BEFORE THE SECRETARY OF COMMERCE

PETITION TO LIST EIGHT SPECIES OF POMACENTRID REEF FISH, INCLUDING THE ORANGE CLOWNFISH AND SEVEN DAMSELFISH, AS THREATENED OR ENDANGERED UNDER THE U.S. ENDANGERED SPECIES ACT

Orange Clownfish (*Amphiprion percula*) photo by flickr user Jan Messersmith

CENTER FOR BIOLOGICAL DIVERSITY
SUBMITTED SEPTEMBER 13, 2012

Notice of Petition

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Pursuant to Section 4(b) of the Endangered Species Act ("ESA"), 16 U.S.C. § 1533(b), Section 553(3) of the Administrative Procedures Act, 5 U.S.C. § 553(e), and 50 C.F.R.§ 424.14(a), the Center for Biological Diversity hereby petitions the Secretary of Commerce and the National Oceanographic and Atmospheric Administration ("NOAA"), through the National Marine Fisheries Service ("NMFS" or "NOAA Fisheries"), to list eight pomacentrid reef fish and to designate critical habitat to ensure their survival.

The Center for Biological Diversity ("Center") is a non-profit, public interest environmental organization dedicated to the protection of imperiled species and their habitats through science, policy, and environmental law. The Center has more than 350,000 members and online activists throughout the United States. The Center and its members are concerned with the conservation of endangered species, including the petitioned damselfish and anemonefish, and the effective implementation of the ESA.

NMFS has jurisdiction over this petition. This petition sets in motion a specific process, placing definite response requirements on NMFS. Specifically, NMFS must issue 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. Petitioner needs not demonstrate that the petitioned action is warranted, rather, Petitioner must only present information demonstrating that such action may be warranted. While Petitioner believes that the best available science demonstrates that listing the eight pomacentrid species as threatened or endangered is in fact warranted, there can be no reasonable dispute that the available information indicates that listing these species as either threatened or endangered may be warranted. As such, NMFS must promptly make a positive initial finding on the petition and commence a status review as required by 16 U.S.C. § 1533(b)(3)(B).

The eight pomacentrid species covered by this petition are:

Amphiprion percula (Orange clownfish)
Chromis atripectoralis (Black-axil chromis)
Chromis viridis (Blue-green damselfish)
Dascyllus albisella (Hawaiian damselfish)
Dascyllus reticulatus (Reticulate damselfish or Two-stripe damselfish)
Microspathodon chrysurus (Yellowtail damselfish or Jewel damselfish)
Plectroglyphidodon dickii (Blackbar devil or Dick's damselfish)
Plectroglyphidodon johnstonianus (Blue-eye damselfish)

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Table of Contents

Executive Summary

The damselfish and anemonefish of the family Pomacentridae are one of the major coral reef fish families evolved to live primarily on coral reefs (Pitkin 2001). Coral dependency within the pomacentrids is high compared with other coral reef fish (Wilson et al. 2008a). Many damselfish species require live coral for shelter, reproduction, habitat, and food as juveniles, adults, or both (Pratchett et al. 2008, Wilson et al. 2008a), while anemonefish are entirely dependent on host anemones in coral reef ecosystems for habitat and protection (Jones et al. 2008). As a result of this coral dependency, numerous studies have documented significant declines in pomacentrid populations following coral habitat loss due to bleaching and other disturbances, with the most severe declines occurring to coral-dwelling and coral-eating specialists (Pratchett et al. 2008, Wilson et al. 2006). Recent laboratory studies have also found that ocean acidification and ocean warming have many direct negative effects on pomacentrids at temperatures and levels of ocean acidity expected within this century, as detailed below.

This petition requests the protection of the orange clownfish (*Amphiprion percula*) and seven damselfish occurring in U.S. waters that are highly threatened by ocean warming and ocean acidification that degrade and destroy their coral reef and anemone habitat, and pose direct threats to these fish by impairing their sensory capabilities, behavior, aerobic capacity, swimming ability, and reproduction. The seven petitioned damselfish are habitat specialists that directly depend on live corals for survival, including shelter, reproduction, recruitment, and food. These damselfish all specialize on sensitive branching corals such as acroporids and pocilloporids which are particularly prone to bleaching, and one damselfish species—*Plectroglyphidodon johnstonianus*—is also an obligate corallivore, which puts these species at particularly high risk from coral habitat loss. The petitioned damselfish species include the yellowtail damselfish (*Microspathodon chrysurus*) that inhabits U.S. waters in Florida and the Caribbean, Hawaiian dascyllus (*Dascyllus albisella*) and blue-eye damselfish (*Plectroglyphidodon johnstonianus*) that inhabit U.S. waters in Hawaii, and four species that inhabit U.S. territorial waters in the Indo-Pacific, including American Samoa and the Northern Mariana Islands: black-axil chromis (*Chromis atripectoralis*), blue-green damselfish (*Chromis viridis*), reticulate damselfish (*Dascyllus reticulatus*), and blackbar devil or Dick's damselfish (*Plectroglyphidodon dickii*).

Rising ocean temperatures pose a profound and growing threat to the petitioned damselfish and clownfish by increasing the frequency and intensity of mass bleaching events that degrade and destroy their coral reef and anemone habitat. Mass coral bleaching events have been shown to result in damselfish population declines (Booth and Beretta 2002, Wilson et al. 2006, 2008b, Graham et al. 2007, 2008, Pratchett et al. 2008), reduced recruitment on bleached and dead corals (Booth and Beretta 2002, Feary et al. 2007b, Bonin et al. 2009, McCormick et al. 2010), increased susceptibility to predation (Coker et al. 2009), lower fish growth rates (Feary et al. 2009), and higher competition for dwindling coral habitat (Coker et al. 2012). Similarly, anemone bleaching has been shown to reduce the densities, reproduction, and recruitment of anemonefish (Jones et al. 2008, Saenz-Agudelo et al. 2011). Globally, coral reefs are facing significant and accelerating coral loss, and the majority of the world's corals are predicted to experience mass bleaching events at frequencies from which they will be unable to recover by the 2020s or 2030s absent significant thermal adaptations by corals and their symbionts (Hoegh-

Guldberg 1999, Sheppard 2003, Donner et al. 2005, 2007, Donner 2009, Burke et al. 2011). Because the branching corals that the petitioned damselfish species rely on are particularly susceptible to bleaching (Marshall and Baird 2000, Loya et al. 2001, McClanahan et al. 2004, 2012, Bonin 2012), mass bleaching events threaten to eliminate much of the petitioned pomacentrid species' essential coral reef habitat before mid-century.

Adding to the impacts of ocean warming, ocean acidification threatens the petitioned pomacentrid reef fish by slowing coral growth rates, increasing susceptibility to bleaching, and weakening the structural integrity of their coral reef habitat. Corals are already experiencing lower calcification rates that have been linked to ocean acidification in the Indo-Pacific and Caribbean regions inhabited by the petitioned reef fish species (Cooper et al. 2008, Gledhill et al. 2008, Bak et al. 2009, De'ath et al. 2009, Bates et al. 2010, Fabricius et al. 2011, Friedrich et al. 2012). Studies projecting the combined impacts of ocean acidification and ocean warming on corals predict that coral erosion will exceed calcification rates at atmospheric CO_2 concentrations of 450 to 500 ppm (Hoegh-Guldberg et al. 2007), and all coral reefs will begin dissolve at CO_2 concentrations of 560 ppm (Silverman et al. 2009). In the Caribbean, a recent study concluded that "coral reef communities are likely to be essentially gone from substantial parts of the Southeast Caribbean by the year 2035" (Buddemeier et al. 2011). In short, due to the synergistic impacts of ocean acidification, mass bleaching, and local impacts, coral reefs are projected to experience "rapid and terminal" declines worldwide at atmospheric CO_2 concentrations 450 ppm, a level that is expected before mid-century (Veron et al. 2009). Thus, the high coral-reef dependency of the petitioned fish species places them at significant risk of extinction due to the accelerating loss and degradation of their essential coral reef habitat due to ocean warming and ocean acidification, in combination with local threats to coral reefs such as overfishing, disease, coastal development, and pollution. According to reef fish scientists, the long-term future of coral reef-dependent fishes is "potentially catastrophic" due to the expected impacts of climate change on their habitat (Pratchett et al. 2011).

In addition to causing habitat loss, ocean acidification and ocean warming directly threaten the survival of the petitioned clownfish and damselfish through a wide array of adverse impacts that are predicted to lead to negative fitness consequences and population declines. Laboratory experiments have shown that ocean acidification at levels expected to occur within this century impairs larval orange clownfish and damselfish sensory abilities and behavior, making it more difficult for them to locate suitable settlement sites on reef habitat and avoid predators. Specifically, ocean acidification disrupts smell, hearing, and behavior of larval orange clownfish (Munday et al. 2009, 2010, Dixson et al. 2010, Nilsson et al. 2012), making larval clownfish attracted to odors from predators and unfavorable habitat (Munday et al. 2010, Dixson et al. 2010). Ocean acidification also impairs the hearing capacity of larval clownfish, which is predicted to have negative effects on settlement success and survival (Simpson et al. 2011). Similarly, research on six damselfish species found that ocean acidification impairs larval damselfish smell, vision, learning, behavior, and brain function, leading to higher risk of mortality. For example, in acidified waters, larval damselfish (1) become attracted to predator odors and display much riskier behaviors, making them more prone to predation; two species suffered a five-fold to nine-fold increase in predation rate at CO_2 levels of 700 to 850 ppm (Munday et al. 2010, Ferrari et al. 2011a); (2) cannot discriminate between habitat olfactory cues, making it more difficult to locate appropriate settlement habitat (Devine et al. 2011); (3)

settle on the reef during dangerous times—the full moon rather than new moon—when they are more vulnerable to predation (Devine et al. 2011); (4) fail to visually recognize or evade important predator species (Ferrari et al. 2012b); (5) cannot learn to respond appropriately to a common predator by watching other fish react or by smelling injured fish, unlike fish under normal conditions (Ferrari et al. 2012a); and (6) suffer disruption of an important neurotransmitter which is thought to result in the sensory and behavioral impairment observed in acidified conditions (Nilsson et al. 2012).

Elevated ocean temperatures also have a suite of adverse effects on damselfish. Laboratory experiments have found that higher temperatures expected within this century interfere with damselfish reproduction by reducing breeding rate, egg size, sperm production, and embryonic survival (Munday et al. 2008, Donelson et al. 2010, Lo-Yat et al. 2010), and reduce swimming performance and aerobic capacity (Nilsson et al. 2009, Johansen and Jones 2011, Donelson et al. 2011), which are likely to result in negative population-level effects.

In addition to threats from greenhouse gas pollution, the orange clownfish and several of the petitioned damselfish may face threats from the global marine aquarium trade. Damselfish and anemonefish are by far the most commonly traded marine ornamental fish species, accounting for ~43% of all fish traded (Zajicek et al. 2009). In 2005, the most commonly imported species into the U.S., numbering more than 900,000 individuals each year, was the petitioned blue-green damselfish (*Chromis viridis*), while the orange clownfish (*Amphiprion percula*) and its sister species *A. ocellaris* were the fifth most imported species into the U.S. with over 400,000 individuals imported each year (Rhyne et al. 2012). Studies suggest that the orange clownfish, black-axil chromis (*Chromis atripectoralis*), and Hawaiian dascyllus (*Dascyllus albisella*) may be threatened by overharvest by the marine aquarium trade in some regions. The widespread and growing trade in coral reef fish and corals adds to the cumulative stresses that the petitioned pomacentrids face from ocean warming and ocean acidification.

Existing regulatory mechanisms have been ineffective in mitigating the principal threats to the petitioned pomacentrid species. Regulatory mechanisms at the national and international level do not require the greenhouse gas emissions reductions necessary to protect the petitioned species from ocean warming, ocean acidification, and other climate change threats, and are therefore inadequate. Regulatory mechanisms for the global marine aquarium trade are also inadequate in the United States and internationally (Tissot et al. 2010).

