PETITION TO LIST THE CAULIFLOWER CORAL (*POCILLOPORA MEANDRINA*) IN HAWAII AS ENDANGERED OR THREATENED UNDER THE ENDANGERED SPECIES ACT

Photo: Scott Godwin/National Park Service

Submitted to the U.S. Secretary of Commerce acting through the National Oceanic and Atmospheric Administration and the National Marine Fisheries Service

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Alaska . Arizona . California . Florida . Minnesota . Nevada . New Mexico . New York . Oregon . Vermont . Washington, DC

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The Center for Biological Diversity ("petitioner") hereby petitions the Secretary of Commerce and the National Oceanic and Atmospheric Administration ("NOAA"), through the National Marine Fisheries Service ("NMFS"), to list the cauliflower coral (*Pocillopora meandrina*) in Hawaii as an endangered or threatened species in a significant portion of its range and designate critical habitat to ensure its recovery pursuant to Section 4(b) of the Endangered Species Act ("ESA"), 16 U.S.C. § 1533(b), section 553(e) of the Administrative Procedure Act, 5 U.S.C. § 533(e), and 50 C.F.R. § 424.14(a).

The Center for Biological Diversity ("the Center') is a national, nonprofit conservation organization dedicated to the protection of endangered species and wild places through science, policy, and environmental law. Among the Center goals is to use the ESA as a powerful tool to preserve imperiled species throughout the United States and abroad and thus conserve and restore biodiversity. The Center has more than 1.5 million members and online activists with a direct interest in ensuring the survival and recovery of imperiled species such as corals. The Center is highly invested in conserving the fragile and impacted coral reef ecosystems of Hawaii and the marine species that depend on them.

NMFS has jurisdiction over this petition because the cauliflower coral, *P. meandrina*, is a marine species. This petition sets in motion a specific legal process, requiring NMFS to make an initial finding as to whether the Petition "presents substantial scientific or commercial information indicating that the petitioned action may be warranted." 16 U.S.C. § 1533 (b)(3)(A). NMFS must make this initial finding "(t)o the maximum extent practicable, within 90 days after receiving the petition." *Id*. The petitioner does not need to demonstrate that listing is warranted, rather, the petitioner must only present information demonstrating that such a listing may be warranted. While the petitioner believes that the best available science demonstrates that listing the cauliflower coral, *P. meandrina* as threatened or endangered in Hawaii as a significant portion of its range is warranted, there can be no reasonable dispute that the available information indicates that listing this species as either threatened or endangered may be warranted. Thus, NMFS should promptly make a positive finding on the Petition and commence a status review as required by the ESA. 16 U.S.C. § 1533 (b)(3)(B).

The Center respectfully submitted this Petition this 14th day of March, 2018.

Suggested citation:

Center for Biological Diversity (2018) Petition to list the cauliflower coral (*Pocillopora meandrina*) in Hawaii as endangered or threatened under the Endangered Species Act. *Center for Biological Diversity*, 52 pp.

Executive Summary

Corals and coral reef ecosystems worldwide are in crisis. Nearly 25% of the world's coral reefs have already been lost, and approximately one-third of all reef-building coral species as at risk of extinction (Carpenter et al. 2008; Veron et al. 2009). Globally, reef-building corals face widespread threats ranging from climate change, habitat destruction, pollution, overfishing, and diseases. In particular, climate change effects such as ocean warming and acidification caused by anthropogenic greenhouse gas emissions threaten the existence of corals and coral reef ecosystems as we know them (Pandolfi et al. 2011; Hughes et al. 2017b). Scientific evidence supports that coral reefs are likely the first ecosystem to collapse at a planetary scale due to climate change (Veron et al. 2009; Hughes et al. 2017b).

This petition seeks to list the cauliflower coral, *Pocillopora meandrina,* as threatened or endangered under the US Endangered Species Act ("ESA"). This coral species has substantially declined over the past years due to mass bleaching events in Hawaii, and it is further locally threatened by land-based pollution, sedimentation, overfishing, and physical damage due human activities. Ocean warming and acidification due to anthropogenic greenhouse emissions pose the most serious short- and long-term threats to the survival this coral species. These threats are already affecting this coral species. Urgent action, including listing under the ESA and substantial cuts in greenhouse gas emissions, is needed now to ensure that this coral species does not become extinct in the foreseeable future.

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1. Introduction

The cauliflower coral (*Pocillopora meandrina*) is a relatively small upright bushy coral species found in shallow reefs throughout the Indo-Pacific and East Pacific. In Hawaii, *P. meandrina* has experienced high bleaching prevalence and subsequent mortality due to thermal stress events during the past four years (2014-2017), the third global bleaching event on record (Eakin 2017). The frequency and severity of thermal stress events in the central Pacific is predicted to increase and it is likely to cause stronger mass coral bleaching, placing *P. meandrina* at high extinction risk in the foreseeable future in a significant portion of its range. In addition, localized sedimentation and pollution from inadequate land-use practices and overfishing of herbivorous fishes have contributed to the decline of shallow water corals in areas of the Main Hawaiian Islands. Thus, the Center petitions NMFS to list the shallow water coral *P. meandrina* as endangered or threatened under the ESA because it is imperiled throughout a significant portion of its range, the Hawaiian Islands.

There is a strong precedent for listing corals under the ESA that show high extinction risk due to climate change. In 2006, NMFS listed the Caribbean species *Acropora cervicornis* and *Acropora palmata* as threatened under the ESA due in part to its vulnerability to ocean warming (NMFS 2006). Further in 2014, NMFS listed 20 additional corals (15 in the U.S. Pacific and 5 in the Caribbean) as threatened under the ESA because warming was the main threat for the continuing survival.

The cauliflower coral warrants ESA listing because of the ongoing and increasing magnitude of climate change threats as the primary contributing factors of the extinction risk. Indeed, the Coral Biological Review Team concluded that the three most important threats increasing extinction risk of corals between now and the end of the century were ocean warming, diseases, and ocean acidification (Brainard et al. 2013 p. 1172). Local threats such as fishing, sedimentation and eutrophication that can cause extirpations at local scales were considered as medium extinction risks (Brainard et al. 2013 p. 1172). Thus, in analyzing extinction risk for *P. meandrina*, NMFS should consider global climate- and ocean change threats (e.g., ocean warming and acidification) and local threats (e.g., land-based pollution, sedimentation, and fishing) that contribute to population decline and extinction in the foreseeable future. In addition, NMFS should consider the cumulative effects and potential synergistic effects among these threats.

NMFS should assess the extinction risk of the cauliflower coral (*P. meandrina*) and protect it under the ESA, including consideration of Hawaii as a significant portion of its range. This coral species occupies the northernmost portion of its range (Johnston et al. 2018 p. 8) and like other coral species it is likely to be severely affected with predicted environmental changes in the near future, such as increasing frequency and magnitude of warming stress events and continued reduction of carbonate saturation state (Hoeke et al. 2011; Bahr et al. 2017 p. 1083).

Urgent action is needed to protect the cauliflower coral, *P. meandrina*, and for this reason NMFS must make a prompt determination that listing may be warranted and undertake a status review.

2. Governing Provisions of the Endangered Species Act

The ESA was enacted in 1973 “to provide a means whereby the ecosystems upon which endangered species and threatened species depend may be conserved, and to provide a program for the conservation of such endangered species and threatened species.” 16 U.S.C. § 1531(b). Protection under the ESA only applies to species that have been listed as endangered or threatened according to the provisions of the statute. Thus listing species that need ESA protection is vital to their conservation.

Specifically, once listed as an “endangered” species, the ESA prohibits the “take” or the killing, capture, or harassment of individual animals, as well as the sale, export, or import of such species. *Id.* §§ 1538(a); 1532(19) (defining “take”). Alternatively, if a species is listed as “threatened”, NMFS “shall issue such regulations as [it] deems necessary and advisable for the conservation of” the species including potentially the same bans applicable to endangered species. Additionally, whenever a U.S. federal agency takes any action that “may affect” a threatened or endangered species, that agency “shall” consult with NMFS regarding those impacts, and the consulting agencies may establish mitigation measures for the project. *Id.* § 1536(a).

2.1 Species Definition under the ESA

The term “species” is broadly defined under the ESA to include “any subspecies of fish, or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature.” 16 U.S.C. § 1532 (16). The ESA provides for the listing of all species that warrant the protections afforded by the Act. NMFS and the U.S. Fish and Wildlife Service (“FWS”) have published a policy to define a “distinct population segment” for the purposes of listing, delisting, and reclassifying species under the ESA. 61 F.R. 4722 (February 7, 1996). Under this policy, a population that is both “discrete” and “significant” can be eligible for listing under the Act. However, the distinct population segment concept does not apply to invertebrates like corals.

2.2 Significant Portion of the Species’ Range

The ESA defines an “endangered species” as any species that is “in danger of extinction throughout all or a significant portion of its range,” 16 U.S.C. § 1532(6), and a “threatened species” as one that “is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range.” 16 U.S.C. § 1532(20). The ESA does not define the phrase “significant portion of its range” (SPR). The 2014, FWS and NMFS (“the Services”) policy interpretation of SPR (79 FR 37577) is invalid and hasbeen vacated in court (see *Center for Biological Diversity et al. v. Jewell 2017, CV-14-02506-TUC-RM*).

NMFS should list a species range-wide as endangered if it faces extinction in a significant portion of its range. It should list a species as threatened if it is likely to become endangered in the foreseeable future in a significant portion of its range. . Thus, if a species is imperiled in an important part of its range it must be protected throughout its range.

While the ESA does not define "foreseeable future," NMFS must use a definition that is reasonable, ensures protection of the petitioned species, and gives the benefit of the doubt regarding any scientific uncertainty to the species of extinction in the future. Generally, the minimum time period that meets these criteria is 100 years. In fact, a 100-year timeframe has been used to project changes to growth and mortality of Hawaiian corals (Hoeke et al. 2011). Because ocean warming and acidification are the foremost threats to the petitioned coral species, NMFS should consider the timeframes used in climate modeling by the latest Fifth Assessment of the Intergovernmental Panel on Climate Change, AR5 (IPCC 2013, 2014). This approach was already used during the latest coral listing process (79 FR 53851). Predictions of climate impacts in the next 100 years or more are routine in the literature, demonstrating that climate impacts within this timeframe are inherently "foreseeable." Therefore, NMFS should use 100 years as the foreseeable future in determining extinction risk for this coral species.

2.3 Climate Change effects guidance

During the reviewing process of this coral species, NMFS must follow the most updated guidance for ESA decisions involving species influenced by climate change (http://www.nmfs.noaa.gov/pr/pdfs/pr_climate_change_guidance_june_2016.pdf). For example:

" For ESA decisions involving species influenced by climate change, NMFS will use climate indicator values projected under the Intergovernmental Panel on Climate Change (IPCC)'s Representative Concentration Pathway 8.5 (RCP 8.5) when data are available. When data specific to that pathway are not available, we will use the best available science that is consistent as possible with RCP 8.5."

"When predicting the future status of species in decisions under ESA sections 4, 7, and 10, NMFS will project climate change effects for the longest timer period over which we can reasonable foresee the effects of climate change on the species' status."

"When addressing the adequacy of existing regulatory mechanisms in status reviews, listing decisions and recovery plans analysis, NMFS will cite to or draw from previous NMFS findings, updated as appropriated in light of development in this area, to describe the adequacy of existing global and national climate change regulatory mechanisms"

2.4 Listing Factors under Section 4(A)(1) of the ESA

Under the ESA, NMFS must make a determination whether a species is endangered or threatened based on the best readily available scientific or commercial information on the following five listing factors, 16 U.S.C. §1533(a)(1):

A. The present or threatened destruction, modification, or curtailment of its habitat or range;
B. Overutilization for commercial, recreational, scientific, or education purposes;
C. Disease or predation;
D. The inadequacy of exiting regulatory mechanism; and
E. Other natural or manmade factors affecting its continue existence.

For a species to be listed under the ESA it needs to only face a substantial threat under one of the above mentioned factors. In addition, any combination of threats that can be considered cumulatively under multiple factors would also support ESA listing.

2.5 90-Day and 12-Month Findings

NMFS is required to determine "to the maximum extent practicable … whether [a] petition presents substantial scientific or commercial information indicating that the petitioned action may be warranted" within 90 days of receiving a petition to list a species. *Id.* § 1533(b)(3)(A). This is also known as the "90-day finding". A "negative" 90-day finding will end the listing process. *Id.* § 1533(b)(3)(C)(ii). A "positive" 90-day finding will lead to a more comprehensive "status review" and to a "12-month finding" that determines, based on the best available scientific and commercial information, whether listing the species as endangered or threatened is warranted, not warranted, or warranted but precluded by other pending listing proposals for higher priority species. *Id.* § 1533(b)(3)(B). A 90-day finding, a not warranted finding, or a warranted but precluded 12-month finding are subject to judicial review. *Id.* § 1533(b)(3)(C)(ii).

For the purposes of the 90-day finding, "substantial information" is defined as "the amount of information that would lead a reasonable person to believe that the measure proposed in the petition may be warranted" 50 C.F.R. § 424.14(b)(1). Under NMFS's regulations a petition presents "substantial information" if it:

i. Clearly indicates the administrative measure recommended and gives the scientific and any common name of the species involved;
ii. Contains detailed narrative justification for the recommended measure; describing, based on available information, past and present numbers and distribution of the species involved and any threats faced by the species;
iii. Provides information regarding the status of the species over all or a significant portion of its range; and
iv. Is accompanied by appropriate supporting documentation in the form of bibliographic references, reprints of pertinent publications, copies of reports or letters from authorities, and maps.

50 C.F.R. §§ 424.14(b)(2)(i)–(iv).

This petition presents substantial information that would lead a reasonable person to believe that listing the cauliflower coral, *P. meandrina,* in Hawaii as threatened or endangered within a significant portion of its range under the ESA may be warranted.

2.6 Reasonable Person Standards

During the initial petition review to make a 90-day finding the ESA does not require "conclusive evidence of a high probability of species extinction" to support a positive outcome. 50 C.F.R. § 424.14(b)(1). Instead, during the initial 90-day review process, NMFS must consider whether a reasonable person could determine that the petition contains substantial information that may warrant a more in-depth status review of the petitioned species. Thus this initial review should be characterized as a "threshold determination." Accordingly, a petition does not need to demonstrate a high likelihood that a species is endangered or threatened during the 90-day finding process, but it need only warrant further review.

2.7 Best Available Scientific and Commercial Data

NMFS is required to make an ESA listing determination for the cauliflower coral (*P. meandrina*) in Hawaii, considering the five listing factors based on the best available scientific and commercial data. 16 U.S.C § 1533(b)(1)(A). NMFS cannot deny a listing for which little information is available if the best available information indicates that the species is endangered or threatened throughout all, or a significant portion of its range.

3. Natural history of the cauliflower coral *P. meandrina*

3.1 Taxonomic classification and issues

In this petition, the taxonomy of the cauliflower coral (*P. meandrina*) was based on the Integrated Taxonomic Information System ("ITIS"),[1] and is as follows:

Kingdom: Animalia
Phylum: Cnidaria
Class: Anthozoa, Ehrenberg, 1834
Order: Scleractinia Bourne, 1900
Suborder: Astrocoeniina Vaughan and Wells, 1943
Family: Pocilloporidae Gray, 1842
Genus: *Pocillopora* Lamarck, 1816
Species: *Pocillopora meandrina* Dana, 1846

The common name is cauliflower coral. Similar species is *Pocillopora eydouxi* (Veron 2000).

The species was first described by Dana (1846). At the beginning of the 1900s, *P. meandrina* was thought to be a similar species as *P. elegans* and *P. verrucosa* (Vaughan 1918). Later, Veron & Pichon (1976) described it as a separate species.

Fine-scale morphological features of the corallite (e.g., oval-convex to styloid, rarely obsolete columelle) and genetic differentiation of nuclear DNA confirms that *P. meandrina* is a unique species (Flot et al. 2008 p. 7; Pinzon & LaJeunesse 2011 p. 318; Schmidt-Roach et al. 2014 p. 1). However, the genus *Pocillopora* is known to have taxonomic issues because its morphological plasticity and due to multiple sympatric species with similar skeletal morphology. Several *Pocillopora* coral species of the eastern Pacific and western Indian Ocean (e.g., *Pocillopora damicornis*) that were thought to be the same species have shown to be genetically different (Flot et al. 2008 p. 7; Pinzón et al. 2013 p. 1599; Schmidt-Roach et al. 2013, 2014 p. 13).

[1] Integrated Taxonomic Information System: *Pocillopora meandrina* Dana, 1846. Taxonomic Serial No.: 53022 https://www.itis.gov/servlet/SingleRpt/SingleRpt?search_topic=TSN&search_value=53022#null

3.2 Species description and identifying characteristics

Colonies of *P. meandrina* are small (up to 30 cm in diameter) forming upright bushes, usually cream, green or pink in color (Fig. 1A). Colonies in shallow waters with high wave energy have curve branches (Fig. 1B). Branches are generally flattened and radiate from the center and initial point of growth (Fig. 1C). Verruca are neat and uniform (Fig. 1D). Species description was extracted from Veron (2000)

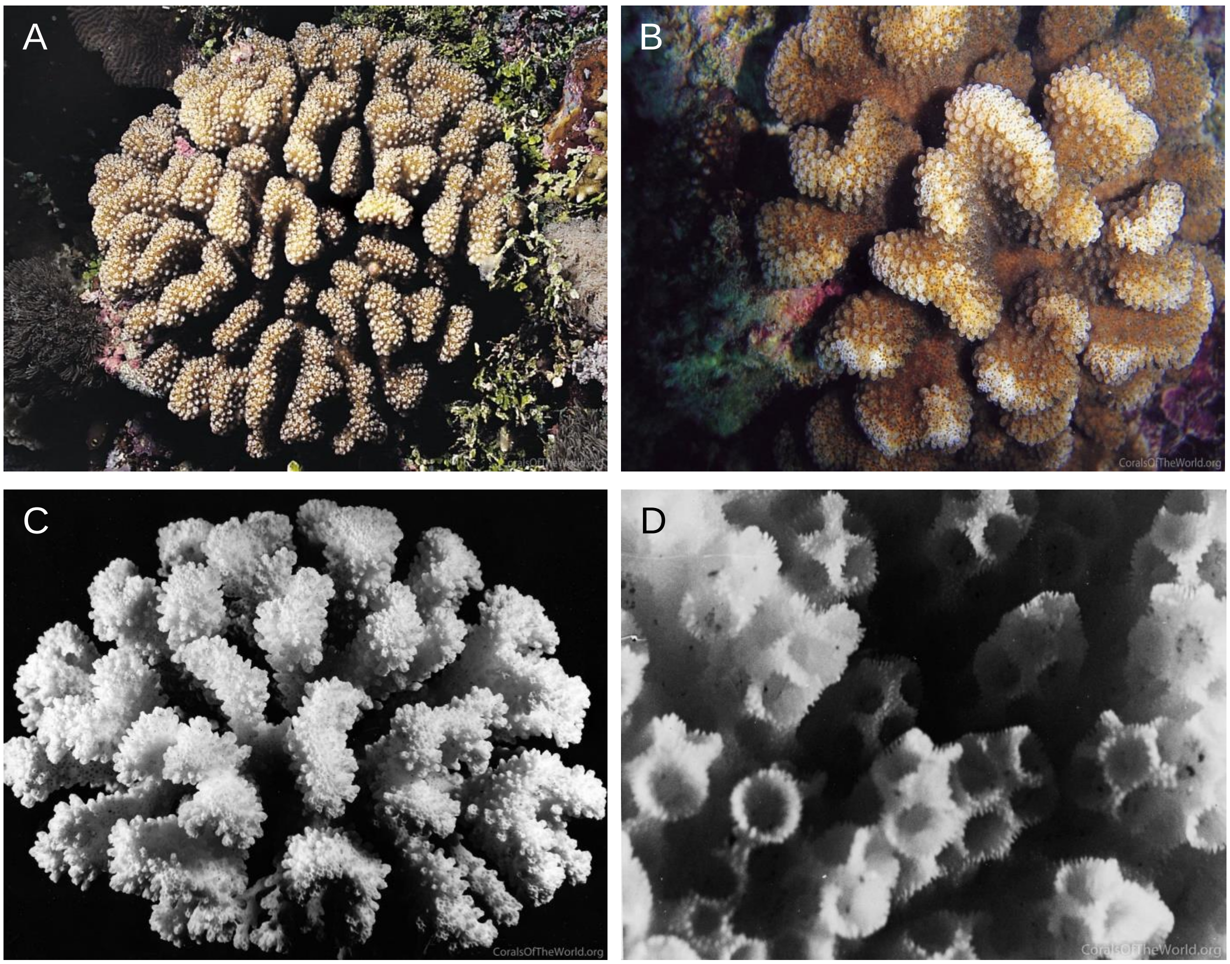


Figure 1 Common colony shapes of *P. meandrina* on shallow reefs. A) Entire colony from Papua New Guinea, photo: Charlie Veron; B) Surface detail from Hawaii, photo: Doug Fenner; C) Colony skeleton from Hawaii, photo: Charlie Veron; D) Close-up showing verrucae, photo: Jim Maragos.

