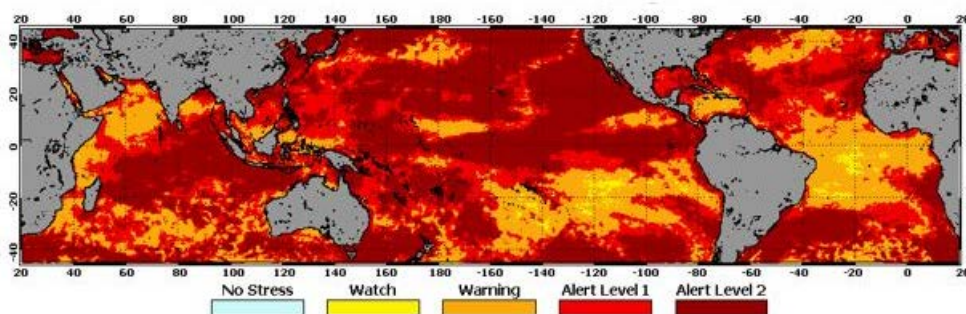
Figure 8. Reef Watch satellite coral bleaching alert area January 2014-December 2016.

exacerbate bleaching impacts by lowering the thermal threshold when corals bleach (i.e., increasing their susceptibility) and/or increasing the duration of impaired growth after a bleaching event (Wooldridge 2009 (Carilli et al. 2009, Wooldridge 2009). Similarly, Bruno et al. (2003) found that nutrient enrichment caused increased disease-associated tissue loss in corals. There are anthropogenic sources (i.e., sewage) of some coral disease-causing bacteria (Patterson et al. 2011). Increased nutrients in runoff promotes algal growth on corals, leading to light blockage to zooxanthellae eventual partial or full mortality of corals (ABRT 2005). Although reefs in the Florida Keys currently experience about ten percent macroalgal cover or less, much of the wider Caribbean Sea may exceed 20 percent cover (Bruno 2008), inhibiting and reducing coral growth and survival. Suspended sediment and sediment deposition act to limit coral growth, feeding patterns, photosynthesis, recruitment, and survivorship. Reductions in long-term water clarity can also reduce the coral photosynthesis to respiration ratio. Telesnicki and Goldberg (1995) and Yentsch et al. (2002) found that elevated turbidity levels did not affect gross photosynthetic oxygen production, but did lead to increased respiration that consumed the products of photosynthesis with little remaining for coral growth.

7.1.3   Information Used in this Status Summary

The most recent reviews on the status of ESA-listed coral species are from the Global Coral Reef Monitoring Network (Jackson et al. 2014), reports generated by NMFS in support of the 2014 listing rule for corals (Brainard et al. 2011, NMFS 2012), and recovery planning documents (NMFS 2015b, a). These reviews, and much of the available published information, precede recent events that are likely to adversely affect ESA-listed corals. Since 2014, coral reef habitats have been subject to elevated ocean surface temperatures (Figure 8, NESDIS, 2017) precipitating a prolonged global bleaching event extending into early 2017 (Hughes et al. 2017).

In addition, the 2017 western Atlantic hurricane season was unusually intense, with four hurricanes over a period of less than two months. Hurricanes Harvey (August 25, category three) and Nate (October 4, category one) struck in the Gulf of Mexico and the category five hurricanes

Irma (August 30) and Maria (September 16) struck Florida and the Caribbean. Evaluation of the impacts of these events is underway. At the time of this writing, the impacts of hurricanes Harvey (category three), Irma and Maria (both category five), and Nate (category 1) on corals is being evaluated. These hurricanes struck the Gulf of Mexico and the Caribbean over a period of less than two months during the 2017.

Atlantic hurricane season. Hurricane Irma churned coastal sediments into the water column and torrential rain carried sediments in runoff from land (Figure 9, Visible Infrared Imaging Radiometer Suite imagery from the Suomi National Polar-orbiting Operational Environmental Satellite System Preparatory Project). As indicated previously, sediment is a direct stressor for many species. Sediment discharges from land and suspension from sediment beds also redistributes sediment-associated contaminants.

Figure 9. Coastal suspended sediment before (left) and after (right) Hurricane Irma.

A rapid assessment of "high value" sites along the Florida Reef Tract affected by Hurricane Irma (NMFS 2017) assigned injury class designations to 57 sites based on degree of damage and potential to successfully mitigate impact (Figure 10). Tier one was assigned to those sites with most severe impacts and/or highest priority for mitigation. Tier two indicate sites with moderate impact and secondary priority for recovery efforts if resources available, and Tier three for sites with minimal impact or not suitable for mitigation. The majority of sites were classified as Tier three (53 percent), followed by Tier two (33 percent), and Tier one (14 percent). Injury types included clogged and damaged sponges, dislodged and fractured corals, heavy sedimentation, burial of stony corals and octocorals, displaced rubble and sand, and fractured substrate. At the Tier three sites, the most common impact was sedimentation. Mitigation was recommended at 14 sites, requiring included unburying and uprighting corals, fragment stabilization and reattachment, and removal of corals from sand. In one case, the recommendation was to rescue fragments/live tissue and transplant them to an alternate site due to extensive rubble movement.

The Global Coral Reef Monitoring Network issued a report describing the impact the 2005 bleaching and hurricane season had on reefs that can inform post 2017 hurricane season expectations (Wilkinson and Souter 2008). While extreme conditions can reduce reefs to rubble, hurricanes can actually provide coral some relief from elevated ocean temperatures. They absorb
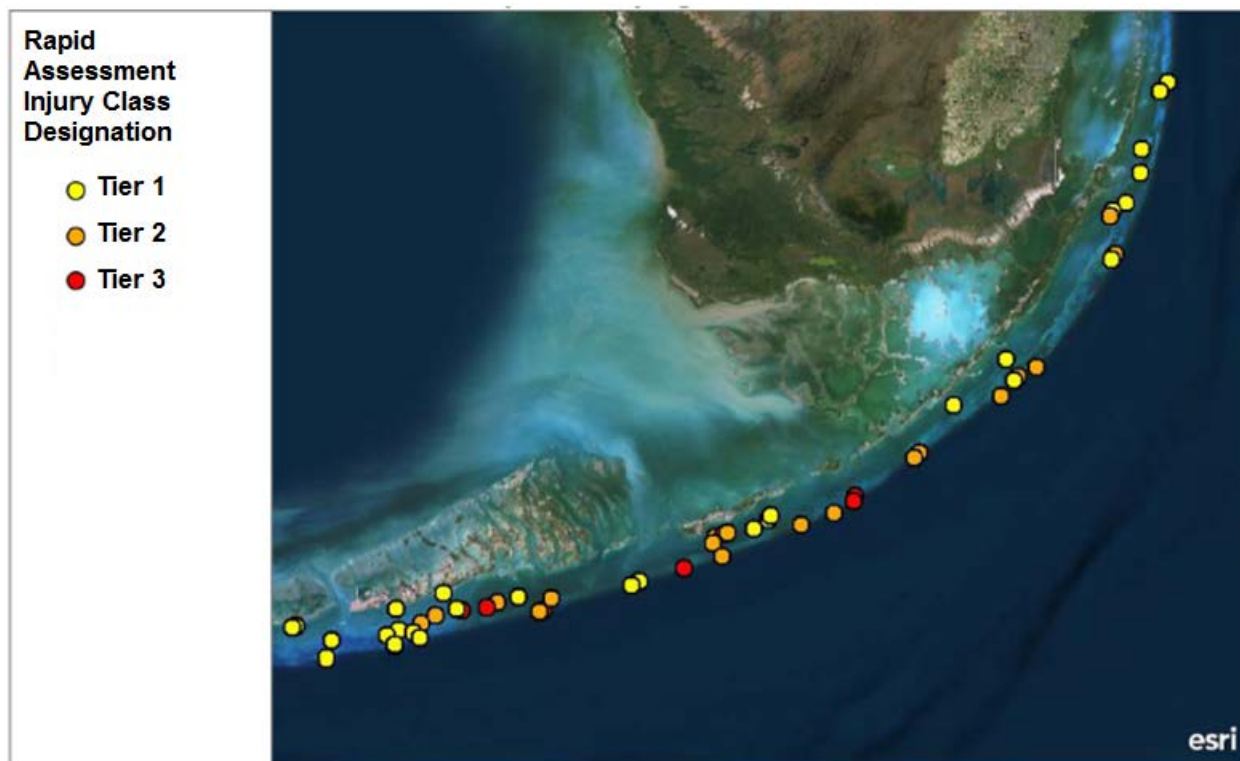
Figure 10. Injury class designations for high value Florida Reef Tract sites affected by Hurricane Irma.

energy from surface waters through the transfer of latent heat (evaporative cooling). They reduce sea surface by inducing local upwelling, bringing deeper, cooler water to the surface. The clouds of a hurricane shade the ocean surface from solar heating allowing the water to cool and reducing light stress. The magnitude of cooling is related to hurricane wind speed and variation of water temperature with depth at each location. In addition, waves and tidal water movements scour some areas exposing hard bottom substrate upon which corals can settle and grow. In other areas, water movement results in the accumulation of unstable sediment and rubble, which is less suitable for coral settlement. Unfortunately, current predictions are for more frequent and intense warming in the Caribbean with the high probability of increased bleaching and coral mortality. Severe coral bleaching is predicted to become a more regular event by 2030, and an annual event by 2100, if the current rate of greenhouse emissions is not reversed. The Status of Caribbean Coral Reefs after Bleaching and Hurricanes in 2005 report concluded:

> *"Coral reefs have experienced these effects of hurricanes and survived for millions of years; however, in light of the rapidly changing climate, the ability of corals to recover*

from severe storms, while facing the combined effects of increasing thermal stress and ocean acidification, could be extinguished."

The following discussion summarizes the currently available species status reviews, adding more recent information, as appropriate and available. The following sections describe common characteristics within the Acropora and Orbicella species before describing the individual listed species within those genera. These sections are followed by discussions of the Dendrogyra and Mycetophyllia ESA-listed species, the pillar coral and rough cactus coral, respectively.

7.2 Threatened *Acropora* Species with Populations in Florida Waters

Elkhorn and staghorn coral were the only large, branching species of coral to build and occupy once extensive complex reef environments within the Atlantic and Caribbean Sea's reef system, forming vast single species thickets. Classical descriptions of Caribbean reef morphology (Figure 4, Goreau, 1959) include an elkhorn zone consisting of elkhorn coral-dominated shallow reef crest habitats less than five meters deep (16 feet) and a staghorn zone consisting of staghorn coral thickets in fore reef shelf areas at seven to 15 meters depth (23 to 49 feet).

Life history of the threatened Acropora species with populations in Florida waters

In elkhorn and staghorn corals, fertilization and development is exclusively external to the parental colonies. These species are hermaphroditic and broadcast spawn packets of eggs and sperm simultaneously on only a few nights (nights two to six after the full moon) during July, August, or September. Some populations may have two spawning events over the course of two months. Large elkhorn and staghorn coral colonies produce proportionally more gametes than small colonies since basal and branch tip tissue are not fertile (Soong and Lang 1992). However, large thickets of healthy corals may have limited sexual reproductive potential if they are composed only of one or few genetically different individuals because viable larvae are only produced when the parents are genetically distinct (Baums et al. 2005b). Embryonic development culminates with the development of planktonic larvae called planulae. Coral planula larvae experience very high mortality from predation or other factors during their planktonic phase (Goreau et al. 1981). Details about the settlement patterns of elkhorn and staghorn corals planula larvae are unknown. However, certain species of crustose coralline algae have been shown to facilitate settlement and post-settlement survival in both species (Ritson-Williams et al. 2010).

In asexual reproduction through fragmentation, physical disturbance from storms or ship groundings usually initiate fragmentation. Reattachment occurs when either live coral tissue on the fragment grows onto suitable substrate or encrusting organisms settle on the dead basal areas of the fragment and cement it to the adjacent substratum (Tunnicliffe 1981).

The many small polyps and branching morphology of these corals optimizes light capture. This morphology is inefficient for zooplankton capture because zooplankton does not uniformly saturate the water column as light does, so densely arrayed polyps cannot be equally nourished through heterotrophy (Porter 1976).

Threats common to the threatened Acropora species with populations in Florida waters

The NMFS 2015 recovery plan for elkhorn and staghorn corals described growing evidence that disease, temperature-induced bleaching, and hurricanes, in combination with each other or with more moderately ranked threats, such as anthropogenic physical damage, nutrients, contaminants, sedimentation, competition, and predation, interact synergistically, exacerbating impacts and affecting the persistence of elkhorn and staghorn coral.

White band disease is thought to be the major factor responsible for the rapid loss of Atlantic *Acropora* species due to mass mortalities in the 1970s and 1980s. White band disease is the only coral disease to date that has been documented to cause major changes in the composition and structure of reefs (Humann and Deloach 2003). Other diseases affect Acroporid corals. In 2011, Sutherland et al. (2011) were able to definitively identify human waste as the source of stressors resulting in white pox disease in elkhorn corals.

There are many reports of coral disease outbreaks following hurricane disturbances. These include Puerto Rico (Bruckner and Bruckner 1997), Navassa (Miller and Williams 2007) the Florida Keys (Williams et al. 2008b), Honduras (Reich et al. 2001), Bonaire, and Curaçao (NMFS 2015b). Mechanism (s) that may explain this linkage of hurricanes and disease impacts are still unknown. Predators of elkhorn and staghorn corals can also serve as vectors for disease (Williams and Miller 2005). High temperature or rapid heating can result in heat shock and alter cellular metabolism within the coral as well as possibly hinder immune response or the ability of zooxanthellae to thrive (Rodriguez-Lanetty et al. 2009, Middlebrook et al. 2010). A link has also been demonstrated between increased coral disease prevalence and/or virulence and increased temperature (Harvell et al. 1999, Patterson et al. 2002). Temperature-induced bleaching was linked to subsequent disease-induced mortality in elkhorn coral following the 2005 bleaching event in the U.S. Virgin Islands (Muller et al. 2008). Ritchie (2006) showed reduced antibiotic activity in the mucus of healthy coral under bleaching conditions. Shifts to a pathogen dominated microbial community can persist long after transient stressful conditions have abated, leading to a long-term loss of innate defenses (Mao-Jones et al. 2010).

The dependence of elkhorn and staghorn corals almost entirely upon symbiotic photosynthesizers for nourishment makes them more susceptible to increases in water turbidity and temperature. Decreases in long-term water clarity can reduce the coral production to respiration ratio below one, meaning the colony is using more energy than is created generated by photosynthesis by the zooxanthellae. Consumption of zooplankton and nutrients from suspended particulate matter may not adequately compensate for reduced light penetration and subsequent reductions in photosynthesis by zooxanthellae. Elkhorn and staghorn corals may not be as resilient following bleaching events as coral species that can compensate through capture of plankton and particulate matter for food (Grottoli et al. 2006). Different strains of symbiotic zooxanthellae (*Symbiodinium* spp.) can confer different thermal and light tolerances to acroporid coral species (Abrego et al. 2009, Ainsworth and Hoegh-Guldberg 2009, Abrego et al. 2010).

While the dominant asexual mode of reproduction for elkhorn and staghorn corals allows rapid recovery from physical disturbances such as storms, reproduction through fragmentation makes recovery from disease or bleaching episodes very difficult because entire colonies or even entire stands are easily killed due to the lack of genetic diversity resulting in nearly identical vulnerability among individuals. The large role of asexual reproduction in both species increases the likelihood that genetic diversity in remnant populations may be very low.

There is substantial evidence to suggest that sexual recruitment of staghorn corals is currently compromised. As broadcast spawners, once colonies become rare, the distance between *Acropora* colonies can limit fertilization success. Reduced colony density in some areas is compounded by low genotypic diversity, indicating that fertilization success and consequently, larval availability, is likely reduced. The reduced potential for exchange of genetic material between populations due to reduced density and increasing inter-population distances can have long-term implications for genetic variability of remaining colonies (Bruckner 2002).

Elkhorn and staghorn corals are particularly susceptible to damage from sedimentation. Synergistic analyses have found that high temperature increases the risk of colony mortality under a variety of sediment loading conditions, but excessive sediment appears to reduce mortality risk under high light and temperature regimes, possibly by reducing exposure to these stressors (Anthony et al. 2007, Boyett et al. 2007). High sediment with otherwise good light and temperature conditions appears to increase colony mortality (Anthony et al. 2007).

Status of threatened Acropora species

Precipitous declines in population density attributed to disease occurred in the 1970s and 1980s. Where quantitative historical data are available, this initial decline is estimated at up to 98 percent reduction in abundance (NMFS 2015b). The best scientific data available show that the current general geographical distribution of elkhorn and staghorn corals has remained unchanged from this historical extent, however, the percentage of reefs where the two species were historically present has declined (Jackson et al. 2014).

Since the 2006 listing of elkhorn and staghorn coral as threatened, continued population declines have occurred in some locations. Some populations of both species have decreased up to an additional 50 percent or more of the remaining population (Lundgren and Hillis-Starr 2008, Muller et al. 2008, Williams et al. 2008a, Colella et al. 2012, Rogers and Muller 2012), likely due in large part to coral mass bleaching in 2005 that led to disease outbreaks. There are small pockets of remnant robust populations in southeast Florida (Vargas-Angel et al. 2003), Honduras (Keck et al. 2005, Riegl et al. 2009) and the Dominican Republic (Lirman et al. 2010). Abundance estimates from the Florida Keys and St. Croix, U.S. Virgin Islands suggested that there are at least hundreds of thousands of elkhorn and staghorn coral colonies in these areas. Absolute abundance within the species range is higher since this species occurs across the Caribbean. Given the frequency of asexual reproduction in *Acropora* species in comparison to sexual reproduction, the genetically effective population size is smaller than abundance estimates (NMFS 2015b).