Under the Endangered Species Act, NMFS is required to designate critical habitat for the damselfish species in U.S. waters concurrent with their listing. Critical habitat is a foundation of the Act's recovery system. Species that have critical habitat protection are approximately twice as likely to have improving population trends as species without critical habitat (Taylor et al. 2005). Critical habitat designation would lead to a suite of benefits for the petitioned reef fish, their coral reef habitat, and thousands of other coral-reef associated species by minimizing stressors to the fish and their habitat from coastal development, pollution, over-fishing, and other threats in the U.S. waters inhabited by these fish.

Congress and the Supreme Court have obliged NFMS to prioritize species survival and recovery, "whatever the cost." See *TVA v. Hill*, 437 U.S. 153, 154 (1978). Given their incalculable intrinsic value, their pivotal role in marine ecosystems, and their importance to human communities, the petitioned clownfish and damselfish species warrant immediate protection under the Endangered Species Act.

Part One of this petition summarizes the natural history and threats to the eight petitioned pomacentrid species. Part Two provides a detailed account of the imminent, high-magnitude suite of threats facing these reef fish and their coral and anemone habitat which jeopardize these species with extinction. The petition clearly demonstrates that, in the context of the ESA's five statutory listing factors, NMFS should promptly list the petitioned clownfish and damselfish species as threatened or endangered.

Part One: Natural History and Threats to the Petitioned Clownfish and Damselfish Species

I. Introduction to Pomacentrids (Damselfish and Anemonefish)

A. Description of the Pomacentrids

The family Pomacentridae, comprised of damselfish and anemonefish, constitutes one of the major coral reef fish families evolved to live primarily on coral reefs (Pitkin 2001). Pomacentrids tend to be small, brightly colored fish (Randall 2005). Most mature pomacentrids are less than 10 to 15 cm in length, and the smallest are 4 to 5 cm in length (Allen 1991). The largest species, which can reach 36 cm in length, belong to the genera *Hypsypops*, *Microspathodon*, and *Parma* and typically inhabit temperate seas (Allen 1991). Adult males tend to be larger than adult females, except in anemonefish where the female is the largest fish in the group. Pomacentrids are highly variable in color ranging from bright hues of orange, yellow, red, and blue to more subdued shades of black, brown, and grey (Allen 1991). Adult color patterns in some species vary based on geographic location, particularly in widely distributed species, and males and females are similarly colored in most species (Allen 1991). The younger stages of many pomacentrids are brightly colored and can appear quite different from adults (Allen 1991). The juveniles of many species have a bright yellow body with neon blue stripes along the head and upper back (Allen 1991).

Morphologically, pomacentrids are characterized by three defining characteristics: a single nostril hole on each side of snout instead of two, scales that extend onto the fins, and an interrupted lateral line (Deloach 1999). They have a high, oval, and laterally compressed body; a single, continuous dorsal fin with eight to 17 spines and 10 to 21 soft rays; an anal fin with two spines and 10 to 16 soft rays; and a caudal fin that is emarginated, forked, or lunate (Randall 2005). Their body is covered with moderately large and ctenoid scales that extend onto the head and basally on the median fins, and the anterior end has tubed scales and often a few pored scales (Randall 2005). Pomacentrids have a small mouth with conical teeth in planktivores and incisor-like teeth in grazers.

B. Taxonomy

The family Pomacentridae belongs to the class Actinopterygii (ray-finned fishes), subclass Neopterygii, infraclass Teleosti, superorder Acanthopterygii, order Perciformes, and suborder Labroidei. The family Pomacentridae consists of 28 genera and approximately 321 species based on Allen (1991); since Allen (1991), 30 new species have been described (Randall 2005). Anemonefish, which are members of the genera *Amphiprion* and *Premnas*, comprise a monophyletic clade of 26 currently recognized species (Ollerton et al. 2007). Two of the 28 previously named species are now recognized as natural hybrids; *Amphiprion leucokranos* and *Amphiprion thiellei* appear to represent variants of crosses between *Amphiprion chryopterus* and *Amphiprion sandaracinos* (Ollerton et al. 2007).

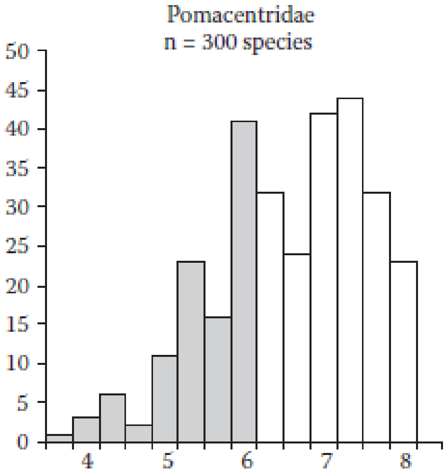
C. Range and Distribution

Pomacentrids occur in all tropical and warm temperate waters, with a few species found in freshwater streams (Allen 1991). The majority of species occur in the Indo-West Pacific with the highest species richness in Australia, Indonesia, and the Philippines (Allen 1991). There are ~268 species in the Indo-West Pacific, 22 in the eastern Pacific, 19 in the Western Atlantic and trans-Atlantic, and 11 in the central and eastern Atlantic (Allen 1991). The 26 species of anemonefish are widespread in the Indo-Pacific but none are found in the Atlantic.

Allen (1991) described three basic patterns of pomacentrid distribution: species that are widely distributed over a large area (e.g., species that occur over much of the Indo-West Pacific); species that have more restricted subregional distributions (e.g., western Indian Ocean, Indo-Australian Archipelago, or smaller areas such as the Red Sea); and species that are endemic to single islands or island groups (e.g., Mauritius-Reunion-Madagascar group, Hawaiian Islands, and Marquesas), including some that are widely scattered and probably represent relict distributions of formerly widespread populations (e.g. *Stegastes insularis* at Christmas Island and Minami Tori Shima 5,000 km to the northeast).

Across four of the major families of coral reef fishes (Acanthuridae, Chaetodontidae, Pomacanthidae and Pomacentridae), Pomacentrids have the highest proportion (34%) of range-restricted species, defined as species with geographic ranges less than 80,000 km^2 (Figure 1) (Pratchett et al. 2008).

Figure 1. The distribution of geographic ranges ($\log_{10}$ km^2) for 300 species of Pomacentridae. Geographic ranges were estimated based on maximum area encompassed within irregular polygons around locations of occurrence, based on data published in Hughes et al. (2002). Grey bars indicate species with "restricted" geographic ranges less than 80,000 km^2.
Source: Pratchett et al. (2008): Figure 7.

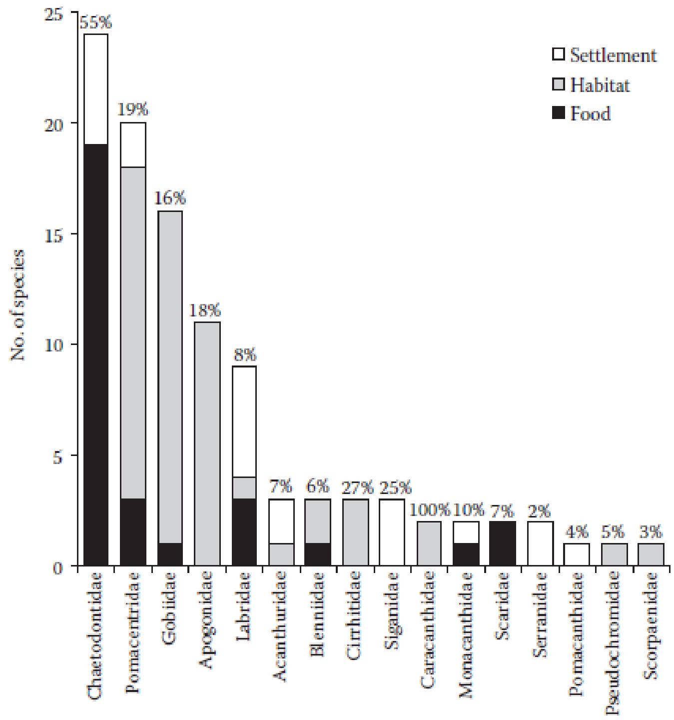


D. Habitat Requirements

Most tropical pomacentrids inhabit coral reefs in coastal or atoll lagoons or adjacent to dropoffs of the outer reef that are exposed to strong currents that transport food (Allen 1991). Temperate pomacentrids mainly occur in rocky areas and three species can be found in brackish estuaries or coastal freshwater streams (Allen 1991). The number of species increases with the variety of habitats and the availability of shelter such as coral formations, caves, crevices, and boulders (Allen 1991). The majority of pomacentrids are territorial, particularly those that forage on the reef bed (Pitkin 2001).

Coral dependency within the pomacentrids is high compared with other coral reef fishes (Wilson et al. 2008a). For example, an estimated ~20% or more of all pomacentrid species on the Great Barrier Reef rely on coral for food or shelter (Figure 2), although this is thought to be an underestimate (Pratchett et al. 2008, Wilson et al. 2008a). Wilson et al. (2008a) found that 40% (8 of 20) of pomacentrid species studied on the Great Barrier Reef live in association with live coral as adults while 53% (7 of 13) of pomacentrid species studied displayed a preference for live coral as juveniles, representing 56% and 60% of all adult and juvenile individuals, respectively. Adults of many species had strong affiliations with branching corals, while juveniles favored both branching and plating corals, which is thought to reflect the sizes of refuge provided by coral types (Wilson et al. 2008a). Some species exhibited ontogenetic changes in microhabitat use, where fish relied on coral habitat as juveniles but not as adults. Wilson et al. (2008) noted that several studies have shown a similar pattern in which some species show strong associations with live corals as juveniles but do not favor corals as adults. The niche breadth of coral-associated juveniles also tends to be narrower than in adult conspecifics (Wilson et al. 2008a).

Figure 2. Coral-dependent fishes on the Great Barrier Reef by family, where numbers indicate the percentage of species in each family considered to be coral dependent relative to the total. When species used corals for more than one purpose, precedence was given to food, then habitat, then recruitment.
Source: Pratchett et al. (2008): Figure 4



Coral-dwelling damselfishes from the genera *Amblyglyphidon*, *Chromis*, *Dascyllus*, *Plectroglyphidodon* , and *Pomacentrus* are commonly associated with the complex, branching acroporid and pocilloporid corals, which provide a high degree of structural complexity, but are often extremely susceptible to bleaching (Pratchett et al. 2008). Many damselfish species retreat to live corals at night or when threatened and otherwise remain in close proximity, feeding above their corals hosts (i.e., *Pomacentrus* spp., *Chromis* spp., *Dascyllus* spp.) (Pratchett et al. 2008). Overall, many damselfish require live coral for shelter, settlement habitat, or food (Pratchett et al. 2008, Wilson et al. 2008a).

Anemonefish are entirely dependent on host anemones for habitat and protection (Jones et al. 2008). All anemonefish live in an obligate, symbiotic association with one or more of 10 species of sea anemones (Anthozoa: Actiniaria) belonging to three unrelated families: *Cryptodendrum adhaesivum, Entacmaea quadricolor*, *Heteractis aurora*, *Heteractis crispa*, *Heteractis magnifica*, *Heteractis mala, Macrodactyla doreensis*, *Stichodactyla gigantea*, *Stichodactyla haddoni*, and *Stichodactyla mertensii* (Ollerton et al. 2007). This mutualism extends from the east coast of Africa and the Red Sea through the Indian Ocean to the western Pacific, and from southeastern Australia to the latitude of Tokyo (Ollerton et al. 2007). Living their entire post-settlement lives within the anemone, anemonefish fish are unaffected by stinging anemone nematocysts because of their mucus coating. The relationship is thought to benefit the fish which are protected from predators by anemone's stinging tentacles that are otherwise lethal to most fish (Ollerton et al. 2007). Anemonefish eggs, which are laid beside the anemone, are likewise protected (Ollerton et al. 2007). Observations and experimental evidence suggest that the relationship is beneficial for some anemones as well (Ollerton et al. 2007). Anemones of some species are protected against anemone predators such as butterflyfish, the ammonia excreted by anemonefish may be used by zooxanthellae that live within the cells of the anemones, and anemonefish movements can increase water circulation around the anemone (Ollerton et al. 2007).