3.3 Life history

P. meandrina is described as a "competitive" species and is likely to recolonize available substrate in disturbed reefs due to relatively faster growth rates than stress tolerant and massive coral species (Darling et al. 2012 p. 3). This life history pattern suggests that *P. meandrina* is an *r* selected species with rapid recruitment as other coral species with similar characteristics (Loya

1976 p. 478). Pocilloporid corals are considered amongst the strongest coral competitors with relatively high growth rates in the Eastern Tropical Pacific region (Glynn 2001 p. 130). For example, the growth rate of these species vary between 3.42 to 4.46 cm per year in Costa Rica (Jiménez & Cortés 2003 p. 190). In general, coral species with high growth rates also exhibit high mortality rates (Glynn 2000 p. 124). Studies have shown that *P. meandrina* growth relatively fast with high recruitment rate (Brown 2004 pp. 76, 89), but it used to be shorted lived in Hawaii due to recolonization by other coral species and high sensitivity to disturbance (Grigg & Maragos 1974 p. 389).

The species is a simultaneous hermaphrodite broadcaster coral that has been documented spawning in Hawaii (Fiene-Severns 1998; Hirose et al. 2001; Marlow & Martindale 2007; Apprill et al. 2009). *P. meandrina* has been observed synchronously spawning four to five days after the full moon during the early mornings of April and May in more less consecutive years (e.g., 1991, 1994, 1995, 1997, 1998) at the Molokini Islet in the southwestern coast of Maui, Hawaii (Fiene-Severns 1998). Based on the few reproduction studies in the wild, spawning events typical last for approximately 15 minutes (Fiene-Severns 1998). *P. meandrina* vertically seeds its eggs with symbiotic zooxanthellae before spawning (Hirose et al. 2001) and thus the planulae larvae contain the symbiotic dinoflagellate, which supplement maternal provisioning through photosynthesis (Marlow & Martindale 2007; Apprill et al. 2009; Baird et al. 2009). We found no studies determining maximum duration of planktonic larvae for *P. meandrina*. However, embryonic development for the species is well studied (Marlow & Martindale 2007). Periodic pulses in recruitment linked to cycles of changes in coral coverage by 10-12 years have been documented in Kahe Point, leeward Oahu in of Hawaii (Coles & Brown 2007).

Because of the branching morphology, asexual reproduction by fragmentation is also common (Glynn & Colley 2008).

3.4 Distribution

The cauliflower coral, *P. meandrina*, is distributed throughout shallow reefs of the tropical and subtropical Indian and Pacific Oceans (Fig. 2). Based on the IUCN account of *P. meandrina*, the species within the Indo-West Pacific, is found in the Central and Northeastern Indian Ocean, the Central Indo-Pacific, Tropical Australia, Southern Japan and the South China Sea, the Oceanic west Pacific, the Hawaiian Islands and Johnston Atoll, and American Samoa region (Hoeksema et al. 2014). In the Eastern Tropical Pacific region, the species occurs from Mexico to Ecuador, including the Galápagos Archipelago (Fig. 2). Specific countries of potential occurrence are American Samoa; Australia; British Indian Ocean Territory; Cambodia; Chile; Christmas Island; Cocos (Keeling) Islands; Colombia; Costa Rica; Ecuador; El Salvador; Fiji; Guadeloupe; Guam; Honduras; India; Indonesia; Japan; Kiribati; Malaysia; Maldives; Marshall Islands; Mauritius; Mexico; Micronesia, Federated States of ; Myanmar; Nauru; New Caledonia; Nicaragua; Northern Mariana Islands; Palau; Panama; Papua New Guinea; Philippines; Samoa; Singapore; Solomon Islands; Sri Lanka; Taiwan, Province of China; Thailand; Tuvalu; United States Minor Outlying Islands; Vanuatu; Viet Nam; Wallis and Futuna (Hoeksema et al. 2014). However, further research may be needed to determine the actual range (Schmidt-Roach et al. 2014).

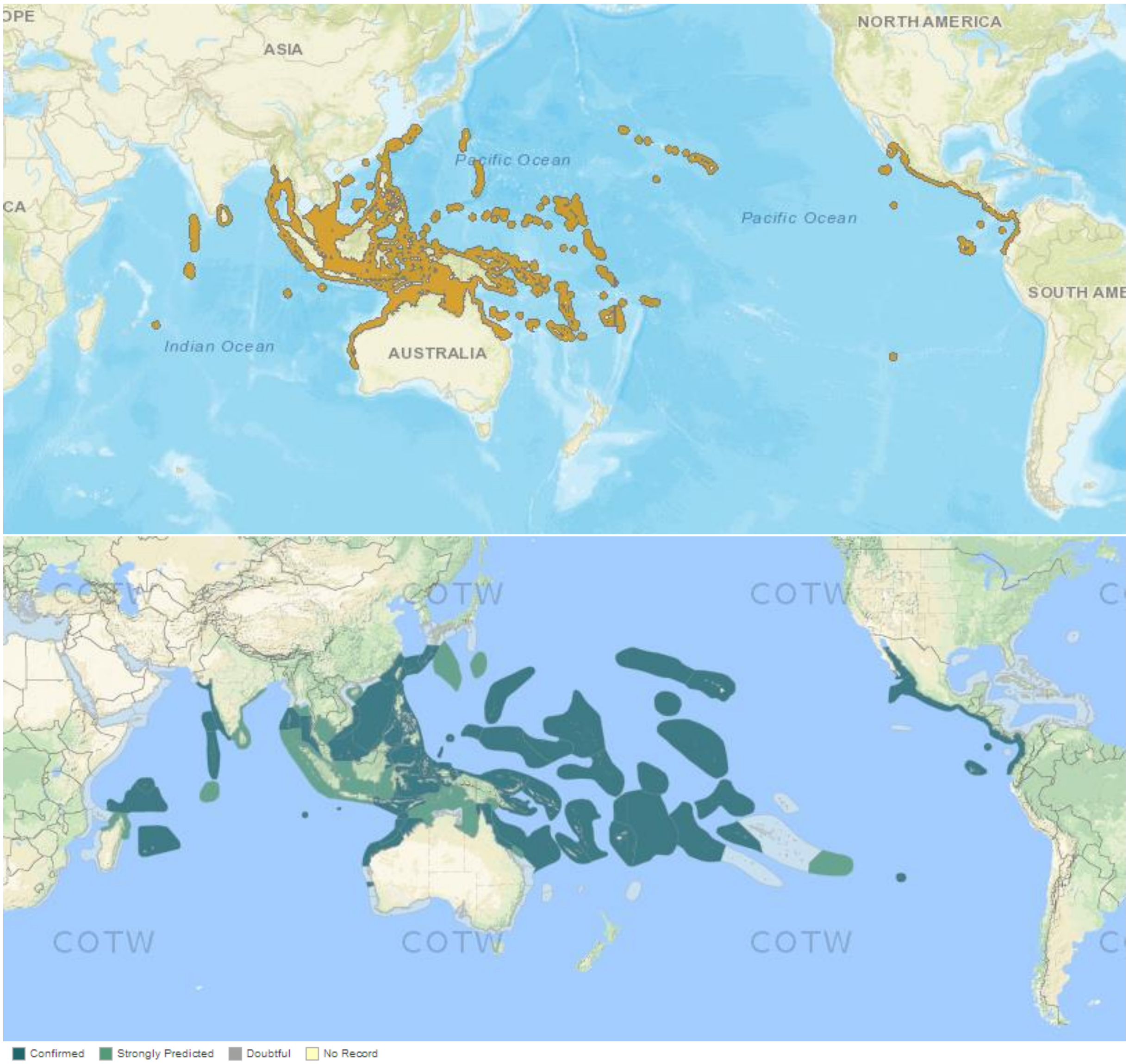


Figure 2 Potential global distribution of the cauliflower coral, *P. meandrina* from IUCN (top panel) http://www.iucnredlist.org/details/full/133095/0 (Hoeksema et al. 2014). Confirmed and predicted records of the species (bottom panel) from Corals of the World (Veron 2000).

3.5 Habitat

In Hawaii, the cauliflower coral, *P. meandrina*, is generally restricted to shallow reef areas that are exposed to relatively high wave energy and warmer water temperatures indicating a reltively high degree of physiological stress (Jokiel 1978). The depth range of the species is 1-27 meters (Hoeksema et al. 2014).

3.6 Abundance and minimum population estimate in Hawaii

Once dominant across Hawaiian coral reefs, today *P. meandrina* may be considered uncommon. The statewide average coverage of *P. meandrina* across 60 permanent transects in Hawaii was 1.5% in the last comprehensive coral reef assessment and monitoring survey conducted in 2012 (Rodgers et al. 2015 p. 5). The only coral species with smaller percent cover was the uncommon coral *Montipora flabellata* with approximately 0.7% cover statewide (Rodgers et al. 2015 p. 5). Percentage cover of *P. meandrina* was 1.89% at shallow stations while only 1.13% at deeper stations (Rodgers et al. 2015 p. 7).

3.7 Current population trend in Hawaii

Coral cover of *P. meandrina* has significantly declined over the last decades. From 1999 to 2012, *P. meandrina* experienced approximately a –36.1 % decline in percentage cover across 60 permanent reef stations in Hawaii to a statewide cover of ~1.5% (Rodgers et al. 2015). Average coral cover of *P. meandrina* across sites of West Hawaii was 0.92% in 2003, 0.84% in 2007, and 0.80% in 2011 (Walsh et al. 2013 p. 89,91,93). Change in coral cover for this species has been variable across Main Hawaiian Islands (Rodgers et al. 2015).

Overall the mean statewide coral cover in Hawaii remained stable from 1999 to 2012 based on data from 60 permanent reef stations collected by the Coral Reef Assessment and Monitoring Program (CRAMP), but variations in coral cover trends for specific areas were observed (Rodgers et al. 2015). For example, coral cover across the Northern Hawaiian Islands was in average 19.9 % with no significant declines between 2000 and 2006 (Friedlander et al. 2008). Higher latitude and relatively cooler water temperature kept these reefs from mass bleaching events during this time (Rodgers et al. 2015). Approximately 40 % of Maui's coral reef stations showed a significant coral cover decline, while 67% of stations at Hawaii Island slightly increase in coral cover (Rodgers et al. 2015).

Several areas showed coral decline from 1999 to 2012 (Brown et al. 2004; Rodgers et al. 2015). During this period, significant coral cover decline was documented at several stations across Maui (Honolua Bay, Mā'alaea Bay, and Papaula Point), mostly associated with crown-of-thorns outbreaks (in 2005), and anthropogenic impacts such as land based pollution (e.g., nutrification), sedimentation, and overfishing (Walsh et al. 2010; Rodgers et al. 2015). In fact, drastic coral cover decline at these sites were so severe that may have induced a total coral reef ecosystem collapse (Walsh et al. 2010). Specifically, coral decline in West Maui has been linked to runoff of land-based nutrients (Smith et al. 2005; Dailer et al. 2010) and low herbivore populations (Walsh et al. 2010). Coral decline at Molokai (from Kamalo to Kamiloloa) has been associated with terrestrial sediment input and coastal sediment transport (Prouty et al. 2010) from improper land management, overgrazing of feral ungulates over the past decades, and dredging in the 1970s (Roberts & Field 2007). However, stations with chronic and recurrent stressors showed much slower coral recovery (e.g., sedimentation and nutrification at Mā'alaea and Honolua Bay, Maui) than stations with sporadic acute stressors (e.g., crown-of-thorns outbreaks at Kanahena Point, Maui) (Rodgers et al. 2015). Localized coral decline in 2002, likely due to physical damage, has also been documented in heavily visited shallow reefs at Hanauma Bay in Oahu, which receives more than 1 million visitors a year (Rodgers et al. 2015). Coral cover also significantly declined at six out of seven monitored sites in northwestern reef of the Hawaii

Island from 2003 to 2011, likely due to a strong winter storm in 2004 and major sediment runoff in 2006 and 2012 (Walsh et al. 2013).

Coral cover significant declined across Hawaii in 2010-2016 (McCoy et al. 2017). Total average of coral cover has declined across the Hawaiian archipelago from 2010 to 2016 by 40-50%, especially in fore reef and lagoon habitats of the Northern Hawaiian Islands such as French Frigate Shoals (55% to 30%), Kure Atoll (15% to 8%), Lisianski Island (37% to 17%), Pearl & Hermes Reef (10% to <5%); as well as the Main Hawaiian Islands such as Hawaii Island (27% to 12%), Kauai (8% to 2.5%), Maui (19% to 11%), Niihau (2.4% to <1%), and Oahu (10% to 6%) (McCoy et al. 2017).

Similarly, total coral cover decline from 2010 to 2016 in islands of American Samoa such as Ofu and Olosega Islands (26% to 18%) (McCoy et al. 2017). During the same period, catastrophic coral cover declines were documented in the Pacific Remote Islands Areas (PRIA) such as Jarvis Islands (24% to <1%) (McCoy et al. 2017). However, no significant change in coral cover during the same period was observed in Lanaii Island and Molokai Island of the Main Hawaiian Islands or in Rose Atoll, Tau Island, and Tutuila Island of American Samoa (McCoy et al. 2017).

4. The cauliflower coral (*P. meandrina*) qualifies as a species under the ESA

The cauliflower coral (*P. meandrina*) qualifies as species under the ESA because is a taxonomic distinct unit (Flot et al. 2008 p. 1; Pinzón et al. 2013 p. 1603; Schmidt-Roach et al. 2014).

5. The cauliflower coral (*P. meandrina*) is threatened or endangered within a significant portion of its range, Hawaii

NMFS should interpret the phrase "significant portion if its range" as a portion of a species range that is biologically significant based on the principles of conservation biology using the concepts of redundancy, resiliency, and representation (the three Rs) (Shaffer & Stein 2000). These concepts can also be analyzed in terms of four population viability characteristics commonly used by NMFS: abundance, spatial distribution, productivity, and diversity of the species. The cauliflower coral is threatened or endangered in a significant portion of its range (Hawaii) because the population across Hawaii faces high extinction risk now or in the foreseeable future. Hawaii is a significant portion of the range of this species because *P. meandrina* has low abundance, occupies the northernmost region of the species geographical distribution, and may be ecologically and genetically isolated from other populations in the Pacific basin. Threats that increase extinction risk are discusses further below.

5.1.1 The cauliflower coral exhibit low abundance across Hawaii

The cauliflower coral is endangered or threatened across a significant portion of its range (Hawaii) based on its low abundance. The abundance of *P. meandrina* across Hawaii was previously discussed (see above). In summary, coral cover of *P. meandrina* is relatively low with a statewide mean of ~ 1.5% (Rodgers et al. 2015). In some areas, mean coral cover for the species is less than 1 %. For example, the average coral cover of *P. meandrina* across sites of

West Hawaii was 0.80% in 2011 (Walsh et al. 2013 p. 93). Given the low abundance of the species across Hawaii, *P. meandrina* faces high extinction risk in a significant portion of its range.

5.1.2 Ecologically isolated from other populations of the Pacific basin

The cauliflower coral in Hawaii may be ecologically isolated from other populations of the Pacific basin due to dispersal limitations of the planulae larvae. As such, Hawaii's cauliflower coral population may show a restricted spatial distribution that could be considered significant.

Dispersal is crucial in the ability of reef-building corals to colonize new space and recover from disturbance. The best scientific evidence shows that *P. meandrina* is a broadcast spawner coral in Hawaii (Fiene-Severns 1998; Riddle 2008), in the Great Barrier Reef (Schmidt-Roach et al. 2012), and likely in French Polynesia (Magalon et al. 2005 p. 2). Unlike most broadcast spawning coral species, gametes of *P. meandrina* are not packaged in egg-sperm bundles, but sperm and eggs are released separated (Schmidt-Roach et al. 2012 p. 2). Eggs are relatively small (~80 µm in diameter), negatively buoyant, and contain algae symbionts (Schmidt-Roach et al. 2012 p. 2). However, nothing is known about the larval biology of this species, such as duration and dispersal capabilities (Schmidt-Roach et al. 2012 p. 3). Genetic subdivision among populations of other *Pocillopora* species on relatively small spatial scales suggests that dispersal of sexual larvae may be limited (Schmidt-Roach et al. 2012 p. 5). It is likely that the small planulae larva of *P. meandrina* may be settling locally, limiting the geographical dispersion across other regions of the Pacific basin. In fact, new scientific evidence based on a molecular technique (e.g., gel-based restriction fragment length polymorphism method) suggests that *P. meandrina* may have a northern range limit to the south of Pearl and Hermes (Johnston et al. 2018 p. 8). If limited distribution exist within the Hawaiian Islands, it is likely that the Hawaiian cauliflower coral population may also be ecological isolated from other regions of the Pacific, making Hawaii a significant portion of the species range.