NMFS expects episodic mass-mortality events caused by bleaching, disease, and/or physical disturbances will continue in the future. Where recruitment has been reported, NMFS expects subsequent mortality rates potentially impede growth into the large, mature colonies (greater than one meter, three feet, in colony diameter) and development of the *Acropora* species thickets that were historically important to habitat structure and reef productivity (Grober-Dunsmore et al. 2007).

Elkhorn and staghorn coral designated critical habitat in Florida waters

Critical habitat units for elkhorn and staghorn coral were designated in 2008 and include portions of Southeastern Florida and the Florida Keys, Puerto Rico, St. Thomas/St. John, and St. Croix. The Florida unit comprises approximately 1,329 square miles of marine habitat (Figure 11, NMFS 2015c); Puerto Rico approximately 1,383 square miles; St. Thomas/St. John approximately 121 square miles; and St. Croix approximately 126 square miles. Thus, the total area covered by the designation is approximately 2,959 square miles. Within the geographic area occupied by a listed species, designated critical habitat consists of specific areas with those physical

Figure 11. Designated critical habitat for elkhorn and staghorn coral in Florida waters.

or biological features essential to the conservation of the species. The feature essential to the conservation of Atlantic acroporid corals is substrate of suitable quality and availability in water depths from the mean high water line to 30 meters to allow for successful sexual and asexual reproduction. Successful sexual and asexual reproduction includes flourishing larval settlement, recruitment, and reattachment of coral 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.

Recovery goals for the threatened Acropora species with populations in Florida waters

The 2015 Elkhorn Coral and Staghorn Coral Recovery Plan contains complete down listing/delisting criteria for each of the two species based on the following recovery objectives and underlying recovery goals (NMFS 2015b).

1. **Ensure population viability:** 1) Preserve abundance (specific to each species); 2) Maintain genotypic diversity; and 3) Properly observe and record recruitment rates.

2. **Eliminate or sufficiently abate global, regional, and local threats**: 1) Develop quantitative recovery criterion through research to identify, treat, and reduce outbreaks of coral disease; 2) Control the local and global impacts of rising ocean temperature and acidification; 3) Reduce the loss of recruitment habitat (if goal 1, criterion 1, preserving abundance, is met then this objective is complete); 4) Reduce sources of nutrients, sediments, and contaminants; 5) Develop and adopt appropriate and effective regulatory mechanisms to abate threats; 6) Reduce impacts of natural and anthropogenic abrasion and breakage; and 7) Reduce impacts of predation.

7.2.1 Elkhorn Coral

Elkhorn coral colonies have frond-like branches, which appear flattened to near round, and typically radiate out from a central trunk and angle upward (Figure 12). Branches are up to approximately 20 inches (50 centimeters) wide and range in thickness from about 1.5 to two inches (four to five centimeters). Individual colonies can grow to at least 6.5 feet (two meters) in height and 13 feet (four meters) in diameter (ABRT 2005).

Figure 12. Elkhorn coral colony.

Goreau (1959) described ten habitat zones on a Jamaican fringing reef from inshore to the deep slope, finding elkhorn coral in eight of the ten zones (Figure 4). Elkhorn coral commonly grows in turbulent water on the fore-reef, reef crest, and shallow spur-and-groove zone (Shinn 1963, Cairns 1982, Rogers et al. 1982b, Miller et al. 2008) within its range (Figure 13, IUCN 2017) in water ranging from approximately three to 15 feet (one to five meters). Elkhorn coral often grows in thickets in fringing and barrier reefs (Jaap 1984, Tomascik and Sander 1987, Wheaton and Jaap 1988). The species formed extensive barrier-reef structures in Belize (Cairns 1982), the greater and lesser Corn Islands, Nicaragua (Lighty et al. 1982), and Roatan, Honduras, and built extensive fringing reef structures throughout much of the Caribbean (Adey 1978). Early studies termed the reef crest and adjacent seaward areas from the surface down to approximately 20 feet (five to six meters) depth the "palmata zone" because of the domination by the species (Goreau 1959, Shinn 1963).
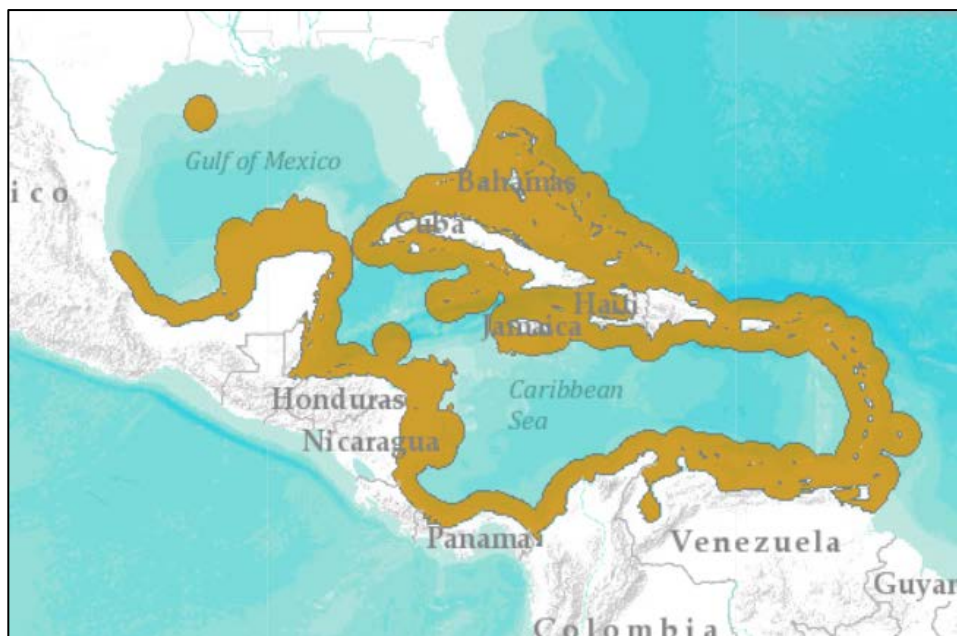
Figure 13. Elkhorn coral range.

Life History. The estimated size at sexual maturity is approximately 250 square inches (1,600 square centimeters), 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).

Depending on the location and size of the colony, physical growth rates for elkhorn coral branches range from approximately four to eleven centimeters (1.6 to 4.3 inches) per year (Vaughan 1915, Jaap 1974, Gladfelter et al. 1978, García U. et al. 1996, Becker and Mueller 2001). Individual colonies can grow to at least two meters in height and four meters in diameter (ABRT 2005).

Annual linear extension has been found to be dependent on the size of the colony (Padilla and Lara 1996), and new recruits and juveniles typically grow at slower rates. Additionally, stressed colonies and fragments may also exhibit slower growth. For example, some fragments at the Fortuna Reefer vessel grounding site at Mona Island, Puerto Rico failed to show any measurable growth over ten years (Bruckner et al. 2008).

As stated previously, reproduction is primarily through fragmentation generating multiple genetically identical colonies. Elkhorn coral can quickly monopolize large spaces of shallow ocean floor through fragmentation. A branch of elkhorn coral may be carried by waves and currents away from the parent colony, and fragments cleaved from the colony may grow into new colonies (Highsmith et al. 1980, Bak and Criens 1982, Highsmith 1982, Rogers et al.

1982a). Distance between genetically identical clones ranges from 0.1 to 100 meters (0.3 to 328 feet), but usually less than 30 meters (98 feet, Baums et al, 2006a). Elkhorn coral fragment wounds healed rapidly (1.59 centimeters of linear growth per month; 0.62 inches per month). Nine months after Tropical Storm Gordon (1994 with landfall in Nicaragua, Jamaica, Cuba, and Florida), 157 of 218 fragments had fused to the sea floor, and protobranches on the fragments grew rapidly (Lirman and Fong 1997).

Population Dynamics. Microsatellite marker analysis of elkhorn coral from eleven locations throughout its geographic range indicate that genetic exchange in the eastern Caribbean with populations in the western Atlantic/Caribbean is limited to absent (Baums et al. 2005b). Puerto Rico is an area of mixing where elkhorn populations show genetic contribution from both regions, though it is more closely connected with the western Caribbean. Models suggest that the Mona Passage between the Dominican Republic and Puerto Rico promotes dispersion of larval and gene flow between the eastern and western Caribbean (Baums et al. 2006b). Within the regions of the Caribbean, the degree of larval exchange appears to be asymmetrical with some locations being entirely self-recruiting and some receiving immigrants from other locations within their region. Because the size and weight of fragments limit the dispersal range of elkhorn (Jackson 1986, Lirman 2000), recovery of damaged areas can be slow where populations of adult colonies (i.e., fragment source) are reduced significantly (Baums et al. 2006a).

Baums et al. (2006a) concluded that the western Caribbean had higher rates of asexual recruitment and that the eastern Caribbean had higher rates of sexual recruitment, possibly because of geographic differences such as the amount of shelf area available as habitat. In that study, the western Caribbean populations were characterized by low genetic variability and lower densities (0.13 ± 0.08 colonies per square meter) relative to the eastern Caribbean populations, which were denser (0.30 ± 0.21 colonies per square meter) and genotypically richer (Baums et al. 2006a).

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 off Navassa Island had unique genotypes (Baums et al. 2006a). Some sites have relatively high genotypic diversity such as in Los Roques, Venezuela (118 unique genotpyes out of 120 samples; Zubillaga et al. 2008) and in Bonaire and Curaçao (18 genotypes of 22 samples and 19 genotypes of 20 samples, respectively; Baums et al. 2006a). In the Bahamas, about one third of the sampled colonies were unique genotypes, and in Panama between 24 and 65 percent of the sampled colonies had unique genotypes, depending on the site (Baums et al. 2006a). This contrasts with a more-recent survey conducted along the coast of Puerto Rico, which found unique genotypes in 75 percent of the samples with high genetic diversity (Mège et al. 2014).

Status. Historically elkhorn coral inhabited most waters of the Caribbean between one to five meters 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 (Goreau 1959, Kornicker and Boyd 1962, Storr 1964, Scatterday 1974, Jaap 1984). 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.

The 2014 report on the status and trends of Caribbean coral reefs (Jackson et al. 2014) indicates that the proportion of reefs where the species occurred declined over the past century (Figure 14, panel A). Since the large mortality events of the 1970s and 1980s, the proportion of reefs where elkhorn coral is a dominant species and percent cover of elkhorn has remained depleted (Figure 14, panels B and C, Jackson et al, 2014).

The overall number of reefs with elkhorn coral present in Caribbean reefs declined during the 1980s from approximately 50 to 30 percent of reefs, remaining relatively stable at 30 percent through the 1990s, thereafter declining to approximately ten percent over 2005-2011 (Jackson et al. 2014). The decline in

Figure 14. Trends in elkhorn population estimates over the past century. Adapted from Jckson et al. 2014. Numbers next to points indicate number of studies.

the total abundance of elkhorn coral is 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 (Mayor et al. 2006, Brainard et al. 2011, Hernandez-Delgado et al. 2011, Rogers and Muller 2012, Downs et al. 2016). It is believed that these effects of stressors act synergistically with one another thereby increasing the overall damage to already- stressed elkhorn colonies that have undergone disturbance by another threat.
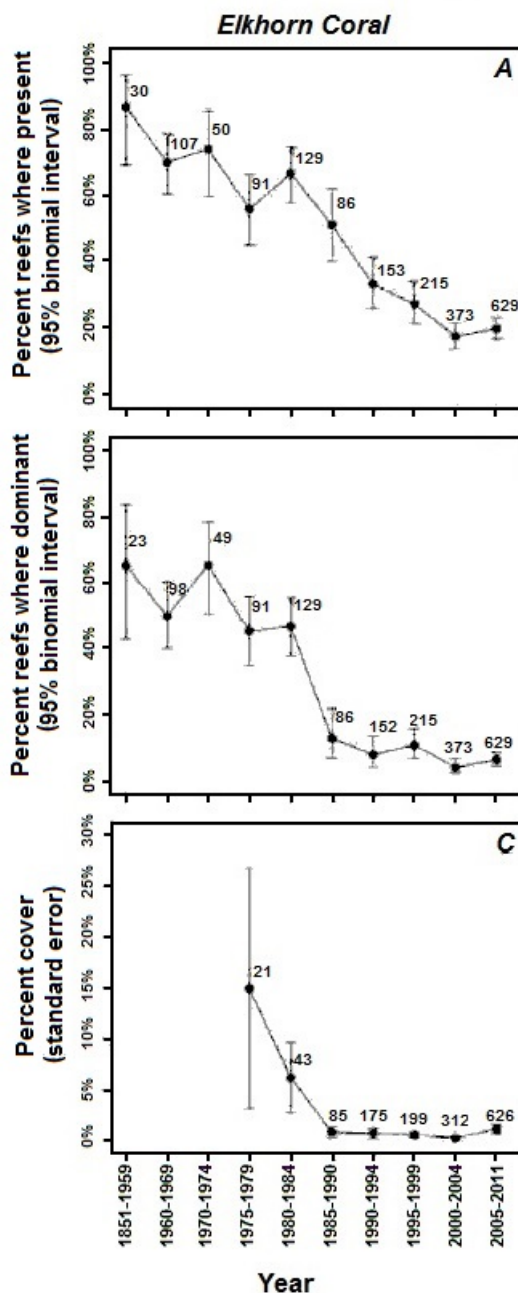
Percent cover appears to have remained relatively stable, albeit it at extremely low levels with a possibly reduced range, since a population crash in the 1980s. However, the proportions of Caribbean sites where elkhorn coral is present and dominant have stabilized since the mid-2000s. In 2012, the population trend appeared to be steady, although there were places where populations continue to decrease and others where there appeared to be modest recovery (Miller et al. 2013).

Elkhorn coral has been locally extirpated from many areas throughout its range (Jackson et al. 2014). Extrapolated elkhorn coral population estimates, and standard errors around those estimates, from stratified random samples across habitat types in the Florida Keys were 0.6 ± 0.5 million colonies in 2005, 1.0 ± 0.3 million colonies in 2007, and 0.5 ± 0.3 million colonies in 2012. Because these estimates are based on random sampling, differences between years may be a function of sampling effort rather than an indication of population trends. Relative to the abundance of other corals in the Florida Keys region, elkhorn coral was among the least abundant, ranking among corals that are naturally rare in abundance despite historically being a dominant species on Florida reefs.

Elasticity analysis from a population model based on data from the Florida Keys has shown that the largest individuals have the greatest contribution to the rate of change in population size (Vardi et al. 2012). The size class distribution of the Florida Keys population included both small and large individuals (> approximately 103 inches, or 260 centimeters), but after 2005 the majority of the colonies were smaller in size. These smallest corals (up to eight inches, or 20 centimeters) had approximately zero to two percent partial mortality during all three survey years. Partial mortality across all other size classes was approximately 20 to 70 percent in 2005, five to 50 percent in 2007, and 15-90 percent in 2012 (Miller et al. 2013).

Colonies monitored in the upper Florida Keys showed a greater than 50 percent loss of tissue as well as a decline in the number of colonies, and a decline in the dominance by large colonies between 2004 and 2010 (Vardi et al. 2012, Williams and Miller 2012). Between 2010 and 2013, elkhorn coral in the middle and lower Florida Keys had mixed trends. Population densities remained relatively stable at two sites and decreased at two sites by 21 percent and 28 percent, respectively (Lunz 2013).

The northern extent of the range in the Atlantic is Broward County, Florida, where it is relatively rare (only a few known colonies), but fossil elkhorn coral reef framework extends into Palm Beach County, Florida. Two colonies of elkhorn coral were discovered in 2003 and 2005 at the Flower Garden Banks, located 100 miles (161 kilometers) off the coast of Texas in the Gulf of Mexico (Zimmer et al. 2006). Since listing, catastrophic mortality events for half of the existing monitored elkhorn coral populations in U.S. waters have included mass-bleaching events in the U.S. Virgin Islands (Lundgren and Hillis-Starr 2008, Muller et al. 2008) and hurricanes/disease in the Florida Keys (Williams et al. 2008a). Williams and Miller (2012) estimated that elkhorn coral would require more than ten years to recover after the 2005 mass mortality event (greater

than 50 percent population. Yet, mass mortalities in this population occur more frequently than every ten years (i.e., 1997-8 and 2005).

7.2.2 Staghorn Coral

Staghorn coral polyps form antler-like colonies with straight or slightly curved, cylindrical branches (Figure 15). The diameter of branches ranges from 0.25-5 centimeters (Lirman et al. 2010), The species can exist as isolated branches, individual colonies up to about 1.5 meters diameter, and thickets comprised of multiple colonies that are difficult to distinguish from one another (ABRT 2005).

Staghorn coral is distributed throughout the Caribbean Sea, the southwestern Gulf of Mexico, and the western Atlantic Ocean (Figure 16, IUCN 2017). 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.

Figure 15. Staghorn coral colony.

1978). This is the northern extent of its current distribution, where it occurs in deeper water (16 to 30 meters; 53 to 98 feet, Goldberg 1973). South and west throughout the coral and hard bottom habitats off Broward County, staghorn coral is distributed across its depth range (5-30 meters), where it historically formed extensive thickets, and southward to Miami-Dade County, the Florida Keys, and the Dry Tortugas (Jaap 1984, Walker and Klug 2014).