E. Diet and Foraging Ecology

Pomacentrids eat a wide variety of plants and animals. Species in the genera *Chromis*, *Dascyllus*, *Lepidozygus*, *Neopomacentrus*, *Pristotis*, and *Teixeirichthys* tend to swim above the reef to consume plankton while using the reef for shelter (Randall 2005). Copepods provide an important food source for planktivores, although other prey species include siphonophores, hydroids, tunicates, polychaetes, serpulids, oligochaetes, small gastropods, nudibranchs, ophiuroids, amphipods, isopods, crabs, and shrimps (Allen 1991). Some species are mostly herbivorous, particularly in the genus *Stegastes*, and eat algae on the coral reef, including blue-green algae, green algae, calcareous and filamentous red algae, and brown algae (Allen 1991). Many territorial grazing damselfish are referred to as "farmers" because they cultivate algal assemblages on reefs. "Intensive" farmers weed unwanted algal species from their territories, encourage the growth of preferred algae, and defend their relatively small territories very aggressively, while "extensive" farmers weed and defend their territories less intensively and maintain relatively large territories with diverse algal assemblages of filamentous and macro-algae (Emslie et al. 2012). Eight pomacentrid species eat live coral, and two of these are considered obligate corallivores including the petitioned species *Plectroglyphidodon johnstonianus* (Cole et al. 2008). Corals are also thought to provide nutritional supplements to many coral reef fish including mucus, tissue, and gametes (Pratchett et al. 2008). Anemonefish eat a variety of zooplankton and algae.

F. Reproductive Ecology and Development

Damselfish have a variety of mating systems, including polygyny where one male may guard the eggs of several females, promiscuity, and monogamy (eol.org). Many damselfish appear to spawn year-round, and spawning activity often increases in early summer (eol.org). Reef-dwelling damselfish typically spawn in synchrony with the lunar cycle, with greatest activity occurring near the full and new moons (eol.org). Some damselfishes spawn within their permanent territory, while others such as planktivorous damselfishes use temporary territories for courtship and spawning (eol.org).

Damselfish have highly specialized breeding behaviors to prepare spawning sites and attract mates (Allen 1991). Several days before spawning, the male (and sometimes the female) begins to groom and tend a nest site. Site choice varies according to species and may include rock ledges, cleaned coral branches, algal turf, empty shells, or the roofs of caves. The male prepares the nest site surface by grazing off algae and invertebrates and removing debris by vigorously fanning the pectoral fins (Allen 1991). The male then engages in courtship behaviors to attract an egg-bearing female to the spawning site. The male often undergoes a temporary dramatic color change during courtship and spawning, makes chirping and grunting sounds, and engages in movements to attract the female including chasing, nipping, and rapid up-and-down swimming (Allen 1991). Spawning often occurs in the morning and can last for more than two hours. The female lays eggs in long rows on the substrate while the male follows and fertilizes the eggs. Depending on the species, each females lays between 50 to 1,000 or more eggs (Allen 1991).

Most damselfish have an incubation period of two to four days, while species with larger eggs such as anemonefish have a longer incubation period of six or seven day (Allen 1991). During incubation the male (and in few cases, the female) guards the eggs until they hatch and is typically very aggressive, frequently attacking larger fishes and especially egg predators such as wrasses and butterflyfishes (Allen 1991). The male meticulously fans the nest, removes dead or diseased eggs to prevent contamination, and generally keeps the nest free of debris to facilitate hatching (Allen 1991). The newborn larvae range in length from 2 to 4 mm depending on the species (Allen 1991). The larvae are left to care for themselves after hatching although larvae of one Indo-Pacific species, *Acanthochromis polycanthus*, are guarded by their parents near the spawning cave for three to six weeks. The nearly transparent larval fish enter a pelagic phase and are transported by the ocean currents for weeks to months depending on species (Allen 1991). *Amphiprion* and *Premnas* have the shortest larval stage from 7 to 14 days; *Chromis* and *Dascyllus* range from 17 to 47 days (20 to 30 days for most); and genera in the subfamily Pomacentrinae (i.e. *Abudefduf, Chrysiptera, Pomacentrus*) range from 13 to 42 days (Thresher and Brothers 1989).

At settlement, the larval fish select a suitable benthic, coral-reef habitat based on chemical and visual cues (Munday et al. 2010). Predation is the most important source of mortality at settlement, whereas competition occurs mostly after settlement. The settlement phase poses a crucial population bottleneck for damselfishes. Newly settled fish assume the juvenile color pattern. The growth rate of juveniles is variable and generally ranges from 5 to 15 mm per month, slowing as they become more mature (Allen 1991).

Anemonefish have a monogamous and protandrous mating system in which male individuals can become female (Allen 1991). Anemonefish most frequently inhabit an anemone as a single, sexually mature male and female pair, along with a group of ambosexual juveniles for which neither sperm nor egg-producing tissues are active. The sexually mature pair stays together for at least a year and sometimes for their entire lifetime. If the female (the largest individual in the group) dies, her male partner develops into a female to take her place, and the largest juvenile grows rapidly and replaces him as the dominant male. The young grow slowly and do not reach sexual maturity until they can replace one of the dominant fish. Anemonefish spawn irregularly (Jones et al. 2008). Spawning occurs at the base of the anemone on a rock surface, or, if the anemone lives on sand, on a surface the fish drag near the anemone (Allen 1991). The male clears the nest site by biting at the tentacles of the anemone until they withdraw, and then leads the female there for spawning. The larvae of anemonefish have a relatively short life span of 8 to 12 days, and larval mortality is extremely high (Jones et al. 2008). Even if anemonefish spawning and recruitment are successful, the unusual social structure results in high mortality of juveniles (Jones et al. 2008). Anemones are thought to provide necessary settlement cues for larvae (Jones et al. 2008).

G. Lifespan

Longevity estimates for tropical pomacentrids vary widely among species, but some individuals may live for 20 years or more (Wilson et al. 2008a). The longer longevity estimates for damselfishes are 10 years for *Pomacentrus wardi*, 11 years for *Dascyllus albisella*, 15 years for *Parma victoriae*, 15 years for *Stegastes altus*, 17 years for *Hypsypops rubicunda*, and 17 years for *Pomacentrus moluccensis* (Buston and García 2007). The longest longevity estimate for a pomacentrid is for female orange clownfish *Amphiprion percula* which may live 30 years, likely due to low mortality because of the protection provided by anemones (Buston and García 2007).

H. Ecological Importance

Many pomacentrid species display territorial behavior which can play an important role in structuring benthic reef communities, especially since territories can cover extensive reef areas and individuals may live for decades (Wilson et al. 2008a). For example, territorial, grazing damselfishes known as “farmers” can shape algal community structure; influence the patterns of coral recruitment, survival, diversity and zonation; and modify the grazing activities of roving herbivores through their aggressive territorial defense (Emslie et al. 2012). Collectively, farmer territories can cover over 90% of the reef substratum in some reef zones (Emslie et al. 2012). Most pomacentrids are also small-bodied and highly abundant, making them major prey items for many reef predators (Wilson et al. 2008a). As summarized by Wilson et al. (2008a), “pomacentrids are an ecologically diverse and important family on coral reefs, and disturbance-induced changes to pomacentrid communities may subsequently affect the composition of benthic communities and reef trophodynamics.”

II. Species Accounts for the Petitioned Clownfish and Damselfish Species: Natural History and Threats

The eight petitioned pomacentrid reef fish species belong to four of the 28 pomacentrid genera: *Amphiprion*, *Chromis*, *Dascyllus*, and *Plectroglyphidodon*, and include one anemonefish and seven damselfish. The damselfish occur in part or entirely in U.S. waters: *Microspathodon chrysurus* inhabits U.S. waters in Florida, the U.S. Virgin Islands, and Puerto Rico; two species inhabit Hawaiian waters exclusively (*Dascyllus albisella*) or in part (*Plectroglyphidodon johnstonianus*); and four species inhabit U.S. territorial waters in the Indo-Pacific: *Chromis atripectoralis, Chromis viridis, Dascyllus reticulatus,* and *Plectroglyphidodon dickii*. The orange clownfish *Amphiprion percula* inhabits non-U.S. waters in the Indo-Pacific.

The primary threats to the petitioned reef fish are ocean warming and ocean acidification resulting from anthropogenic greenhouse gas pollution. Ocean warming and acidification are rapidly degrading and destroying these species' coral reef and anemone habitat through the increasing frequency and intensity of mass bleaching events and reductions in coral calcification and structural integrity, respectively. Laboratory experiments have found that ocean acidification at levels expected within this century also directly harms the orange clownfish and damselfish by impairing the sensory capacities and behavior of larval fish, which hinders their ability to find settlement sites and avoid predators. Elevated ocean temperatures have adverse effects on damselfish reproduction, swimming ability, and aerobic capacity.

Among reef fish, the petitioned pomacentrids are particularly vulnerable to coral habitat loss from ocean warming and acidification because they are coral habitat specialists that depend on live coral for shelter, reproduction, recruitment, and food. All the petitioned damselfish specialize on branching corals such as acroporids and pocilloporids which are particularly prone to bleaching, and one species—*P. johnstonianus*—is also an obligate corallivore.

The petitioned damselfish were selected based on the following criteria: (1) they occur in U.S. waters, including territorial waters; (2) they depend on live coral for shelter, reproduction, recruitment, and/or food, which makes them highly vulnerable to coral habitat loss and degradation due to ocean warming and ocean acidification; (3) they are habitat specialists that rely on branching corals which are particularly susceptible to bleaching; and (4) scientific studies provide evidence that the species is harmed by the direct or indirect effects of ocean warming and ocean acidification.

The species accounts below include a species description and a summary of the distribution, habitat requirements, natural history, and threats faced by each species. A detailed description of the suite of threats facing each species is provided in Part Two of the petition. The abundance and population trends of the petitioned species are unknown, and none of the petitioned species has been assessed by the International Union for Conservation of Nature and Natural Resources (IUCN).

A. Anemonefish

1. *Amphiprion percula* (Orange clownfish)

Species Description: The orange clownfish is orange with three white bands on the head, middle, and tail regions, with the middle band bulging forward toward the head centrally. Black stripes separate the orange and white coloration on the body, which can range from thick black lines to patches of black on the sides. The fins have black tips. There are 30 to 38 pored scales with no interruptions along the lateral line. The dorsal fins contain a total of 9 or 10 spines. The standard length of the male is 3.6 cm, while the female is larger at a standard length of 4.6 cm, and the maximum reported total length is 11 cm (Florida Museum of Natural History 2011). The orange clownfish is distinguished from its sibling species *A. ocellaris* by its larger black bands, fewer spines in its dorsal fin, and an allopatric distribution although there may be some overlap in Tomini Bay in Sulawesi (Timm et al. 2008).

Distribution: The species is found in the Indo-Pacific in Queensland, Australia, and Melanesia including northern Great Barrier Reef, northern New Guinea, New Britain, Solomon Islands and Vanuatu. It is not known from New Caledonia and the Fiji Islands, although there is one record from the latter area (FishBase.org). See Figure 3.

Habitat: The orange clownfish is a non-migratory species that inhabits lagoon and seaward reefs at depths ranging from 1 to 15 m (Florida Museum of Natural History 2011). It lives in a symbiotic association with three species of anemones: *Heteractis crispa, Heteractis magnifica,* and *Stichodactyla gigantea* (Ollerton et al. 2007). A study of anemonefish in Papua New Guinea found that the density of *A. percula* was highest at sites where there were high densities of their host anemone, *H. magnifica*, and low abundance of an anemonefish species, *A. perideraion,* that uses the same host anemone species (Elliott and Mariscal 2001). Larger diameter anemones support larger orange clownfish (Elliot and Mariscal 2001).

Natural History:

Diet and foraging ecology: Orange clownfish forage on algae and plankton as well as obtaining food from the host anemone. The anemonefish will eat the dead tentacles of the anemone and any leftover food from the stinging tentacles of the anemone (Florida Museum of Natural History 2011).