5.1.3 Genetically distinct from other populations outside Hawaii

The cauliflower coral in Hawaii may be also genetically distinct from other populations across the Pacific and thus Hawaii could be considered a significant portion of the species range. The best available science supports the hypothesis that pocilloporid corals are genetically isolated within regions of the Indo-Pacific (Combosch et al. 2008 p. 1307; Pinzón et al. 2013 p. 1599). For example, Combosch et al. (2008) showed that the eastern Pacific pocilloporids were genetically isolated from other regions of the Indo-Pacific by the "Eastern Pacific Barrier". The coral biological review team determined that at least one eastern Pacific pocilloporid species, *Pocillopora elegans*, was a distinct species from corals identified as *P. elegans* in the central and western Pacific and reported extinction risk as separated species (Brainard et al. 2011). Gene flow in *P. meandrina* has been shown to be restricted at a large geographical scale in the South Pacific (Magalon et al. 2005 p. 5). *P. meandrina* show genetic differentiation between populations at large geographic scale of thousands of kilometers likely due to large-scale bleaching events (Magalon et al. 2005 p. 1). For example, genetic differentiation between populations of *P. meandrina* have been detected between Tonga and the Society Islands in the South Pacific (Magalon et al. 2005 p. 3). Although, there are not genetic studies differentiating cauliflower corals of Hawaii from the rest of the Pacific basin, this is a possibility based on evidence for other Pocilloporid species.

6. Threats to the species and factors for listing

Under the ESA, a species must be listed if it is in danger of extinction or threatened by possible extinction in all or a significant portion of its range now or in the foreseeable future. 16 U.S.C. § 1533(a)(1). In making this determination, the agency must rely "solely on the best scientific and commercial data available" and analyze the species' status in light of five statutory listing factors:

1. the present or threatened destruction, modification, or curtailment of its habitat or range;
2. overutilization for commercial, recreational, scientific or educational purposes;
3. disease and predation;
4. the inadequacy of existing regulatory mechanisms; and
5. other natural or manmade factors affecting its continued existence.

16 U.S.C. §§ 1533(a)(1)(A)-(E); 50C.F.R. §§ 424.11(c)(1)-(5).

The cauliflower coral (*P. meandrina*) in Hawaii is threatened by at least four of the five listing factors: present modification of its habitat, disease and predators, the inadequacy of existing regulatory mechanisms, and other natural or manmade factors.

6.1 Present or threatened destruction, modification, or curtailment of its habitat or range

Human pressures within the Archipelago are concentrated in the Main Hawaiian Islands, which face challenges relating to land-based pollution, coastal development, tourism, reef fisheries, and the aquarium trade (Friedlander et al. 2008). Sediment runoff from agricultural and coastal development is a major problem for the coral reef ecosystems of the Main Islands, that combine with overfishing created ideal conditions for the proliferation of native as well as invasive algae populations (Jokiel et al. 2004; Friedlander et al. 2008). The commercial aquarium trade is now the region's biggest reef fishery (Friedlander et al. 2008; Nadon et al. 2015). Reef fisheries extract ~ 990,000 specimens annually, ~76% of which from the island of Hawaii (Friedlander et al. 2008). Seine net fisheries in the Main Islands report catch rate declines of 35% between 1966 and 2006 (Friedlander et al. 2008). Health of fish stocks varies among islands and is negatively correlated with human population density (Friedlander et al. 2008; Williams et al. 2008, 2015; Jouffray et al. 2015). For example, intense fishing pressures near the dense population center of Oahu have decimated apex predators and dramatically decreased overall reef fish biomass (Friedlander et al. 2008; Williams et al. 2008, 2015). Human pressure on the Northwest Hawaiian Island reefs is considerable less than on the populated Main Hawaiian Islands (Friedlander et al. 2008; Williams et al. 2008, 2015). As such reef fish assemblages in this more isolated section of the Archipelago are significantly healthier than in the Main Hawaiian Islands (Friedlander et al. 2008; Williams et al. 2008, 2015).

6.1.1 Land-based source pollution

Land-based source pollution across Hawaii is localized and can result in significant threats at local scales, especially in semi-enclose bays with slow water exchange. Sedimentation from improper land use practices is considered a major threat to corals in the Main Hawaiian Islands. Decline in coral cover has been directly associated with chronic and acute sedimentation during periods of high rainfall in areas such as Kawaihae (West Hawaii) (Walsh et al. 2013), Maalaea and Honolua Bay (Maui) (Rodgers et al. 2015). In the western Indian Ocean, *Pocillopora* has been identified as a sediment-intolerant genus (McClanahan & Obura 1997). Collapsed reefs in Costa Rica due to excess sedimentation were once dominated by pocilloporid corals (Rogers 1990). Additionally, *Pocillopora* have been found to be susceptible to mortality from freshwater inputs in Hawaii (e.g., Kaneohe bay, Oahu) (Jokiel et al. 1993) and elevated nutrients can cause physiological and reproductive negative effects (Koop et al. 2001; Cox & Ward 2002; Villanueva et al. 2006). Local land-based source pollution can act synergistically with global stressors such as warming and reduce resilience to bleaching (Carilli et al. 2009).

6.1.2 Sewage pollution

Sewage pollution negatively affects coral reefs in Hawaii, particularly in areas close to high human population densities (Yoshioka et al. 2016). Sewage pollution is a complex stressor that introduces diverse pollutants including toxic chemicals, heavy metal, nutrients (nitrogen and phosphorous), and microbial pathogens that negatively impact coral reefs (Wear & Thurber 2015). Pollutants may facilitate the emergence of diseases through nutrient enrichment and favoring the proliferation of microorganisms that are usually nutrient-limited in coral reefs (Bruno et al. 2003; Harvell et al. 2007; Vega Thurber et al. 2014).

In some areas of Hawaii, untreated sewage can enter coastal waters and directly affect corals. For example, in Puako, West Hawaii, raw sewage enter the coral reef ecosystem through old cesspool systems, porous volcanic bedrock, and submarine groundwater discharge (Knee et al. 2010). Combined with other stressors such as fishing, land-based pollution, and climate change, coral cover in the Puako region have drastically decline from 80% in the 1970s to 32% in 2010 (Walsh et al. 2013). Further coral decline has been documented in other studies and Puako is one of four regions most impacted in West Hawaii (Couch et al. 2014; Yoshioka et al. 2016). The dominant coral in the region (*Porites lobata*), exhibit a high prevalence (~14%) of growth anomalies, likely due to pollution (Yoshioka et al. 2016). Growth anomalies in *Porites* are among the most prevalent coral disease on Hawaii Island (Aeby et al. 2011; Couch et al. 2014). The high prevalence of diseases and low coral cover in this region has been linked to sewage input to the reef (Yoshioka et al. 2016).

6.1.3 Oxybenzone pollution from sunscreens

Sunscreens are widely used in Hawaii because tourism and are affecting corals. Oxybenzone is a chemical found in many sunscreen and other personal-care products sold in the United States designed to protect people from UV radiation. Oxybenzone and derivatives are emerging environmental contaminants of concern regularly found in wastewater effluents, rivers, lakes and coastal waters (Gago-Ferrero et al. 2011; Vidal-Dorsch et al. 2012; Agüera et al. 2013; Tovar-

Sánchez et al. 2013; Richardson & Ternes 2014; Rodríguez et al. 2015). Scientists have grown increasingly concerned that oxybenzone poses an extreme hazard to freshwater and marine wildlife, including corals (Eichenseher 2006; Coronado et al. 2008; Blitz & Norton 2008; Blüthgen et al. 2012; Downs et al. 2015; McCoshum et al. 2016).

Recent scientific research has shown that oxybenzone pollution is a substantial threat to corals, and may cause coral bleaching by promoting viral infections (Danovaro et al. 2008; Downs et al. 2015). Given the wide use of oxybenzone, scientists estimate that between 6,000-14,000 tons of sunscreen, containing up to 10 % of oxybenzone, enter coral reefs around the world every year (McCoshum et al. 2016). Snorkelers and divers visiting coral reefs are the primary source and each year put at least 40% of coastal coral reefs at risk (McCoshum et al. 2016). In many popular near-shore swimming areas, oxybenzone concentration can reach alarming levels (Tashiro & Kameda 2013; Bargar et al. 2015; Rodríguez et al. 2015; Bratkovics et al. 2015).

Oxybenzone harms corals, and the concentration of this pollutant at highly visited coastal areas is sufficiently high to kill coral larvae and affect coral tissue. Oxybenzone is insoluble in water and become highly concentrated in the water column, water surface, or in sediments when released from sunscreen lotions. Scientists have documented relatively high concentration of oxybenzone at popular swimming and snorkeling locations in the Hawaiian Islands. For example, oxybenzone levels in Hawaii popular beaches range from 0.8 to 19.2 µg/L (11,300 part per trillion (ppt) at Waikiki beach, 4,780 ppt at Waimea bay, and 568 ppt at Ko Olina cove).[2] High levels of oxybenzone pollution have also been found in popular snorkeling areas in the U.S. Virgin Islands (75µg/L to 1.4 mg/L), American Samoa and the Florida Keys (Bargar et al. 2015; Downs et al. 2015).

Previous studies have found that sunscreens at very low concentrations cause rapid coral bleaching by promoting viral infections (Danovaro et al. 2008). Therefore, sunscreens can potentially induce coral bleaching in coastal areas where tourism and the use of sun blocking lotions are relatively high (Danovaro et al. 2008). Recent studies have also documented genotoxic effects, and skeletal disruption in coral larvae and culture primary cells at concentrations far below these ambient levels with significant harm to corals (Downs et al. 2015). For example, coral larvae deformations are seen at oxybenzone concentrations as low as 6.5 µg/L, and the lethal concentration when 50 % of coral larvae are killed (LC_{50}) is as low as 139 µg/L (Downs et al. 2015). Studies have shown that oxybenzone affect the reproductive capacity of corals, turning them into "zombies" by making them sterile and impairing recruitment (Danovaro et al. 2008; Downs et al. 2015).

The combination of ongoing stressors with chemical pollution such as high concentration of oxybenzone in the water can act synergistically amplifying the negative effects of these stressors and further reducing reproductive capacity of corals. This is because coral larvae and juvenile corals (i.e., recruits) are more sensitive to the toxicological effects of pollution and negative stressors such as high temperature and ocean acidification than adult corals (Negri &

[2] *See,* Downs, Int'l Coral Reef Symposium; *see also*, Booth, H.S., M.M. Manning. 2016 The Sunscreen Sheen: An Assessment of the Presence and Quantity of Organic UV-Filters in the Waters off Waikiki Beach (Poster Presentation #40, ID #: 464) Int'l Coral Reef Symposium, Honolulu Hawaii.

Hoogenboom 2011; Kroeker et al. 2013). Oxybenzone from sunscreens is therefore, another threat that contributes to reproductive failure in corals and can be easily avoided.

In January 25, 2017, Hawaii's Senator Will Espero introduced a bill (SB1150 SD2 HD3)[3] that would ban the sale of sunscreens containing oxybenzone within the state of Hawaii.

6.1.4 Marine debris

Discarded monofilament fishing lines have been documented to cause death of *P. meandrina* at popular cast fishing sites in Oahu, Hawaii (Yoshikawa & Asoh 2004). For example, ~65 % of *P. meandrina* colonies out of 129 colonies surveyed at the Kaka'ako Waterfront State Recreation Area, Kewalo Honolulu in March of 1998 had fishing lines on their surface and 80% of these affected colonies showed partial mortality or were dead due to abrasion (Yoshikawa & Asoh 2004).

6.2 Over-utilization for commercial, recreational, scientific, or educational purposes

6.2.1 International trade

Due to the relatively small size (30 cm in diameter), the cauliflower coral, *P. meandrina*, is indiscriminately targeted by collectors for the aquarium trade (https://trade.cites.org/).

However, there is no evidence of overutilization of *P. meandrina* for commercial, recreational, scientific or educational purposes in Hawaii. Harvesting of pocilloporid corals do occur throughout the geographical range. For example, in the Eastern Tropical Pacific pocilloporid harvesting is a major threat along the continental coast and has eliminated pocilloporid corals from Acapulco (Mexico), Bahia Culebra (Costa Rica), Taboga Island (Panama), and in some areas of Ecuador (Glynn 2001).

Approximately 126 records of *P. meandrina* were listed on the CITES Trade Database (https://trade.cites.org/) from 1985 to 2016, mostly for trade. However, ~1,818 records of unidentified *Pocillopora* spp. were also listed indicating that the actual number of *P. meandrina* records could be higher. From the 126 records, ~31,425 corals (+ 17,663 kg) were reported as imports and ~48,366 corals (+ 29,965 kg) as exports (see Appendix). Approximately 43% of this trade was listed as live specimens. From these, the United States imported ~26,792 corals (+213 kg), but only exported 51 corals (see Appendix). It is unknown whether any export or import of *P. meandrina* were coming from or going to Hawaii.

6.3 Disease or predation

6.3.1 Disease

Diseases that induce tissue loss affecting colonies of *P. meandrina* have been documented in the east and west coast of Hawaii. Tissue loss generally progresses from one side of the colony with old algae-covered skeleton to denuded skeleton to sloughing and into apparently healthy tissue

[3] http://www.capitol.hawaii.gov/measure_indiv.aspx?billtype=SB&billnumber=1150&year=2017

(Walsh et al. 2010). This tissue loss appear to originate from the base of each branch as a clear branch that slowly progresses towards the top (Walsh et al. 2010). Coral senescence reaction has also been observed in *P. meandrina* (Walsh et al. 2010). However, *Pocillopora* has been identified as the genus with lower disease prevalence at least in the Northwestern Hawaiian Islands (Aeby 2009). Other pocilloporid corals have been documented with diseases. For example Pocilloporids on the Great Barrier Reef have been occasionally documented with common diseases (Willis et al. 2004). Pathogens such as *Vibrio coralliilyticus* have been found on bleached *P. damicornis* during thermal stress in Zanzibar, enhancing production of a lytic extracellular protease (Ben-Haim et al. 2003; Tout et al. 2015). In general, pocilloporid are not particularly vulnerable to most diseases known that affect corals, but chronic skeletal growth anomalies (eroding, hyperplasia and neoplasia), black-band disease, yellow-band disease, pink-line syndrome, and white-band disease have been occasionally documented (Brainard et al. 2011).

6.3.2 Predation

Pocillopora corals are commonly preyed on by several coralivorous taxa including fishes, crabs, gastropods, seastars, and urchins (Glynn 2003). Predation of *P. meandrina* by the corallivorous snail, *Drupella* spp., the asteroid *Culcita novaeguineae,* and the crown-of-thorns startfish, *Acanthaster planci*, has been documented in Hawaii (Glynn & Krupp 1986), especially after bleaching events (Kramer et al. 2016). *Pocillopora* coral species are among the most commonly preyed on by crown-of-thorns (Glynn 1985; Pratchett 2001). Crown-of-thorns sea stars usually target juvenile corals of *Pocillopora* because the lack of crustacean symbionts (e.g., a crab and a snapping shrimp) that defend the host coral colonies (Glynn 1976; Coles 1980). Predation by these sea stars can reduce recruitment success of *Pocillopora* populations. In addition, *Pocillopora* corals are also prey of other corallivorous gastropod *Jenneria pustulata* (Glynn 1976) and fishes (Cole et al. 2008; Jayewardene et al. 2009).

6.4 The inadequacy of existing regulatory mechanisms

Existing regulatory mechanisms are inadequate to address the principal threats to *P. meandrina*. Regulatory mechanisms addressing greenhouse gas emissions and impacts to this coral species from associated ocean warming and ocean acidification are woefully inadequate. Unless strong near-term emissions reductions are implemented in short order at the national and international levels, it is likely that *P. meandrina* will be committed to extinction. This section reviews regulatory mechanisms addressing greenhouse gas emissions and other threats to corals and coral reef ecosystems.

6.4.1 Regulatory mechanisms addressing greenhouse gas emissions, climate change, and ocean acidification are ineffective

Greenhouse gas emissions pose the primary threat to the continued existence of *P. meandrina* principally through impacts from ocean warming and acidification, and yet are among the least regulated threats. The best available science indicates that the current average atmospheric CO_2 concentration of ~405 ppm is already detrimental to hundreds of coral species, and that atmospheric CO_2 concentrations must be reduced to at most 350 ppm, and perhaps much lower

(300-325 ppm CO_2), to adequately reduce the synergistic threats of ocean warming, ocean acidification, and other impacts (Hoegh-Guldberg et al. 2007a; Donner & others 2009; Veron et al. 2009; Frieler et al. 2013). To reach a 350 ppm CO_2 target, the United States and developed countries must achieve or exceed the upper end of the reduction range of 25% to 40% below 1990 levels by 2020 (IPCC 2013, 2014). A target that is realistically impossible under the current CO_2 trajectory and near future predictions of 408 ppm in 2017 on average (Betts et al. 2016). In fact, we are already committed to levels of ocean warming and acidification that will certainly harm corals and other marine organism in coral reefs (Donner & others 2009; Matthews & Weaver 2010; Frieler et al. 2013). Regulatory mechanisms at the national and international level do not adequately address the impacts from climate change and ocean acidification to the petitioned coral species, nor require the greenhouse gas emissions reductions necessary to protect the petitioned coral species from further decline and future extinction.

6.4.2 United States climate initiatives are ineffective

Regulatory mechanisms in the United States are inadequate to effectively address climate change (73 FR R 28287-28288). While existing laws including the Clean Air Act, Energy Policy and Conservation Act, Clean Water Act, Endangered Species Act, and others provide authority to executive branch agencies to require greenhouse gas emissions reductions from virtually all major sources in the U.S., the federal government is currently not implementing these legal mechanisms. While full implementation of these environmental laws, particularly the Clean Air Act, would provide an effective and comprehensive greenhouse gas reduction strategy, due to their non-implementation, existing regulatory mechanisms must be considered inadequate to protect the petitioned coral species from climate change and ocean acidification.

In the absence of federal leadership, state and local governments have taken the lead in measures to reduce greenhouse gas emissions. While certainly a step in the right direction, unfortunately, these measures on their own are insufficient to prevent the extinction of the petitioned coral species. For example, the strongest law enacted to date is the California Global Warming Solutions Act of 2006 (Cal. Health and Safety Code § 38501(a)). This law it is the nation's first mandatory cap on a state's overall greenhouse gas emissions and requires the reduction of greenhouse gas emissions to 1990 levels by the year 2020 (Cal. Health and Safety Code § 38550). Although this law is a promising first step, it is grossly insufficient on its own to slow ocean warming and ocean acidification significantly enough to ensure the survival of imperiled corals.

6.4.3 International climate initiatives are also ineffective

The latest international regulatory mechanisms addressing greenhouse gas emissions are the United Nations Framework Convention on Climate Change and the Paris Agreement. The Paris Agreement's central objective is:

> "[T]o strengthen the global response to the threat of climate change by keeping a global temperature rise well below 2ºC above pre-industrial levels within this century, and to pursue efforts to limit the temperature increase even further to 1.5ºC. Additionally, the agreement aims to strengthen the ability of countries to deal with the impacts of climate

change. To reach these ambitious goals, appropriate financial flows, a new technology framework and an enhanced capacity building framework will be put in place, thus supporting action by developing countries and the most vulnerable countries, in line with their own national objectives. The Agreement also provides for enhanced transparency of action and support through a more robust transparency framework."
(Paris Agreement 2016).