Staghorn coral naturally occurs on spur and groove, bank reef, patch reef, and transitional reef habitats, as well as on limestone ridges, terraces, and hard bottom habitats (Goldberg 1973, Gilmore and Hall 1976, Cairns 1982, Davis 1982, Jaap 1984, Wheaton and Jaap 1988, Muller et
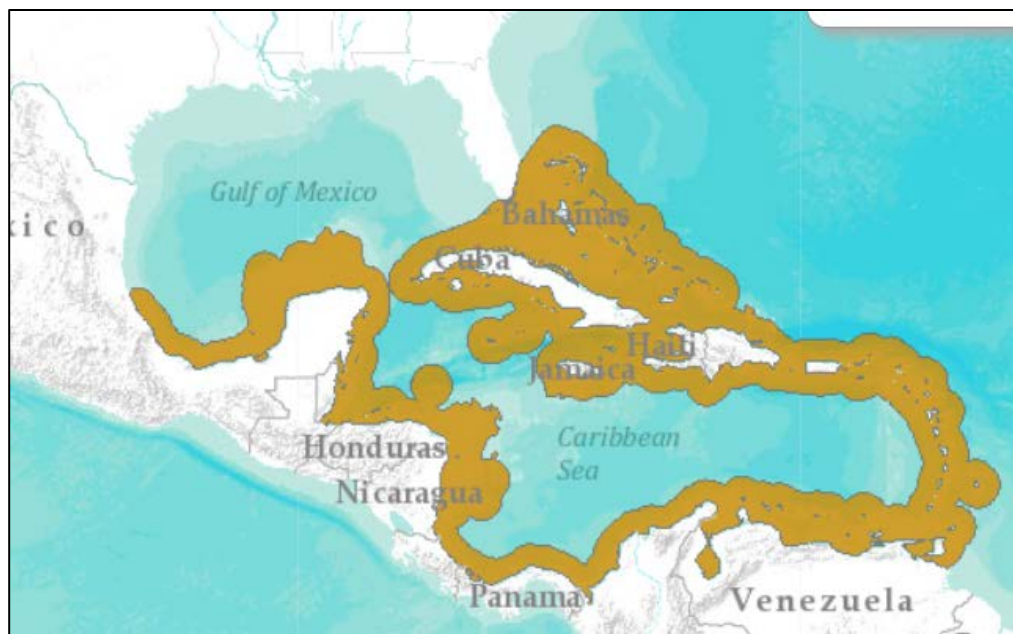
Figure 16. Staghorn coral range.

al. 2008). This habitat heterogeneity moderates vulnerability to extinction because disturbances in thermal regimes and ocean chemistry usually vary over location, habitat, and time, so stressor events, or events with combined stressors, are not likely to be experienced by all populations simultaneously. Historically staghorn coral grew in thickets in water ranging from approximately five to 20 meters in depth, though it has been found to approximately 60 meters in specific sites (Davis 1982, Jaap 1984, Schuhmacher and Zibrowius 1985, Wheaton and Jaap 1988, Jaap et al. 1989). At the northern extent of its range, it grows in deeper water, 16-30 meter depth (Goldberg 1973). In the Florida Keys, staghorn coral occurs in various habitats but is now most prevalent on patch reefs as opposed to the deeper fore-reef habitats (i.e., five to 22 meters) where it was formerly abundant (Miller et al. 2008).

Precht and Aronson (2004) suggest that staghorn coral only recently re-occupied its historic range coincident with climate warming. They based this idea on the presence of large thickets off Fort Lauderdale, Florida, which were discovered in 1998 and had not been reported in the 1970s or 1980s (Precht and Aronson 2004). However, because staghorn coral colonies were present in Palm Beach County, north of Fort Lauderdale, in the early 1970s (Goldberg 1973), these thickets may have been present, but undetected, prior to 1998.

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.

The skeletal growth rate for staghorn coral has been reported to range from three to 11.5 centimeters, or one to five inches (Vaughan 1915, Shinn 1966, Jaap 1974, Shinn 1976, Gladfelter et al. 1978, Becker and Mueller 2001). This growth rate is relatively fast in comparison to other scleractinian corals and historically enabled these species to construct significant reef structures in several locations throughout the Atlantic/Caribbean (Adey 1978). During daylight, calcium carbonate ($CaCO_3$) accretion occurs on all of the skeletal elements of staghorn coral; at night, the activity is limited to crystal formation at the extending tips of skeletal elements.

Staghorn coral is a hermaphroditic broadcast spawning species. The spawning season occurs several nights after the full moon in July, August, or September. Spawning is dependent on the location and timing of the full moon and, rarely, 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, which facilitate larval settlement and post-settlement survival (Ritson-Williams et al. 2010). As stated previously, while reproduction through

fragmentation can allow for rapid post disturbance recolonization, reduced genetic diversity limits the capacity of elkhorn to repopulate spatially dispersed sites.

Population dynamics. (Miller et al. 2013) extrapolated population abundance of staghorn coral in the Florida Keys and Dry Tortugas from stratified random samples across habitat types. Population estimates and standard errors of staghorn coral in the Florida Keys were 10.2 ± 4.6 million colonies in 2005, 6.9 ± 2.4 million colonies in 2007 and 10.0 ± 3.1 million colonies in 2012. Population estimates in the Dry Tortugas were 0.4 ± 0.4 million colonies in 2006 and 3.5 ± 2.9 million colonies in 2008, though the authors note their sampling scheme in the Dry Tortugas was not optimized for staghorn coral. Because these estimates were based on random sampling, differences between years more likely reflects variability due to the sampling design than population trends. Populations in both the Florida Keys and Dry Tortugas were dominated by small colonies less than 12 inches (30 centimeters) in diameter. Partial mortality appeared highest in 2005 and lowest in 2007.

Fragmentation, coupled with a relatively fast skeletal growth rate, facilitates potential spatial competitive superiority for staghorn coral relative to other corals and other benthic organisms (Shinn 1976, Neigel and Avise 1983, Jaap et al. 1989). Staghorn coral historically was one of the dominant species on most Caribbean reefs, forming large, single-species thickets referred to as the staghorn zone in classical descriptions of Caribbean reef morphology (Goreau 1959). Massive, Caribbean-wide mortality, attributed to 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 hurricanes, disease outbreaks, and mass bleaching events has added to the decline of staghorn coral (Brainard et al. 2011). In locations where quantitative data are available (Florida, Jamaica, U.S. Virgin Islands, Belize), between the 1970s and early 2000s there was a reduction of approximately 92 to greater than 97 percent (ABRT 2005) of the staghorn coral population.

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

While cover of staghorn coral on a Jamaican reef increased from 0.6 percent in 1995 to 10.5 percent in 2004 and to 44 percent in 2005 (Idjadi et al. 2006), it collapsed after the 2005 bleaching event and subsequent disease to less than 0.5 percent in 2006 (Quinn and Kojis 2008). A cold water die-off across the lower to upper Florida Keys in January 2010 resulted in the complete mortality of all staghorn coral colonies at 45 of the 74 reefs surveyed (61 percent, Schopmeyer et al. 2012). Walker et al. (2012) report increasing size of two thickets (expansion of up to 7.5 times the original size of one of the thickets) monitored off southeast Florida, but also noted that cover within monitored plots concurrently decreased by about 50 percent highlighting the dynamic nature of staghorn coral distribution via fragmentation and re-attachment. Lidz and Zawada (2013) observed 400 colonies of staghorn coral along 44 miles (70.2 kilometers) 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.

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 low gene flow occurred across the greater Caribbean such that populations greater than approximately 500 kilometers apart are genetically different from each other. This is consistent with studies conducted on other Caribbean corals showing that gene flow is restricted at spatial scales over 500 kilometers or 310 miles (Fukami et al. 2004, Baums et al. 2005b, Brazeau et al. 2005).

Fine-scale genetic differences have been detected at reefs separated by as little as two kilometers, suggesting that gene flow in staghorn coral may not occur at much smaller spatial scales (Vollmer and Palumbi 2007, Garcia Reyes and Schizas 2010). This fine-scale population structure was greater when considering genes of elkhorn coral were found in staghorn coral due to back-crossing of the hybrid *A. prolifera* with staghorn coral. Populations in Florida and Honduras are genetically distinct from each other and other populations in the U.S. Virgin Islands, Puerto Rico, Bahamas, and Navassa (Baums et al. 2010), indicating little to no larval connectivity overall. However, some potential connectivity between the U.S. Virgin Islands and Puerto Rico was detected and also between Navassa and the Bahamas (Baums et al. 2010). Both the Baums et al. (2010) and Vollmer and Palumbi (2007) acroporid population studies suggest that no population is more or less significant to the status of these species and there is limited ability of reefs to seed one another over large distances.

Status. Historically, staghorn coral was one of the primary constructors of mid-depth ten to 15 meter reef terraces in the western Caribbean, including Jamaica, the Cayman Islands, Belize, and some reefs along the eastern Yucatan peninsula (Adey 1978). The 2014 report on the status and trends of Caribbean coral reefs (Jackson et al. 2014) indicates that the proportion of reefs where the species occurred declined over the past century (Figure 17, panel A). Since the large mortality events of the 1970s and 1980s, the proportion of reefs where staghorn is a dominant species has declined and percent cover of staghorn has remained at a depleted level (Figure 17, panels B and C, Jackson et al. 2014).

The overall number of reefs with staghorn coral present declined during the 1980s from approximately 50 to 30 percent of reefs, remaining relatively stable at 30 percent through the 1990s, then decreasing to approximately 20 percent of the reefs in 2000 to 2004 and to approximately ten percent in 2005-2011 (Jackson et al. 2014). As of 2015, populations appeared to consist mostly of isolated colonies or small thickets of 0.5 to one meter (1.6 to three feet) across compared to the vast thickets once prominent throughout its range (NMFS 2015b).

A stratified random survey designed to detect *Acropora* species colonies along the south, southeast, southwest, and west coasts of Puerto Rico identified staghorn coral in 21 out of 301 stations between 2011 and 2013 (García Sais et al. 2013).

Figure 17. Historical trend in staghorn coral decline. Adapted from Jackson et al. (2014). Numbers next to points indicate number of stations.

Staghorn coral was also observed at 16 sites outside of the surveyed area. The largest colony was 60 centimeters and density ranged from one to ten colonies per fifteen square meters (García Sais et al. 2013).
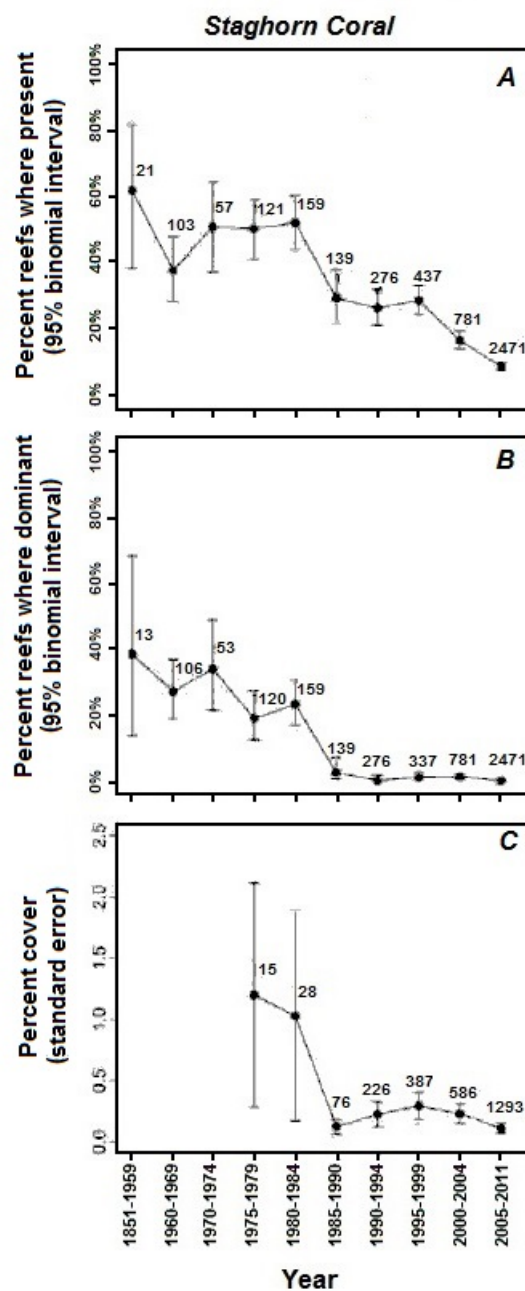
54

The 2015 recovery plan indicates that there has been no evidence of range constriction, though loss of staghorn coral at the reef level has occurred (NMFS 2015b). 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 continue to occur, percent benthic cover and proportion of reefs where staghorn coral is dominant remained stable over its range from the mid-1980s to 2011 (Figure 17, panel B, Jackson et al, 2014). Currently staghorn corals in the Florida Keys occur primarily in patch reefs as opposed to their former abundance in deeper fore reef habitats (Miller et al. 2008).

7.3 Threatened *Orbicella* Species with Populations in Florida Waters

On September 10, 2014, NMFS listed three star coral species as threatened (79 FR 53851): boulder star coral (*Orbicella franksi*), lobed star coral (*Orbicella annularis*), and mountainous star coral (*Orbicella faveolata*). These three species are in the *Orbicella* spp. complex, formerly in the genus *Montastraea*. The complex has been found at depths to 90 m. It is dominant on mesophotic reefs in Puerto Rico and the U.S. Virgin Islands at depths of 30 to 45 meters, but also occurs in waters as deep 90 meters in these locations. In classical descriptions of Caribbean reefs, the *Orbicella* species were the dominant members of reef "buttress" and "annularis" zones (Goreau 1959, Bak 1977).

Recent work has reclassified these three species to the genus *Orbicella* (Budd et al. 2012). The complex was originally considered a single species –*Orbicella annularis*– with varying growth forms ranging from columns, to massive boulders, to plates. In the early 1990s, Weil and Knowlton (1994) suggested the partitioning of these growth forms into separate species, resurrecting the previously described taxa, *Montastraea (*now *Orbicella*) *faveolata*, and *Montastraea* (now *Orbicella*) *franksi.* The 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 *Orbicella annularis* complex into three species. Studies published prior to 1994, before *Orbicella annularis* was split into the three separate species, reflect the species complex. Many studies published in subsequent years also discuss the species as a complex due to their similarity in ecology and appearance. Information reported below reflects knowledge of the species complex. Where species-specific information is available, it is reported in a separate section for each species.

Life history of threatened Orbicella with populations in Florida waters

The *Orbicella* species are slow growing and can live for centuries, becoming sexually mature after many years. The *Orbicella* species are reported to have the lowest larval recruitment rates among the western Atlantic reef-building corals (Szmant-Froelich 1985). 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 timing of the full moon and location. Boulder star coral spawning is reported to be about one to two hours earlier than lobed star coral and mountainous star coral. All three species are largely self-incompatible (Knowlton et al. 1997, Szmant et al. 1997). Fertilization success measured in

the field was generally below 15 percent for all three species, as it was closely linked to the number of colonies concurrently spawning (Levitan et al. 2004). In Puerto Rico, minimum size at reproduction for the *O. annularis* complex was 83 square centimeters. The star coral complex has growth rates ranging from 0.06-1.2 centimeters per year and averaging approximately one-centimeter linear growth per year (Szmant-Froelich 1985).

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 *O. annularis* complex in the past (Bruckner 2012a). Only a single recruit of *Orbicella* species was observed over 18 years of intensive observation of approximately 12 square meters of reef in Discovery Bay, Jamaica (Hughes and Tanner 2000). Smith and Aronson (2006) examined 384 permanent monitoring quadrats (237 square meters) in the lower Florida Keys between 1998 and 2003 and reported only 18 *Orbicella* species recruits. Many other studies throughout the Caribbean also report negligible to absent recruitment of the complex (Bak and Engel 1979, Rogers et al. 1984, Edmunds et al. 2011). The ability of the complex to dominate with such low recruitment rates has been described as a storage effect whereby large, old colonies are able to persist and maintain the population until favorable conditions for recruitment occur (Edmunds and Elahi, 2007). However, potential problems may exist for species employing storage effects if favorable conditions for recruitment occur so infrequently that they fall outside the life span of the colony (Foster et al., 2013).

While the longevity and life history strategy of the *O. annularis* complex buffer the taxa against stressful periods and allow species to remain relatively abundant, the listing of the complex is based on the reduction in that buffering capacity due to recent population declines and partial mortality, particularly in large colonies.

Population dynamics of threatened Orbicella with populations in Florida waters

The *O. annularis* complex often makes up the largest proportion of coral cover on Caribbean reefs (Burns 1985, Nemeth et al. 2008, Stokes et al. 2010, Bruckner 2012b). However, numerous examples of population decline in terms of cover, abundance, and condition are identified in the 2014 listing determination. Population declines in the Florida Keys between the late 1970s and 2003 was approximately 80 to 95 percent, with further losses during the 2012 cold weather event (Dustan and Halas 1987, Dupont et al. 2008, Smith et al. 2008). Since 1995, disease, mass bleaching events (1995, 1998, 2005, 2009), predation, and increased competition by other benthic organisms have contributed to a declining trend in this complex (Bruckner and Bruckner 2006b, a, Miller et al. 2006, Edmunds and Elahi 2007, Bruckner and Hill 2009, Rogers et al. 2009, Hughes et al. 2017). Bleaching events were often followed by disease outbreaks, a stressor scenario considered to be the principle cause of *Orbicella* species mortality (Bruckner 2012a). These corals are susceptible to disease and can harbor multiple infections simultaneously (Bruckner and Bruckner 2006a).

Decadal-scale declines across the remote islands of Navassa, Mona, and Desecheo in the central Caribbean affected 85 percent of colonies found there (Miller and Williams 2007, Bruckner and

Hill 2009, Jackson et al. 2014). In the U.S. Caribbean (U.S. Virgin Islands and Puerto Rico), an 80 to 90 percent decline has been reported over the past two decades (Edmunds and Elahi 2007, Miller et al. 2009, Hernandez-Pacheco et al. 2011). While percent cover was reportedly stable in Curaçao in the mid-1970s, an 85 percent increase in partial mortality was observed between 1998 and 2005 (Bak and Luckhurst 1980, Bruckner and Bruckner 2006b). Between 1975 and 1998 at Glovers Reef in Belize, a 38 to 75 percent decline in relative cover occurred, with a further 40 percent decline since (Mcclanahan and Muthiga 1998, Huntington et al. 2011). Colonies in Colombia were stable between 1998 and 2003 although demographic changes imply some degree of decline (Bruckner 2012a).