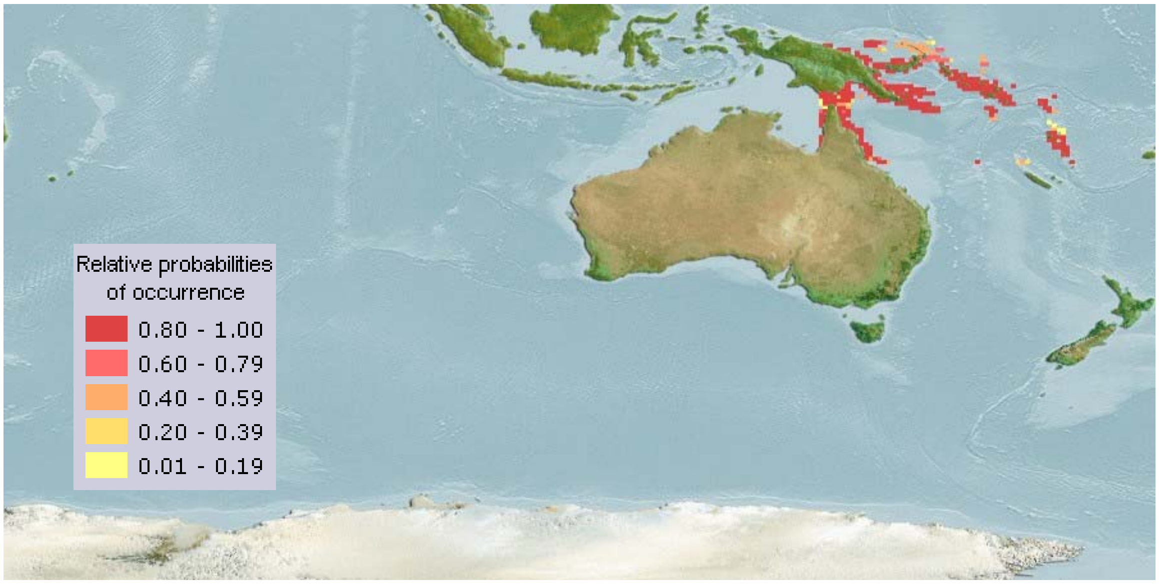
Reproductive ecology: The mating system in the orange clownfish is monogamous and protandrous, meaning that all fish develop into males first and can become females later. Breeding takes place within the host anemone in a social group of non-related individuals consisting of a dominant breeding pair and zero to four non-breeding fish, with a strict size-based dominance hierarchy (Buston and Elith 2011). The female is the largest fish, the male is second largest, and non-breeding individuals get progressively smaller down the hierarchy (Buston and Elith 2011). If the female in the social group dies, the male changes sex and assumes the position vacated by the female, while the largest nonbreeding fish inherits the position vacated by the sex-changing male (Buston and Elith 2011). Reproductive success in clownfish appears to be unrelated to the presence and number of nonbreeders, but is positively related to female growth, female size, and male breeding experience (Buston and Elith 2011).

Reproduction occurs throughout the year (Buston and Elith 2011). The male clownfish prepares a nest site on a patch of bare rock within the protection of the anemone's tentacles by cleaning the surface with his mouth. During spawning, the female releases eggs as she brushes the surface of the nest, making multiple passes and followed by the male who externally fertilizes the eggs. The female lays from one hundred to over one thousand eggs depending on her size and age. After spawning occurs, the male cares for the nest by mouthing and fanning the eggs to oxygenate them during the incubation period, removing dead eggs, and guarding against predators. Incubation takes 6 to 7 days, the larvae hatch and enter a short pelagic larval phase of 8 to 12 days, and the larval clownfish settle to the bottom in search of a host anemone (Buston et al. 2007). The larvae appear to use olfactory settlement cues indicating the presence of reefs beginning at seven-days post-hatching, which may facilitate initial dispersal away from natal reefs followed by settlement to new reef habitat (Dixson et al. 2011). A study of larval orange clownfish in Papua New Guinea found that successful dispersal of larval clownfish declines with distance, where larvae are most likely to successfully disperse within one meter of their natal site (Buston et al. 2012). Anemone saturation influences the likelihood of recruitment (Elliott and Mariscal 2001). Once the larvae settle on an anemone, the interactions with the anemone stimulate the young fish to produce a protective mucous coating to prevent it from being stung (Florida Museum of Natural History 2011).

Lifespan: A study in New Guinea found that the expected lifespan for a female orange clownfish is an extraordinary 30 years, which is two times greater than the longevity estimated for any other coral reef pomacentrid and six times greater than expected for a fish of the same size (Buston and García 2007). The study hypothesized that the protection from predators afforded by anemones favors delayed senescence and the extremely long lifespan in this species (Buston and García 2007).

Threats: The orange clownfish is threatened by ocean acidification and ocean warming caused by anthropogenic greenhouse gas pollution, as well as harvest by the global marine aquarium trade. Ocean acidification at levels expected to occur within this century impairs larval orange clownfish senses of smell and hearing and results in abnormal behaviors, which harm their ability to find settlement sites and avoid predators (Munday et al. 2009, 2010, Dixson et al. 2010, Nilsson et al. 2012). Ocean warming and ocean acidification are degrading and destroying the coral reef ecosystems on which the clownfish depends. Ocean warming has been documented to bleach host anemones, which can lead to reductions in anemone abundance and size (Hattori 2002, Jones et al. 2008, Frisch and Hobbs 2009, Saenz-Agudelo et al. 2011, Hill and Scott 2012) and reduce the densities, reproduction, and recruitment of anemonefish (Jones et al. 2008, Saenz-Agudelo et al. 2011). The global marine aquarium trade also poses a threat to the orange clownfish through overharvest. The orange clownfish and its sister species *A. ocellaris* were the fifth most commonly imported marine ornamental fish into the U.S. in 2005, with more than 400,000 individuals imported per year (Rhyne et al. 2012). Studies provide evidence of population declines of anemonefish resulting from overharvesting due to the marine aquarium trade (Shuman et al. 2005, Jones et al. 2008).

Figure 3. Map of the distribution of *Amphiprion percula*.
Source: Computer Generated Map for *Amphiprion percula* (un-reviewed). www.aquamaps.org, version of Aug. 2010. Web.



B. Damselfish: Caribbean and Indo-Pacific Species Occurring in U.S. Waters

2. *Chromis atripectoralis* (Black-axil chromis)

Species Description: Blue-green, shading to white ventrally; attains 11 cm (Randall 2005). Similar to *C. viridis* but distinguished by the black base (axil) of the pectoral fin and more branched pectoral rays--usually 17 or 18 in *C. atripectoralis* and 15 or 16 in *C. viridis*, although the number of pectoral rays overlaps in the two species (Froukh and Kochzius 2008).

Distribution: This species occurs at the Ryukyu Islands to Great Barrier Reef and Lord Howe Island; east to the islands of Oceana except the Hawaiian Islands, Marquesas, and Pitcairn Islands; west in Indian Ocean to Maldives and Seychelles (Randall 2005). *C. atripectoralis* occurs almost in the same region as *C. viridis* but not in the Red Sea (Froukh and Kochzius 2008). It occurs in U.S. territorial waters in American Samoa and the Northern Mariana Islands (Allen 1991: Table 2). See Figure 4.

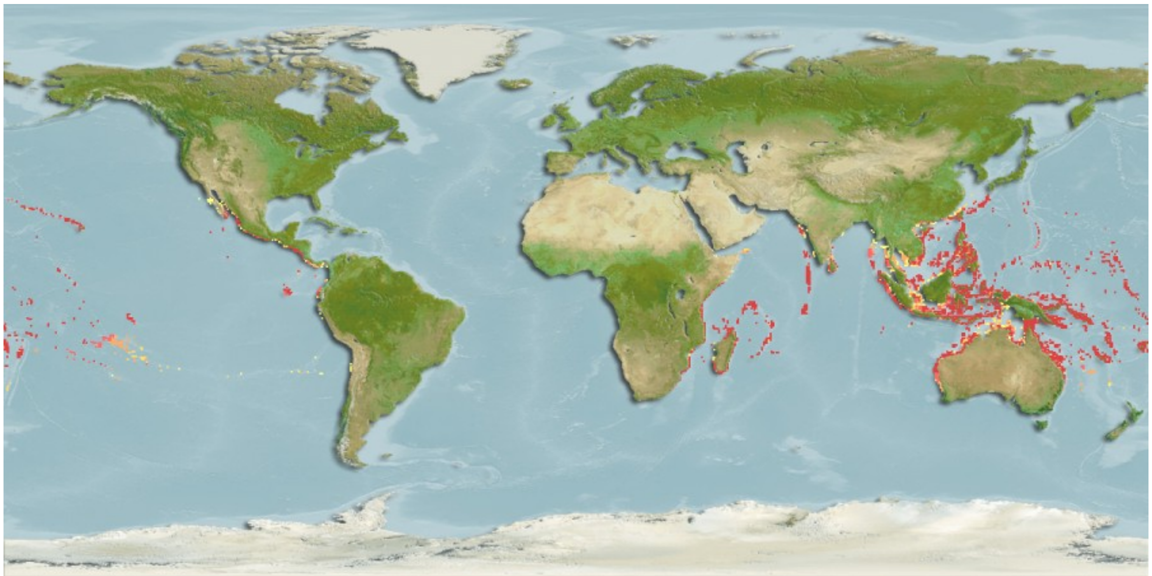
Habitat: *C. atripectoralis* is closely associated with branching corals, especially *Acropora* and *Pocillipora*, for shelter, reproduction (i.e., laying eggs on coral branches), and recruitment (Lewis 1998, Randall 2005, Wilson et al. 2008a). It inhabits a depth range of 2 to 15 meters (Randall 2005). Lewis (1998) noted that larvae settle directly into live corals and are found in close association with living coral throughout their adult life. Wilson et al. (2008a) found that adults prefer branching corals while juveniles use branching corals and plating corals almost equally. Wilson et al. (2008a) also found a significant relationship between live coral cover and the abundance of *C. atripectoralis*, where branching coral cover explained a large percentage (35%) of the variance in abundance, more than in any of the 22 pomacentrid species examined.

C. atripectoralis was also the only species for which there was a significant relationship between branching coral cover and juvenile abundance (Wilson et al. 2008a).

Natural History: Adults are typically seen in foraging aggregations above corals where they feed on zooplankton including copepods, shrimp nauplii, and mysids (Randall 2005).

Threats: *C. atripectoralis* is threatened by the loss and degradation of its coral reef habitat due to temperature-induced mass bleaching events and ocean acidification, as well as direct harms to essential functions due to ocean warming and acidification, such as reduced aerobic capacity. In a study of the effects of coral loss on reef fish abundance on the Great Barrier Reef, Lewis (1998) documented significant in declines in abundance of *C. atripectoralis*, which was one of five species out of 53 fish species studied that declined, attributed to its close association with live coral. Similarly, in a study of coral reef fish abundance following coral loss at seven reefs on the Great Barrier Reef where coral cover declined by more than fifty percent, Wilson et al. (2008a) found that *C. atripectoralis* consistently showed larger declines in abundance following coral loss than pomacentrids with wider niche breadths. Finally, Nilsson et al. (2009) found that elevated temperature significantly reduced the respiratory scope of *C. atripectoralis*, where its scope for oxygen uptake decreased from 300% at 29ºC to 178% at 33ºC.

Figure 4. Map of the distribution of *Chromis atripectoralis.*
Source: Computer Generated Map for *Chromis atripectoralis* (un-reviewed). www.aquamaps.org, version of Aug. 2010. Web.



3. *Chromis viridis* (Blue-green damselfish)

Species Description: Blue-green shading to white ventrally with a blue line from the front of the snout to the eye; the courting and nesting male changes hue to yellow with black posteriorly (Randall 2005). It reaches 10 cm (Randall 2005). *C. viridis* looks similar to *C. atripectoralis*, but is generally smaller and lacks the black area on the inside of the pectoral fin base (Fishbase.org).