The effectiveness of the Paris Agreement in reducing CO_2 emissions and keeping global temperature below 2ºC above pre-industrial levels is still too early to evaluate. Thus, this international regulatory mechanism must be considered as currently ineffective.

6.4.4 State regulations

In Hawaii, state law prohibits breaking or damaging stony corals (H.A.R. 13-95-70). The sale of stony corals native to Hawaii is also prohibited. *Id*. Hawaii law also imposes fines on vessels that run aground on coral reefs. *Id*. Nonetheless, this state law is unable to fully address key threats to coral reefs in Hawaii. Although the protections offered by Hawaii state law are important, they are inadequate to protect the cauliflower coral. ESA listing would afford greater protection from a variety of threats and greater deterrents from harming this coral.

Management actions at the local level have been established for several years in some areas across the Hawaiian Islands to reduce localized sedimentation, nutrients runoff and increase the abundance of herbivorous fishes. These stressors directly and indirectly contribute to coral decline via smothering and by blocking potential available space for coral recruitment. For example, the seasonal blooms of the invasive alga *Acanthofora spicifera* in bays of the west coast of Maui, which has shown to smoother corals, led to creation of the Kahekili Herbivore Fisheries Management Area where removal of herbivories fishes has been prohibited (Walsh et al. 2010). For example, most of the decline in coral cover at Papaula in Maui since 2009 coincided with a drastic increase of *A. spicifera* (Walsh et al. 2010).

State regulations are insufficient to protect this coral species.

6.4.5 Federal regulations

6.4.5.1 Federal acts, initiative and regulations are ineffective

The Coral Reef Conservation Act, passed in 2000, requires NMFS to develop a national coral reef action strategy, initiate a matching grants program for reef conservation, and create a conservation fund to encourage public–private partnerships (US Commission on Ocean Policy 2004). H.R. 5821, Coral Reef Conservation Act Reauthorization and Enhancement Amendments of 2016, was introduced in the US House of Representatives in July 14 2016 (See https://www.congress.gov/bill/114th-congress/house-bill/5821). However, the bill was referred to the Subcommittee on Environment on November 30, 2016 and has been stalled since then.

The Coral Reef Conservation Act can be viewed as Congress' acknowledgement that the government needs and is willing to fund assistance from outside organizations. While this effort funds research, mapping, and monitoring efforts, it is also inadequate to address the multifaceted threats facing corals and coral reefs. Congress has not (and will likely not) mandated any projects focus on bleaching, global warming or ocean acidification. The law does not provide any enforceable regulatory measures that will protect coral habitat or protect corals from direct human threats or the overarching threats of global warming and ocean acidification. Therefore, even if this Act passes it will be ineffective in protecting imperiled corals.

The US Coral Reef Task Force ("Task Force") was established in 1998 by Presidential Executive Order 13089, which mandates that federal agencies (1) use their programs and authorities to protect and enhance US coral reef ecosystems; and (2) to the extent permitted by law, ensure that any authorized, funded, or executed actions will not degrade the conditions of these ecosystems (Maurin and Bobbe 2009 (US Coral Reef Task Force Federal Member Coral Profiles)). The Task Force currently consists of 12 Federal agencies; 7 U.S. states, territories, commonwealths (Commonwealths of the Northern Mariana Islands and Puerto Rico, States of Florida and Hawaii, and the Territories of Guam, American Samoa, and the US Virgin Islands); and the three US Freely Associated States (Federated States of Micronesia, Republic of the Marshall Islands, and the Republic of Palau – all non-voting members). *Id*. Task Force responsibilities include (1) overseeing implementation of Executive Order 13089 and developing and implementing efforts to map and monitor US coral reefs; (2) researching reef decline and its solutions; (3) minimizing and mitigating coral reef degradation from pollution, overfishing, and other causes; and promoting international conservation and sustainable reef use. *Id*. The Task Force is an important step toward the conservation of corals and it has acknowledged the severity of the threats faced by coral reefs including glob al warming. However, the Task Force has not responded to threats by taking measures to prevent long term threats to corals from global warming and ocean acidification. Primarily, the Task Force has focused on research, monitoring, and reporting without needed action to protect corals. Moreover, the objectives set out by the Executive Order do not mandate any federal agency action because they are framed as creating a policy, which is simply a guiding principle or procedure and is not legally binding or enforceable (Exec. Ord. 13089 §§ 2 & 6). The Order explicitly denies the creation of any right, substantive or procedural, enforceable in law or equity by a party against the U.S., its agencies, and its officers. *Id*. § 6. Additionally, the ability of the Task Force to carry out its goals is limited by discretionary appropriations. *Id.* § 3.

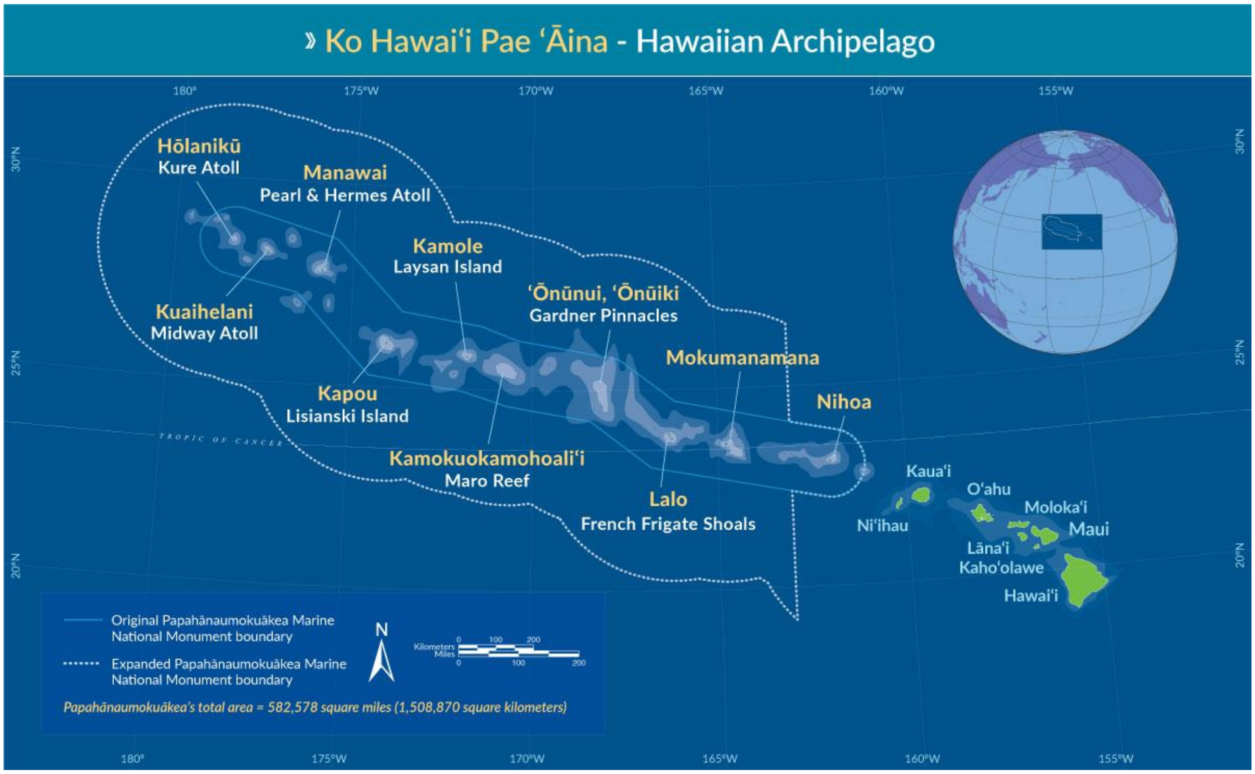
The United States Coral Reef Initiative (USCRI) of 1996. Created as a platform for U.S. support of national and international coral reef conservation efforts, the USCRI's goal "is to strengthen and fill the gaps in existing efforts to conserve and 165 sustainably manage coral reefs and related ecosystems (sea grass beds an d mangrove forests) in U.S. waters." The USCRI consists of federal, state, territorial and commonwealth governments, the nation's scientific community, the private sector, and other organizations. The National Oceanic and Atmospheric Administration (NOAA), one of the prime federal agency contributors to the USCRI, has worked to reduce pollution, create marine protected areas, educate import and export officials to identify corals, monitor and research coral reefs. The U.S. Coral Reef Initiative, whose achievements are primarily attributed to NOAA and its partners, has filled some of the gaps left open by inadequate state and congressional statutes in terms of coral reef monitoring and the ability to effect change within local communities across the nation. The USCRI and NOAA can locate bleaching events and measure their severity, but their role is merely one of reaction, not action.

The USCRI and NOAA have only enacted tools to chart the results of increasing greenhouse gases emissions and global warming. There exist no efforts to tackle the issue of how to reduce, mitigate, and adapt to global warming.

The US All-Islands Coral Reef Initiative is "a collaboration of marine resource managers from state, commonwealth, territorial agencies and freely associated states working together with federal agencies to conserve and protect coral reefs in the United States." (http://www.allislandscorals.org/). It was established in 1994 and includes governor-appointed representatives from American Samoa, Commonwealth of the Northern Marianas, Guam, Hawai'i, Puerto Rico, the U.S. Virgin Islands, and Florida, as well as Affiliate Members from the freely associated states of the Federated States of Micronesia, the Republic of the Marshall Islands, and the Republic of Palau. *Id.* Most of these efforts are focused on direct human threats to corals, and they do not have a comprehensive approach to addressing greenhouse gas emissions and ocean acidification.

The Marine Protection, Research, and Sanctuaries Act, which includes the **Ocean Dumping Act** and the **National Marine Sanctuaries Act (NMSA**), was passed in 1972. 33 U.S.C. §§ 1401 et seq. The Ocean Dumping Act seeks to regulate ocean discharge and limit or prevent the dumping of any material that would adversely affect (1) human health, welfare, or amenities; or (2) the ecological systems or economic potential of the marine environment (US Commission on Ocean Policy 2004a). The NMSA authorizes NMFS to designate marine sanctuaries and promulgate conservation and management regulations for those areas. *Id*. The NMSA includes a provision that allows NMFS to fund habitat restoration within sanctuaries, including coral reefs, with cost recovery from responsible parties. Recovered funds may be used to restore the damaged habitats or other habitats within national marine sanctuaries. *Id*. There are currently 13 sanctuaries managed under the National Marine Sanctuaries Program, at least five of which contain coral communities (US Commission on Ocean Policy 2004a). Coral research, monitoring, and management activities are conducted in these sanctuaries. *Id*. see also http://sanctuaries.noaa.gov/ . The NMSA has no provisions for projects designed to prevent physical or long-term chronic damages to reefs from global warming, ocean acidification, nutrient overloading, or diseases (US Commission on Ocean Policy 2004a). The continued loss of corals in marine sanctuaries indicates that the designations alone are not sufficient to arrest the decline and encourage the recovery of species. The designation of a sanctuary and its boundaries does not lessen the key threats such as bleaching and impaired calcification. Bleaching will occur whether or not a reef is within a sanctuary. Thus, while the designation of marine sanctuaries and other marine protected areas is crucial to prevent some forms of direct human damage (e.g., overfishing), yet the designation cannot protect corals from larger long-term global threats.

US National Marine Monuments are also managed under the NMFS National Marine Sanctuaries Program. Some activities are prohibited in National Monuments thus affording limited protections to corals. The Northwestern Hawaiian Islands were designated a Marine National Monument in 2006 (71 FR 36443), amended in February 28, 2007 and named the part of the Papahānaumokuākea Marine National Monument. On August 26 of 2016, President Obama substantially expanded the monument to a total area of 1,508,870 square kilometers, nearly the size of the Gulf of Mexico, (see Map). Although the protection of the Northwestern Hawaiian Island may help fish assemblages it does not protect corals from the main drivers of decline, climate change. Other Marine National Monuments that contain coral reefs include the Marianas Trench National Monument in the Northern Marianas Islands (74 FR 1557, Jan. 6, 2009), the Pacific Remote Islands Marine National Monument (74 FR 1565, Jan. 6, 2009), and Rose Atoll Marine National Monument in American Samoa (74 FR 1577, Jan. 6, 2009). Similar to marine sanctuaries, the regulatory mechanisms within National Monuments are also inadequate to protect imperiled corals.



Map. Expansion of the Papahānaumokuākea Marine National Monument.

Executive Order 13,158 created an advisory committee for coordinating and strengthening a coordinated system of Marine Protected Areas ("MPAs"). Corals within MPAs benefit from the management of uses with in the designated areas. There are 207 Marine Protected Areas ("MPAs") encompassing coral reef ecosystems within the United States (Waddell & Clarke 2008). Approximately 76% of these MPAs are multiple use areas allowing some level of resource extraction throughout the entire site, and almost one quarter of them were established for the explicit purpose of continued extraction (Waddell & Clarke 2008). Take of marine resources is prohibited in part or all of 49 of these MPAs. Approximately 86% of these MPAs are permanent, whereas protections are only temporarily provided at 14% of the sites (Waddell & Clarke 2008). Management plans have been approved for only 42 MPAs, illustrating the challenge of long-term plan development (Waddell & Clarke 2008). Enforcement, funding, management capacity, monitoring, and public support have been identified as key problems at a majority of the MPAs (Waddell & Clarke 2008). From all these MPAs, ~45% support some sort of ongoing monitoring activity, and some level of enforcement effort is reported at 74% of the sites (Waddell & Clarke 2008). Like the regulatory mechanisms for marine sanctuaries, and national monuments other marine protected areas are good at protecting reefs from some of the direct human threats, but are inadequate to address the global threats posed by climate change and ocean acidification. Additionally, these various forms of protecting marine areas are all managed differently and most do not completely remove direct human impacts, for example many designated areas still permit various forms of fishing that can adversely impact coral reefs.

There are other U.S. laws that could be brought to bear to further protect corals, however, thus far the implementation of these laws does not provide much protection for imperiled corals. In large part, these laws have not yet been employed to the benefit of corals. Even if fully utilized they would still provide only small protection for corals. Moreover, they cannot sufficiently address the key long-term threats to corals. Instead these laws provide a patchwork of environmental laws that are important, but inadequate to provide a safety net for corals. These laws include:

- **Magnuson–Stevens Fishery Conservation and Management Act**, established sovereign federal rights to all fishery resources within 200 miles of US shores. 16 U.S.C. §§ 1801 et seq.
- **The Coastal Zone Management Act** of 1972 (16 U.S.C. §§ 1451 et seq.) provides technical assistance and financial incentives for sustainable state management of coastal areas (US Commission on Ocean Policy 2004b).
- **The Lacey Act** of 1900 prohibits trade of wildlife (including all invertebrates) that is illegally harvested, possessed, transported, or sold. 16 USC §§ 3371-3378
- **The National Environmental Policy Act** (42 U.S.C. §§ 4321 et seq.), which requires the federal government to thoroughly analyze the environmental impacts of any federal action that could significantly affect the environment, including those in coral reef habitat.
- **The Clean Water Act** (33 U.S.C. §§ 1251 et seq.), which regulates the discharge of dredged or fill materials into U.S. waters.
- **The Sikes Act** (16 U.S.C. § 670), which requires the U.S. Department of Defense to provide for conservation and rehabilitation of natural resources on military installations, which in some locations include corals.

All these federal laws have powerful tools that Hawaii can use to protect corals, however, corals are rarely considered under these laws during analysis of potential harming activities

6.4.6 Foreign and international regulations

The International Coral Reef Initiative ("ICRI"), established in 1994, is a voluntary informal network consisting of several countries, including the US, as well as US and international non-governmental organizations. The ICRI has built important international partnerships, has raised international public awareness of the coral reef crisis, and has called on nations to reduce greenhouse gas emissions, but it lacks the ability to mandate change in this sector. The Global Coral Reef Monitoring Network, one of the operating units of the ICRI, aims to develop and support consistent regional ecological and socioeconomic monitoring networks for coral reefs and to disseminate the results of monitoring efforts at local, regional, and global scales. See http://www.gcrmn.org/about.aspx .

The Convention on International Trade in Endangered Species of Wild Fauna and Florida, regulate trade of corals. Several coral species, including *P. meandrina* are listed on Appendix II of CITES. However, the collection of these species for que aquarium trade is unregulated with no quotas or size limits and requires regulation under fisheries management (Hoeksema et al. 2014). As a result, thousands of live corals have been traded unregulated among countries over the past decades. Furthermore, CITES listing can only provide protection against global trade in imperiled species and does not regulate the other threats facing corals.

Other international agreements could potentially provide added protection to corals, however, since they are aimed generally at protecting the environment or ocean resources they have not been fully applied to the protection of coral reefs. For example, **the Convention on Biological Diversity** and **United Nations Convention on Law of the Sea** both have general provisions that aim for environmental conservation and therefore could affect the protection of coral reefs. However, international treaties are rarely binding and have not thus far afforded coral reefs needed protections.

In sum, the various state, federal, and international regulatory mechanisms are inadequate to protect corals. These measures are important for the conservation of coral reefs and can effectively reduce some of the direct human threats to corals. Nonetheless, these regulatory mechanisms form a patchwork of approaches and offer only small protection to corals. Meanwhile, the overarching threats to the corals from bleaching due to increased ocean temperatures and ocean acidification are largely unaddressed. The existing protections for corals either do not address ocean warming and acidification, or they only provide research and monitoring of those impacts.

Protection under the ESA for this imperiled coral species will provide comprehensive protections for which no other regulatory mechanisms can substitute. The threats facing this coral species are particularly troublesome because of their interrelated nature. The effects of these threats are synergistic, indicating that addressing each threat independently will not be sufficient to conserve this species.

6.5 Other natural or manmade factors affecting its continued existence

6.5.1 Ocean warming and acidification

Ocean warming and acidification is directly related to the increase in atmospheric carbon dioxide (CO_2) emissions globally. Atmospheric CO_2 concentrations reached average annual levels of over 402.9 parts per million (ppm) globally in 2016 (Blunden & Arndt 2017) which is higher than at any point during the last 800,000 years (Lüthi et al. 2008). Over the past 200 years, the global oceans have absorbed approximately 25 % of the anthropogenic CO_2 released to the atmosphere (IPCC 2014). Approximately 2.6 Giga tones of CO_2 per year (i.e., 26% of total emissions) entered the global oceans in the last decade (Le Quéré et al. 2016).