Whole colony death was less commonly observed than partial tissue mortality among coral colonies at 185 sites in five countries (Bahamas, Bonaire, Cayman Islands, Puerto Rico, and St. Kitts and Nevis). Partial tissue mortality was three to nearly four times higher in *Orbicella* species than observed in other species, with other species colonizing exposed *Orbicella* species skeleton alongside the remaining colony polyps. The primary cause for mortality was described as bleaching events that were followed by outbreaks of disease. Additional mortality was attributed to increased predation of live coral and removal of live tissue by damselfish cultivating algal lawns on the *Orbicella* species skeleton, bioerosion by sponges, and increased competition with other species (Bruckner 2012a).

Surveys in Bonaire in 2008 showed that the *O. annularis* complex dominated coral cover in depths less than 20 meters and cover was similar to that reported in 1982. However, all sites surveyed showed signs of disease and partial mortality in a large number of the massive colonies, and many were reduced to a patchwork of live tissue and dead areas colonized by algae (Stokes et al., 2010). In 2011, *O. annularis* complex was still the dominant coral taxa. Most colonies were between 30 and 80 centimeters in diameter and there was a notable absence of colonies less than ten centimeters in diameter (as measured by the skeleton, not live tissue) and an absence of recruits. Whole mortality was observed in 4.5 percent of colonies. Surviving colonies had a mean of 28 percent partial mortality, with larger surviving colonies divided into average of 6.6 tissue remnants. The reefs at Bonaire are still in relatively good condition and several sites contained a high abundance of large, unblemished *O. annularis* complex colonies (Bruckner, 2012c).

In surveys of juvenile corals (less than four centimeters in diameter) on nine reefs in the Florida Keys between 1993 and 1994, density of members of the *Orbicella* complex ranged between 0.02 and 0.04 juvenile corals per square meter on six of the nine reefs. Density of *Orbicella* species juveniles was correlated with non-juvenile *Orbicella* species density and with depth (Chiappone and Sullivan 1996). Between 1999 and 2009, overall cover of *Orbicella* species in the Florida Keys on the deep and shallow fore-reefs but remained stable on patch reefs (Ruzicka et al. 2013). A comparison of 1995 and 2005 surveys of *Orbicella* species at 13 patch reefs in the Florida Keys reported ten sites had between five and 40 percent more dead areas (Gischler 2007). The 2010 cold-water event reduced cover of *Orbicella* species from 4.4 percent to 0.6 percent on four patch reefs in the upper and middle Florida Keys. Greater than 50 percent of

Orbicella species colonies across all size classes suffered lethal or severe mortality, and 93 percent of all *Orbicella* species colonies surveyed suffered complete or partial mortality. The complex suffered the highest mortality of all coral species affected (Colella et al. 2012). Using *Orbicella* species population metrics observed in in 1998 and 2003 at four sites in the Lower Florida Keys, (Smith and Aronson 2006) developed size-transition matrices that forecasted a steady population decline over 15 years driven by insufficient recruitment and low growth rates of smaller size classes. The forecast indicated a drop below 50 percent of initial population size within seven to ten years of their 2006 publication (i.e., between 2013 and 2016).

Coral condition in a four-kilometer area on the south side of St. John, U.S. Virgin Islands has been monitored since 1994. Density of juvenile colonies of the *O. annularis* complex increased from 0.07 juveniles per square meter to 0.15 juveniles per square meter in 2008, and continued at 0.12 juveniles per square meter in 2009. The increased density did not extend outside the initial survey area. While not possible to distinguish the species in the field, the authors conclude juveniles were most likely *O. annularis* due to the abundance of *O. annularis* on adjacent reefs and the rarity of the presence of the other two species in water less than nine meters (Edmunds et al. 2011).

At Yawzi Point, St. John, U.S. Virgin Islands, the percentage of total coral cover declined by more than 50 percent between 1987 to 1998, from 45 percent to 20 percent. In 1988, 94 percent of the coral cover at Yawzi was *O. annularis* complex mostly *O. annularis* (97 percent), with a few colonies of *O. faveolata* (six percent). Despite a reduction in total cover, *O. annularis* complex remained spatially dominant in 1998 at 96 percent of the coral cover (Edmunds, 2002). Coral cover at this site again declined an additional 65 percent between 1999 and 2011 to seven percent cover, with *O. annularis* complex remaining dominant at 77 percent of the coral cover (Edmunds, 2013).

At Tektite Reef, St. John, U.S. Virgin Islands, total coral cover increased from 32 percent in 1987 to 43 percent in 1998 but then decreased to 29 percent in 2011 (Edmunds, 2002; Edmunds, 2013). In 1988, 79 percent of the complex was *O. annularis*, with lesser amounts of *O.* faveolata (one percent) and *O.* franksi (21 percent)(Edmunds, 2002). Greater than 72 percent of coral cover at the site was *O. annularis* complex in all survey years (Edmunds, 2013).

Surveys of the Flower Garden Banks between 1974 and 1980 found cover of *O. annularis* complex between approximately 23 and 40 percent in areas less than 36 meters depth (Bright et al., 1984). Species from the *O. annularis* complex were the dominant corals between 2002 and 2003 at 32 percent cover (Aronson et al., 2005). In random surveys between 2002 and 2006, *O. annularis* complex (predominantly *O. franksi*) dominated coral cover in the Flower Garden Banks comprising between 27 and 40 percent of the benthic cover (Hickerson et al., 2008). In permanent photo quadrats (eight square meter total), cover of *O. annularis* complex (as measured by planar surface area of individual colonies) fluctuated between approximately 20 and 45 percent cover in the East Flower Gardens between 1992 and 2006 with periods of sharp increase and decrease in cover (Hickerson et al., 2008). Cover in west Flower Gardens was

between 22 and 40 percent over the same period and had less annual variability and a generally increasing or stable trend through time (Hickerson et al., 2008).

Surveys of five sites in the Mexican Yucatan in 1985 and 2005 revealed a decrease in relative cover of *O. annularis* complex. At four out of the five sites, cover of *O. annularis* complex decreased from approximately 50 to 60 percent in 1985 to approximately ten to 25 percent in 2005. The fifth site had a less dramatic decrease in relative cover from approximately 35 percent to 30 percent cover during this 20-year interval. Disease appeared to be the main cause of decline, but hurricanes may have also played a role (Harvell et al., 2007).

The abundance and population trends summary in the 2014 listing determination for the *O. annularis* complex states that while these corals were historically dominant in Caribbean fore-reef sites, reports of recent declines in *O. annularis* complex cover informed the listing determination. Major declines in the species cover range from approximately 50 to 95 percent in locations including Puerto Rico, Belize, the Florida Keys, Mexico, and the U.S. Virgin Islands, and lower levels of decline (five to 33 percent) have been reported at individual sites within some of these same locations. However, there have also been reports of more stable percent cover trends (e.g., Bonaire) and periods of increase (e.g., Flower Garden Banks). Observed declines in total coral cover in the Caribbean, since the major decline of *Acropora* spp. in the 1980s, have often been a result of the decline of the *O. annularis* complex because the taxa can make up a large proportion of the total coral cover. Despite decreases, the *O. annularis* complex continues to be reported as the dominant coral taxa, albeit at times its relative dominance has decreased to a lower percentage of the total coral cover (e.g., Curaçao, U.S. Virgin Islands).

Status of threatened Orbicella species with populations in Florida waters

At the time of the listing, the only comprehensive abundance data in the Caribbean were for the three *Orbicella* species, partially because they historically made up a predominant part of live coral cover. Even for these species, the time series data are often of very short duration (they were not separated as sibling species until the early 1990s and many surveys continue to report them as "*Orbicella annularis* complex'') and cover a very limited portion of the species range (e.g., the time series only monitors a sub-section of a single national park).

Decadal-scale declines across the remote islands of Navassa, Mona, and Desecheo in the central Caribbean impacted 85 percent of colonies found there. In the U.S. Caribbean (U.S. Virgin Islands and Puerto Rico), an 80 to 90 percent decline has been reported over the past two decades in the star coral complex. Percent cover was reportedly stable in Curaçao in the mid-1970s, but an 85 percent increase in partial mortality occurred between 1998 and 2005. Between 1975 and 1998 at Glovers Reef in Belize, a 38 to 75 percent decline in relative cover occurred with a further 40 percent decline since. Colonies in Colombia were stable between 1998 and 2003 although demographic changes imply some degree of decline. Surveys of population structure

across five countries found a significant increase in small ramets[8] less than 500 square centimeter while the proportion of large (1,500 to 30,000 square centimeter), completely live colonies declined by 51 to 57 percent (Bruckner 2012a).

In 1998 the *O. annularis* complex covered more of the benthos than any other coral taxon at nine monitored sites off Mona and Desecheo Islands, Puerto Rico: 47 percent on reefs off Desecheo Island and 32 percent off Mona Island. In 2008 live cover of *O. annularis* complex ranged from zero to 14 percent with 95 percent decline off Desecheo Island and 78 percent decline off Mona Island. This was accompanied by large changes in the size frequency distribution and extent of partial mortality, with size structure remaining constant. The amount of living tissue declined by 55 percent due to partial mortality affecting medium and large colonies, with an increase in the number of colonies with small (less than ten centimeters in diameter) tissue remnants. Sponges and macroalgae colonized newly exposed area, and sponges appeared to be preventing re-sheeting of tissue remnants. No *Orbicella* species spp. recruits were observed during the ten-year study off Mona (Bruckner and Hill 2009).

The NMFS' 2014 listing summarizes the status of the *Orbicella* complex as follows. Based on population estimates, there are at least tens of millions of *Orbicella* complex colonies in surveyed areas. Absolute abundance is higher than these estimates because the species occur in many other un-surveyed locations throughout their range. However, these species have undergone major declines mostly due to warming-induced bleaching and disease. Despite high declines, the species is still common and remains one of the most abundant species on Caribbean reefs. The *Orbicella* species are highly susceptible to a number of threats, and cumulative effects of multiple threats have likely contributed to decline and exacerbate vulnerability to extinction. There is evidence of synergistic effects of threats for this species including disease outbreaks following bleaching events and reduced thermal tolerance due to chronic local stressors stemming from land-based sources of pollution. 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 these species are limited to areas with high, localized human impacts and predicted increasing threats. The life history characteristics of large colony sizes and long life spans have enabled the *Orbicella* species to remain relatively persistent despite slow growth and low recruitment rates, thus moderating vulnerability to extinction. However, the buffering capacity of *Orbicella* species life history strategies that allowed these species to remain abundant has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. This buffering is expected to decrease as colonies shift to smaller size classes. The depth range for the *Orbicella* species moderates vulnerability to extinction over the foreseeable future because deeper areas of their range will usually have lower temperatures than surface waters, and acidification is generally predicted to accelerate most in waters that are deeper and cooler than

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[8] A ramet is a tissue isolate that is genetically identical but physiologically separate from the parent colony

those in which the species occurs. The *Orbicella* species also occur in most reef habitats and this moderates vulnerability to extinction over the foreseeable future because different types of reef environments are predicted, on local and regional scales, to experience highly variable thermal regimes and ocean chemistry at any given point in time.

In Florida, the percent cover data from four fixed sites have shown the *O. annularis* complex to have declined in absolute cover from five to two percent in the Lower Keys between 1998 and 2003 (Fig. 6.5A) and was accompanied by five to 40 percent colony shrinkage and virtually no recruitment (Smith et al., 2008). Earlier studies from the Florida Keys indicated a 31 percent decline of *O. annularis* complex absolute cover between 1975 and 1982 (Dustan and Halas, 1987) at Carysfort Reef and > 75 percent decline (from over six percent cover to less than one percent) across several sites in Biscayne National Park between the late 1970s and 1998–2000 (Dupont et al., 2008). Taken together, these data imply extreme declines in the Florida Keys (80–95 percent) between the late 1970s and 2003, and it is clear that further dramatic losses occurred in this region during the cold weather event in January 2010. Decline of star corals in the Florida Keys between the late 1970s and 2003 was approximately 80 to 95 percent, with further losses during the 2010 cold weather event.

Designated critical habitat for threatened Orbicella species with populations in Florida waters

At this time, critical habitat has not been designated for members of the *Orbicella* species complex.

Recovery goals for threatened Orbicella species with populations in Florida waters

No final recovery plan currently exists for the *Orbicella* species; however a recovery outline or the five Caribbean coral species listed as threatened in 2014 was published in 2015 to serve as interim guidance to direct preliminary short and longer term recovery efforts, including recovery planning, until a final recovery plan is developed and approved. The document lists the following recovery goals:

Short Term:

- Improve understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling

- Improve understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility

- Reduce locally manageable stress and mortality sources (e.g., acute sedimentation, nutrients, contaminants, and over-fishing)

- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit *Orbicella* species spp

Longer Term:

- Develop and implement U.S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats

- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment

7.3.1 Boulder Star Coral

Boulder star coral is distinguished by large, unevenly arrayed polyps that give the colony its characteristic irregular surface (Figure 18). Colony

Figure 18. Boulder star coral colony.

form is variable, and the skeleton is dense with poorly developed annual bands. Colony diameter can reach up to five meters with a height of up to two meters. Boulder star coral occurs in the western Atlantic and throughout the Caribbean, including the Bahamas, Flower Garden Banks,

Figure 19. Boulder star coral range.

and the entire Caribbean coastline (Figure 19, IUCN 2017, Weil and Knowton 1994).

Life history. Boulder star coral is reported to be the slowest growing of the three species in the *Orbicella* complex (Brainard et al. 2011). They grow slowest in deep or murky waters. Of 351

colonies of boulder star coral tagged in Bocas del Toro, Panama, larger colonies were noted to spawn more frequently than smaller colonies between 2002 and 2009 (Levitan et al. 2011).

Population Dynamics. 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 2005 boulder star coral ranked 26th most abundant out of 47 coral species, in 2009, the species ranked 32nd out of 43, and in 2012, 33rd out of 40 species. Extrapolated population estimates and standard errors around those estimates from stratified random surveys were 8.0 ± 3.5 million colonies in 2005, 0.3 ± 0.2 million colonies in 2009, and 0.4 ± 0.4 million 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 ten to 20 centimeters and approximately 20 to 30 centimeters. Partial colony mortality ranged from zero 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 and standard errors were 79 ± 19 million colonies in 2006 and 18.2 ± 4.1 million 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 to 30 centimeters 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 ten to 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 to 90 centimeters) appeared to be less than in 2006. The number of colonies in the largest size class of greater than 90 centimeters 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 2006b). 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).

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 or at least partial mortality of colonies; colony abundance appears to be stable in other areas (Bruckner 2012a).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.

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 meters, colonies nearby spawned more synchronously when within five meters of each other than farther spaced colonies, regardless of genotype. At distances greater than five meters apart, spawning was random between colonies (Levitan et al. 2011).

Boulder star coral is found 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 five to 50 meters, with the complex reported to 90 meters. *Orbicella* species are a common, often dominant, component of Caribbean mesophotic reefs (e. g., > 30 meters), suggesting the potential for deep refugia for boulder star coral.

Status. In addition to the status information described for the Orbicella complex in section 7.3, the NMFS 2014 listing provides the following information specific to the status of boulder star coral. Percent cover has declined between 37 to 90 percent over the past several decades at reefs at Jamaica, Belize, Florida Keys, Bahamas, Bonaire, Cayman Islands, Curaçao, Puerto Rico, U.S. Virgin Islands, and St. Kitts and Nevis. Abundance has decreased in some areas between 19 to 57 percent, and shifts to smaller size classes have occurred in locations such as Jamaica, Colombia, Bahamas, Bonaire, Cayman Islands, Puerto Rico, U.S. Virgin Islands, and St. Kitts and Nevis At some reefs, a large proportion of the population is comprised of non-fertile or less reproductive size classes. Several population projections indicate population decline in the future is likely at specific sites, and local extirpation is possible within 25 to 50 years at conditions of high mortality, low recruitment, and slow growth rates.

7.3.2   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 (Figure 20).

Lobed star coral occurs throughout the western Atlantic Ocean and greater Caribbean Sea including the Flower Garden Banks, but may be absent from Bermuda (Figure 21, IUCN 2017). Lobed

Figure 20. Lobed star coral colony.
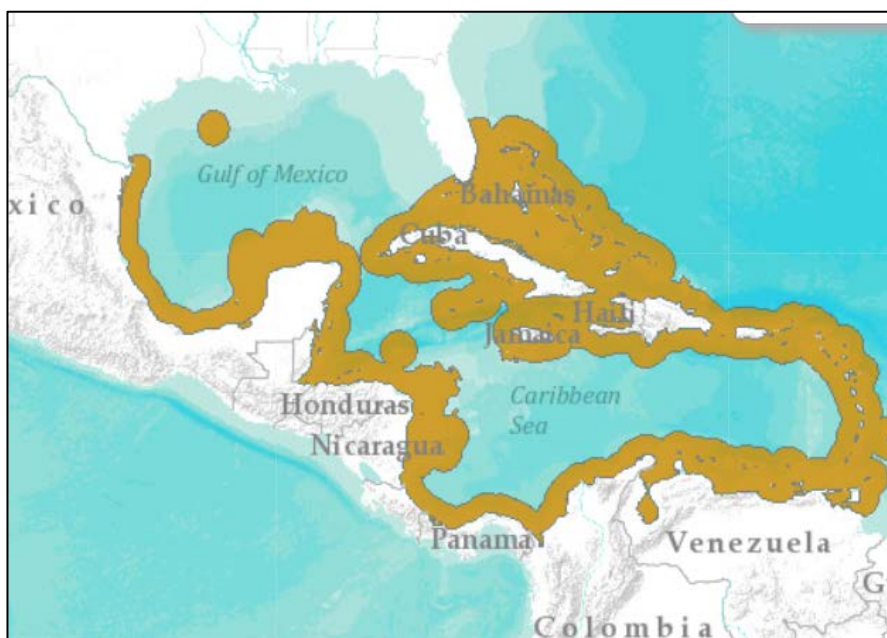
Figure 21. Lobed star coral range.

star coral is reported from most reef environments in depths of approximately 1.5 to 66 feet (0.5 to 20 meters, Weil and Knowton, 1994).