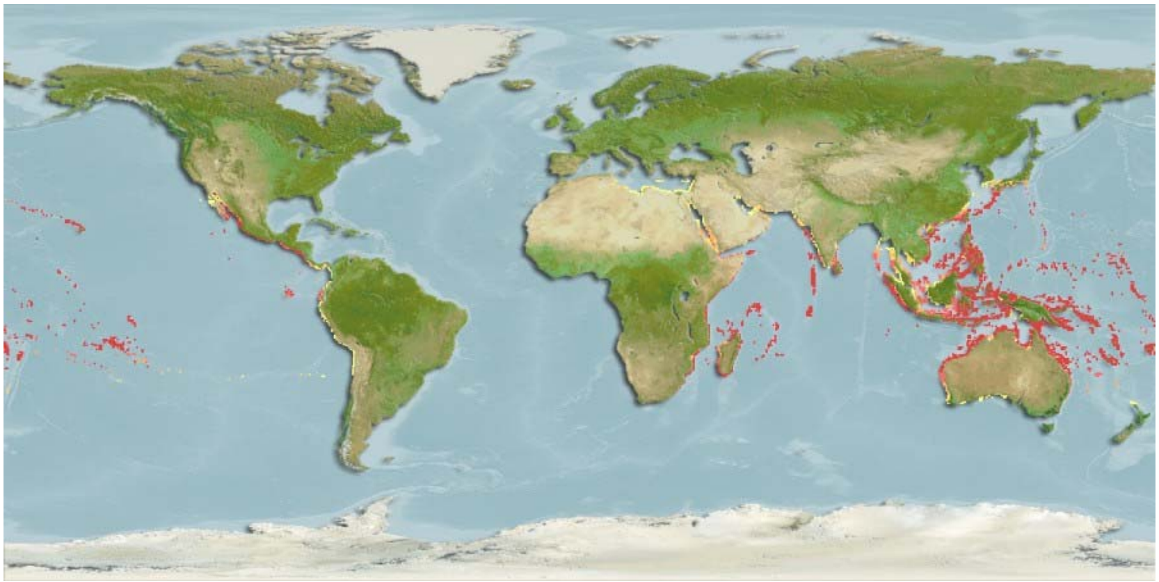
Distribution: This species occurs in the Red Sea and east coast of Africa to the Line Islands and the Tuamotu archipelago; Ryukyu Islands to Great Barrier Reef and New Caledonia (Randall 2005). It occurs in U.S. territorial waters in American Samoa and the Northern Mariana Islands (Allen 1991: Table 2). See Figure 5.

Habitat: *C. viridis* inhabits shallow, protected inshore and lagoon reefs. Many studies have reported its close association with a narrow set of branching coral species as juveniles and adults (Allen 1991, Booth 2002, Lecchini et al. 2005, Froukh and Kochzius 2008, Ben-Tzvi et al. 2008). Allen (1991) reported that this species is closely associated with branching corals, especially *Acropora,* at a depth range of 1 to 12 m. Randall (2005) noted that it is "typically found on shallow coral reefs in protected waters, retreating to spaces among branching corals for shelter, the young in more closely branched corals." According to Fishbase, adults are found in large aggregations above thickets of branching *Acropora* corals in sheltered areas such as subtidal reef flats and lagoons, and juveniles are closely tied to individual coral heads (Fishbase.org). On the Great Barrier Reef, Booth (2002) found that *C. viridis* recruits aggregate in large numbers on isolated branching tabulate acroporid corals. A study in the Red Sea found that larvae preferred to settle on the branching coral *Acropora eurystoma* (Ben-Tzvi et al. 2008). Lewis (1998) reported that the larvae settle directly into live corals and are found in close association with living coral throughout the adult life. Finally, Lecchini et al. (2005) found that *C. viridis* juveniles and adults shelter in a relatively narrow range of branching coral habitats.

Natural History: *C. viridis* is a planktivore that feeds manly on copepods and crustacean larvae in large aggregations above branching corals, staying close to the shelter of reef (Randall 2005). Males prepare the nest for spawning which is shared with several females (Fishbase.org). Spawning involves a large number of eggs which hatch in 2 to 3 days. Males guard the nest and ventilate the fertilized eggs with their caudal fins, and feed on eggs which do not hatch (Fishbase.org). This species is oviparous, with distinct pairing during breeding. Its eggs are demersal and adhere to the substrate (Fishbase.org). The larvae settle directly onto live coral (Lewis 1998). In the Red Sea, Ben-Tsivi et al. (2008) found that larvae have a strong preference for settling on *Acropora* coral and prefer colonies that host consepecific adults and juveniles.

Threats: *C. viridis* is threatened by the loss and degradation of its coral reef habitat due to temperature-induced mass bleaching events and ocean acidification, as well as direct harms to essential functions due to ocean warming and acidification, such as reduced aerobic capacity. *C. viridis* is closely associated with live, branching coral habitat, and its populations have been shown to decline sharply following the loss of live coral habitat from bleaching and other disturbances (Nilsson et al. 2009). In a survey of a portion of the Great Barrier Reef that experienced bleaching during the 1997-1998 mass bleaching event, Booth and Beretta (2002) found that numbers of *C. viridis* collapsed after the bleaching event, attributed to the destruction of live coral coverage combined with a takeover of the coral structures by algae. In a study of the effects of coral loss on reef fish abundance on the Great Barrier Reef, Lewis (1998) documented significant in declines in abundance of *C. viridis*, which was one of five species out of 53 fish species studied that declined, attributed to its close association with live coral. Finally, habitat choice experiments found that *C. viridis* larvae preferred live coral cover to degraded and algal-covered coral, and that the density of late-stage larvae was significantly lower on degraded and algal-covered coral compared to live coral (Feary et al. 2007b). The global marine aquarium trade may also pose a threat from overharvest, as *C. viridis* was the most commonly imported marine ornamental fish into the U.S. in 2005, with more than 900,000 individuals imported per year (Rhyne et al. 2012).

Figure 5. Map of the distribution of *Chromis viridis*.
Source: Computer Generated Map for *Chromis viridis* (un-reviewed). www.aquamaps.org, version of Aug. 2010. Web.



4. *Dascyllus albisella* (Hawaiian dascyllus)

Species Description: *D. albisella* is a small, deep-bodied fish reaching a maximum length of approximately five inches (Stevenson 1963a). Adults are pale or dark with white spots fading with age, while juveniles are black with a white spot on each side and a turquoise spot on the head (Stevenson 1963a). Nuptial coloration of males is white except for the black caudal and anal fins and margin of the dorsal fin (Randall 1985).

Distribution: This species is endemic to the U.S., occurring at the Hawaiian Islands and Johnston Atoll (Danilowicz 1995a, Asoh and Yoshikawa 2002). See Figure 6.

Habitat: The juveniles *D. albisella* are closely associated with branching coral heads, particularly pocilloporids, for recruitment and shelter (Allen 1991, Booth 1992, Randall 1985). Adults are found in very shallow, protected water over coral and rocky bottoms, where they form feeding aggregations over reefs (Randall 1985, Allen 1991). Larval fish settle primarily on branching coral heads, and recruits and juveniles remain closely associated with branching coral heads until individuals reach a size threshold (65-70 mm) and become sexually mature (Booth 1992). Allen (1991) noted that juveniles usually associate with small heads of *Pocillopora* coral or sometimes with sand-dwelling anemone *Heteractis malu*. Similarly, Randall (1985) reported that young fish often take shelter among branches of *Pocillopora* coral and are occasionally commensal with the sand-dwelling anemone *Marcanthia cookei.* Stevenson (1963b) also described the occasional commensal relationship with *Marcanthia cookei.*

Natural history:

Territorial behavior: Adults school above the reef as planktivores (Booth 1992), although males guard nest territories against conspecific males throughout the year, and chase potential egg predators out of the nest during courtship and when guarding eggs (Danilowicz 1995a). Recruits and juveniles remain closely associated with branching coral heads until individuals attain 65-70 mm TL, when they become sexually mature and join the nearby but spatially distinct adult population of schooling adults (Booth 1992). This spatial segregation of juvenile and adult stages contrasts with other species of *Dascyllus* in which social groups of adults and juveniles cohabit in the same coral heads (Booth 1992). *D. albisella* also exhibits an occasional commensal relationship with the anemone *Maracanthia cookei*; it appears to engage in an acclimatization process with the anemone after which it takes refuge among the tentacles of the anemone host (Stevenson 1963b).

Diet and foraging behavior: *D. albisella* is a planktivore that feeds on the larvae of mysids, shrimp, and crab; copepods; pelagic tunicates; and other zooplankton, sometimes as much as 20 feet above the bottom (Randall 1985). A study at Johnston Atoll found that damselfish spent most (more than 85%) of their time feeding, and ate primarily caridean zoea and copepods (Mann and Sancho 2007). When currents were low, damselfish foraged higher in the water column where the flux of plankton was generally greater, with larger damselfish foraging higher than small damselfish; however, at higher current speeds, damselfish foraged closer to the bottom, with less of a difference in the depth distribution of differently sized damselfish (Mann and Sancho 2007).

Reproductive ecology: In Hawaii, spawning occurs throughout the year, with the highest spawning activity from June to September or October (Asoh and Yoshikawa 2002). Spawning occurs cyclically and synchronously every 5 to 7 days, with all spawning in each cycle occurring over 2 to 3 days (Asoh 2003). Increasing temperature appears to cue the initiation of spawning, while decreasing temperature cues the cessation of spawning (Danilowicz 1995b). Both males and females spawn repeatedly over a season with various partners (Asoh and Yoshikawa 2002). Fecundity increases exponentially with body length, and spawning duration increases linearly with body length (Asoh 2003). Males prepare nests by removing debris from an area of the reef substratum, and attract females to their nesting sites with signal jumps (lateral body undulations while swimming up and down in the water column) and vocalizations (Danilowicz 1995a). Females travel to a male nest site and lay an average of 25,000 demersal eggs during the morning hours (Danilowicz 1995a). Eggs are tended, aerated, and guarded by the male until embryos hatch which occurs at dusk on the fourth day of development at water temperatures ranging between 26ºC to 29ºC (Asoh 2003).

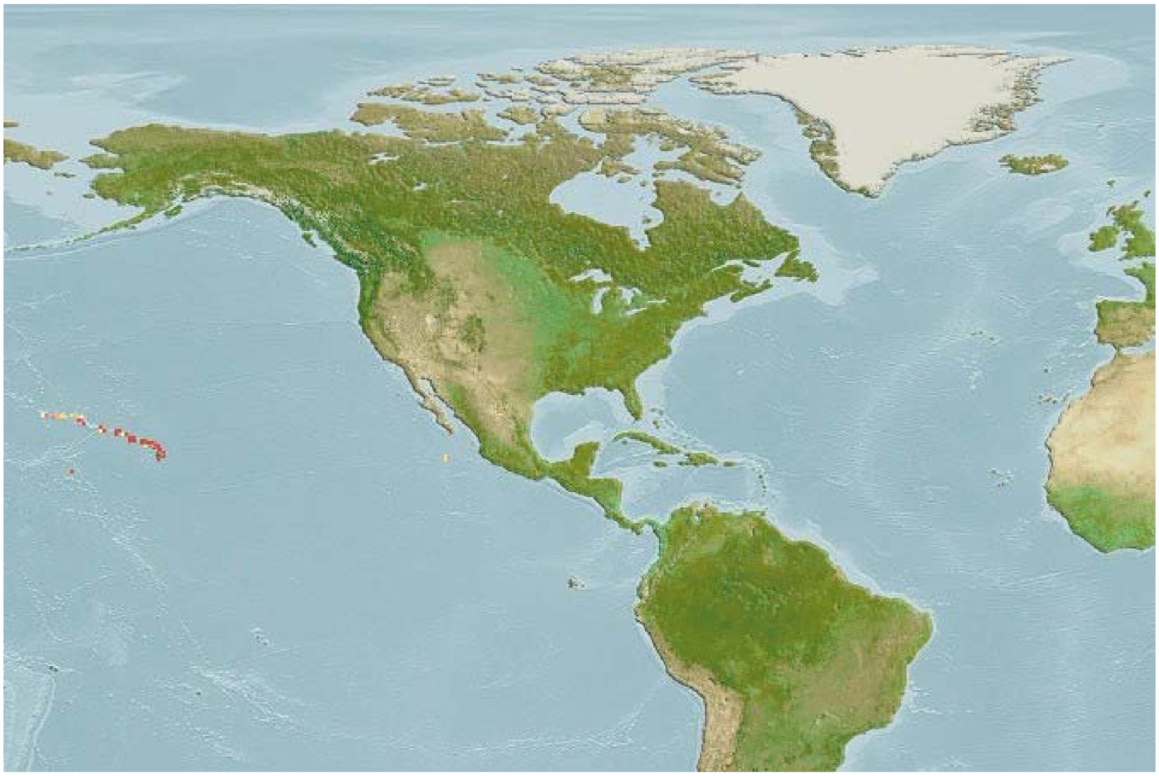
Settlement generally occurs after a 25 to 29-day pelagic phase, and larvae settle primarily on branching coral heads (Booth 1992). Larvae appear to prefer coral heads with larger groups of conspecific juveniles and use chemical and visual cues to find settlement habitat (Booth 1992). Newly settled fish are 10-15 mm total length, and appear to reach maturity at 65 to 70 mm TL

(Booth 1992). This species is thought to be intermediate between pre- and post-maturational sex-changing species (Danilowicz 1995a). Groups of juveniles live in transient groups on small, branching coral heads and do not interact with adults (Booth 1992).