Atmospheric CO_2 emissions also contribute to warming of the oceans (IPCC 2013). The year 2016 surpassed 2015 as the warmest year in 137 years of recordkeeping since 1880 (Blunden & Arndt 2017). The average global temperature across land and ocean surfaces in 2016 was +0.94

ºC above the 20th century average (NCEI 2017). The annual average global temperature for the oceans was the warmest on record at +0.75 ºC in 2016 for the third consecutive year since 2014. Since 1955, the global oceans have absorbed over 90% of the excess heat trapped by greenhouse gas emissions (Levitus et al. 2012).

As the global oceans uptake the excess of CO_2, seawater chemistry profoundly changes and the oceans become more acidic (Doney et al. 2009). The average pH of the global surface ocean has already decreased by 0.1 units (from 8.2 to 8.1 pH units), which represents 30 % increase acidity and 10% decrease in carbonate ion concentration in comparison with pre-industrial levels (Feely et al. 2004; Doney et al. 2009). Changes in ocean chemistry are unprecedented in the geological record with acidification rates faster than in the past ~300 million years, a period that includes three major mass extinctions, and 96% of marine species extinct (Hönisch et al. 2012). Anthropogenic CO_2 emissions will further reduce surface ocean pH by 0.3 to 0.5 units on average by 2100, and regional changes may be even more severe (Caldeira & Wickett 2005; McNeil & Matear 2006). At an atmospheric CO_2 level of 560 ppm, pH would drop 0.24 units to ~7.9 and most ocean surface waters would be adversely undersaturated with respect to aragonite (Veron et al. 2009). If CO_2 levels reach 788 ppm, ocean pH would drop 0.3 or 0.4 units mounting to a 100–150% change in acidity, and tropical surface concentrations of carbonate would decline by 45% (Orr et al. 2005; Meehl et al. 2007).

Reduction of carbonate ion concentration due to ocean acidification also reduces calcium carbonate saturation state globally (Jiang et al. 2015). Reduction of calcium carbonate saturation is already affecting entire coral reef ecosystem, particularly impacting marine species that rely on calcium carbonate to produce shells and skeletons (Andersson et al. 2011). Aragonite saturation states, the main forms of calcium carbonate that is used by scleractinean corals has already declined globally and it is predicted to further decline by the end of the century (Cao & Caldeira 2008).

Anthropogenic climate change is directly threatening reef-building scleractinean corals by: 1) mass coral mortality associated with coral bleaching due to increasingly frequent thermal stress events and 2) decreased calcification rates and reproduction success due to increasing atmospheric carbon dioxide that causes ocean acidification by decreasing sea water pH and aragonite saturation state (Hoegh-Guldberg et al. 2007a; Carpenter et al. 2008).

6.5.1.1 Ocean warming kills corals

Ocean warming has caused large-scale coral bleaching events resulting in mas coral mortality and associated reef decline regionally and globally in the past decades with increasing frequency and severity (Hoegh-Guldberg et al. 2007a; Pandolfi et al. 2011; Hughes et al. 2017a), and is projected to continue (Hoeke et al. 2011). The longest global bleaching event on record, which currently continues, started in 2014, and has affected more coral reefs than any previous worldwide bleaching event (Eakin 2017). For example, the Australian Great Barrier Reef recently experienced a catastrophic mass bleaching event, where more than 90% of its 2,300 km reef tract was considerable affected and over 30% of corals subsequently died (Hughes et al. 2017a). This mass bleaching events have equally affected degraded and pristine reefs (i.e., no or minimal human impacts), the latter were thought to be resistant to bleaching (Hughes et al.

2017a). Recurrent thermal stress events on corals appear to affect pristine, protected, and degraded reef equally (Hughes et al. 2017a).

Coral reefs around Hawaii have experienced several thermal stress events leading to localized coral bleaching in 1996 (Main Hawaiian Islands), 2002 and 2004 (Northwest Islands), and widespread bleaching in 2014, 2015, and 2016 (Jokiel & Brown 2004; Kenyon & Brainard 2006; Rodgers et al. 2015, 2017; Bahr et al. 2015; Kramer et al. 2016). The most affected coral genera during all these bleaching events have been *Pocillopora, Porites,* and *Montipora*.

Sea surface temperatures across coastal waters of Hawaii have increased +1.15ºC over the past six decades (Jokiel & Brown 2004). The consecutive bleaching event of 2014 and 2015 has been unprecedented in scale and magnitude (Fig. 4). Coral mortality due to bleaching across the Hawaiian archipelago is predicted to increase due to higher incidence of thermal stress events in the future (Rodgers et al. 2015), especially in the northern end of the archipelago (Hoeke et al. 2011). Relatively higher incidence and severity of coral bleaching due to thermal stress events has been already observed in the northern Hawaiian atolls (Kenyon et al. 2006).

Studies of projected changes to coral growth and mortality, based on modeling of current emission scenarios, predict that it is extremely unlikely that viable coral populations will exist in shallow waters of the Hawaiian archipelago by 2100, if corals are not able to increase their thermal tolerance to future levels of heat stress (Hoeke et al. 2011). In fact, Hoeke et al. (2011) predicted precipitous declines in coral cover in the northern Hawaiian region between 2030 and 2050, while steady decline over the century in the southern region. The capacity of some coral communities to obtain limited thermal tolerance to near future heat stress could be through selection of more thermally tolerant algal endosymbionts (Baker et al. 2004), succession of more resistant or resilient corals species (Grottoli et al. 2014), or a combination of both. The ability of coral communities to adapt to future thermal stress, in particular in Hawaii, is unknown and is subject to debate (Hoegh-Guldberg 2012). Today, there is little evidence that corals can adapt or acclimate to the current and projected frequency and magnitude of environmental change (Baskett et al. 2009; Rodolfo-Metalpa et al. 2014; Edmunds et al. 2014). In fact, recent evidence of widespread coral bleaching in areas that have experienced recurrent mas bleaching indicates that the capacity of coral communities to withstand increasing frequent and more severe thermal stress is very limited (Hughes et al. 2017a). The severity of such mass bleaching events already exceeds the ability of coral species and reefs to recover from future thermal stress (Eakin et al. 2010).

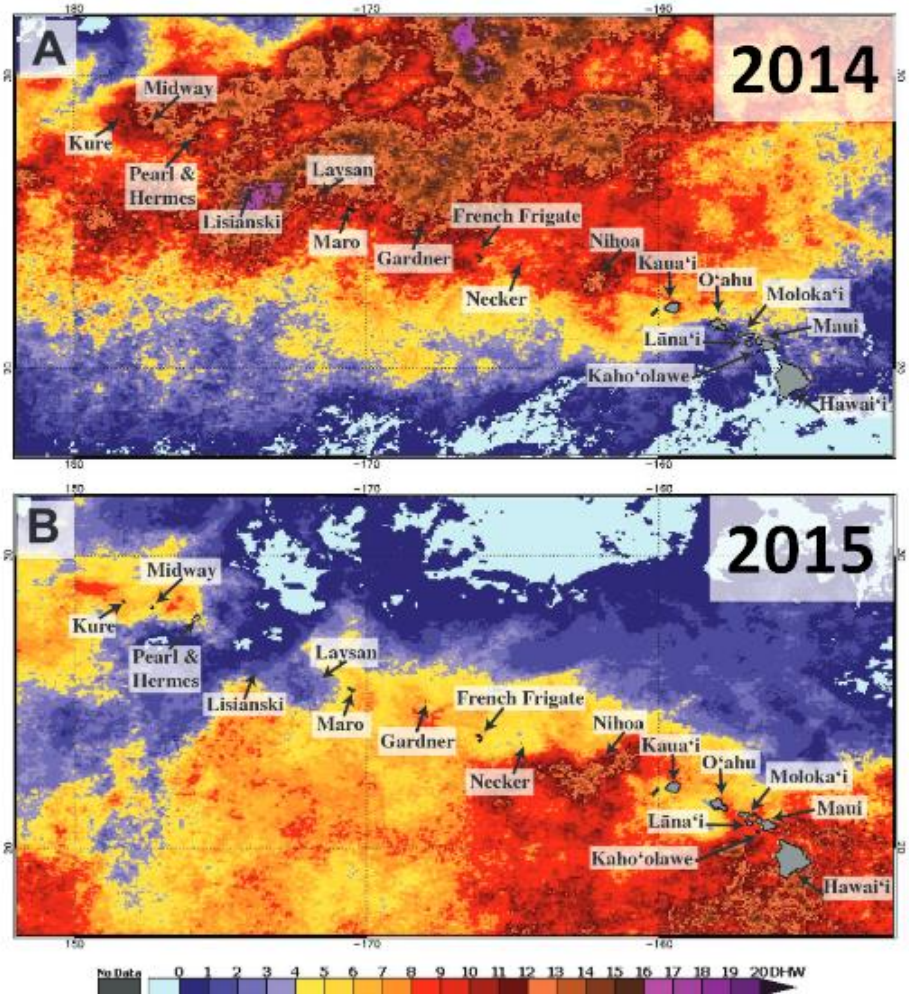


Figure 3. Maximum degree heating weeks for the Hawaiian archipelago in 2014 and 2015. Degree heating weeks (DHW) show how much heat stress has accumulated in a specific area over the past 12 weeks (3 months) by adding up any temperature exceeding the bleaching threshold during that time period. When DHW reaches 4°C-weeks, significant coral bleaching is likely, especially in more sensitive species. When DHW is 8°C-weeks or higher, widespread bleaching and mortality from thermal stress may occur. Note that heat stress affected more the Northern Hawaiian Islands in 2014 while the Main Hawaiian Islands in 2015. *Figure after Couch et al. (2016).*

Several species within the *Pocillopora* genus are recognized as being susceptible to bleaching during thermal stress. For example, field experiments in the Great Barrier Reef have shown that *Pocillopora damiconis* responds as severely as acroporid corals to increasing temperature (Marshall & Baird 2000). Similarly, *Pocillopora* spp. were among the most sensitive coral species to bleaching after *Acropora* and branching *Porites* in the western Indian Ocean (McClanahan et al. 2007). In the first reported bleaching events due to warming in the 1980s, pocilloporid corals suffered high mortality in the eastern tropical Pacific, ranging from 51% at Caño Island, 76-85% in Panama, to 97-100% in the Galapago Islands (Glynn 1983, 1990; Glynn et al. 1988). Controlled laboratory experiments have also shown the highly susceptibility of

Pocillopora corals to thermal stress (Glynn & D'Croz 1990; Hueerkamp et al. 2001; D'Croz & Maté 2004). Bleaching in corals can lead to mortality or physiological stress that lower fecundity and increase susceptibility to diseases (Ward et al. 2002; Bruno et al. 2007; Muller et al. 2008).

The cauliflower coral *P. meandrina* is particularly vulnerable to bleaching due to thermal stress. During the 2014 mass bleaching event ~ 20% of the colonies show signs of bleaching across the entire Hawaiian archipelago (Couch et al. 2016 p. 20). In some regions such as the east coast of Lisianski Island *P. meandrina* disappeared, from 1.05 % cover in 2014 to 0 % cover in 2015 (Couch et al. 2016 p. 34). During the fall (August to November) of 2015, an extreme and prolonged thermal stress event occurred within coastal waters of West Hawaii, where water temperatures exceed 30ºC (Eakin 2017). This event led to a mass coral bleaching event that resulted in catastrophic coral mortality for reefs of West Hawaii, with approximately half of the coral cover loss from 2014 (Kramer et al. 2016). Survey results from 24 permanent monitoring sites from South Kona to North Kohala indicated that mean coral bleaching prevalence during the thermal stress event was ~53% (45 - 66% range), resulting in an average of ~50% coral mortality between 2014 and 2016 (Kramer et al. 2016). During this event, *P. meandrina* showed major bleaching prevalence and subsequently catastrophic mortality across the leeward reefs of the Hawaii Island (Kramer et al. 2016). Severe post-bleaching mortality was detected for *P. meandrina* after the bleaching event in this area. Approximately, ~96% of colonies suffered partial post-bleaching mortality (> 5% tissue loss) and ~78% of colonies showed total post-bleaching mortality (=100% tissue) (Kramer et al. 2016). Total post-bleaching mortality further increased to ~90% of the colonies affected in Kealakekua Bay by May of 2016 (Kramer et al. 2016).

The 2015 thermal stress also affected *P. meandrina* in other areas of the Hawaiian Islands. Approximately, 45% of corals in the Hanauma Bay Nature Preserve in Oahu showed signs of bleaching in October of 2015, with a subsequent mortality rate of 9.8% (Rodgers et al. 2017). In this area, *P. meandrina* was one of two coral species with the highest bleaching prevalence with more ~ 64% of the colonies affected followed by a 1.3% mortality rate (Rodgers et al. 2017). Bleaching prevalence was highly variable in space due to localized environmental gradients such as warm water accumulation (Rodgers et al. 2017).

6.5.1.1 Ocean acidification affects corals

Ocean acidification in response to increasing atmospheric carbon dioxide (CO_2) emission is well recognized to reduce growth and survivorship in corals and associated species. Laboratory and field experiments have shown that as the pCO_2 of seawater increases, calcium carbonate production considerably declines slowing the growth, calcification rates, and survival of adult and juvenile corals (Langdon et al. 2003; Jokiel et al. 2008; Albright et al. 2010a; Gray et al. 2012; Castillo et al. 2014; Comeau et al. 2015). Similarly, ocean acidification also reduces calcification rate and reproduction success of crustose coralline algae (Kuffner et al. 2008; Diaz-Pulido et al. 2012), a crucial reef group that facilitate coral settlement and contribute to coral reef accretion (Diaz-Pulido et al. 2009; Webster et al. 2013; Fabricius & De'ath 2014). Thus, by affecting crustose coralline algae, ocean acidification would reduce settlement rates on corals (Webster et al. 2013; Fabricius et al. 2017). As ocean acidification becomes more severe, reef-building corals and crustose coralline algae will continue to show reduced calcification rates and reef accretion will be further reduced (Pandolfi et al. 2011; Andersson & Gledhill 2013; Albright et al. 2016).

Corals are already experiencing lower calcification rates linked to ocean acidification in the Pacific and Caribbean (Cooper et al. 2008; De'ath et al. 2009; Bak et al. 2009; Fabricius et al. 2011). Experimental and field studies show that the impact ocean acidification on calcification rates varies among coral species (Edmunds 2012; Comeau et al. 2013). A recent experimental study rearing corals under warming (2 ºC above normal maximum summer temperature) and ocean acidification (high pCO_2 twice present-day) conditions showed that corals of Hawaii reduced calcification rates (in *Porites compressa*, *Pocillopora damicornis*, *Fungia scutaria,* and *Montipora capitata*) in the high-temperature treatment (Bahr et al. 2016). The synergistic effect of warming water and ocean acidification caused mortality in at least one coral species, *P. compressa* (Bahr et al. 2016). This study indicates that biological response to temperature and pCO_2 elevation is species specific.

Studies have shown that calcification of entire coral reef ecosystem at different spatial scales is already declining in response to changes in water chemistry (De'ath et al. 2009; Comeau et al. 2015; Albright et al. 2016). For example, reduced calcification is already being detected across coral reef of the Great Barrier Reefs (De'ath et al. 2009). On a global scale, models show that most reefs are already calcifying 20-40% slower today compared with their pre-industrial rates, and that 30% of the world's coral reefs have decreased their gross calcification by 60-80% compared with pre-industrial rates (Silverman et al. 2009). Calcification rates of reef-building corals are predicted to decrease 40% further with increasing pCO_2 by the end of the century (Kleypas & Langdon 2006; Hofmann et al. 2010; Erez et al. 2011; Gattuso et al. 2015).

Ocean acidification also disrupts reproduction and compromises recruitment success of several studied coral species (Albright et al. 2010a; Albright 2011c; Doropoulos et al. 2012a; Chan & Connolly 2013), affecting the capacity of reef ecosystems to recover (Anthony et al. 2011b). Because ocean acidification effects have been quantified for relatively few coral species, the impact on *P. meandrina* should be inferred from congeneric or confamilial species, as previously done during the listing process of 82 candidate corals (Brainard et al. 2013).

Pocilloporid corals are affected by ocean acidification. Experimental studies of two-week incubation of *P. meandrina*, in which water pH was reduced to 7.8, as forecasted for the year 2100, show that calcification rate decreased by more than 50% at 27ºC, although no change was observed at higher temperature (29ºC) (Muehllehner & Edmunds 2008). Larvae of *P. damicornis* incubated for 6 hours at high CO_2 concentrations (950 µatm) and elevated water temperatures (30°C) showed depressed metabolic rates which could affect dispersal, settlement and development (Rivest & Hofmann 2014). Growth rates of *P. elegans* and *P. damicornis* in the eastern Pacific has declined 25%-3-% over the past 30 years due to ocean acidification, and extension rate varies in species along a calcium carbonate saturation state gradient (Manzello 2010). These studies demonstrate that the effects of ocean acidification are temperature dependent. New studies are needed to determine how the effects of ocean acidification in shallow coastal waters varies with light intensity, temperature and daily pH fluctuations, and increased nutrients and organic matter inputs (Jokiel et al. 2016).

The response of corals to ocean acidification varies among coral species. However, on average changes in aragonite saturation state of one unit result in a 20% decrease in calcification rate (Kleypas & Langdon 2006). Studies have demonstrated a strong positive correlation between aragonite saturation and coral calcification (Andersson et al. 2011). Optimal coral growth occurs when aragonite saturation state is greater than 4.0; levels of 3.0 to 3.5 are marginal or low; and 3.3 is generally considered the critical threshold for coral growth, below which considerably decrease in calcification rates occurs and reef accretion shifts to dissolution (Kleypas & Langdon 2006; Hoegh-Guldberg et al. 2007b). Field studies confirm that reef accretion approaches to zero or becomes negative at aragonite saturation below 3.3 when atmospheric CO_2 approaches 480 ppm (Kleypas & Langdon 2006). This is confirmed by observations that reefs with net accretion today under approximately 400 ppm in atmospheric CO_2 are restricted to waters where aragonite saturation exceeds 3.3 (Hoegh-Guldberg et al. 2007b). In sum, the best available science shows that aragonite saturation state below 3.5 is sub-optimal for coral growth and corals cease growth and begin dissolution below 3.3.

Coral reefs in Hawaii are already affected by ocean acidification. The aragonite saturation state in surface waters surrounding Hawaii may have declined more than - 0.5 units since the industrial revolution (Cao & Caldeira 2008) (Fig. 4). Prior to the industrial revolution, most coral reefs (98.4%) worldwide were found near open ocean waters with aragonite saturation state above 3.5 (Cao & Caldeira 2008). Today, Hawaii's surface waters already experience sub-optimal conditions. By mid-century models show that coral reef would be within waters with carbonate saturation below 3.0 and 2.75, a point at which coral growth and accretion is substantially reduced (Cao & Caldeira 2008) (Fig. 4). At atmospheric CO_2 concentration of 450 to 500 ppm, predicted by mid-century, reef erosion will exceed reef accretion (Hoegh-Guldberg et al. 2007a), changing the structure of many coral reefs and increasing the extinction risk of vulnerable coral species. At CO_2 concentration of over 500 pm coral reef ecosystem could disappear as we know them (Hoegh-Guldberg et al. 2007a; Hughes et al. 2017b). Such near future prediction for the middle of the century (only 33 years from now) will have tremendous implications for Hawaii coral reefs.