Life history. 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.

Population Dynamics. Lobed star corals are the third most abundant coral species by percent cover in permanent monitoring stations in the U.S. Virgin Islands. 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).

Lobed star coral was historically considered to be one of the most abundant species in the Caribbean (Weil and Knowton 1994). Percent cover has declined 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, U.S. Virgin Islands, 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, U.S. Virgin Islands, and St. Kitts and Nevis. At some reefs, a large proportion of the population is comprised of non-fertile or less-reproductive size classes. Several population

projections indicate population decline in the future is likely at specific sites, and local extirpation is possible within 25-50 years at conditions of high mortality, low recruitment, and slow growth rates. 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.

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 and standard errors from stratified random samples were 5.6 million ± 2.7 million in 2005, 11.5 million ± 4.5 million in 2009, and 24.3 million ± 12.4 million 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 and standard error was 0.5 million ± 0.3 million 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 ten square meters. 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 ten square meters 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 (0.34 colonies per approximately ten square meters) compared to Palm Beach and Broward Counties (ten square meters; 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 ten square meters (Wagner et al. 2010). Off southwest Cuba on remote reefs, lobed star coral densities and standard deviations were 0.31 ± 0.46 per approximately ten meters transect on 38 reef-crest sites and 1.58 ± 1.29 colonies per approximately ten meters 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 100 centimeters (Alcolado et al. 2010).

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

Population trends are available from a number of studies. In a study of sites inside and outside a marine protected area in Belize, lobed star coral cover declined significantly over a 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).

Cover of lobed star coral at Yawzi Point, St. John, U.S. Virgin Islands 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 (Edmunds and Elahi 2007). This decline continued between 1994 and 1999 during a time of two 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 one square meter between 1988 and 2003, due mostly to the death and fission of medium-to-large colonies (≥ 151 square centimeters). Meanwhile, the population size class structure shifted between 1988 and 2003 to a higher proportion of smaller colonies in 2003 (60 percent less than 50 square centimeters in 1988 versus 70 percent in 2003) and lower proportion of large colonies (six percent greater than 250 square centimeters in 1988 versus three percent in 2003. 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).

Asexual fission and partial mortality can lead to multiple clones of the same colony. The percentage of unique individuals varies by location, ranging 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.

Status. In addition to the status information described for the Orbicella complex in section 7.3, the NMFS 2014 listing provides the following information specific to the status of lobed star coral. 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. Percent cover has declined between 37 to 90 percent over the past several decades at reefs at Jamaica, Belize, Florida Keys, Bahamas, Bonaire, Cayman Islands, Curaçao, Puerto Rico, U.S. Virgin Islands, and St. Kitts and Nevis. Abundance has decreased in some areas between 19 to 57 percent, and shifts to smaller size classes have occurred in locations such as Jamaica, Colombia, Bahamas, Bonaire, Cayman Islands, Puerto

Rico, U.S. Virgin Islands, and St. Kitts and Nevis on some reefs, a large proportion of the population is comprised of non-fertile or less reproductive size classes.

7.3.3 Mountainous Star Coral

Mountainous star coral belongs to the taxonomic family of Merulinidae, a group of stony corals whose hard exoskeletons are highly fused and lack paliform lobes. (Weil and Knowton 1994). Mountainous star coral grows in heads or sheets, the surface of which may be smooth or have keels or bumps (Figure 22). The skeleton is much less dense than in the other two star coral species. Colony diameters can reach up to ten meters with heights of four to five meters.

Figure 22. Mountainous star coral colony.

Mountainous star coral is found in the western Atlantic and throughout the Caribbean (Figure 23, IUCN 2017). Mountainous star coral has been reported in most reef habitats and is often the most abundant coral at ten to twenty meters in fore-reef environments. The depth range of mountainous star coral has been reported as approximately 0.5-40 meters, though the complex has been reported to depths of 90 meters. Star coral species are a common, often dominant component of Caribbean
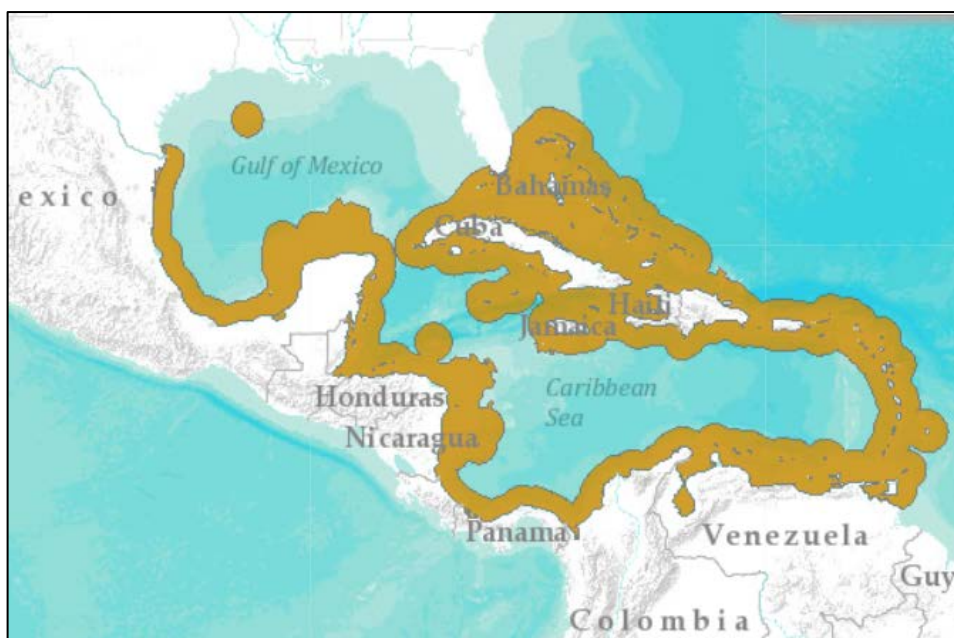

Figure 23. Mountainous star coral range.

mesophotic reefs (e.g., > 30 meters), suggesting the potential for deep refugia for mountainous star coral.

Life history. 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 0.3 and 1.6 centimeters per year (Tomascik and Logan 1990, Cruz-Piñón et al. 2003, Villinski 2003, Waddell 2005). Graham and van Woesik (2013) report that after partial colony mortality, 44 percent of the remaining small colony mountainous star coral produced smaller eggs than those of mature corals. The number of eggs produced per unit area by 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 three to fifteen percent remaining alive after 30 days. Post-settlement survivorship was much lower than the 29 percent observed for elkhorn coral after seven 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 complex in the past (Bruckner 2012a). Large colonies in the complex maintain the population until favorable recruitment conditions occur. However, poor conditions can influence the frequency of recruitment events. While the life history strategy of the star coral 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.

Population Dynamics. Population trend data exists for several locations. At nine sites off Mona and Desecheo Islands, Puerto Rico, no species extirpations were noted at any site over ten 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 six 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 seven percent in 2008.

Extrapolated population estimates and standard errors from stratified random samples in the Florida Keys were 39.7 ± eight million colonies in 2005, 21.9 ± seven million colonies in 2009, and 47.3 ± 14.5 million colonies in 2012. The greatest proportion of colonies tended to fall in the ten to 20 centimeter and 20 to 30 centimeter size classes in all survey years, but there was a fairly large proportion of colonies in the greater than 90 centimeter-size class. Partial mortality of the colonies was between ten 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 and standard errors, were 36.1 ± 4.8 million colonies in 2006 and 30 ± 3.3 million colonies in 2008. The size classes with the largest proportion of colonies were ten to 20 centimeter and 20 to 30 centimeter, but there a fairly large proportion of colonies were greater-than-90 centimeters. Partial mortality of the colonies ranged between approximately two 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 the U.S. Virgin Islands, the reproductive performance of O. faveolata was assessed over a five-week period at three depth ranges of five to ten meters, 15–22 meters, and 35–40 meters. The results showed that corals at the upper edge of the mesophotic zone 35–40 meters were more fecund and produced more eggs than those at shallower depths (Holstein et al. 2015).

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 five genetic markers, but the populations were not stratified by geography, indicating they were shared among the three regions (Baums et al. 2010). Of ten 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.

In a survey of 31 sites in Dominica between 1999 and 2002, mountainous star coral was present at 80 percent of the sites at one to ten 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 in three meters to five meters depth spanning 149 miles (240 kilometers) in the Florida Keys, mountainous star coral was the third most abundant coral species comprising seven 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 ten percent of the 76.5 percent coral cover on reefs 32 to 40 meters, and partial mortality due to bleaching, disease, and predation were rare at monitoring stations (Precht et al. 2005).

Colony density typically ranges from approximately 0.1 to 1.8 colonies per ten square meters 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 ten square meters (Wagner et al. 2010). On remote reefs off southwest Cuba, densities of mountainous star coral, and associated standard errors were 0.12 ± 0.20 colonies per ten meter transect on 38 reef-crest sites and 1.26 ± 1.06 colonies per ten meter 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 ten square meters and was highest on mid-channel reefs followed by

70

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 ten square meters compared to 0.11 colonies per ten square meters in Palm Beach and Broward Counties (Burman et al. 2012).

Status. In addition to the status information described for the Orbicella complex in section 7.3, the NMFS 2014 listing provides the following information specific to the status of mountainous star coral. Population decline has occurred over the past few decades with a 65 percent loss in mountainous star coral cover across five countries. Losses from Mona and Descheo Islands, Puerto Rico include a 36 to 48 percent reduction in abundance and a decrease of 42 to 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.

7.4 Pillar Coral

Pillar coral forms cylindrical columns on top of encrusting bases. Colonies are generally grey-brown in color and may reach approximately ten feet (three meters) in height. Polyps' tentacles remain extended during the day, giving columns a furry appearance (Figure 24, IUCN 2017). 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 25). Brainard et al. (2011) identified a single known colony in Bermuda that was in poor condition. There is fossil evidence of the presence of the species off Panama less than 1,000 years ago, but it has been reported as absent today (Florida Fish and Wildlife

Figure 24. Pillar coral colony.

Conservation Commission 2013). Pillar coral inhabits most reef environments in water depths ranging from approximately three to 75 feet (one to 25 meters), but it is most common in water between approximately 15 to 45 feet (five to 15 meters) deep (Goreau and Wells 1967, Cairns 1982, Acosta and Acevedo 2006).

Figure 25. Pillar coral range.

7.4.1 Life History

Average growth rates of approximately 1.8-2.0 centimeters per year in linear extension have been reported within the Florida Keys (Hudson and Goodwin 1997), 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[9] 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 (Waddell and Clarke 2008, Neely et al. 2013) 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.

7.4.2 Population Dynamics

Information on pillar coral status and populations dynamics is spotty throughout its range. Pillar coral is currently uncommon to rare throughout Florida and the Caribbean. Low abundance and

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

infrequent encounter rate in monitoring programs result in small samples sizes. The low cover of this species renders monitoring data difficult to extrapolate to clear 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, coupled with no observed sexual recruitment, indicate that natural recovery potential from mortality is low.

Comprehensive and systematic census and monitoring have not been conducted outside of Florida. Thus, the status and populations dynamics must be inferred from the few locations where data exist. 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). In the U.S. Virgin Islands, seven percent of 26 monitored colonies 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).

Pillar coral is uncommon but conspicuous with scattered, isolated colonies. It is rarely found in aggregations. In coral surveys, it generally has a rare encounter rate, low percent cover, and low density. 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 one and 0.1 colonies per ten square meters (approximately 100 square feet), respectively, between 2005 and 2010 in surveys from West Palm Beach to the Dry Tortugas (Burman et al. 2012).

Density of pillar corals in other areas of the Caribbean is also low and on average less than 0.1 colonies per ten square meters. The average number of pillar coral colonies in remote reefs off southwest Cuba was 0.013 ± 0.045 colonies per ten meters (approximately 32 feet) 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 square kilometers (0.6 square miles) for an overall density of approximately 0.000017 colonies per ten square meters (approximately 100 square feet, Acosta and Acevedo 2006).

Benthic cover is generally less than one 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). At permanent monitoring stations in the U.S. Virgin Islands, pillar coral was observed in low abundance at ten of 33 sites and ranged in cover from less than 0.05 to 0.22 percent where present (Smith 2013). 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 one of

seven fringing reefs surveyed off Barbados, and cover was 2.7 ± 1.4 percent (Tomascik and Sander 1987).

The National Coral Reef Monitoring Program surveyed Caribbean reefs (see https://www.coris.noaa.gov/monitoring/). In Puerto Rico, pillar coral density surveyed between 2008 and 2016 ranged from zero to 0.3 colonies per square meter with an average density of 0.03 colonies per ten square meters (approximately 100 square feet); it occurred at four percent of the sites. Pillar coral cover surveyed between 2001 and 2016 ranged between zero and four percent with an average of 0.02 percent. In the U.S. Virgin Islands, surveys between 2002 and 2015 found that pillar coral density ranged between zero and 0.3 colonies per square meter with an average density of 0.01 colonies per ten square meters (approximately 100 square feet); it occurred in three percent of the sites surveyed. Cover, surveyed between 1992 and 2015, ranged from zero to six percent with an average cover of 0.03 percent.

7.4.3 Status

Pillar coral appears to have some susceptibility to ocean warming, though there are conflicting characterizations of the susceptibility of pillar coral to bleaching. Some locations experienced high bleaching of up to 100 percent of pillar coral colonies during the 2005 Caribbean bleaching event (Oxenford et al. 2008) while others had a smaller proportion of colonies bleach (e.g., 36 percent; Bruckner and Hill 2009). Reports of low mortality after less severe bleaching indicate potential resilience, though mortality information is absent from locations that reported high bleaching frequency. Although bleaching of most coral species is spatially and temporally variable, understanding the susceptibility of pillar coral is further confounded by the species' rarity and, hence, low sample size in any given survey.

Pillar coral is sensitive to cold temperatures. In laboratory studies of cold shock, pillar coral had had the most severe bleaching of the three species tested at 12 degrees Celsius (Muscatine et al. 1991). During the 2010 cold water event in the Florida Keys, pillar coral experienced 100 percent mortality on surveyed inshore reefs, while other species experienced lower mortality (Kemp et al. 2011).

Pillar coral is susceptible to black band disease and white plague, though impacts from white plague are likely more extensive because of rapid progression rates (Brainard et al. 2011). Disease appears to be present in about three to four percent of pillar coral populations in locations surveyed (Acosta and Acevedo 2006, Ward et al. 2006). Because no studies have tracked disease progression in pillar coral, the effects of disease are uncertain at both the colony and population level. However, the reported low partial mortality and large colony size suggest that individual colonies are less likely to suffer complete colony mortality from a given disease exposure.

Pillar coral may be more sensitive to turbidity due to the high reliance of the species on nutrition from photosynthesis (Brainard et al. 2011) and as evidenced by the geologic record (Hunter and Jones 1996). However, pillar coral appears to be moderately capable of removing sediment from

its tissue (Brainard et al. 2011). Pillar coral survival may also be susceptible to nutrient enrichment as evidenced by its absence from eutrophic sites in Barbados (Brainard et al. 2011). There is uncertainty about whether its absence is a result of eutrophic conditions or a result of its naturally uncommon or rare occurrence. Pillar coral likely has some susceptibility to sedimentation and nutrient enrichment. However, the available information does not support a more precise description of its susceptibility to this threat.

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 meters, 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. In addition, low sexual recruitment is likely to inhibit recovery potential from mortality events, further exacerbating its vulnerability to extinction. Pillar coral is likely to decrease in abundance in the future with increasing threats.

7.4.4 Status within Florida Waters

Pillar coral's average percent cover was 0.002 on patch reefs and 0.303 in shallow offshore reefs in annual surveys of 37 sites in the Florida Keys between 1996 and 2003 (Somerfield et al. 2008). In stratified random surveys from Palm Beach County to the Dry Tortugas, Florida, between 2005 and 2010, pillar coral was seen only on the ridge complex and mid-channel reefs at densities of approximately one and 0.1 colonies per approximately ten square meters, respectively (Burman et al. 2012). Average number of pillar coral colonies in remote reefs off southwest Cuba was 0.013 ± 0.045 colonies per approximately ten meter transect, and the species ranked sixth rarest out of 38 coral species (Alcolado et al. 2010). In surveys of the upper Florida Keys in 2011, pillar coral was the second rarest out of 37 coral species and encountered at one percent of sites (Miller et al. 2011).

7.4.5 Designated Critical Habitat

Critical habitat has not been designated for pillar coral.

7.4.6 Recovery Goals

No final recovery plans currently exists for pillar coral; however a recovery outline or the five Caribbean coral species newly listed as threatened in 2014 was published in 2015 to serve as interim guidance to direct preliminary short and longer term recovery efforts, including recovery

planning, until a final recovery plan is developed and approved. The document lists the following recovery goals:

Short Term:

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

Longer Term:

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

7.5 Rough Cactus Coral

Rough cactus coral forms a thin, encrusting plate that is weakly attached to the substrate. Rough cactus coral is a taxonomically distinct (i.e., separate species), though difficult to distinguish in the field from other *Mycetophyllia* species (Figure 26, IUCN 2017). The maximum colony size of the species is 50 centimeters in diameter. While rough cactus coral occurs in the western Atlantic Ocean and throughout the wider Caribbean Sea (Figure 27, IUCN 2017), it has not been reported in the Flower Garden Banks

Figure 26. Rough cactus coral colony.