Lifespan: The lifespan for *D. albisella* has been estimated at 11 years (Buston and García 2007).

Threats: *D. albisella* is threatened by the loss and degradation of its coral reef habitat due to temperature-induced mass bleaching events and ocean acidification, as well as direct harms to essential functions due to ocean warming and acidification. The dependence of *D. albisella* on live branching *Pocillopora* coral for larval settlement and juvenile habitat makes this species highly vulnerable to coral habitat loss and degradation due to temperature-induced mass coral bleaching and ocean acidification. As found by DeMartini et al. (2010), *Pocillopora* corals in the Hawaiian Islands are the most susceptible to bleaching and sedimentation stressors, which threatens *D. albisella* juveniles which are obligately associated with these branching corals. *D. albisella* is also a preferred target species in Hawaii for the global marine aquarium trade (Stevenson et al. 2011).

Figure 6. Map of the distribution of *Dascyllus albisella*.
Source: Computer Generated Map for *Dascyllus albisella* (un-reviewed). www.aquamaps.org, version of Aug. 2010. Web.



5. *Dascyllus reticulatus* (Reticulate dascyllus or Two-stripe damselfish)

Species Description: Palish blue-grey, the edges of the scales narrowly black, with a blackish bar anteriorly on the body continuing as a broad outer border on the spinous portion of dorsal fin; attains 8.5 cm in length (Randall 2005).

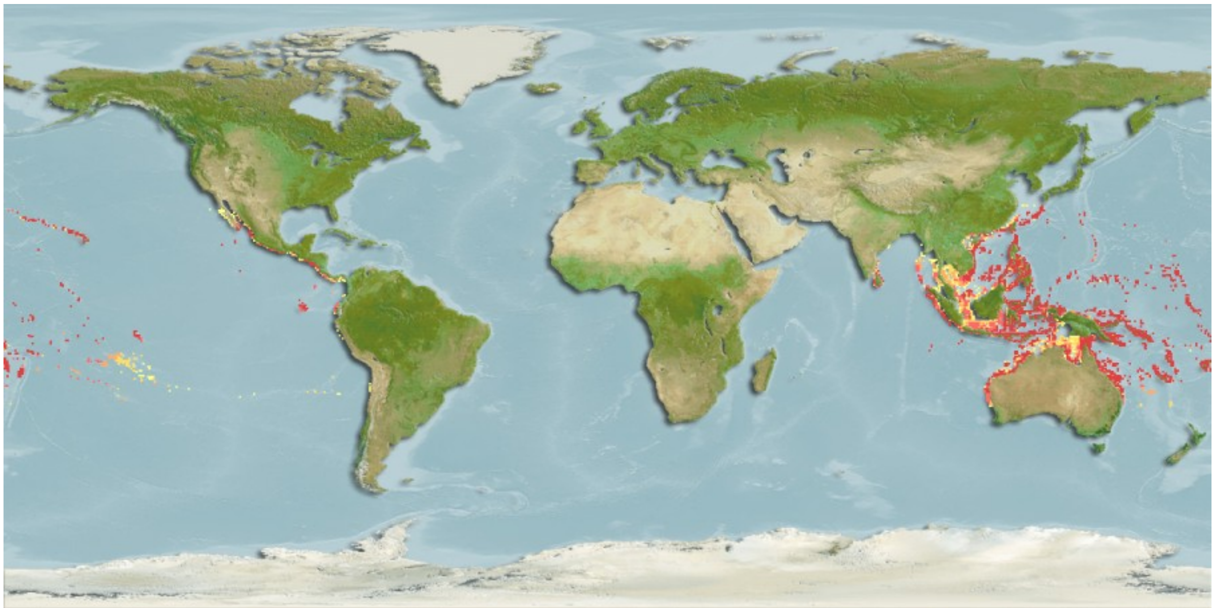
Distribution: This species occurs in Southern Japan to Great Barrier Reef, Lord Howe Island, and New Caledonia, east to the Tuamotu archipelago, Pitcairn Islands, and Micronesia; western Australia, Cocos-Keeling Islands, and Adnaman Sea in Indian Ocean (Randall 2005). It occurs in U.S. territorial waters in American Samoa and the Northern Mariana Islands (Allen 1991: Table 2). See Figure 7.

Habitat: *D. reticulatus* is closely associated with branching corals as juveniles and adults (Wilson et al. 2008a). Randall (2005) reported that this species occurs in colonies closely associated with branching corals which it uses for shelter, at depths of 1 to 50 meters (Randall 2005). Allen (1991) noted that it inhabits branching coral on onshore and offshore reef at a depth range 1 to 50 m. Sweatman (1983) found that it occurs in social groups in living colonies of branching corals, with *Pocillopora damicornis* a commonly inhabited species at Lizard Island on the Great Barrier Reef. Lewis (1998) and Sweatman (1983) noted that the larvae settle directly into live branching corals and are found in close association with living coral throughout the adult life.

Natural History: *D. reticulatus* is a planktivore that feeds on zooplankton a short distance above corals (Sweatman 1983, Randall 2005). The species is hermaphroditic and the male prepares a nest by cleaning a rock or coral surface with its mouth where the female will attach demersal eggs (Fishbase.org). Larval *D. reticulatus* prefer to settle on live coral where conspecifics predominate (Sweatman 1983).

Threats: *D. reticulatus* is threatened by the loss and degradation of its coral reef habitat due to temperature-induced mass bleaching events and ocean acidification, as well as direct harms to essential functions due to ocean warming and acidification, such as reduced aerobic performance and swimming ability. In a study of the effects of coral loss on reef fish abundance on the Great Barrier Reef, Lewis (1998) documented significant in declines in abundance of *D. reticulatus*, which was one of five species out of 53 fish species studied that declined, attributed to its close association with live coral. Johansen and Jones (2011) found that warmer water temperatures at 3ºC over ambient levels reduced the aerobic performance and swimming ability of *D. reticulatus*. Specifically, *D. reticulatus* showed a 28% reduction in critical swimming speed, a 51% reduction in gait transition, a 43% reduction in maximum oxygen consumption, and a 49% reduction in aerobic scope. The study concluded that “these results demonstrate the physiological inability of several coral reef fishes to uphold the performance required within the habitats they currently occupy. A possible outcome is therefore a reduction in species abundance and a shift in distribution ranges as these species are forced into more sheltered habitats befitting their reduced swimming performance or into higher latitudes where their performance can be upheld.”

Figure 7. Map of the distribution of *Dascyllus reticulatus.*



Source: Computer Generated Map for *Dascyllus reiculatus* (un-reviewed). www.aquamaps.org, version of Aug. 2010. Web.

6. *Microspathodon chrysurus* (Yellowtail damselfish or Jewel damselfish)

Species description: *M. chrysurus* has a deep blue to black body with a yellow tail, although body color can vary from brownish to violet shades (Bohlke and Chaplin 1993). The head, back, and dorsal fin of adults are accented with bright blue speckles. Juveniles are dark blue with a translucent to whitish tail that changes to yellow with maturity, with brilliant blue dots covering the body and fins (Bohlke and Chaplin 1993). They are larger members of the Pomacentridae that can attain up to 20 cm in length (Sikkel and Kramer 2006). They are distinct in having movable rather than fixed teeth on the jaws, and a deep notch in the pre-orbital bordering the upper jaw (Bohlke and Chaplin 1993).

Distribution: This species occurs in the Western Atlantic including Bermuda, southern Florida, and the Caribbean Sea (Allen 1991) to the northern coast of South America, including the Gulf of Mexico (Bohlke and Chaplin 1993). It occurs in U.S. and U.S. territorial waters of Florida, the U.S. Virgin Islands, and Puerto Rico. See Figure 8.

Habitat: The yellowtail damselfish inhabits shallow coral reefs at a depth range of 1 to 10 m (Allen 1991). Many sources have described the association of adults with *Acropora* corals and juveniles with *Millepora* corals. Deloach (1999) reported that adult yellowtail damselfish inhabit elkhorn corals where females are found over widespread territories on reef crests while males typically occupy deeper zones of elkhorn coral rubble, and juveniles settle in shallow patch reefs where they may inhabit blade fire coral *Millepora complanata.* Allen (1991) noted that juveniles are usually seen among branches of the yellow stinging coral *Millepora.* A study of microhabitat use in the U.S. Virgin Islands found that adults are significantly associated with *Acropora* corals and total live coral cover (Tolimieri 1998). Similarly, Wilkes et al. (2008) found an association of adults with live staghorn coral in the Dry Tortugas. Finally, Sikkel and Kramer (2006) described yellowtail damselfish habitat as the spur and groove zone of fringing reefs and adjacent offshore patch reefs to a depth of about 10 meters.

Natural History:

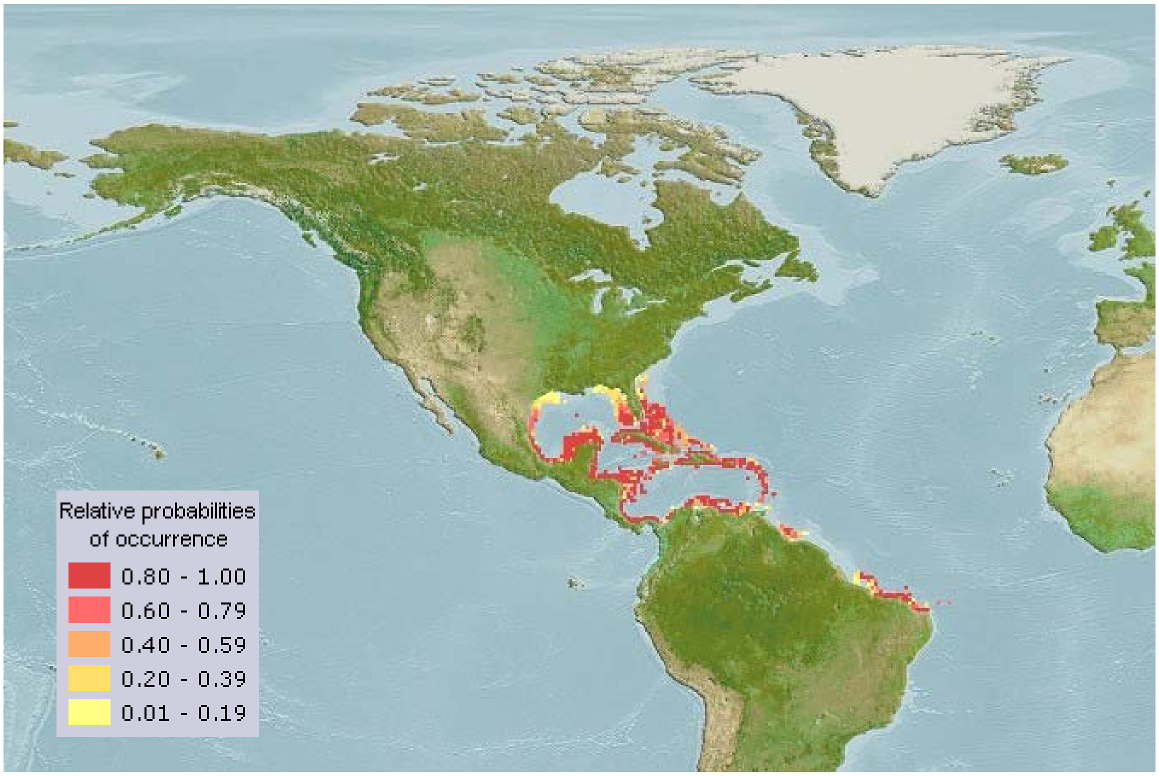
Territorial behavior: Adults of both sexes occupy non-overlapping, often contiguous territories and defend them vigorously against conspecifics and less so against other species, except near male nests (Sikkel and Kramer 2006). Territories of females tend to be shallower and closer to shore than those of males (Sikkel and Kramer 2006). Yellowtail damselfish territories are often superimposed on the territories of smaller-bodied damselfish species, especially the dusky and threespot damselfish; yellowtail damselfish may benefit by pre-empting food from these smaller cohabitants and from the aggressive defense provided by these smaller fish against other herbivores (Gutierrez 1998, Deloach 1999). Female yellowtail damselfish spend nearly all their time within the boundaries of their territories but occasionally leave to visit cleaners or to spawn (Sikkel and Kramer 2006).