Ocean acidification conditions observed in Hawaii reefs already affect calcification and physiology of important reefs species such as corals and crustose coralline algae. A long term (10 months) mesocosm experiment conducted in Kaneohe Bay, Oahu, with simulations of pCO_2 expected for this century showed that crustose coralline algae cover can decrease by 86% with 250% lower rate of calcification under acidified conditions (Jokiel et al. 2008). Similarly, coral calcification of *Montipora capitata* declined 15% to 20% under acidified conditions (Jokiel et al. 2008). Because crustose coralline algae play an important role in coral reef ecosystems enhancing coral recruitment and promoting reef accretion (Bak 1976; Webster et al. 2013), a change in their abundance will have negative effects on the entire ecosystem.

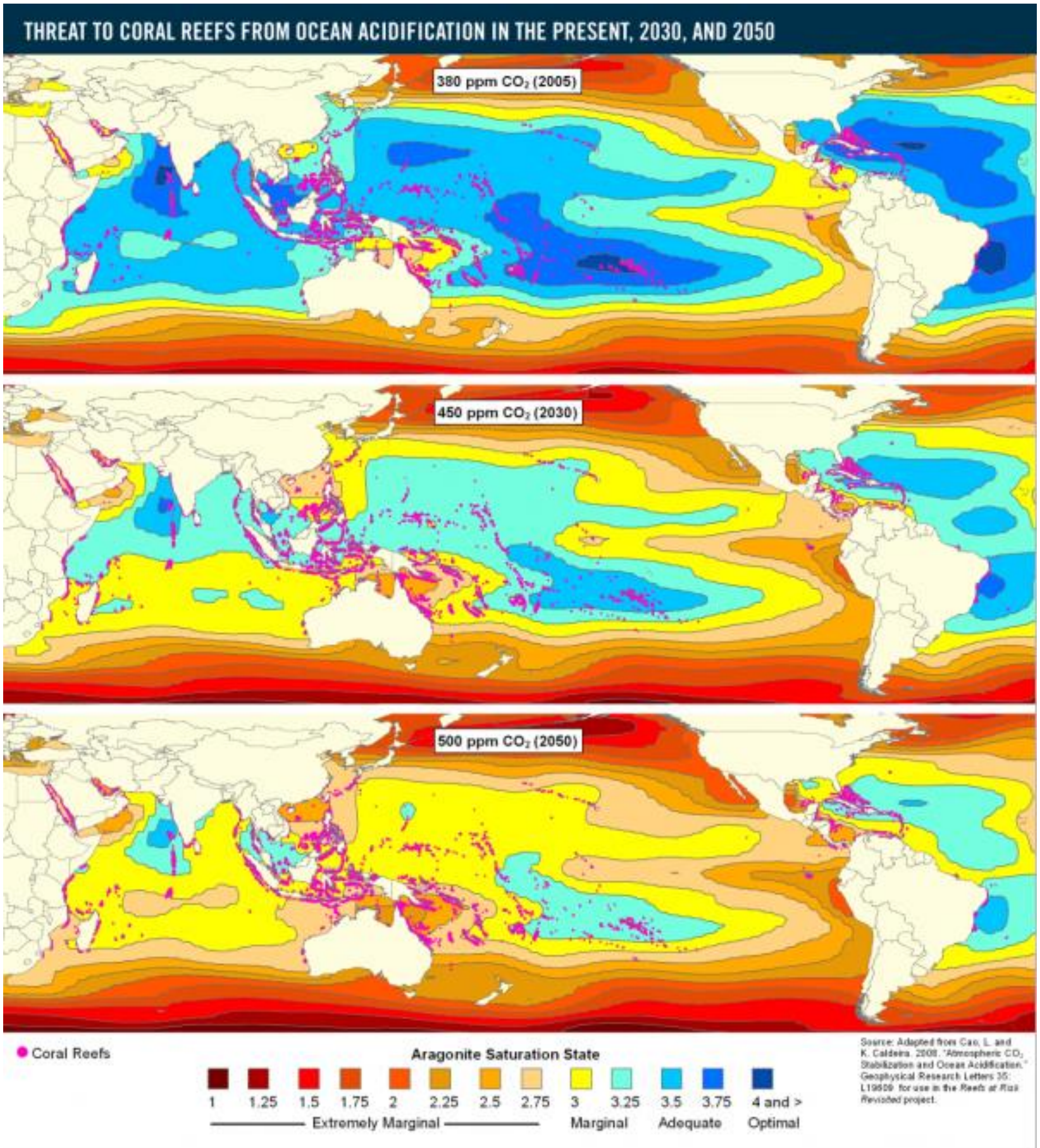


Figure 4. Sea surface aragonite saturation state for 2005, 2030, and 2050. Note that at 500 ppm of CO_2 predicted for 2050, aragonite saturation state around Hawaii waters is mostly marginal (3.0) and extremely marginal (2.75). Figure after Cao and Caldeira (2008)

Ocean acidification also disrupts other physiological processes in corals such as photosynthesis and reproduction. Some coral species can maintain calcification rates under acidified conditions by spending relatively high amount of energy and compromising other biological functions such as nutrient assimilation. For example, under low pH combined with high temperature, the scleractinian coral *Stylophora pistillata* can significantly decrease inorganic phosphorous and nitrogen uptake which are important elements to maintain metabolic functions (Godinot et al.

2011). A laboratory experiment on the effects of acidification on symbiotic algae in *Porites australiensis,* found that acidified seawater significantly decrease fluorescence yield of the coral and thus affecting calcification (Iguchi et al. 2012). In *Acropora millepora,* acidified conditions lead to the loss of more than 50% of their symbiotic algae decreasing both photosynthesis and respiration rates (Kaniewska et al. 2012). Thus, ocean acidification can cause major cellular and physiological impacts on reef building corals.

Ocean acidification affects all early life history stages of corals. In the threatened Caribbean elkhorn coral *Acropora palmata*, low pH conditions affect three fundamental stages needed for successful coral recruitment: (*i*) larval availability by reducing fertilization, (*ii*) reducing settlement success, and (*iii*) impeding postsettlement growth (Albright et al. 2010b; Albright 2011a, 2011b). The compounding effect of these impacts translates to 52-73% reduction in the number of larval settlers on the reef and elevated rates of postsettlement mortality (Albright et al. 2010b). Other coral species also show similar responses to low pH. For example, recruits of the mustard coral *Porites astreoides*, are strongly affected by acidic conditions (Albright & Langdon 2011). In this study, larval metabolism declined by 27 % and 63 % under acidification levels expected by mid-century and end-of-century, respectively. Settlement was also reduced up to 45% and 60% at the mid and end-of-century acidification levels, respectively (Albright & Langdon 2011). Similarly, a study of *Acropora digitifera* larvae showed that even when fertilization is successful under low pH conditions metamorphosis can be halted (Nakamura et al. 2011). In fact, under low pH conditions coral larvae lose their preference for settlement on crustose coralline algae, which is normal conditions enhance settlement (Doropoulos et al. 2012b).

6.5.2 Intensification of storms

Coral reefs of Hawaii periodically experience intense and short term powerful storms, and *P. meandrina* has been affected with temporal declines after acute storm events (Coles & Brown 2007). Wave disturbance have been recognized as important factor in structuring coral assemblages throughout the Hawaiian Islands (Grigg 1983, 1994). Although coral reefs are adapted to withstand acute disturbance events, hurricanes has contributed to coral reef decline in shallow areas of some regions, e.g., Caribbean (Gardner et al. 2005; Crabbe et al. 2008; Edwards et al. 2011). Strong storms physically damage corals by breaking off colonies, severing branches, and increasing acute sedimentation and pollution flux from rivers and land-based runoff during and after storms that result in widespread mortality (Gardner et al. 2005). Immediate mortality to colonies and fragments from severe storms is often high, and damaged populations are more susceptible to subsequent disturbances (Dollar 1982).

The number of strong hurricanes (category 4 and 5) has steadily increased over the past four decades across all ocean basins, but specially in the North Pacific, Indian and Southwest Pacific Oceans (Webster et al. 2005; Holland & Bruyère 2014). Furthermore, future projections indicate that ocean warming will cause an increase in the average intensity of tropical cyclones towards stronger storms by the end of the century (Knutson et al. 2010; Manganello et al. 2014). Thus, stronger storms are a potential threat to shallow coral reefs in Hawaii. Corals that are under stress from ocean warming and acidification will be particularly vulnerable to storm incidence (Veron et al. 2009).

6.5.3 Combined impacts of stressors

Multiple stressors can affect corals simultaneously and interactivity. Ocean acidification and warming can lower coral reef resilience, impairing growth, and increasing mortality (Anthony et al. 2011a; Bozec & Mumby 2015). Under warming and acidification forecasted within this century, coral abundance is predicted to decrease further by more than 50% even with high herbivory and good water quality (Anthony et al. 2011a). If grazing is reduced algae may become dominant sooner under low pH conditions (Anthony et al. 2011a). While some corals are more sensitive to ocean acidification than others (Castillo et al. 2014), warming will exacerbate the impacts of acidification (Pandolfi et al. 2011; Putnam et al. 2012; Harvey et al. 2013). Ocean warming may also act synergistically with coral diseases (or increase disease prevalence) reducing health condition and survival (Bruno & Selig 2007). Other stressors such as pollution and sedimentation can further diminish the ability of corals to adjust to future climate change conditions (Biscéré et al. 2015). Growing evidence suggests that corals that suffer physiological stress from poor water quality (Carilli et al. 2009; De'ath & Fabricius 2010) are more sensitive to high water temperature. Thus, action must be taken to reduce local stressors and thus conserve corals by increasing the odds against acidification and warming which threaten coral reefs beyond the capacity for recovery.

7. Critical Habitat

The ESA mandates that, when NMFS lists a species as endangered or threatened, the agency generally must also concurrently designate critical habitat for that species. Section 4(a)(3)(A)(i) of the ESA states that, "to the maximum extent prudent and determinable," NMFS: shall, concurrently with making a determination . . . that a species is an endangered species or threatened species, designate any habitat of such species which is then considered to be critical habitat . . . . " (16 U.S.C. § 1533(a)(3)(A)(i) and § 1533(b)(6)(C))

The ESA defines the term "critical habitat" to mean: (i) The specific areas within the geographical area occupied by the species, at the time it is listed… , on which are found those physical or biological features (I) essential to the conservation of the species and (II) which may require special management considerations or protection; and (ii) Specific areas outside the geographical area occupied by the species at the time it is listed …, upon a determination by the Secretary that such areas are essential for the conservation of the species. (16 U.S.C. § 1532(5)(A).

The Center expects that NMFS will comply with this unambiguous mandate and designate critical habitat concurrently with the listing of the petitioned coral species. We believe that all current and historic areas occupied by this species meet the criteria for designation as critical habitat and must therefore be designated as such.

8. Requested designation and conclusion

The Center hereby requests that the National Marine Fisheries Service list the cauliflower coral (*P. meandrina*) as endangered or threatened in a significant portion of its range (Hawaii) under the ESA. NMFS must promptly make a positive 90-day finding on this Petition, initiate a status review, and expeditiously proceed toward listing and protecting this coral species.

Drastic reductions of global anthropogenic emissions are unlikely. Thus, coral reef managers should embrace strategies that maximize the little resilience of coral reef ecosystems to global climate change by reducing local and regional impacts (e.g., pollution, overfishing, and habitat degradation) on imperiled corals (West & Salm 2003; Donner et al. 2005; Hughes et al. 2010), slow down the widespread ecological shift to non-coral dominated reefs (Jouffray et al. 2015), and prevent the extinction of the most vulnerable coral species (Carpenter et al. 2008). Listing under the ESA provides a series of legal regulatory mechanisms that can facilitate coral resilience by reducing local impacts, promote active restoration efforts, and buy time while global carbon emissions are reduced.

We look forward to the official response as required by the ESA.

9. **References** (all references are included in a flash drive with a hard copy of this petition)

Aeby GS. 2009. Baseline levels of coral disease in the Northwestern Hawaiian Islands. Coral Health and Disease in the Pacific: Vision for Action:168.

Aeby GS, Williams GJ, Franklin EC, Kenyon J, Cox EF, Coles S, Work TM. 2011. Patterns of coral disease across the Hawaiian archipelago: relating disease to environment. PloS one **6**:e20370.

Agüera A, Bueno MJM, Fernández-Alba AR. 2013. New trends in the analytical determination of emerging contaminants and their transformation products in environmental waters. Environmental Science and Pollution Research **20**:3496–3515.

Albright R. 2011a. Reviewing the Effects of Ocean Acidification on Sexual Reproduction and Early Life History Stages of Reef-Building Corals. Journal of Marine Biology **2011**:1–14.

Albright R. 2011b. Effects of Ocean Acidification on Early Life History Stages of Caribbean Scleractinian Corals.

Albright R. 2011c. Reviewing the Effects of Ocean Acidification on Sexual Reproduction and Early Life History Stages of Reef-Building Corals, Reviewing the Effects of Ocean Acidification on Sexual Reproduction and Early Life History Stages of Reef-Building Corals. Journal of Marine Biology, Journal of Marine Biology **2011**, **2011**:e473615.

Albright R et al. 2016. Reversal of ocean acidification enhances net coral reef calcification. Nature **531**:362–365.

Albright R, Langdon C. 2011. Ocean acidification impacts multiple early life history processes of the Caribbean coral Porites astreoides. Global Change Biology **17**:2478–2487.

Albright R, Mason B, Miller M, Langdon C. 2010a. Ocean acidification compromises recruitment success of the threatened Caribbean coral Acropora palmata. Proceedings of the National Academy of Sciences **107**:20400–20404.

Albright R, Mason B, Miller M, Langdon C. 2010b. Ocean acidification compromises recruitment success of the threatened Caribbean coral Acropora palmata. Proceedings of the National Academy of Sciences **107**:20400–20404.

Andersson AJ, Gledhill D. 2013. Ocean acidification and coral reefs: effects on breakdown, dissolution, and net ecosystem calcification. Annual Review of Marine Science **5**:321–348.

Andersson AJ, Mackenzie FT, Gattuso J-P. 2011. Effects of ocean acidification on benthic processes, organisms, and ecosystems. Ocean acidification **8**:122–153.

Anthony KRN, Maynard J a., Diaz-Pulido G, Mumby PJ, Marshall P a., Cao L, Hoegh-Guldberg O. 2011a. Ocean acidification and warming will lower coral reef resilience. Global Change Biology **17**:1798–1808.

Anthony KRN, Maynard JA, Diaz-Pulido G, Mumby PJ, Marshall PA, Cao L, Hoegh-Guldberg O. 2011b. Ocean acidification and warming will lower coral reef resilience. Global Change Biology **17**:1798–1808.

Apprill A, Marlow HQ, Martindale MQ, Rappé MS. 2009. The onset of microbial associations in the coral Pocillopora meandrina. The ISME Journal **3**:685.

Bahr KD, Jokiel PL, Rodgers KS. 2015. The 2014 coral bleaching and freshwater flood events in Kāneʻohe Bay, Hawaiʻi. PeerJ **3**:e1136.

Bahr KD, Jokiel PL, Rodgers KS. 2016. Relative sensitivity of five Hawaiian coral species to high temperature under high-pCO2 conditions. Coral Reefs **35**:729–738.

Bahr KD, Jokiel PL, Rodgers KS. 2017. Seasonal and annual calcification rates of the Hawaiian reef coral, Montipora capitata, under present and future climate change scenarios. ICES Journal of Marine Science **74**:1083–1091.

Baird AH, Guest JR, Willis BL. 2009. Systematic and biogeographical patterns in the reproductive biology of scleractinian corals. Annual Review of Ecology, Evolution, and Systematics **40**:551–571.

Bak R. 1976. The growth of coral colonies and the importance of crustose coralline algae and burrowing sponges in relation with carbonate accumulation. Netherlands Journal of Sea Research **10**:285–337.

Bak RPM, Nieuwland G, Meesters EH. 2009. Coral Growth Rates Revisited after 31 Years: What is Causing Lower Extension Rates in Acropora Palmata? Bulletin of Marine Science **84**:287–294.

Baker AC, Starger CJ, McClanahan TR, Glynn PW. 2004. Coral reefs: Corals' adaptive response to climate change. Nature **430**:741–741.

Bargar TA, Alvarez DA, Garrison VH. 2015. Synthetic ultraviolet light filtering chemical contamination of coastal waters of Virgin Islands national park, St. John, US Virgin Islands. Marine Pollution Bulletin **101**:193–199.

Baskett ML, Gaines SD, Nisbet RM. 2009. Symbiont diversity may help coral reefs survive moderate climate change. Ecological Applications **19**:3–17.

Ben-Haim Y, Zicherman-Keren M, Rosenberg E. 2003. Temperature-regulated bleaching and lysis of the coral Pocillopora damicornis by the novel pathogen Vibrio coralliilyticus. Applied and Environmental Microbiology **69**:4236–4242.

Betts RA, Jones CD, Knight JR, Keeling RF, Kennedy JJ. 2016. El Nino and a record CO2 rise. Nature Climate Change **6**:806–810.

Biscéré T, Rodolfo-Metalpa R, Lorrain A, Chauvaud L, Thébault J, Clavier J, Houlbrèque F. 2015. Responses of Two Scleractinian Corals to Cobalt Pollution and Ocean Acidification. PLoS ONE **10**:e0122898.
Blitz JB, Norton SA. 2008. Possible environmental effects of sunscreen run-off. Journal of the American Academy of Dermatology **59**:898.
Blunden J, Arndt DS. 2017. State of the Climate in 2016. Bulletin of the American Meteorological Society **98**:Si–S277.
Blüthgen N, Zucchi S, Fent K. 2012. Effects of the UV filter benzophenone-3 (oxybenzone) at low concentrations in zebrafish (Danio rerio). Toxicology and applied pharmacology **263**:184–194.
Bozec Y-M, Mumby PJ. 2015. Synergistic impacts of global warming on the resilience of coral reefs. Philosophical Transactions of the Royal Society B: Biological Sciences **370**:20130267.
Brainard R, Birkeland C, Eakin CM, McElhany P, Miller MW, Patterson M, Piniak GA. 2011. Status review report of 82 candidate coral species petitioned under the US endangered species act. US Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center. Available from https://pdfs.semanticscholar.org/c710/846b9f0b4424376d69b4f6246b9114ed2283.pdf (accessed September 18, 2017).
Brainard RE, Weijerman M, Eakin C, McElhany P, Miller MW, Patterson M, Piniak GA, Dunlap MJ, Birkeland C. 2013. Incorporating climate and ocean change into extinction risk assessments for 82 coral species. Conservation Biology **27**:1169–1178.
Bratkovics S, Wirth E, Sapozhnikova Y, Pennington P, Sanger D. 2015. Baseline monitoring of organic sunscreen compounds along South Carolina's coastal marine environment. Marine Pollution Bulletin **101**:370–377.
Brown E. 2004. Reef coral populations: spatial and temporal differences observed on six reefs off west Maui. Available from http://scholarspace.manoa.hawaii.edu/bitstream/10125/11973/1/uhm_phd_4425_r.pdf (accessed September 30, 2017).
Brown EK, Cox E, Jokiel PL, Rodgers SK, Smith WR, Tissot BN, Coles SL, Hultquist J. 2004. Development of benthic sampling methods for the Coral Reef Assessment and Monitoring Program (CRAMP) in Hawai'i. Pacific Science **58**:145–158.
Bruno JF, Petes LE, Drew Harvell C, Hettinger A. 2003. Nutrient enrichment can increase the severity of coral diseases. Ecology Letters **6**:1056–1061.
Bruno JF, Selig ER. 2007. Regional decline of coral cover in the Indo-Pacific: timing, extent, and subregional comparisons. PLoS ONE **2**:e711.
Bruno JF, Selig ER, Casey KS, Page CA, Willis BL, Harvell CD, Sweatman H, Melendy AM. 2007. Thermal stress and coral cover as drivers of coral disease outbreaks. PLoS biology **5**:e124.
Cao L, Caldeira K. 2008. Atmospheric CO2 stabilization and ocean acidification. Geophysical Research Letters **35**:L19609. American Geophysical Union.
Carilli JE, Norris RD, Black BA, Walsh SM, McField M. 2009. Local Stressors Reduce Coral Resilience to Bleaching. PLoS ONE **4**:e6324.
Carpenter KE et al. 2008. One-third of reef-building corals face elevated extinction risk from climate change and local impacts. Science **321**:560–563.