(Gulf of Mexico) or in Bermuda. It inhabits reef environments in water depths of five to ninety meters, including shallow and mesophotic habitats (e.g., > 30 meters).

Figure 27. Rough cactus coral range.

7.5.1 Life History

Rough cactus coral is a hermaphroditic brooding[10] species. Colony size at first reproduction is greater than 100 square centimeters. 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 U.S. Virgin Islands, although adults were observed on the adjacent reef (Rogers and Garrison 2001).

7.5.2 Population Dynamics

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 square meters in Florida and at 0.8 colonies per 100 meters transect in Puerto Rico sites sampled by the Atlantic and Gulf Rapid Reef Assessment (Veron 2002, Wagner el al., 2010, 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 (U.S. Virgin Islands/NOAA Center for Coastal Monitoring and Assessment randomized monitoring stations) show Mycetophyllia ferox cover to be consistently less occasional observations up to two percent and no apparent temporal trend (Brainard et al. 2011).

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

Dustan (1977) proposes that rough cactus coral was much more abundant in the upper Florida Keys in the early to mid- 1970s (the methods are not well described for that study) than current observations, but that it was highly affected by disease.

According to the International Union for the Conservation of Nature Species Account and the species database of the Convention on International Trade in Endangered Species of Wild Fauna and Flora, 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 federal waters where rough cactus coral has been observed and recorded (cited in Brainard et al. 2011): Dry Tortugas National Park; Virgin Islands National Park/Virgin Islands Coral Reef National Monument; Florida Keys National Marine Sanctuary; Navassa Island National Wildlife Refuge; Biscayne National Park; Buck Island Reef National Monument, St. Croix.

On reefs where rough cactus coral is found, it generally occurs at abundances of less than one colony per approximately ten square meters 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.

7.5.3   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 five to 90 meters 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.

7.5.4   Status within Florida Waters

Disease has resulted in population declines over the past several decades in the Florida Keys. Rough cactus coral is uncommon (Veron 2000), constituting < 0.1 percent of coral colonies and occurs at densities < 0.8 colonies per ten square meters in Florida (Wagner et al. 2010). Dustan (1977) suggests that rough cactus coral was much more abundant in the upper Florida Keys in the early to mid to 1970s than currently. No bleached rough cactus coral colonies were observed during the 2005 mass coral bleaching event in Florida (Wagner et al. 2010). Rough cactus coral are susceptible to acute and subacute white plague. Dustan (1977) reported dramatic impacts from this disease to the population in the upper Florida Keys in the mid-1970s. He also reported that the rate of disease progression was positively correlated with water temperature and measured rates of disease progression up to three millimeters daily.

7.5.5   Designated Critical Habitat

Critical habitat has not been designated for rough cactus coral.

7.5.6   Recovery Goals

No final recovery plan currently exists for rough cactus coral; however a recovery outline or the five Caribbean coral species newly listed as threatened in 2014 was published in 2015 to serve as interim guidance to direct preliminary short and longer term recovery efforts, including recovery planning, until a final recovery plan is developed and approved. The document lists the following recovery goals:

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, and 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

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 CFR 402.02). The key purpose of the *Environmental Baseline* is to describe the condition of the ESA-listed species and designated critical habitat within the action area and the consequences of that condition without the action.

Baseline conditions nationwide are reflected within Florida. Flather et al. (1998) identified habitat loss and alien species as the two most widespread threats to endangered species, affecting more than 95 percent and 35 percent of listed species, respectively. For example, the net effect of human-altered hydrology creates conditions that increase stormwater runoff; transporting land based pollutants into surface waters, reducing the filtration of stormwater runoff through wetlands prior to reaching surface waters.

Increases in polluted runoff has been linked to a loss of aquatic species diversity and abundance, including many important commercial and recreational fish species. Non-point source pollution has also contributed to coral reef degradation, fish kills, seagrass bed declines, and algal blooms, including blooms of toxic algae. In addition, many shellfish bed and swimming beach closures can be attributed to polluted runoff. As discussed in EPA's latest National Coastal Condition Report, non-point sources have been identified as one of the stressors contributing to coastal water pollution (USEPA 2012).

With its mean elevation above sea level of 30 meters and porous limestone aquifers, Florida is uniquely vulnerable to sea level rise associated with climate change. Expansion of inland tidal marshes replacing lowland coastal forests over the last 120 years was demonstrated along the Big Bend of Florida (Raabe and Stumpf 2016). Temperature records between 1878 and 2012 for Florida Keys coral reef habitats indicate an increase of 0.8ºC in the last century (Kuffner et al. 2015).

The Intergovernmental Panel on Climate Change estimated that average global land and sea surface temperature has increased by 0.85°C (± 0.2) since the late 1800s, with most of the change occurring since the mid-1900s (IPCC 2013). This temperature increase is greater than what would be expected given the range of natural climatic variability recorded over the past 1,000 years (Crowley and Berner 2001). All species discussed in this opinion are presently, or are likely to be, affected by the direct and indirect effects of global climatic change. Global climate change stressors, including consequent changes in land use, are major drivers of ecosystem alterations (USEPA 2008). Climate change is projected to have substantial direct effects on individuals, populations, species, and the community structure and function of marine, coastal, and terrestrial ecosystems in the foreseeable future (McCarty 2001, IPCC 2002, Parry et al. 2007, IPCC 2013). A northward shift in loggerhead nest placement was reported for Melborne Beach,

Florida, the largest U.S. rookery for this species (Reece et al. 2013). Climate change is most likely to have its most pronounced effects on species whose populations are already in tenuous positions (Williams et al. 2008b). Increasing atmospheric temperatures have already contributed to changes in the quality of freshwater, coastal, and marine ecosystems and have contributed to the decline of populations of endangered and threatened species (Mantua et al. 1997, Karl et al. 2009, Littell et al. 2009).

Increasing surface water temperatures can cause the latitudinal distribution of freshwater and marine fish species to change: as water temperatures rise, cold and warm water species will spread northward (Hiddink and Ter Hofstede 2008, Britton et al. 2010). Climate-mediated changes in the global distribution and abundance of marine species are expected to reduce the productivity of the oceans by affecting keystone prey species in marine ecosystems such as phytoplankton, krill, and cephalopods. For example, climate change may reduce recruitment in krill by degrading the quality of areas used for reproduction (Walther et al. 2002). Aquatic nuisance species invasions are also likely to change over time, as oceans warm and ecosystems become less resilient to disturbances (USEPA 2008). Invasive species that are better adapted to warmer water temperatures could outcompete native species that are physiologically geared towards lower water temperatures; such a situation currently occurs along central and northern California (Lockwood and Somero 2011). Warmer water stimulates biological processes, which can lead to environmental hypoxia. Oxygen depletion in aquatic ecosystems can result in anaerobic metabolism increasing, thus leading to an increase in metals and other pollutants being released into the water column (Staudinger et al. 2012).

Ocean acidification, as a result of increased atmospheric carbon dioxide, can interfere with numerous biological processes in corals including fertilization, larval development, settlement success, and secretion of skeletons (Albright et al. 2010). In addition to global warming, acidification poses another significant threat to oceans because many major biological functions respond negatively to increased acidity of seawater. Photosynthesis, respiration rates, growth rates, calcification rates, reproduction, and recruitment may be negatively impacted with increased ocean acidity (London 2005). Kroeker et al. (2010) review of 139 studies quantifying ocean acidification effects determined that the effects were variable depending on species, but effects were generally negative, with calcification being one of the most sensitive processes.

Aquatic species, especially marine species, already experience stress related to the impacts of rising temperature. Corals, in particular, demonstrate extreme sensitivity to even small temperature increases. When sea temperatures increase beyond a coral's limit, the coral "bleaches" by expelling the symbiotic organisms that not only give coral its color, but also provide food for the coral through their photosynthetic capabilities. According to (Hoegh-Guldberg 2010), bleaching events have steadily increased in frequency since the 1980s.

The baseline condition of Florida's aquatic resources is described in detail in the 2014 Integrated Water Quality Assessment for Florida (FDEP 2014). The following paragraphs are derived from that document. There are 88,250 kilometers of streams and rivers, 127,239 kilometers of ditches

and canals, and 45,838 square kilometers of freshwater and tidal wetlands in Florida (Figure 2). Florida's coastline ranks second in length only to Alaska. Florida's low relief, coupled with its geologic history, has created unique hydrogeologic features making groundwater quality particularly critical to surface water quality.

8.1 Human Alterations of Surface Waters

The Port of Miami was recently dredged to accommodate the newest generation of freighters. Among sediment impacts assessed, the most severe is for a sedimentation assessment site located 200 m north of the dredged channel. This assessment characterized 81 percent of the points surveyed as 'sediment over hard bottom' compared to one percent at the corresponding reference site. Overall, increased sediment accumulation and associated biological responses (e.g., higher prevalence of partial mortality of corals) extended up to 700 m from the channel (Miller et al. 2016, NMFS 2016).

8.2 Pollutants

In aquatic environments, sediments provide essential habitat but, at the same time, may be a source of contamination and recycled nutrients. Sediment contaminants, such as trace metals, organic pesticides, excess nutrients, and other contaminants such as 4-nonylphenol, accumulate over time from upland discharges, the decomposition of organic material, and atmospheric deposition. Periodic water quality monitoring cannot fully evaluate aquatic ecosystems, as it is not usually designed to assess the cumulative impact of sediment contaminants. Knowledge of a site's sediment quality is important for environmental managers in evaluating future restoration and dredging projects. Unlike many water column constituents, Florida has no criteria for sediment and no statutory authority to establish criteria.

8.3 Aquatic Invasive Species

Aquatic invasive species are aquatic organisms introduced into new habitats and subsequently produce harmful impacts on the natural resources in and human uses of these ecosystems (http://www.anstaskforce.gov). Not all non-native (also called alien or nonindigenous) species are considered invasive. Overall, there have been 374 documented invasive species in U.S. waters, 150 of which have arrived since 1970 (PEW 2003). The Nonindigenous Aquatic Species database lists 53 non-native species reported in Florida's brackish and marine waters. These data are preliminary or provisional and are subject to revision. This opinion refers to them because this data base contains the best science available at the time of writing this opinion. Among the species identified, 16 species have established populations in one or more of the estuaries and coastal areas of Florida. The presence of established populations for 32 of the species is unknown and five species failed to establish populations. Many of the fish species are aquarium releases and some of the established populations were actually stocked as forage fish (e.g., shad and blueback herring). The lionfish, originally from the indo-pacific is a particularly harmful invasive fish species in Florida's waters. Lionfish are a major predator on commercial and sport fish species and the herbivorous fish species that are important to controlling algal growth on

coral reefs (Lesser and Slattery 2011, Albins and Hixon 2013, Cote et al. 2013). Their presence in reef systems has been associated with severe declines in fish abundance (Albins and Hixon 2008). Initial observations in the mid-1980s are attributed to aquarium releases. They are established in coastal waters from North Carolina to South America. Lionfish have invaded the Loxahatchee estuary (i.e., Jupiter Inlet on the Atlantic coast of Florida). Over 200 young-of-year individuals ranging from 23 to 185 millimeters were collected over a one-year survey period. They were primarily associated with man-made structures and associated debris along the shoreline as far as 5.5 kilometers inland (Anderson et al. 2014).

Introduced aquatic invasive species are one of the main sources of risk to ESA-listed species, second only to habitat loss (Wilcove et al. 1998). They have been implicated in the endangerment of 48 percent of the species listed under ESA (Czech and Krausman 1997). The USFWS considers invasive species to be a significant contributing factor in determining the "threatened" or "endangered" status of many native species (OTA 1993, Ruiz et al. 1997). Invasive species affect aquatic environments in many different ways. They can reduce native species abundance and distribution, and reduce local biodiversity by out-competing native species for food and habitat. They may displace food items preferred by native predators, disrupting the natural food web. They may alter ecosystem functions. Exotic plants can clog channels and interfere with recreational fishing and swimming. Introduced non-native algal species combined with nutrient overloading may increase the intensity and frequency of algal blooms. An overabundance of algae can lead to depleted DO. Oxygen depletion can result in "dead zones," murky water, seagrass and coral habitat degradation, and large-scale fish kills (Deegan and Buchsbaum 2005).

8.4 Harmful Algal Blooms

Florida monitors for Harmful Algal Blooms (HABs) in fresh, estuarine, and marine waters. Blooms can occur any time of year in Florida, due to its subtropical climate. A suite of unique taxa that can bloom under particular physical, chemical, and biological conditions cause the HABs. The drivers of some HABs are well understood, while the drivers of other HABs, such as the red tide organism *Karenia brevis*, are still unclear. While HABs can occur naturally, they are frequently associated with elevated nutrient concentrations. Toxins may be produced by HABs toxins that contaminate shellfish or finfish, making them unsuitable for human consumption. They can also affect plant and animal communities. The Gulf of Mexico Alliance, a partnership between Alabama, Florida, Louisiana, Mississippi, and Texas, is working to increase regional collaboration to enhance the Gulf's ecological and economic health. Reducing the effects of HABs is one of its water quality priorities.

Freshwater cyanobacteria (or blue-green algae) blooms have received increased attention in recent years because of their potential to produce toxins that can harm humans, livestock, domestic animals, fish, and wildlife. While blooms of cyanobacteria can occur naturally, they are frequently associated with elevated nutrient concentrations, slow-moving water, and warm temperatures. Cyanotoxins are bioactive compounds naturally produced by some species of

cyanobacteria that can damage the liver (hepatotoxins), nervous system (neurotoxins), and skin (dermatotoxins) of humans and other animals. Potentially toxigenic cyanobacteria have been found statewide in Florida's rivers, streams, lakes, and estuaries. There are also concerns that freshwater cyanotoxins can be transported into coastal systems. The results of the Cyanobacteria Survey Project (1999 to 2001), managed by the Harmful Algal Bloom Task Force at the FWCC Fish and Wildlife Research Institute, indicated that the taxa *Microcystis aeruginosa, Anabaena* spp., and *Cylindrospermopsis raciborskii* were dominant, while species with the genera *Aphanizomenon, Planktothrix, Oscillatoria*, and *Lyngbya* were also observed statewide but not as frequently. Cyanotoxins (microcystins, saxitoxin [STX], cylindrospermopsins, and anatoxin) were also found statewide (Williams 2007). Other cyanobacteria of concern in Florida are reported in Abbott et al. (2009).

Other bloom-forming marine species can be divided into two categories: toxin-producing species and taxa that form blooms associated with other problems, such as low oxygen concentrations, physical damage to organisms, and general loss of habitat. Potential toxin-producing planktonic marine HAB species include the diatom group *Pseudo-nitzschia* spp.; the dinoflagellates *Alexandrium monilatum, Takayama pulchella, K. mikimotoi, K. selliformis, Karlodinium veneficum, Prorocentrum minimum, P. rhathymum*, and *Cochlodinium polykrikoides*; and the prymnesiophytes *Prymnesium* spp. and *Chrysochromulina* spp., and the raphidophyte *Chattonella* sp. (Abbott et al. 2009). Many of these species are associated with fish or shellfish kills in various ecosystems around the world (Landsberg 2002). Additionally, benthic cyanobacteria and macroalgae blooms have been observed on Florida's coral reefs and have been associated with mortality and disease events involving various organisms (Lapointe et al. 2004, Paul et al. 2005, Richardson et al. 2007).

Although many HAB species have been observed at bloom levels in Florida (Phlips et al. 2011), uncertainty remains over the relative toxicity of the specific strains. In addition to ichthyotoxic HAB species that directly cause fish kills, the list of HAB species linked to hypoxia or other density-related issues (e.g., allelopathy, physical damage to gills of fish) is extensive and includes almost any species that reaches exceptionally high biomass. Examples include the widespread bloom-forming planktonic dinoflagellate *Akashiwo sanguinea*, in the Indian River Lagoon and the St. Lucie Estuary, and the cyanobacterium *Synechococcus* in Florida Bay (Phlips et al. 2010, Phlips et al. 2011). Many fish kills, particularly those occurring in the early morning hours, are due to low dissolved oxygen levels in the water associated with the algal blooms and are not necessarily the result of toxins.

Another important issue associated with HABs is the loss or alteration of overall habitat quality. Prolonged and intense coastal eutrophication can result in domination by a select few species, resulting in a loss of diversity and alteration of food web structure and function. For example, during major *Pyrodinium* blooms, 80 percent to 90 percent of total phytoplankton biomass is attributable solely to this species (Phlips et al. 2006). Similar domination by a single species

occurs in benthic ecosystems, where massive blooms of green and red macroalgae have periodically over-run some shallow habitats of the Florida coast (Lapointe and Bedford 2007).

8.5 Aquatic Impairments

Florida's most recent EPA-approved 303(d) list of impaired waters is for the year 2010.[11] The 2010 data indicate 8,418 miles out of 51,858 assessed rivers and streams are threatened or impaired with the top five impairment causes identified as dissolved oxygen, mercury in fish tissue, fecal coliform, chlorophyll-a, and impaired biological communities where the proximate stressor is unknown. A total of 5,176 square miles of bays and estuaries were also listed as impaired and the top five impairments, in order of most to least common, are mercury in fish tissue, fecal coliform, dissolved oxygen, chlorophyll-a, and copper.