Diet and foraging behavior: The yellowtail damselfish is a territorial grazer that feeds primarily on epilithic microalgae and associated fauna (Sikkel and Kramer 2006). It harvests epiphytic diatoms that grow on its protected algal lawns (Deloach 1999). It is also a facultative corallivore (Cole et al. 2008) that is known to feed on the polyps of *Millepora* corals (Species-identification.org).

Reproductive ecology: The spawning period for *M. chrysurus* peaks for four to five weeks in February to March and again in July to August, with decreasing activity in autumn (Deloach 1999). Females spawn at regular three-day intervals from three days before to three weeks after the full moon during the extended breeding season (Sikkel and Kramer 2006). Males prepare one to five nest sites within their territories by cleaning off algae and invertebrates from the surfaces of coral rubble (often elkhorn) or the bases of gorgonian stems (Pressley 1980, Deloach 1999). Spawning occurs to the first one to three hours of daylight (Sikkel and Kramer 2006). Males begin courtship shortly before sunrise, and male color changes to pale gray (Pressley 1980). Females leave their territories, swimming typically from 5 to 120 meters to male territories (Sikkel and Kramer 2006). When spawning, females swim in circles and figure eight patterns around the nest, and lay adhesive eggs an expanding patch over the nest surface (Pressly 1980). The male remains nearby and occasionally passes his ventral surface close to the newly deposited eggs to fertilize them (Pressley 1980). The male occasionally swims away from the nest site to chase away conspecifics, forage, and drive away potential egg predators (Pressley 1980). Spawning generally takes an hour to complete, after which the female returns to her territory and the male guards the eggs from predators (Pressley 1980). The size of egg patches averages 76 cm^2 and contains approximately 19,000 eggs (Pressley 1980). The embryos hatch approximately five days after fertilization (Pressley 1980), and enter a 21 to 27 day pelagic phase, after which the larvae settle in shallow patch reefs often inhabited by *Millepora* coral which makes up much of the early diet (Deloach 1999).

Threats: *M. chrysurus* is threatened by the loss and degradation of its coral reef habitat due to temperature-induced mass bleaching events and ocean acidification, as well as direct harms to essential functions due to ocean warming and acidification. The dependence of juveniles on live branching *Millepora* and adults on live branching *Acropora* corals makes this species highly vulnerable to coral habitat loss and degradation due to temperature-induced mass coral bleaching and ocean acidification. In addition, *M. chrysurus* inhabits Caribbean coral reef ecosystems which are among the most threatened in the world (Wilkinson and Souter 2008, Eakin et al. 2010, Burke et al. 2011, Jackson et al. 2012), putting this species at added risk. The Caribbean has the largest proportion of corals in IUCN high-extinction-risk categories (Carpenter et al. 2008). Average live coral cover has declined from 50 to 60% in the 1970s to just 8% today, with the major causes of coral decline including bleaching, overfishing, pollution, and disease (Jackson et al. 2012). These massive losses of live coral cover were dominated by the devastation of branching acroporid corals in the region. The Caribbean also suffered massive coral losses in response to the 2005 mass bleaching event which led to the bleaching of over 80% of corals, where more than 40% died at many sites (Eakin et al. 2010). The ongoing, high-magnitude threats and low resiliency of Caribbean coral reefs pose significant threats to *M. chrysurus*.

Figure 8. Map of the distribution of *Microspathodon chrysurus*.
Source: Computer Generated Map for (un-reviewed). www.aquamaps.org, version of Aug. 2010. Web.



7. *Plectroglyphidodon dickii* (Blackbar devil or Dick's damselfish)

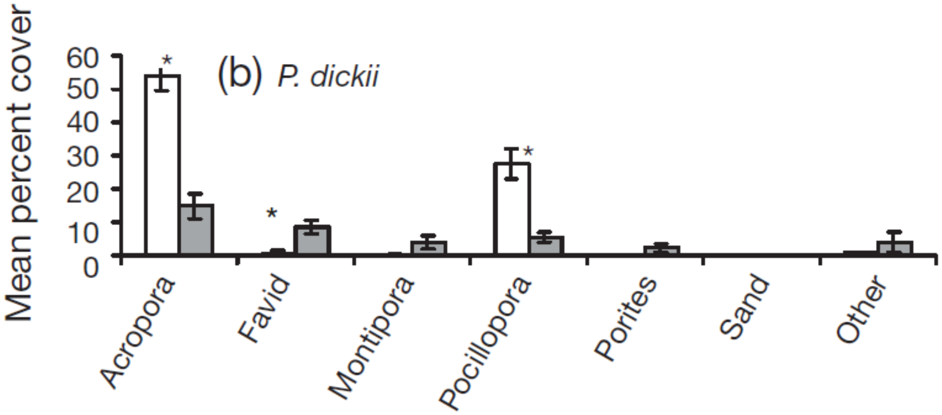
Species Description: Light brown damselfish, with a sharp black band toward the posterior end, with a white back end and tail; it attains 8.5 cm (Randall 2005).

Distribution: This species occurs throughout most of Indo-Pacific from the Red Sea and east coast of Africa to islands of French Polynesia, and Ryukyu Islands to New South Wales and Lord Howe (Randall 2005). It occurs in U.S. territorial waters in American Samoa (Allen 1991: Table 2). See Figure 10.

Habitat: Adults inhabit coral-rich areas of clear lagoon and seaward reefs, and many sources report their association with live, branching *Pocillopora* and *Acropora* corals. Randall (2005) reported that it is typically found in colonies taking refuge in branching corals in isolated reefs on sand bottom of lagoons and bays in less than 12 meters. Allen (1991) reported that it is associated with branching corals on inshore and lagoon reefs, with a depth range 1 to 12 meters. Wilson et al. (2008a) also reported the strong association of adults with branching corals.

A detailed study of habitat use in Papua New Guinea found that *P. dickii* territories were dominated by live coral (~80% cover) and had twice the coral cover than found in areas adjacent to their territories and more than twice the amount of live coral than found in territories of two other sympatric damselfish species (*Plectroglyphidodon lacrymatus* and *Stegastes nigricans*). *P. dickii* territories also contained significantly higher proportional cover of live *Acropora* and *Pocillopora* corals than outside their territories (Figure 9).

Figure 9. Mean percent cover of live coral assemblages inside and adjacent to (outside) territories of *P. dickii*. Significance at $p < 0.007$ using multiple Kruskal-Wallis tests. Error bars = 95% CL.



Natural History:

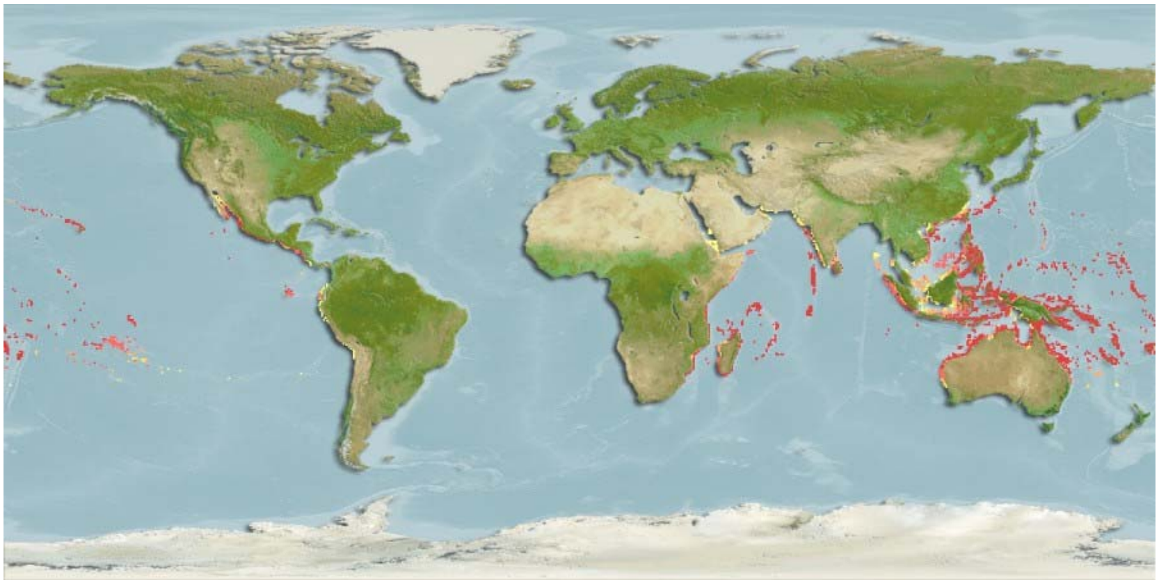
Territorial behavior: *P. dickii* establishes territories on live, branching corals, mainly of the genera *Acropora* and *Pocillopora* (Jones et al. 2006). They defend their territories against large, mobile grazers and corallivores, which can enhance the survivorship of corals that might otherwise be damaged by scraping and excavating grazers and can result to an increase in overall coral diversity (Emslie et al. 2012).

Diet and foraging behavior: P. dickii is a territorial grazer that maintains distinct algal farms on small sections of live coral branches and feeds primarily on filamentous algae and associated small benthic invertebrates within its territories (Walsh et al. 2012). It is classified as an "intensive farmer" that weeds unwanted algal species from its territories, encourages the growth of preferred algae, and defend its territories very aggressively against large, mobile grazers and corallivores (Emslie et al. 2012). Analysis of stomach contents of *P. dickii* in Papua New Guinea found that it is primarily a herbivore that selectively consumes a narrow range of algae found in its territory, with diatoms as the most important food source, followed by gelids, blue-green algae, and the red alga *Polysiphonia* (Jones et al. 2006). It is also considered a facultative corallivore (Cole et al. 2008), although Jones et al. (2006) reported that it actively kills coral polyps without consuming them to increase the area for algal growth inside its territory.

Reproductive ecology: P. dickii uses small sections of the coral branches as nest sites (Jones et al. 2006).

Threats: *P. dickii* is threatened by the loss and degradation of its coral reef habitat due to temperature-induced mass bleaching events and ocean acidification, as well as direct harms to essential functions due to ocean warming and acidification. In a study of the effects of loss of live coral cover on the abundance of coral reef fish in Fiji, Wilson et al. (2008b) found that *P. dickii* declined significantly following loss of *Acropora* coral cover. Similarly, large declines in hard coral cover at Hoskyn Island in the Great Barrier Reef were followed by declines of *P. dickii* (Australian Institute of Marine Science 2012).

Figure 10. Map of the distribution of *Plectroglyphidodon dickii*.
Source: Computer Generated Map for *Plectroglyphidodon dickii* (un-reviewed). www.aquamaps.org, version of Aug. 2010. Web.



8. *Plectroglyphidodon johnstonianus* (Blue-eye damselfish)

Species Description: *P. johnstonianus* has a pale yellowish-gray body with a very broad black posterior bar, a head that is gray dorsally shading to yellowish-gray ventrally, a violet-blue line on the sides of snout, and lavender scales rimming the eyes (Randall 2005).

Distribution: This species occurs on the East coast of Africa to Hawaiian Islands, French Polynesia, and Pitcairn Islands; Ryukyu Islands and Ogasawara islands to Great Barrier Reef, Lord Howe, and Norfolk Island (Randall 2005). It occurs in U.S. waters in Hawaii, and U.S. territorial waters in American Samoa and the Northern Mariana Islands (Allen 1991: Table 2). See Figure 11.