Castillo KD, Ries JB, Bruno JF, Westfield IT. 2014. The reef-building coral Siderastrea siderea exhibits parabolic responses to ocean acidification and warming. Proceedings of the Royal Society of London B: Biological Sciences **281**:20141856.
Chan NCS, Connolly SR. 2013. Sensitivity of coral calcification to ocean acidification: a meta-analysis. Global Change Biology **19**:282–290.
Cole AJ, Pratchett MS, Jones GP. 2008. Diversity and functional importance of coral-feeding fishes on tropical coral reefs. Fish and Fisheries **9**:286–307.
Coles SL. 1980. Species diversity of decapods associated with living and dead reef coral Pocillopora meandrina. Marine Ecology Progress Series:281–291.
Coles SL, Brown EK. 2007. Twenty-five years of change in coral coverage on a hurricane impacted reef in Hawai ‘i: the importance of recruitment. Coral Reefs **26**:705–717.
Combosch DJ, Guzman HM, Schuhmacher H, Vollmer SV. 2008. Interspecific hybridization and restricted trans-Pacific gene flow in the Tropical Eastern Pacific Pocillopora. Molecular Ecology **17**:1304–1312.
Comeau S, Carpenter RC, Lantz CA, Edmunds PJ. 2015. Ocean acidification accelerates dissolution of experimental coral reef communities. Biogeosciences **12**:365–372.
Comeau S, Edmunds PJ, Spindel NB, Carpenter RC. 2013. The responses of eight coral reef calcifiers to increasing partial pressure of CO2 do not exhibit a tipping point. Limnology and Oceanography **58**:388–398.
Cooper TF, De’ath G, Fabricius KE, Lough JM. 2008. Declining coral calcification in massive Porites in two nearshore regions of the northern Great Barrier Reef. Glob. Change Biol. **14**:529–538.
Coronado M, De Haro H, Deng X, Rempel MA, Lavado R, Schlenk D. 2008. Estrogenic activity and reproductive effects of the UV-filter oxybenzone (2-hydroxy-4-methoxyphenyl-methanone) in fish. Aquatic Toxicology **90**:182–187.
Couch CS, Burns J, Kanoelani S, Gutlay T-N, Geiger E, Eakin M, Kosaki R. 2016. Causes and Consequences of Mass Bleaching in Papahānaumokuākea. Honolulu, HI. Available from https://nmspapahanaumokuakea.blob.core.windows.net/papahanaumokuakea-prod/media/archive/council/meetings/2017/couch_2016_icrs_pmnm_bleachingtalk.pdf.
Couch CS, Garriques JD, Barnett C, Preskitt L, Cotton S, Giddens J, Walsh W. 2014. Spatial and temporal patterns of coral health and disease along leeward Hawai’i Island. Coral Reefs **33**:693–704.
Cox EF, Ward S. 2002. Impact of elevated ammonium on reproduction in two Hawaiian scleractinian corals with different life history patterns. Marine Pollution Bulletin **44**:1230–1235.
Crabbe MJC, Martinez E, Garcia C, Chub J, Castro L, Guy J. 2008. Growth modelling indicates hurricanes and severe storms are linked to low coral recruitment in the Caribbean. Marine environmental research **65**:364–368.
Dailer ML, Knox RS, Smith JE, Napier M, Smith CM. 2010. Using δ 15 N values in algal tissue to map locations and potential sources of anthropogenic nutrient inputs on the island of Maui, Hawai ‘i, USA. Marine Pollution Bulletin **60**:655–671.
Danovaro R, Bongiorni L, Corinaldesi C, Giovannelli D, Damiani E, Astolfi P, Greci L, Pusceddu A. 2008. Sunscreens cause coral bleaching by promoting viral infections. Environmental health perspectives **116**:441.
Darling ES, Alvarez-Filip L, Oliver TA, McClanahan TR, Côté IM. 2012. Evaluating life-history strategies of reef corals from species traits. Ecology Letters **15**:1378–1386.

D'Croz L, Maté JL. 2004. Experimental responses to elevated water temperature in genotypes of the reef coral Pocillopora damicornis from upwelling and non-upwelling environments in Panama. Coral Reefs **23**:473–483.
De'ath G, Fabricius K. 2010. Water quality as a regional driver of coral biodiversity and macroalgae on the Great Barrier Reef. Ecological Applications **20**:840–850.
De'ath G, Lough JM, Fabricius KE. 2009. Declining Coral Calcification on the Great Barrier Reef. Science **323**:116–119.
Diaz-Pulido G, Anthony K, Kline DI, Dove S, Hoegh-Guldberg O. 2012. Interactions between ocean acidification and warming on the mortality and dissolution of coralline algae. Journal of Phycology **48**:32–39.
Diaz-Pulido G, Harii S, McCook LJ, Hoegh-Guldberg O. 2009. The impact of benthic algae on the settlement of a reef-building coral. Coral Reefs **29**:203–208.
Dollar SJ. 1982. Wave stress and coral community structure in Hawaii. Coral Reefs **1**:71–81.
Donner SD, others. 2009. Coping with commitment: projected thermal stress on coral reefs under different future scenarios. PLoS One **4**:e5712.
Donner SD, Skirving WJ, Little CM, Oppenheimer M, Hoegh-Guldberg OVE. 2005. Global assessment of coral bleaching and required rates of adaptation under climate change. Global Change Biology **11**:2251–2265.
Doropoulos C, Ward S, Diaz-Pulido G, Hoegh-Guldberg O, Mumby PJ. 2012a. Ocean acidification reduces coral recruitment by disrupting intimate larval-algal settlement interactions. Ecology Letters **15**:338–346.
Doropoulos C, Ward S, Diaz-Pulido G, Hoegh-Guldberg O, Mumby PJ. 2012b. Ocean acidification reduces coral recruitment by disrupting intimate larval-algal settlement interactions. Ecology letters:338–346.
Downs CA et al. 2015. Toxicopathological Effects of the Sunscreen UV Filter, Oxybenzone (Benzophenone-3), on Coral Planulae and Cultured Primary Cells and Its Environmental Contamination in Hawaii and the U.S. Virgin Islands. Archives of Environmental Contamination and Toxicology **70**:265–288.
Eakin CM et al. 2010. Caribbean Corals in Crisis: Record Thermal Stress, Bleaching, and Mortality in 2005. PLoS ONE **5**:e13969.
Eakin CM. 2017. Global Coral Bleaching 2014-2017: Status and an Appeal for Observations. NOAA Coral Reef Watch Program. Available from https://coralreefwatch.noaa.gov/satellite/analyses_guidance/global_coral_bleaching_2014-17_status.php.
Edmunds PJ. 2012. Effect of pCO2 on the growth, respiration, and photophysiology of massive Porites spp. in Moorea, French Polynesia. Marine biology **159**:2149–2160.
Edmunds PJ et al. 2014. Persistence and Change in Community Composition of Reef Corals through Present, Past, and Future Climates. PLoS ONE **9**:e107525.
Edwards HJ, Elliott IA, Eakin CM, Irikawa A, Madin JS, Mcfield M, Morgan JA, Van WOESIK R, Mumby PJ. 2011. How much time can herbivore protection buy for coral reefs under realistic regimes of hurricanes and coral bleaching? Global Change Biology **17**:2033–2048.
Eichenseher T. 2006. The cloudy side of sunscreens. Environmental science & technology **40**:1377.
Erez J, Reynaud S, Silverman J, Schneider K, Allemand D. 2011. Coral calcification under ocean acidification and global change. Pages 151–176 Coral reefs: an ecosystem in transition.

Springer. Available from http://link.springer.com/10.1007/978-94-007-0114-4_10 (accessed October 1, 2017).
Fabricius K, De'ath G. 2014. Environmental factors associated with the spatial distribution of crustose coralline algae on the Great Barrier Reef. Coral Reefs **19**:303–309.
Fabricius KE, Langdon C, Uthicke S, Humphrey C, Noonan S, De'ath G, Okazaki R, Muehllehner N, Glas MS, Lough JM. 2011. Losers and winners in coral reefs acclimatized to elevated carbon dioxide concentrations. Nature Climate Change **1**:165–169. Nature Publishing Group.
Fabricius KE, Noonan SHC, Abrego D, Harrington L, De'ath G. 2017. Low recruitment due to altered settlement substrata as primary constraint for coral communities under ocean acidification. Proc. R. Soc. B **284**:20171536.
Fiene-Severns P. 1998. A note on synchronous spawning in the reef coral Pocillopora meandrina at Molokini Islet, Hawai ʻi. Reproduction in Reef Corals Results of the 1997 Edwin W. Pauley Summer Program in Marine Biology **4**:22–24.
Flot J-F, Magalon H, Cruaud C, Couloux A, Tillier S. 2008. Patterns of genetic structure among Hawaiian corals of the genus Pocillopora yield clusters of individuals that are compatible with morphology. Comptes Rendus Biologies **331**:239–247.
Friedlander A, Aeby G, others. 2008. The state of coral reef ecosystems of the main Hawaiian Islands. The state of coral reef ecosystems of the United States and Pacific freely associated states,. Available from http://ccmaserver.nos.noaa.gov/ecosystems/coralreef/coral_report_2005/MHI_Ch9_C.pdf (accessed October 23, 2015).
Frieler K, Meinshausen M, Golly A, Mengel M, Lebek K, Donner SD, Hoegh-Guldberg O. 2013. Limiting global warming to 2 °C is unlikely to save most coral reefs. Nature Climate Change **3**:165–170.
Gago-Ferrero P, Díaz-Cruz MS, Barceló D. 2011. Occurrence of multiclass UV filters in treated sewage sludge from wastewater treatment plants. Chemosphere **84**:1158–1165.
Gardner TA, Cote IM, Gill JA, Grant A, Watkinson AR. 2005. Hurricanes and Caribbean coral reefs: impacts, recovery patterns, and role in long-term decline. Ecology **86**:174–184.
Gattuso J-P et al. 2015. Contrasting futures for ocean and society from different anthropogenic CO2 emissions scenarios. Science **349**:aac4722-aac4722.
Glynn PW. 1976. Some physical and biological determinants of coral community structure in the eastern Pacific. Ecological monographs **46**:431–456.
Glynn PW. 1983. Extensive "bleaching"and death of reef corals on the Pacific coast of Panama. Environmental Conservation **10**:149–154.
Glynn PW. 1985. El Nino-associated disturbance to coral reefs and post disturbance mortality by Acanthaster planci. Mar. Ecol. Prog. Ser **26**:4.
Glynn PW. 1990. Coral mortality and disturbances to coral reefs in the tropical eastern Pacific. Elsevier oceanography series **52**:55–126.
Glynn PW. 2000. El Nino-Southern oscillation mass mortalities of reef corals: a model of high temperature marine extinctions? Geological Society, London, Special Publications **178**:117–133.
Glynn PW. 2001. Eastern Pacific coral reefs: new revelations in the twentieth century. Atoll Research Bulletin:119–160.
Glynn PW. 2003. Coral communities and coral reefs of Ecuador. Latin American coral reefs. Elsevier, Amsterdam:449–472.

Glynn PW, Colley SB. 2008. Survival of brooding and broadcasting reef corals following large scale disturbances: is there any hope for broadcasting species during global warming. Pages 1–5 Proceedings of 11th international coral reef symposium, Ft. Lauderdale (session 11). Available from http://nsuworks.nova.edu/cgi/viewcontent.cgi?filename=72&article=1000&context=occ_icrs&type=additional (accessed September 27, 2017).
Glynn PW, Cortés-Núñez J, Guzmán-Espinal HM, Richmond RH. 1988. El Niño (1982-83) associated coral mortality and relationship to sea surface temperature deviations in the tropical eastern Pacific. Mortalidad de corales asociada con El Niño (1982-83) y relación con las desviaciones de la temperatura superficial del mar en el Pacífico oriental tropical. Pages 237–243 Proceedings of the 6th International Coral Reef Symposium, Australia. Available from http://www.sidalc.net/cgi-bin/wxis.exe/?IsisScript=oet.xis&method=post&formato=2&cantidad=1&expresion=mfn=002207 (accessed September 28, 2017).
Glynn PW, D'Croz L. 1990. Experimental evidence for high temperature stress as the cause of El Nino-coincident coral mortality. Coral Reefs **8**:181–191.
Glynn PW, Krupp DA. 1986. Feeding biology of a Hawaiian sea star corallivore, Culcita novaeguineae Muller & Troschel. Journal of Experimental Marine Biology and Ecology **96**:75–96.
Godinot C, Houlbrèque F, Grover R, Ferrier-Pagès C. 2011. Coral Uptake of Inorganic Phosphorus and Nitrogen Negatively Affected by Simultaneous Changes in Temperature and pH. PLoS ONE **6**:e25024.
Gray SEC, DeGrandpre MD, Langdon C, Corredor JE. 2012. Short-term and seasonal pH,pCO2and saturation state variability in a coral-reef ecosystem. Global Biogeochemical Cycles **26**:GB3012.
Grigg RW. 1983. Community structure, succession and development of coral reefs in Hawaii. Marine Ecology Progress Series:1–14.
Grigg RW. 1994. Community structure, succession and development of coral reefs in Hawaii. A Natural History of the Hawaiian Islands: Selected Readings II **196**. Available from https://books.google.com/books?hl=en&lr=&id=SFLoKRq-joEC&oi=fnd&pg=PA196&dq=Community+structure,+succession+and+development+of+coral+reefs+in+Hawaii.&ots=3FWybMqdBJ&sig=k0lvv5I4oCiCAgHsWdiLrY2Nb3g (accessed September 30, 2017).
Grigg RW, Maragos JE. 1974. Recolonization of hermatypic corals on submerged lava flows in Hawaii. Ecology **55**:387–395.
Grottoli AG, Warner ME, Levas SJ, Aschaffenburg MD, Schoepf V, McGinley M, Baumann J, Matsui Y. 2014. The cumulative impact of annual coral bleaching can turn some coral species winners into losers. Global Change Biology **20**:3823–3833.
Harvell D, Jordán-Dahlgren E, Merkel S, Rosenberg E, Raymundo L, Smith G, Weil E, Willis B. 2007. Coral disease, environmental drivers, and the balance between coral and microbial associates. Oceanography **20**:172–195.
Harvey BP, Gwynn-Jones D, Moore PJ. 2013. Meta-analysis reveals complex marine biological responses to the interactive effects of ocean acidification and warming. Ecology and Evolution **3**:1016–1030.
Hirose M, Kinzie R, Hidaka M. 2001. Timing and process of entry of zooxanthellae into oocytes of hermatypic corals. Coral Reefs **20**:273–280.

Hoegh-Guldberg O et al. 2007a. Coral reefs under rapid climate change and ocean acidification. Science **318**:1737–1742.
Hoegh-Guldberg O. 2012. The adaptation of coral reefs to climate change: Is the Red Queen being outpaced? Scientia Marina **76**:403–408.
Hoegh-Guldberg O, Mumby P, Hooten A, Steneck R, Greenfield P, Gomez E, Harvell C, Sale P, Edwards A, Caldeira K. 2007b. Coral reefs under rapid climate change and ocean acidification. Science **318**:1737. American Association for the Advancement of Science.
Hoeke RK, Jokiel PL, Buddemeier RW, Brainard RE. 2011. Projected Changes to Growth and Mortality of Hawaiian Corals over the Next 100 Years. PLoS ONE **6**:e18038.
Hoeksema BW, Rogers A, Quibilan MC. 2014. Pocillopora meandrina: The IUCN Red List of Threatened Species 2014: e.T133095A54195662. International Union for Conservation of Nature. Available from http://www.iucnredlist.org/details/full/133095/0 (accessed September 28, 2017).
Hofmann GE, Barry JP, Edmunds PJ, Gates RD, Hutchins DA, Klinger T, Sewell MA. 2010. The Effect of Ocean Acidification on Calcifying Organisms in Marine Ecosystems: An Organism-to-Ecosystem Perspective. Annual Review of Ecology, Evolution, and Systematics **41**:127–147.
Holland G, Bruyère CL. 2014. Recent intense hurricane response to global climate change. Climate Dynamics **42**:617–627.
Hueerkamp C, Glynn PW, D’Croz L, Maté JL, Colley SB. 2001. Bleaching and recovery of five eastern Pacific corals in an El Niño-related temperature experiment. Bulletin of Marine Science **69**:215–236.
Hughes TP et al. 2017a. Global warming and recurrent mass bleaching of corals. Nature **543**:373–377.
Hughes TP et al. 2017b. Coral reefs in the Anthropocene. Nature **546**:82–90.
Hughes TP, Graham NAJ, Jackson JBC, Mumby PJ, Steneck RS. 2010. Rising to the challenge of sustaining coral reef resilience. Trends in Ecology & Evolution **25**:633–642.
Iguchi A, Ozaki S, Nakamura T, Inoue M, Tanaka Y, Suzuki A, Kawahata H, Sakai K. 2012. Effects of acidified seawater on coral calcification and symbiotic algae on the massive coral Porites australiensis. Marine Environmental Research **73**:32–36.
IPCC. 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate ChangeStocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
IPCC. 2014. Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Barros, V.R., C.B. Field, D.J. Dokken, M.D. Mastrandrea, K.J. Mach, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L. White (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
Jayewardene D, Donahue MJ, Birkeland C. 2009. Effects of frequent fish predation on corals in Hawaii. Coral Reefs **28**:499–506.