8.6 Monitoring and Point Sources for Florida Waters

The assessments discussed in section 8.1.4 above do not include all possible substances that may impair a water body. Most importantly, this monitoring does not look for the presence of 4-nonylphenol. A search of the Florida Department of Environmental Protection STORET public access database did not return any data indicating that 4-nonylphenol was monitored for in sewers, rivers and streams, ocean, and bay and estuaries. A search for 4-nonylphenol in EPA's database pf monitored discharges[12] did not identify facilities reporting 4-nonylphenol levels in Florida. Finally, a search of the National Water Quality Monitoring Council's Water Quality Portal, which integrates monitoring data collected by the United States Geological Survey (USGS), the EPA, and the National Water Quality Monitoring Council,[13] returned just over 495 reports between 2001 and 2013 for 4-nonylphenol and nonylphenol ethoxylates at 287 stations in Florida ground waters and surface waters. Among these, only 63 sampling events report analytes above the detection limits. Detection limits ranged from 0.5 to 20 micrograms 4-nonylphenol per liter, with most 327 out of 495 detection limits at 5 micrograms per liter. This detection limit is close to the proposed criteria for 4-nonylphenol in some freshwaters (1.7 or 6.6 micrograms per liter, depending on water classification) and above the proposed 4-nonylphenol criterion of 1.7 micrograms per liter for marine waters and certain high value freshwaters. There were only 18 monitoring events reported for marine waters, and half of these had a detection limit of 5 micrograms per liter, the other half 0.5 micrograms per liter. Among the 216 sampling events for 4-nonylphenol in streams, 169 had a detection limit of 5 micrograms per liter or below. The

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[11] accessed 12/14/2015 at
iaspub.epa.gov/apex/waters/f?p=ASKWATERS:V_WO_CURRENT_IMPAIRMENTS_LIST:::::P4_OWNER:ATTAINS

[12] National Pollutant Discharge Elimination System monitoring reports https://cfpub.epa.gov/dmr/data_explorer.cfm

[13] Databases include EPA STORET, the USGS National Water Information System and the U.S. Department of Agriculture Sustaining the Earth's Watersheds, Agricultural Research Data System.

remaining sampling events were of groundwater, with detection limits of 6.3 micrograms per liter or lower.

In 2014, EPA added 4-nonylphenol to the list of chemicals included in Toxic Chemical Release Reporting required under section 313 of the Emergency Planning and Community Right-to-Know Act of 1986 and section 6607 of the Pollution Prevention Act of 1990. While relatively few facilities in the nation discharge 4-nonylphenol in reportable quantities under the rule (USEPA 2013), nearly 100 facilities in Florida are among the industrial classes that potentially discharge 4-nonylphenol under National Pollution Discharge Elimination (NPDES) permits. Only a few of these facilities discharge to waters where ESA-listed species under NMFS jurisdiction occur. Ten are located within coastal sub watersheds. Nine potential dischargers are in sub-watersheds where smalltooth sawfish have been observed, four facilities discharge within sub-watersheds where designated critical habitat for Johnson's seagrass occurs, and a single facility occurs in a sub-watershed where designated critical habitat for elkhorn and staghorn coral occurs. None of the Florida's permitted facilities are expected to discharge 4-nonylphenol to the St. Marys River or St. Johns rivers where shortnose and Atlantic sturgeon may occur.

In September of 2018, EPA expects to finalize a significant new use rule under section 5(a)(2) of the Toxic Substances Control Act for nonylphenols and nonylphenol ethoxylates. The significant new use rule would require persons who intend to manufacture (including import) or process these chemical substances to notify EPA at least 90 days before commencing that activity. The required notification would provide EPA with the opportunity to evaluate the intended use and, if necessary, to prohibit or limit that activity before it occurs to prevent unreasonable risk to human health or the environment.

9   EFFECTS OF THE ACTION ON SPECIES ANALYZED

Thus far, the analysis determined that ESA-listed coral species of the Caribbean are likely to respond to exposures to 4-nonylphenol at the proposed criteria. Section 7 regulations define "effects of the action" as the direct and indirect effects of an action on the species or designated 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). As defined previously, indirect effects are those that are caused by the proposed action and are later in time, but are reasonably certain to occur.

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 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.

9.1 Direct Toxicity

Survival: The marine invertebrate mortality NOECs and LOECs fall at the lower range of NOECs and LOECS reported for freshwater invertebrates, so only the marine data are considered here. The minimum observed marine invertebrate mortality observation is a NOEC of 6.7 micrograms 4-nonylphenol per liter opossum shrimp (Ward and Boeri 1991a). This is nearly 4-fold the proposed standard of 1.7 micrograms 4-nonylphenol per liter. The few marine invertebrate species for which data are available are not closely related taxonomically to corals. NMFS searched for data more representative of ESA-listed corals and found an adult and embryo survival data for the freshwater cnidarian, hydra. The studies reported an LC10 value for embryos of 21 micrograms 4-nonylphenol per liter (lethal to 10 percent of exposed) and an LC10 for adult hydra at concentrations 67 micrograms per liter (Pachura et al. 2005, Pachura-Bouchet et al. 2006). The studies did not identify NOECs, so they can not confirm whether or not the 4-nonylphenol criteria would negatively affect survival, only that ten percent mortality would occur in embryos and adults at concentrations 15 and 39-fold the criterion for predominantly marine waters, respectively. It is reasonable to expect that exposures at the criterion would result in insignificant effects. While the reported marine invertebrate mortality NOECs are four fold to an order of magnitude above the proposed 4-nonylphenol standard (6.7 to 130 micrograms 4-nonylphenol per liter), the limited amount of marine invertebrate data and the taxonomic distance of species for which data are available contributes uncertainty into NMFS' determination.

In summary, the available toxicity data indicate that exposures to the 4-nonylphenol at or below the proposed criterion would not result in in significant effects on survival. Therefore, NMFS concludes that EPA's approval of FDEP's proposed 4-nonylphenol criteria for predominantly marine waters is not likely to adversely affect survival of ESA-listed coral species through direct mortality.

Fitness: The few available NOECs and LOECs for growth and development-associated responses in marine invertebrates (0.01 to 30 micrograms per liter) were comparatively lower and less broad than those for freshwater invertebrates (0.05 to 500 micrograms per liter) were, so only the saltwater data were considered. The NOECs ranged from 0.01 micrograms 4-nonylphenol per liter (normal copepod maturation, Marcial et al, 2003) to 30 micrograms 4-nonylphenol per liter (normal mysid sex ratio, Hirano et al 2009) and LOECs ranged from 0.1 micrograms 4-nonylphenol per liter (delayed copepod maturation, Marcial et al, 2003) to 30 micrograms 4-nonylphenol per liter (12 percent of exposed mysids did not mature, Hirano et al 2009). The copepod data are an order of magnitude lower than the proposed standard for predominantly marine waters, yet copepods are taxonomically distant to corals, making them a poor surrogate species. LOECs and NOECs for opossum shrimp (length NOEC at 0.3 micrograms 4-nonylphenol per liter and LOEC at 1.0 micrograms 4-nonylphenol per liter, Hirano et al, 2009) and Australian barnacle maturation rate (LOEC at 0.6 micrograms 4-nonylphenol per liter, Billinghurst et al. 2001) are also below the 4-nonylphenol standard. There are no comparable data to from other studies (i.e., same species group and life stages) to suggest whether these observations are outliers. 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 4-nonylphenol. Our understanding of role of estrogens in coral species is a growing area (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 streroidal 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 for that an estrogen mimic such as 4-nonylphenol would have reproductive effects on coral, mechanism (s) for such effects have not been established.

The lowest reported marine invertebrate fitness NOEC and LOEC are for progeny counts in opossum shrimp. Ward and Boeri (1991a) report a NOEC of 6.7 micrograms 4-nonylphenol per liter and LOEC of 9.1 micrograms 4-nonylphenol per liter. NMFS looked for, but did not find, additional data on the effects on 4-nonylphenol on fecundity of species that would serve as suitable surrogates for coral. The reported marine invertebrate fecundity NOECs discussed above are four fold to more than an order of magnitude above the proposed 4-nonylphenol standard (6.7 to 130 micrograms 4-nonylphenol per liter), the limited amount of marine invertebrate data and the taxonomic distance of species for which data are available contributes uncertainty into NMFS' determination. However, recent work from the National Center for Coastal Science in Charleston, South Carolina evaluated the effects of nonylphenol on fertilization success in

88

mountainous star coral using the Organization for Economic Co-operation and Development testing guidelines. The nominal exposure concentrations of 0, 0.05, 1, 100, and 300 milligrams 4-nonylphenol per liter did not closely bracket FDEP's proposed criteria of 1.7 milligrams per liter. The 1 milligram per liter exposure did not differ significantly from the control while exposures at 100 micrograms per liter 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 microgram per liter, which is below the proposed criterion. The proximity of the NOEC to the proposed criterion of 1.7 micrograms 4-nonylphenol per liter, the underlying variability of the data contributed by the different genetic crosses tested, and the absence of exposures between 1 and 100 micrograms per liter makes 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 these data for mountainous star coral likely underestimate the potential for toxic effects of 4-nonylphenol to the more sensitive species of corals, such as the *Acropora (*see discussion comparing toxicity of two different coral families in Section 3.1.1).

While there are no published 4-nonylphenol toxicity data for species that are taxonomically closely related to coral, the available toxicity data indicate that adverse effects on growth and development occur in other marine invertebrates exposed to 4-nonylphenol at concentrations below the proposed criterion for predominantly marine waters. In addition, unpublished toxicity data indicated 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 Acropora. Taken together, this information leads NMFS to conclude that EPA's approval of the proposed criterion is likely to adversely affect the fitness of ESA-listed coral species.

9.2 Indirect Toxicity

While all coral species can capture and consume prey, most coral polyps also contain zooxanthellae, symbiotic algae that support the coral's energy budget and calcium carbonate secretion. These symbionts are particularly important for *Acropora* sp., with their many small polyps and branching morphology optimizing light capture. This morphology is inefficient for zooplankton capture because zooplankton does not uniformly saturate the water column as light does, so densely arrayed polyps cannot be equally nourished through heterotrophy (Porter 1976). While the dependence of elkhorn and staghorn corals almost entirely upon symbiotic photosynthesizers for nourishment makes them more susceptible to increases in water turbidity and temperature, it actually makes them less susceptible to dietary exposure to toxicants. Indirect effects based on zooxanthellae response also need to be considered. The only data on 4-nonylphenol effects on marine algal species in ECOTOX are for 4-nonylphenol effects on the abundance of diatoms. Diatom cell numbers declined by ten percent at 12.5 micrograms 4-nonylphenol per liter and by 50 percent at 32 micrograms 4-nonylphenol per liter (Ward and

Boeri 1990b). These values are well above the proposed 4-nonylphenol standard for predominantly marine waters of 1.7 micrograms 4-nonylphenol per liter. Data were not found for the effects on invertebrates consuming plankton that had accumulated 4-nonylphenol. Complex mesocosm studies do report changes in freshwater zooplankton and phytoplankton communities (Severin et al. 2003, Hense et al. 2005), but the NOEC for these related studies was 30 micrograms per liter.

In summary, dietary exposures are not expected to be significant due to coral reliance on zooxanthellae, accumulation to toxic levels is extremely unlikely to occur such that it is discountable, and adverse effects on phytoplankton populations are extremely unlikely to occur such that they are discountable. Therefore, NMFS concluded that EPA's approval of the 4-nonylphenol criterion for predominantly marine waters is not likely to adversely affect due to indirect effects through indirect toxicity.

9.3 Risk Analysis

The exposure and response analyses in the preceding sections determined that the individual fitness of ESA-listed coral species under NMFS's jurisdiction is likely to be adversely affected by exposures to 4-nonylphenol at the proposed marine criterion. Specifically, NMFS concluded that exposure to 4-nonylphenol at FDEP's proposed marine 4-nonylphenol criterion may reduce the fitness of corals through effects to growth and development because toxicity data indicate that effects occur in other marine invertebrates at concentrations below the proposed criteria. In addition, NMFS concluded that exposure to 4-nonylphenol at FDEP's proposed marine criterion may reduce fitness of corals through decreased fecundity because toxicity data for fertilization success indicated the potential for reproductive effects in a coral species known to be robust to toxicant exposures (Mountainous star coral) relative to other listed coral species. During consultation, we could find limited information regarding the relative differences in toxicant sensitivity among the seven species of ESA-listed corals considered.

Successful recruitment of larvae into sexually reproducing adults is the only means by which genetically unique individuals enter a population, thereby maintaining or increasing genotypic diversity. Genotypic diversity is important for stress resilience and disease causing organisms (Foret et al. 2007, Baskett et al. 2010, Schopmeyer et al. 2012). The development of planula into adult polyps is a critical step in establishing genetically unique colonies. Asexual reproduction in stony corals is dependent on growth. It occurs through budding, with new polyps forming from a parent polyp, or through fragmentation, with broken pieces attaching to suitable substrate, forming a new, but genetically identical, colony (Brainard et al. 2011). The potential consequences of the dominance of genetically identical colonies include poor to no reproductive success, because these species do not self-fertilize and increased susceptibility to stress events for which that clone is not adapted. Additionally, severe fragmentation, as commonly observed after storms, may limit future sexual reproduction by reducing the biomass of colonies and shifting energy allocation from reproduction to regeneration (i.e., growth, Jackson, 1986). Taking the above information into consideration, toxicants impairing sexual reproduction,

growth and development can clearly threaten the persistence and recovery of ESA-listed coral populations under NMFS jurisdiction.

9.4 Uncertainty within the Risk Analysis

The limited availability of data for toxicant effects on coral species has and will continue to present a substantial information gap when evaluating proposed water quality criteria or proposed pollutant discharge permits. In this analysis, it was necessary to rely on data for 4-nonylphenol effects on an unrelated marine copepod, Phylum Arthropoda, to infer effects on coral, which belong to the Phylum Cnidaria. The data gaps for corals are substantial.

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 CFR 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.

Florida's population has grown steadily throughout the past several decades. From 2000 to 2012, Florida's population grew at an annual average rate of 1.5 percent, adding on average 259,600 residents annually (U.S. Census Bureau 2012). Florida is currently the fourth most populous state in the U.S. (~ 20 million residents) and is expected to continue to grow in the decades to come (FDOT 2014). In addition to the large and growing resident population, Florida is the top travel destination that attracts large numbers of tourists and vacationers each year. In 2015 an estimated 101.5 million people visited Florida, an increase of 19 percent since 2011 (Visit Florida official website http:// http://www.visitflorida.org/). General resource demands in Florida are expected to increase as a result of population growth (both resident and visitors), as well as the anticipated increase in the average standard of living in Florida. These demands are particularly high in coastal areas which have higher population densities and greater resource consumption compared to other parts of the state.

The future intensity of specific non-Federal activities in the action area is molded by difficult-to-predict future economy, funding levels for restoration activities, and individual investment decisions. In addition, the need to for communities to adapt to climate change and recover from severe climatic events will influence how wetlands, inland surface waters, and coastal areas are managed. Due to their additive and long-lasting nature, the adverse effects of non-Federal activities that are stimulated by general resource demands, and driven by changes in human population density and standards of living, are likely to compound in the future. Specific human activities that may contribute to declines in the abundance, range, and habitats of ESA-listed species in the action area include the following: urban and suburban development; shipping; infrastructure development; water withdrawals and diversion; recreation, including off-road vehicles and boating; expansion of agricultural and grazing activities, including alteration or clearing of native habitats for domestic animals or crops; and introduction of non-native species which can alter native habitats or out-compete or prey upon native species.

Activities which degrade water quality will continue into the future. These include conversion of natural lands, land use changes from low impact to high impact activities, water withdrawals, effluent discharges, the progression of climate change, the introduction of nonnative invasive species, and the introduction of contaminants and pesticides. Under Section 303(c) of the Clean Water Act, individual states are required to adopt WQSs to restore and maintain the chemical, physical, and biological integrity of the nation's waters. EPA must approve of state WQSs and this approval is subject to ESA section 7 consultation, which is the purpose of this Opinion. While some of the stressors associated with non-federal activities that degrade water quality will

be directly accounted for in section 7 consultations between NMFS and EPA, some may be accounted for only indirectly, while others may not be accounted for at all. In particular, many non-point sources of pollution, which are not subject to Clean Water Act NPDES permit and regulatory requirements, have proven difficult for states to monitor and regulate. Non-point source pollution have been linked to loss of aquatic species diversity and abundance, coral reef degradation, fish kills, seagrass bed declines and toxic algal blooms (Gittings et al. 2013). Non-point sources of pollution are expected to increase in Florida as the human population continues to grow. Florida will need to address increases in non-point source pollution in the future to meet the state's approved WQS and designated water body use goals. Given the challenges of monitoring and controlling non-point source pollution and accounting for all the potential stressors and effects on listed species, chronic stormwater discharges will continue to result in aggregate impacts.

Bycatch of ESA-listed species in commercial and recreational fishing gear (discussed in the "Baseline" section) will also continue into the future. The 1995 Florida net ban outlawed the use of entangling nets (i.e., gill and trammel nets) and restricted other forms of nets (i.e., seines, cast nets, and trawls) in state waters (nine nautical miles from the Gulf coast and three nautical miles from the Atlantic coast). This law has greatly reduced bycatch of listed species in state managed fisheries (NMFS/USFWS 2009). Sawfish and sturgeon may still occasionally be captured incidentally in Florida's state waters in pound nets, fyke/hoop nets, fish traps, shrimp trawls, hook and line fisheries, and other allowed gears types. NMFS is not aware of any proposed or anticipated changes in non-federally managed fisheries that would substantially change the impacts each fishery has on listed species and the analysis in this opinion.

Commercial and recreational vessel activity in Florida waters is likely to increase in the future with increases in population size, tourism, and average standard of living. As a result, the cumulative effects of vessel strikes involving sturgeon are also expected to continue to increase.

11 INTEGRATION AND SYNTHESIS

The *Integration and Synthesis* section is the final step in our assessment of the risk posed to species and designated critical habitat as a result of implementing the proposed action. In this section, we add the effects of the action from the *Risk Analysis* section of this opinion (Section 9.1) to the *Environmental Baseline* (Section 8) and the *Cumulative Effects* (Section 10) to formulate the agency's 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 designated critical habitat for the conservation of the species. These assessments are made in full consideration of the status of the species and designated critical habitat (Section 7).

The 4-nonylphenol criteria are based on EPA's 2005 recommended chronic criteria values for 4-nonylphenol, which are intended to be implemented as four-day average concentrations (USEPA 2005). Florida's adoption of the chronic criteria as never-to-be-exceeded concentrations is more protective than adopting the criteria as recommended by EPA. Yet from current monitoring data, it appears that monitoring that would detect violations of this criterion is unlikely. EPA's BE states: "Florida could not locate any nonylphenol sampling data in their state water quality database. Florida hypothesizes that the concentrations are below detection, but does not have data to support their hypothesis." The absence of data of any kind is not equivalent to a "looked for but did not find" conclusion that 4-nonylphenol is not present.

The risk analysis concluded that the gaps in data available for 4-nonylphenol effects on coral or suitable surrogate species, taken with data shared by Dr. Cheryl Woodley on 4-nonylphenol effects on fertilization in mountainous star coral (C. Woodley, NOAA National Center for Coastal Science, pers. comm. to P. Shaw-Allen, NMFS OPR, November 2, 2017), and the greater sensitivity of marine invertebrates to 4-nonylphenol suggests that exposure to the 4-nonylphenol at the marine criterion may result in adverse effects in populations of elkhorn coral, staghorn coral, rough cactus coral, pillar coral, lobed star coral, mountainous star coral, and boulder star coral. Accordingly, this integration and synthesis focuses on whether exposure to 4-nonylphenol at FDEP's proposed criterion for predominantly marine waters is likely to reduce appreciably the likelihood of both the survival and recovery of ESA-listed coral species in the wild by reducing their numbers, reproduction, or distribution.

Our discussion of threats common to coral species (Section 7.1.2) described why sexual reproduction is important to the resilience of coral colonies. The resulting planula are also the only life cycle phase that disperse over long distances, genetically linking populations and providing potential to re-populate depleted areas (Hughes and Tanner 2000, Jackson et al. 2014, NMFS 2015b). The environmental baseline (Section 8) describes the aftermath of the 2005 Caribbean widespread coral bleaching event and record hurricane season resulting in coral cover reduced to less than 12 percent on many reefs (Rogers et al. 2008). Since 2014, coral reef habitats around the world have been subject to elevated ocean surface temperatures (Figure 8) precipitating a prolonged global bleaching event extending into early 2017 (Hughes et al. 2017).

In addition to these events, dredging of the Port of Miami and Hurricane Irma resulted in severe localized sedimentation damage to Florida reef tracts. Sediment plumes resulting from Hurricane Irma and Maria also affected reefs of Puerto Rico and the Virgin Islands. Conclusions of a report published by the Global Coral Reef Monitoring Network and Reef and Rainforest Research Centre indicated that while coral reefs have persisted for millions of years under hurricane impacts, the ability of corals to recover from severe storms, while facing the combined effects of increasing thermal stress and ocean acidification due to climate change, could be extinguished.

The impacts of recent hurricane and dredging damage are reported to be highly localized. NMFS takes this to mean that coral colonies from nearby areas with lesser impacts could contribute to recovery. The status of the species section for coral (Section 7) reported small pockets of remnant robust populations of staghorn and elkhorn coral in southeast Florida (Vargas-Angel et al. 2003), Honduras (Keck et al. 2005, Riegl et al. 2009) and the Dominican Republic (Lirman et al. 2010). There have also been reports of more stable percent cover trends (e.g., Bonaire) and periods of increase (e.g., Flower Garden Banks). Despite decreases, the *O. annularis* complex continues to be reported as the dominant coral taxa. Pillar coral appears to be extirpated from portions of the Florida Reef Tract adjacent to populated areas of Dade County, but healthy populations occur along the Florida Keys. Like pillar coral, rough cactus coral is usually uncommon to rare throughout its range. Its abundance, combined with spatial variability in threats across the species' range, moderate vulnerability to extinction because these threats are non-uniform. Post Hurricane Irma recovery efforts for Florida reefs have been completed or are underway. Impact assessment identified 14 percent of high value areas assessed that were a high priority for triage mitigation and 56 percent that were minimal impact or not an ideal reef community for triage. The post 2017 hurricane season status of reefs in the broader Caribbean is not yet known.

These events underscore a current need for recovery of the species through growth and successful recruitment of new coral fragments and planulae. The stressors of the action section (Section 3.1) explained that nonylphenol mimics 17β-estradiol, which 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 reduced reproductive potential and growth (Tarrant et al. 2004). Unpublished data also indicate reduced fertilization at high 4-nonylphenol concentrations, 100 micrograms per liter, in a relatively robust coral species (C. Woodley, NOAA National Center for Coastal Science, pers. comm. to P. Shaw-Allen, NMFS OPR, November 2, 2017). While the exact mechanism and response threshold for estrogenic effects of 4-nonylphenol in ESA-threatened coral species are unknown, exposures leading to effects on reproduction and growth would be expected to impede recovery from hurricane damage and thermal stress, two major threats to this species group.

While data reporting a response threshold for adverse effects of 4-nonylphenol on coral species were not found, evidence indicates that 17β-estradiol has a role in sexual reproduction of coral and data point to a greater sensitivity to 4-nonylphenol effects on development and growth

among other marine invertebrates. Corals are therefore expected to be susceptible to the effects of estrogenic chemicals such as 4-nonylphenol. Data for other marine invertebrates indicate effects on growth and development occur at concentrations below the proposed criterion. Sexual reproduction, and subsequent development and growth of newly established polyps, is important to the resilience and long-range distribution of coral species. After disturbances like hurricanes, disease outbreaks, and bleaching, genetic diversity and growth are also important to the recovery of colonies and reefs. Repeated, recent, and increasingly more frequent thermal stress and hurricane disturbances magnifies the importance of sexual reproduction and growth, but these events appear to result in localized effects, such that colonies in lesser impacted areas, and human triage efforts, can contribute to recovery.

The types and magnitude of responses for the marine invertebrate are helpful in determining whether effects translate to effects to the species. The study evaluating effects on Australian barnacle (Billinghurst et al. 2001) observed disruption in the timing of larval development, but did not indicate any abnormalities in final development. Among the harpacticoid copepod studies, the Marcial et al (2003) study reported that control organisms exposed matured from nauplii to copepod stage in 5.6+/- 0.5 days while those exposed at the LOEC of 1 microgram 4-nonylphenol per liter matured in 6.4 +/- 0.7 days. For the opossum shrimp data, growth at the LOEC of 1 microgram 4-nonylphenol per liter was reduced by 5 to 10 percent compared to controls (Hirano et al. 2009). The significance of such effects on growth and development represents an erosion in the fitness of exposed organisms, but is not expected to result in species-level effects.

12 CONCLUSION

After reviewing the current status of the ESA-listed species, 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 likely to adversely affect, but not likely to jeopardize the continued existence or recovery of elkhorn coral, staghorn coral, rough cactus coral, pillar coral, lobed star coral, mountainous star coral, boulder star coral.

13 INCIDENTAL TAKE STATEMENT

Section nine 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.

Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. 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.

13.1   Amount or Extent of Take

Section 7 regulations require NMFS to specify the impact, i.e., the amount of extent, of any incidental take of endangered or threatened species; that is, the amount or extent, of such incidental taking on the species (50 CFR § 402.14 (i)(1)(i)). A "surrogate (e.g., similarly affected species or habitat or ecological conditions) may be used to express the amount or extent of anticipated take provided that the biological opinion or ITS: Describes the causal link between the surrogate and take of the listed species, explains why it is not practical to express the amount or extent of anticipated take or to monitor take-related impacts in terms of individuals of the listed species, and sets a clear standard for determining when the level of anticipated take has been exceeded." (50 C.F.R. § 402.14).

The proposed action is anticipated to cause incidental take because EPA proposes to approve a water quality criterion for 4-nonylphenol for predominantly marine waters of Florida that is greater than exposure concentrations reported to cause adverse effects in the growth and development of non-coral marine invertebrates. The only data available for the effects of 4-nonylphenol for coral is for a moderately toxicant-tolerant coral species and these data did not test exposure concentrations useful in evaluating the criterion (Section 8.1). Use of this proposed criterion by FDEP in its water quality regulatory actions (e.g., NPDES permit effluent limitations, 305(b) assessments) therefore may result in incidental take of ESA-listed coral species under NMFS jurisdiction. Specifically, these incidental takes are anticipated to include:

- Reduced generation of viable planula

- Reduced recruitment of genetically unique individuals into the adult population

- Reduced distribution and recruitment into unoccupied habitat

- Reduced reestablishment of coral fragments during post disturbance recovery

Florida currently does not have a criterion for 4-nonylphenol in surface waters, so FDEP water quality regulation efforts cannot identify waters with levels of 4-nonylphenol known to be

harmful to aquatic life or place effluent limits on facilities that may discharge 4-nonylphenol at concentrations that would be harmful to aquatic life. While exposures to 4-nonylphenol at and below the criterion may adversely affect ESA-listed coral, the absence of any 4-nonylphenol criterion may currently be resulting in greater amounts of take of ESA-listed corals under NMFS jurisdiction than would occur if the proposed criterion were implemented.

Incidental take under the 4-nonylphenol criterion for predominantly marine waters cannot be accurately quantified or monitored as a number of individuals because the action area includes the entire Florida reef tract and data do not exist that would allow us to quantify how many individuals of each species and life stage exist in affected waters, especially considering that the numbers of individuals vary with environmental conditions, and changes in population size due to recruitment and mortality. In addition, currently we have no means to detect or determine which impairments to reproduction, development, and growth are due to the water quality under a Florida 4-nonylphenol criterion versus other natural and anthropogenic environmental stressors. Because we cannot quantify the amount of take, we will use a water quality measure reflecting the presence of substances that are or will degrade into nonylphenol for the extent of authorized take as a surrogate for the amount of authorized take.

The EPA's approved standard method for the measurement of nonylphenols (USEPA, 2016), ASTM method D7065-11, can quantify nonylphenol at concentrations above five micrograms per liter, this is the method's reporting limit. This reporting limit is more than twice FDEP's proposed criterion of 1.7 micrograms 4-nonylphenol per liter for predominantly marine waters. Using this method, waters containing nonylphenol concentrations between 1.7 and five micrograms per liter may not be identified as impaired. To confirm that the specified amount or extent of take is not exceeded, a more sensitive method is needed to quantify nonylphenol in waters where ESA-listed corals occur or may recolonize.

The specified amount or extent of incidental take of ESA-listed coral species is the presence of 4-nonylphenol in waters of the Florida Reef tract at concentrations that are equal to or below FDEP's proposed criterion of 1.7 micrograms 4-nonylphenol per liter. Authorized take of ESA-listed coral species will have been exceeded if nonylphenol is found to occur in Florida Reef Tract waters at concentrations above FDEP's proposed criterion of 1.7 micrograms 4-nonylphenol per liter, using appropriately sensitive methodology (i.e., use ASTM method D7485-09[14] when concentrations are below five micrograms per liter.

This reflects Florida's intended level of protection for aquatic life in predominantly marine waters and ensures that exceedances will be detected and addressed, thereby minimizing take of

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[14] In 2012, this methodology was proposed for inclusion into 40 CFR Parts 136: Guidelines Establishing Test Procedures for the Analysis of Pollutants Under the Clean Water Act; Analysis and Sampling Procedures. However, EPA decided to postpone approval for general use until: *"...completion of a full interlaboratory validation study designed to fully characterize the performance of these methods across multiple laboratories and matrices."* Currently, EPA's Region five laboratory has this analysis capability.

ESA-listed coral. The EPA will need to ensure that the criterion is incorporated into effluent limitations and that monitoring occurs to identify and address waters where ambient concentrations of 4-nonylphenol exceed the criterion. Consideration should also be given to the presence of 4-nonylphenol precursors, nonylphenol ethoxylates, at concentrations that potentially result in future 4-nonylphenol concentrations above the criterion.

13.2   Effects of Take

During consultation as documented in this opinion, NMFS determined that the amount or extent of anticipated take is not likely to result in jeopardy to the ESA-listed coral species with populations in Florida waters.

13.3   Reasonable and Prudent Measures

The measures described below are nondiscretionary, and must be undertaken by EPA so that 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, reasonable and prudent measures, 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.

"Reasonable and prudent measures" are nondiscretionary measures to minimize the amount or extent of incidental take (50 C.F.R. §402.02). NMFS believes the reasonable and prudent measures described below are necessary and appropriate to minimize the impacts of incidental take on threatened and endangered species:

NMFS believes all measures described as part of the proposed action, together with the RPM described below, are necessary and appropriate to minimize the likelihood of incidental take of ESA-listed species due to implementation of the proposed action:

1) The EPA will use its authorities to ensure that a 4-nonylphenol criterion that minimizes take of ESA-listed coral species is applied and implemented in waters where these species occur or may recolonize.

2) The EPA will ensure that the extent of incidental take is not exceeded through monitoring 4-nonylphenol in waters where ESA-listed corals occur or may recolonize.

13.4   Terms and Conditions

To be exempt from the ESA prohibitions of take, 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 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.

1) The following term and condition implements reasonable and prudent measure 1:

   a) Within one month of the signature on this opinion, EPA will write a letter to the FDEP, and copy NMFS. The letter will:

      i) Inform FDEP that exposures to 4-nonylphenol at or below the proposed criterion of 1.7 micrograms per liter for predominantly marine waters may adversely affect ESA-listed coral species under NMFS jurisdiction.

      ii) Indicate that EPA is approving the proposed 4-nonylphenol criteria, including the application of the criterion of 1.7 micrograms per liter for predominantly marine waters in waters where ESA-listed corals occur or may recolonize.

      iii) Inform FDEP that if data becomes available suggesting that exposures to 4-nonylphenol at the criterion of 1.7 micrograms in predominantly marine waters is likely to adversely affect ESA-listed corals under NMFS jurisdiction, EPA will reinitiate consultation with NMFS on these effects.

      iv) As such, EPA's approval does not foreclose either the formulation by NMFS, or the implementation by the EPA, of any alternatives that might be determined in the reinitiated consultation to be needed to comply with section 7(a)(2).

      v) Require that effluent limitations for industries that use 4-nonylphenol and nonylphenol ethoxylates and discharge into a mixing zone that includes coral reef habitat include an end of pipe restriction for 4-nonylphenol at 1.7 micrograms per liter.

2) The following term and condition implements reasonable and prudent measure 2:

   a) In the same letter required in the first term and condition, EPA will explain to FDEP that:

      i) Unauthorized take of ESA-listed species is prohibited under section 9 of the ESA and this prohibitions applies to all individuals, organizations, and agencies subject to United States jurisdiction.

      ii) Monitoring data for nonylphenol is required to determine whether take covered by this opinion has been exceeded, and EPA will need to report the results of monitoring to NMFS.

      iii) Strongly encourage FDEP to design and implement, within the next 18 months, an effective monitoring program for the Florida Reef Tract using an analytical method that can quantify 4-nonylphenol at the criterion in order to establish whether waters where ESA-listed corals occur or may recolonize comply with the criterion, and

proceed accordingly. Coordinate with NOAA coral experts in the design of this monitoring program. The program will:

  (a) Sunset when 4-nonylphenol concentrations are demonstrated to be at or below the criterion, or is achieved in cases where water quality is impaired (since nonylphenol production and use is being phased out).

  (b) Include periodic, adaptive monitoring to detect whether previously undiscovered discharges to these waters are occurring.

  (c) Include a monitoring trigger if a new discharge or significant new use is authorized.

b) If within three years, FDEP does not establish a program and initiate the monitoring described in item iii above, EPA will work with NMFS to design a monitoring program that will enable them to ensure that take is not exceeded.

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 designated critical habitat, to help implement recovery plans or develop information (50 C.F.R. §402.02).

Actions or measures that could also minimize or avoid adverse effects of Florida's proposed 4-nonylphenol criterion for predominantly marine waters on ESA-listed coral species under NMFS jurisdiction include:

  (1) Coordinate with coral experts to close gaps in our understanding of the effects of 4-nonylphenol and its ethoxylate precursors[15] on reef species biology, ecology, and recovery by conducting toxicity testing using coral and other reef species.
  (2) Coordinate with state and federal agencies that carry out water quality monitoring in Florida marine waters where coral occur or could reestablish to sample and analyze for 4-nonylphenol other endocrine disruptors to close gaps (see Section 8.1.5) in our understanding of endocrine disruptors that are likely to occur in water, sediment, or biota.
  (3) Use information gained in items 1) and 2) above, along with up-to-date toxicity data, to determine whether coral reef species are at risk from exposure to endocrine disruptors.
  (4) If the analysis in item 3) above indicate species are currently at risk or may be at risk in the future, coordinate with private, state, and federal stakeholders to develop and implement actions that minimize or prevent such risks.

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[15] Optimally, such work would evaluate a suite endocrine disrupting substances in addition to 4-nonylphenol and nonylphenol ethoxylates.

In order for NMFS' Office of Protected Resources 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 designated 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. 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, including any new information suggesting that 4-nonylphenol at concentrations below 1.7 micrograms per liter in marine waters are likely to cause greater reductions in fitness, and greater adverse impacts on populations and species, than identified in this opinion.

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. For example, reinitiation would be triggered upon listing additional marine invertebrates as threatened or endangered under the ESA or upon designation of critical habitat that includes toxicant-sensitive biological features or water quality requirements related to pollutants.

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117

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