Jiménez C, Cortés J. 2003. Growth of seven species of scleractinian corals in an upwelling environment of the eastern Pacific (Golfo de Papagayo, Costa Rica). Bulletin of Marine Science **72**:187–198.
Johnston EC, Forsman ZH, Toonen RJ. 2018. A simple molecular technique for distinguishing species reveals frequent misidentification of Hawaiian corals in the genus Pocillopora. PeerJ **6**:e4355.
Jokiel P, P. Jury C, Kuffner I. 2016. Coral Calcification and Ocean Acidification.
Jokiel PL. 1978. Effects of water motion on reef corals. Journal of Experimental Marine Biology and Ecology **35**:87–97.
Jokiel PL, Brown EK. 2004. Global warming, regional trends and inshore environmental conditions influence coral bleaching in Hawaii. Global Change Biology **10**:1627–1641.
Jokiel PL, Hunter CL, Taguchi S, Watarai L. 1993. Ecological impact of a fresh-water “reef kill” in Kaneohe Bay, Oahu, Hawaii. Coral Reefs **12**:177–184.
Jokiel PL (Paul L., Brown EK, Friedlander A, Rodgers SK, Smith WR. 2004. Hawai’i Coral Reef Assessment and Monitoring Program: Spatial Patterns and Temporal Dynamics in Reef Coral Communities. Pacific Science **58**:159–174.
Jokiel PL, Rodgers KS, Kuffner IB, Andersson AJ, Cox EF, Mackenzie FT. 2008. Ocean acidification and calcifying reef organisms: a mesocosm investigation. Coral Reefs **27**:473–483.
Jouffray J-B, Nyström M, Norström AV, Williams ID, Wedding LM, Kittinger JN, Williams GJ. 2015. Identifying multiple coral reef regimes and their drivers across the Hawaiian archipelago. Philosophical Transactions of the Royal Society B: Biological Sciences **370**:20130268.
Kaniewska P, Campbell PR, Kline DI, Rodriguez-Lanetty M, Miller DJ, Dove S, Hoegh-Guldberg O. 2012. Major cellular and physiological impacts of ocean acidification on a reef building coral. PloS one **7**:e34659.
Kenyon JC, Aeby GS, Brainard RE, Chojnacki JD, Dunlap MJ, Wilkinson CB. 2006. Mass coral bleaching on high-latitude reefs in the Hawaiian. Page 643 Proceedings of 10th International Coral Reef Symposium. Available from https://pifsc-www.irc.noaa.gov/library/pubs/Kenyon_etal_ICRS10_2006.pdf (accessed September 21, 2017).
Kenyon JC, Brainard RE. 2006. Second recorded episode of mass coral bleaching in the Northwestern Hawaiian Islands. Atoll Research Bulletin **543**:505–523.
Kleypas JA, Langdon C. 2006. Coral Reefs and Changing Seawater Carbonate Chemistry. Coral Reefs and Climate Change: Science and Management:73–110.
Knee KL, Street JH, Grossman> EE, Boehm AB, Paytan A. 2010. Nutrient inputs to the coastal ocean from submarine groundwater discharge in a groundwater-dominated system: Relation to land use (Kona coast, Hawaii, U.S.A.). Limnology and Oceanography **55**:1105–1122.
Knutson TR, McBride JL, Chan J, Emanuel K, Holland G, Landsea C, Held I, Kossin JP, Srivastava AK, Sugi M. 2010. Tropical cyclones and climate change. Nature Geoscience **3**:157–163.
Koop K et al. 2001. ENCORE: the effect of nutrient enrichment on coral reefs. Synthesis of results and conclusions. Marine pollution bulletin **42**:91–120.
Kramer KL, Cotton SP, Lamson MR, Walsh WJ. 2016. Bleaching and catastrophic mortality of reefbuilding corals along West Hawai’i Island: findings and future directions. Pages 229–

241 13th International Coral Reef Symposium. Honolulu, HI. Available from https://sgmeet.com/icrs2016/static/files/ICRS-2016-Abstract-Book.pdf (accessed September 18, 2017).
Kroeker KJ, Kordas RL, Crim R, Hendriks IE, Ramajo L, Singh GS, Duarte CM, Gattuso J-P. 2013. Impacts of ocean acidification on marine organisms: quantifying sensitivities and interaction with warming. Global Change Biology **19**:1884–1896.
Kuffner IB, Andersson AJ, Jokiel PL, Rodgers KS, Mackenzie FT. 2008. Decreased abundance of crustose coralline algae due to ocean acidification. Nature Geoscience **1**:114–117.
Langdon C, Broecker WS, Hammond DE, Glenn E, Fitzsimmons K, Nelson SG, Peng T-H, Hajdas I, Bonani G. 2003. Effect of elevated CO2 on the community metabolism of an experimental coral reef. Global Biogeochemical Cycles **17**:1011.
Le Quéré C et al. 2016. Global Carbon Budget 2016. Earth System Science Data **8**:605–649.
Loya Y. 1976. The Red Sea coral Stylophora pistillata is an r strategist. Nature **259**:478–480.
Lüthi D et al. 2008. High-resolution carbon dioxide concentration record 650,000–800,000 years before present. Nature **453**:379–382.
Magalon H, Adjeroud M, Veuille M. 2005. Patterns of genetic variation do not correlate with geographical distance in the reef-building coral Pocillopora meandrina in the South Pacific. Molecular Ecology **14**:1861–1868.
Manganello JV et al. 2014. Future Changes in the Western North Pacific Tropical Cyclone Activity Projected by a Multidecadal Simulation with a 16-km Global Atmospheric GCM. Journal of Climate **27**:7622–7646.
Manzello DP. 2010. Coral growth with thermal stress and ocean acidification: lessons from the eastern tropical Pacific. Coral Reefs **29**:749–758.
Marlow HQ, Martindale MQ. 2007. Embryonic development in two species of scleractinian coral embryos: Symbiodinium localization and mode of gastrulation. Evolution & development **9**:355–367.
Marshall PA, Baird AH. 2000. Bleaching of corals on the Great Barrier Reef: differential susceptibilities among taxa. Coral reefs **19**:155–163.
Matthews HD, Weaver AJ. 2010. Committed climate warming. Nature Geoscience **3**:142–143.
McClanahan TR, Ateweberhan M, Graham NAJ, Wilson SK, Sebastian CR, Guillaume MM, Bruggemann JH. 2007. Western Indian Ocean coral communities: bleaching responses and susceptibility to extinction. Marine Ecology Progress Series **337**:1–13.
McClanahan TR, Obura D. 1997. Sedimentation effects on shallow coral communities in Kenya. Journal of Experimental Marine Biology and Ecology **209**:103–122.
McCoshum SM, Schlarb AM, Baum KA. 2016. Direct and indirect effects of sunscreen exposure for reef biota. Hydrobiologia **776**:139–146.
McCoy K, Heenan A, Asher J, Ayotte PM, Gorospe K, Gray A, Lino K, Zamzow J, Williams I. 2017. Pacific Reef Assessment and Monitoring Program. Data Report Ecological monitoring 2016 — reef fishes and benthic habitats of the main Hawaiian Islands, Northwestern Hawaiian Islands, Pacific Remote Island Areas, and American Samoa. PIFSC Data Report DR-17-001. Joint Institute for Marine and Atmospheric Research University of Hawaii at Manoa, Pacific Islands Fisheries Science Center National Marine Fisheries Service, Honolulu, HI. Available from https://www.pifsc.noaa.gov/library/pubs/DR-14-003.pdf.
Meehl GA et al. 2007. Global climate projections. Climate change **283**. Available from http://www.southwestclimatechange.org/node/902 (accessed July 17, 2015).

Muehllehner N, Edmunds PJ. 2008. Effects of ocean acidification and increased temperature on skeletal growth of two scleractinian corals, Pocillopora meandrina and Porites rus. Pages 57–61 Proceedings of the 11th International Coral Reef Symposium, Ft. Lauderdale, Florida. Available from https://www.researchgate.net/profile/Nancy_Muehllehner/publication/236053541_Effects_of_ocean_acidification_and_increased_temperature_on_skeletal_growth_of_two_scleractinian_corals_Pocillopora_meandrina_and_Porites_rus/links/00b49515f0c373775d000000.pdf (accessed September 27, 2017).

Muller EM, Rogers CS, Spitzack AS, Van Woesik R. 2008. Bleaching increases likelihood of disease on Acropora palmata (Lamarck) in Hawksnest Bay, St John, US Virgin Islands. Coral Reefs **27**:191–195.

Nadon MO, Ault JS, Williams ID, Smith SG, DiNardo GT. 2015. Length-based assessment of coral reef fish populations in the Main and Northwestern Hawaiian Islands. PLoS One **10**:e0133960.

Nakamura M, Ohki S, Suzuki A, Sakai K. 2011. Coral larvae under ocean acidification: survival, metabolism, and metamorphosis. PLoS ONE **6**:e14521.

Negri AP, Hoogenboom MO. 2011. Water Contamination Reduces the Tolerance of Coral Larvae to Thermal Stress. PLOS ONE **6**:e19703.

NMFS. 2006. Endangered and threatened species: final listing determinations for elkhorn coral and staghorn coral. Federal Register **76**:26852–26861.

Orr JC et al. 2005. Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. Nature **437**:681–686.

Pandolfi JM, Connolly SR, Marshall DJ, Cohen AL. 2011. Projecting coral reef futures under global warming and ocean acidification. Science **333**:418–422.

Paris Agreement. 2016. The Paris Agreement - main page. Available from http://unfccc.int/paris_agreement/items/9485.php (accessed September 29, 2017).

Pinzon JH, LaJeunesse T. 2011. Species delimitation of common reef corals in the genus Pocillopora using nucleotide sequence phylogenies, population genetics and symbiosis ecology. Molecular Ecology **20**:311–325.

Pinzón JH, Sampayo E, Cox E, Chauka LJ, Chen CA, Voolstra CR, LaJeunesse TC. 2013. Blind to morphology: genetics identifies several widespread ecologically common species and few endemics among Indo-Pacific cauliflower corals (Pocillopora, Scleractinia). Journal of Biogeography **40**:1595–1608.

Pratchett MS. 2001. Influence of coral symbionts on feeding preferences of crown-of-thorns starfish Acanthaster planci in the western Pacific. Marine Ecology Progress Series **214**:111–119.

Prouty NG, Field ME, Stock JD, Jupiter SD, McCulloch M. 2010. Coral Ba/Ca records of sediment input to the fringing reef of the southshore of Moloka'i, Hawai'i over the last several decades. Marine Pollution Bulletin **60**:1822–1835.

Putnam HM, Mayfield AB, Fan TY, Chen CS, Gates RD. 2012. The physiological and molecular responses of larvae from the reef-building coral Pocillopora damicornis exposed to near-future increases in temperature and pCO2. Marine Biology **160**:2157–2173.

Richardson SD, Ternes TA. 2014. Water Analysis: Emerging Contaminants and Current Issues. Analytical Chemistry **86**:2813–2848.

Riddle D. 2008. Coral reproduction, part one: A natural coral spawning in Hawaii, The cauliflower coral (Pocillopora meandrina). Advanced Aquarist's Online Magazine **7**:10–16.

Rivest EB, Hofmann GE. 2014. Responses of the Metabolism of the Larvae of Pocillopora damicornis to Ocean Acidification and Warming. PLOS ONE **9**:e96172.

Roberts L, Field ME. 2007. Ridge to reef: Sediment shed to the island: People, land, and the reefs of South Moloka'i. The coral reef of South Moloka 'i, Hawai 'i: Portrait of a sedimentthreatened fringing reef. US Geological Survey Sci entific Investigations Report **5101**:133–128.

Rodgers KS, Bahr KD, Jokiel PL, Richards Donà A. 2017. Patterns of bleaching and mortality following widespread warming events in 2014 and 2015 at the Hanauma Bay Nature Preserve, Hawai'i. PeerJ **5**:e3355.

Rodgers KS, Jokiel PL, Brown EK, Hau S, Sparks R. 2015. Over a Decade of Change in Spatial and Temporal Dynamics of Hawaiian Coral Reef Communities. Pacific Science **69**:1–13.

Rodolfo-Metalpa R, Hoogenboom MO, Rottier C, Ramos-Esplá A, Baker AC, Fine M, Ferrier-Pagès C. 2014. Thermally tolerant corals have limited capacity to acclimatize to future warming. Global Change Biology:n/a-n/a.

Rodríguez AS, Sanz MR, Rodríguez JB. 2015. Occurrence of eight UV filters in beaches of Gran Canaria (Canary Islands). An approach to environmental risk assessment. Chemosphere **131**:85–90.

Rogers CS. 1990. Responses of coral reefs and reef organisms to sedimentation. Marine ecology progress series. Oldendorf **62**:185–202.

Schmidt-Roach S, Miller KJ, Andreakis N. 2013. Pocillopora aliciae: a new species of scleractinian coral (Scleractinia, Pocilloporidae) from subtropical Eastern Australia. Zootaxa **3626**:576–582.

Schmidt-Roach S, Miller KJ, Lundgren P, Andreakis N. 2014. With eyes wide open: a revision of species within and closely related to the Pocillopora damicornis species complex (Scleractinia; Pocilloporidae) using morphology and genetics. Zoological Journal of the Linnean Society **170**:1–33.

Schmidt-Roach S, Miller KJ, Woolsey E, Gerlach G, Baird AH. 2012. Broadcast spawning by Pocillopora species on the Great Barrier Reef. PLoS One **7**:e50847.

Silverman J, Lazar B, Cao L, Caldeira K, Erez J. 2009. Coral reefs may start dissolving when atmospheric CO2 doubles. Geophysical Research Letters **36**:L05606.

Smith JE, Runcie JW, Smith CM. 2005. Characterization of a large-scale ephemeral bloom of the green alga Cladophora sericea on the coral reefs of West Maui, Hawai'i. Marine Ecology Progress Series **302**:77–91.

Tashiro Y, Kameda Y. 2013. Concentration of organic sun-blocking agents in seawater of beaches and coral reefs of Okinawa Island, Japan. Marine pollution bulletin **77**:333–340.

Tout J, Siboni N, Messer LF, Garren M, Stocker R, Webster NS, Ralph PJ, Seymour JR. 2015. Increased seawater temperature increases the abundance and alters the structure of natural Vibrio populations associated with the coral Pocillopora damicornis. Frontiers in microbiology **6**. Available from https://www.ncbi.nlm.nih.gov/pmc/articles/PMC4435422/ (accessed September 27, 2017).

Tovar-Sánchez A, Sánchez-Quiles D, Basterretxea G, Benedé JL, Chisvert A, Salvador A, Moreno-Garrido I, Blasco J. 2013. Sunscreen products as emerging pollutants to coastal waters. PloS one **8**:e65451.

Vega Thurber RL, Burkepile DE, Fuchs C, Shantz AA, McMinds R, Zaneveld JR. 2014. Chronic nutrient enrichment increases prevalence and severity of coral disease and bleaching. Global Change Biology **20**:544–554.

Veron JE. 2000. Coral of the World. Townsville: Australian Institute of Marine Science.

Veron JEN, Hoegh-Guldberg O, Lenton TM, Lough JM, Obura DO, Pearce-Kelly P, Sheppard CRC, Spalding M, Stafford-Smith MG, Rogers AD. 2009. The coral reef crisis: The critical importance of<350ppm CO2. Marine Pollution Bulletin **58**:1428–1436.

Vidal-Dorsch DE, Bay SM, Maruya K, Snyder SA, Trenholm RA, Vanderford BJ. 2012. Contaminants of emerging concern in municipal wastewater effluents and marine receiving water. Environmental Toxicology and Chemistry **31**:2674–2682.

Villanueva RD, Yap HT, Montano MNE. 2006. Intensive fish farming in the Philippines is detrimental to the reef-building coral Pocillopora damicornis. Marine Ecology Progress Series **316**:165–174.

Waddell JE, Clarke AM. 2008. The state of coral reef ecosystems of the United States and Pacific Freely Associated States: 2008. Available from http://aquaticcommons.org/14919/ (accessed September 30, 2017).

Walsh W, Cotton SP, Barnett C, Couch C, Preskitt L, Tissot B, Osada-D'Avella K. 2010. Long-term monitoring of coral reefs of the Main Hawaiian Islands. NOAA Coral Reef Conservation Program, Hawai'i Division of Aquatic Resources. Available from http://www.malamamaunalua.org/wp-content/uploads/DLNR-DAR-2010.pdf (accessed September 22, 2017).

Walsh W, Cotton SP, Barnett C, Couch C, Preskitt L, Tissot B, Osada-D'Avella K. 2013. Long-term monitoring of coral reefs of the Main Hawaiian Islands. NOAA Coral Reef Conservation Program, Hawai'i Division of Aquatic Resources. Available from http://www.malamamaunalua.org/wp-content/uploads/DLNR-DAR-2010.pdf (accessed September 22, 2017).

Ward S, Harrison P, Hoegh-Guldberg O. 2002. Coral bleaching reduces reproduction of scleractinian corals and increases susceptibility to future stress. Pages 1123–1128 Proceedings of the Ninth International Coral Reef Symposium, Bali, 23-27 October 2000,. Available from http://www.coremap.or.id/downloads/ICRS9th-MS-6-Ward.pdf (accessed September 27, 2017).

Wear SL, Thurber RV. 2015. Sewage pollution: mitigation is key for coral reef stewardship. Annals of the New York Academy of Sciences **1355**:15–30.

Webster NS, Uthicke S, Botté ES, Flores F, Negri AP. 2013. Ocean acidification reduces induction of coral settlement by crustose coralline algae. Global Change Biology **19**:303–315.

Webster PJ, Holland GJ, Curry JA, Chang H-R. 2005. Changes in tropical cyclone number, duration, and intensity in a warming environment. Science **309**:1844–1846.

West JM, Salm RV. 2003. Resistance and resilience to coral bleaching: implications for coral reef conservation and management. Conservation Biology **17**:956–967.

Williams ID, Baum JK, Heenan A, Hanson KM, Nadon MO, Brainard RE. 2015. Human, oceanographic and habitat drivers of central and western Pacific coral reef fish assemblages. PLoS ONE **10**:e0120516.

Williams ID, Walsh WJ, Schroeder RE, Friedlander AM, Richards BL, Stamoulis KA. 2008. Assessing the importance of fishing impacts on Hawaiian coral reef fish assemblages along regional-scale human population gradients. Environmental Conservation **35**:261–272.

Willis BL, Page CA, Dinsdale EA. 2004. Coral disease on the great barrier reef. Pages 69–104 Coral health and disease. Springer. Available from http://link.springer.com/chapter/10.1007/978-3-662-06414-6_3 (accessed September 27, 2017).

Yoshikawa T, Asoh K. 2004. Entanglement of monofilament fishing lines and coral death. Biological Conservation **117**:557–560.

Yoshioka RM, Kim CJ, Tracy AM, Most R, Harvell CD. 2016. Linking sewage pollution and water quality to spatial patterns of Porites lobata growth anomalies in Puako, Hawaii. Marine pollution bulletin **104**:313–321.