Habitat: *P. johnstonianus* inhabits passes and outer reefs at a depth of 2 to 12 meters, and is often associated with *Acropora* or *Pocillopora* corals (Allen 1991). Randall (2005) reported that this species inhabits exposed coral reefs, generally 2 to 18 meters deep, and is closely associated with corals of the genera *Acropora* and *Pocillopora*, especially *P. eydouxi;* its territory may be a single large head of this coral or adjacent heads. It is considered highly dependent on live coral for shelter, food, and reproduction (Cole et al. 2008, Emslie et al. 2012).

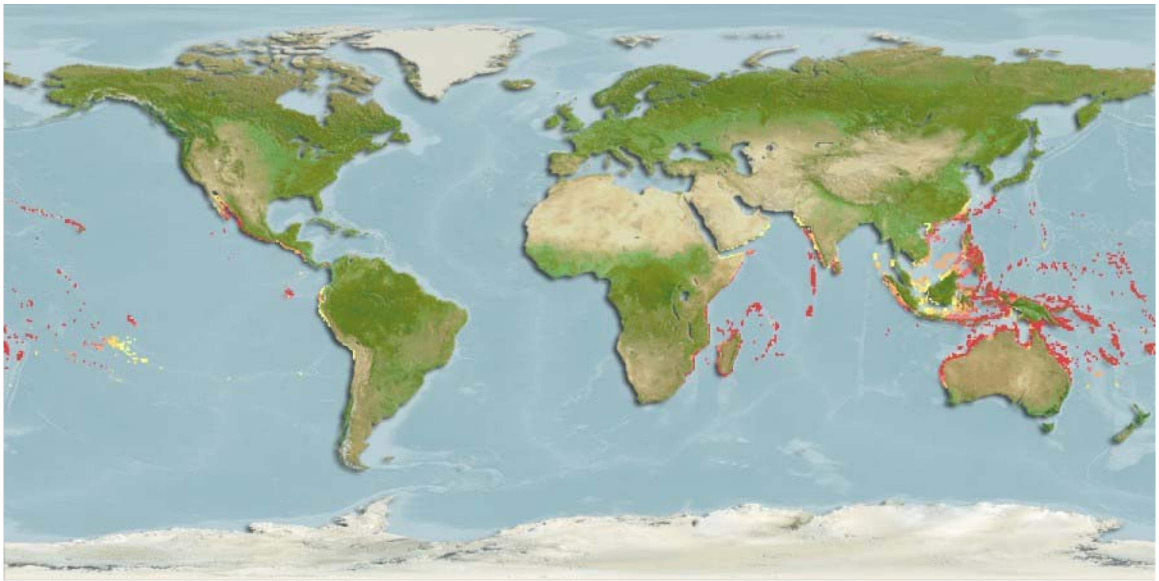
Natural History:

Territorial behavior: *P. johnstonianus* is classified as an ‘‘indeterminate’’ algal farming species, which refers to species that defend their territories less aggressively than other farming species, weed less intensively, and have more subtle effects on the composition of algal assemblages within their territories (Emslie et al. 2012).

Diet and foraging ecology: The blue-eye damselfish is one of two damselfish species that is considered an obligate corallivore (Cole et al. 2008). *Acropora* and *Montipora* corals are major dietary items, while *Pocillopora* and *Porites* corals are moderately used dietary items (Cole et al. 2008). Randall (2005) similarly reported that this species feeds mainly on coral polyps.

Threats: *P. johnstonianus* is threatened by the loss and degradation of its coral reef habitat due to temperature-induced mass bleaching events and ocean acidification, as well as direct harms to essential functions due to ocean warming and acidification. *P. johnsonianus* is highly dependent on live coral cover, and particularly branching *Acropora* and *Pocillopora* corals for shelter, food, and reproduction. Several studies have found that *P. johnstonianus* declines in abundance in response to loss of live coral cover. A meta-analysis of studies that documented the effects of disturbance-mediated coral loss on coral reef fishes found that *P. johnstonianus* declined significantly and consistently across multiple study locations (Wilson et al. 2006). In a study of the effects of loss of live coral cover on the abundance of coral reef fish in Fiji, Wilson et al. (2008b) found that *P. johnstonianus* declined significantly following loss of *Acropora* coral cover. Similarly, large declines in hard coral cover at Hoskyn Island in the Great Barrier Reef were followed by declines of *P. johnstonianus* (Australian Institute of Marine Science 2012).

Figure 11. Map of the distribution of *Plectroglyphidodon johnstonianus*.
Source: Computer Generated Map for *Plectroglyphidodon johnstonianus* (un-reviewed). www.aquamaps.org, version of Aug. 2010. Web.



Part Two: The Petitioned Reef Fish Are Threatened or Endangered Based on the Endangered Species Act Listing Factors

I. Criteria for Listing Species as Endangered or Threatened under the Endangered Species Act and the Time Horizon for the Foreseeable Future

A. Listing Criteria

Under the ESA, 16 U.S.C. § 1533(a)(1), NMFS is required to list a species for protection if it is in danger of extinction or threatened by possible extinction in all or a significant portion of its range. In making such a determination, NMFS must analyze the species' status in light of five statutory listing factors, relying "solely on the best scientific and commercial data available," 16 U.S.C. § 1533(b)(1)(A):

> (A) the present or threatened destruction, modification, or curtailment of its habitat or range;
> (B) overutilization for commercial, recreational, scientific, or educational purposes;
> (C) disease or predation;
> (D) the inadequacy of existing regulatory mechanisms;
> (E) other natural or manmade factors affecting its continued existence.

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

A species is "endangered" if it is "in danger of extinction throughout all or a significant portion of its range" due to one or more of the five listing factors. 16 U.S.C. § 1531(6). A species is "threatened" if it is "likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range." 16 U.S.C. § 1531(20).

Under the ESA, a "species" includes any species, subspecies, or a "distinct population segment" of a vertebrate species. 16 U.S.C. § 1532(16). As noted in the individual species accounts above, each of the petitioned fish is recognized as a distinct species.

B. The Foreseeable Future

While the ESA does not define the "foreseeable future," NMFS must use a definition that is reasonable, that ensures protection of the petitioned species, and that gives the benefit of the doubt regarding any scientific uncertainty to the species. As explained below, the minimum time period that meets these criteria for the petitioned pomacentrid reef fish species is the period through 2100.

Because climate change and ocean acidification are foremost threats to the petitioned fish species, NMFS should consider the timeframes used in climate modeling. 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."

As a primary example of the feasibility of a 100-year time frame, the Intergovernmental Panel on Climate Change ("IPCC"), a foremost world scientific authority on climate change, has provided climate change projections through 2100 under a range of plausible emissions scenarios. For the Fourth Assessment, the IPCC performed an unprecedented internationally coordinated climate change experiment using 23 models by 14 modeling groups from 10 countries to project future climate conditions. This large number of models ranging from simple to complex, running the same experiments, provided both quantification of future climate conditions through the end of this century and the uncertainty of the results. As stated by the IPCC itself, climate projections run through the end of the 21st century under different emissions scenarios, and accompanied by the range of uncertainty, were provided in its 2007 Fourth Assessment Report specifically because of their policy-relevance:

> Advances in climate change modeling now enable best estimates and *likely* assessed uncertainty ranges to be given for projected warming for different emission scenarios. Results for different emission scenarios are provided explicitly in this report to avoid loss of this policy-relevant information. Projected global average surface warmings for the end of the 21st century (2090–2099) relative to 1980–1999 are shown in Table SPM.3. These illustrate the differences between lower and higher SRES emission scenarios, and the projected warming uncertainty associated with these scenarios.
> (IPCC 2007: 13).

The IPCC Fifth Assessment will use a suite of new emissions pathways, the Representative Concentration Pathways (RCPs), which similarly provide updated, high-resolution datasets for emissions trajectories and impacts analysis through 2100, with estimates of uncertainty (Vuuren et al. 2011). The RCPs are supplemented with extensions (Extended Concentration Pathways) which allow climate modeling experiments through the year 2300 (Vuuren et al. 2011), further demonstrating the foreseeability of climate change impacts.

As additional support for defining the foreseeable future through the year 2100, NMFS determined in three recent listing-related decisions that the year 2100 represents the most appropriate, science-based timeline for the foreseeable future for considering climate change threats to species. In December 2010, NMFS defined the foreseeable future for assessing climate change impacts to species as the end of the 21st century in the proposed listing rules for the ringed seal and bearded seal, which are both threatened by climate change:

> [N]MFS scientists have revised their analytical approach to the foreseeability of threats and responses to those threats, adopting a more threat specific approach based on the best scientific and commercial data available for each respective threat. For example, because the climate projections in the Intergovernmental Panel on Climate Change's (IPCC's) *Fourth Assessment Report* extend through the end of the century (and we note the IPCC's *Fifth Assessment Report*, due in 2014, will extend even farther into the future), we used those models to assess impacts from climate change through the end of the century. We continue to recognize that the farther into the future the analysis extends, the greater the inherent uncertainty, and we incorporated that limitation into our assessment of

> the threats and the species' response. For other threats, where the best scientific and commercial data does not extend as far into the future, such as for occurrences and projections of disease or parasitic outbreaks, we limited our analysis to the extent of such data. We believe this approach creates a more robust analysis of the best scientific and commercial data available. (75 Fed. Reg. 77477, 77497; emphasis added)

The proposed rules for these seals further conclude that climate projections through the end of the 21st century "currently form the most widely accepted version of the best available data about future conditions":

> NMFS scientists have recognized that the physical basis for some of the primary threats faced by the species had been projected, under certain assumptions, through the end of the 21st century, and that these projections currently form the most widely accepted version of the best available data about future conditions. In our risk assessment for ringed seals, we therefore considered all the projections through the end of the 21st century to analyze the threats stemming from climate change. (75 Fed. Reg. 77482, 77503; emphasis added).

The status reviews for these seals also concluded that the end of the 21st century is the foreseeable future for climate change threats for these species (Cameron et al. 2010, Kelly et al. 2010). For example, the ringed seal status review stated:

> In this review of ringed seal population status, the BRT recognized that the physical basis for some of the primary threats faced by the species have been projected, under certain assumptions, through the end of the 21st century, and that these projections currently form the most widely accepted version of the best available information about future conditions. Therefore, in the risk assessment that follows, the BRT used the full 21st-century projections as the basis for the foreseeability of threats stemming from climate change. (Kelly et al. 2010: 43; emphasis added).

NMFS reaffirmed this determination for "foreseeable future" in its peer-reviewed *Status Review Report* for 82 coral species petitioned under the U.S. Endangered Species Act (Brainard et al. 2012). In this decision, NMFS scientists stated that "the year 2100 was used as the time horizon for this risk evaluation because this century was the timeframe over which the BRT [biological review team] had access to reasonable, scientifically vetted predications of key threats and their impacts," and "the IPCC collection of CO_2 emissions scenarios and climate models provided projections with adequate confidence to the year 2100":

> There is no formal definition in the U.S. Endangered Species Act for the term "foreseeable future" as used in the legal description of "threatened." However, agency policy guidance recommends linking the time horizon for the risk evaluation to the timeframe over which it is possible to scientifically predict the impact of the threats (U.S. Department of Interior, 2009). Both the petition and the BRT determined that climate change and ocean acidification probably pose

> significant extinction risk threat to corals. The year 2100 was used as the time horizon for this risk evaluation because this century was the timeframe over which the BRT had access to reasonable, scientifically vetted predictions of key threats and their impacts (see Chapter 3). In particular, the BRT determined that the Intergovernmental Panel on Climate Change (IPCC) collection of CO_2 emissions scenarios and climate models provided projections with adequate confidence to the year 2100 to reasonably support their use in evaluating Critical Risk Thresholds for the candidate coral species. Much of the scientific information available on the potential impacts of ocean acidification on corals has likewise been based on IPCC CO_2 emission scenarios and model projections. (Brainard et al. 2012: 100).

These determinations by NMFS scientists that climate projections through 2100 represent the best-available science on the foreseeable future for assessing climate change threats provides a solid basis for applying the same foreseeable future timeline to the petitioned fish species.

Additionally, in planning for species recovery, NMFS and its sister agency, the U.S. Fish and Wildlife Service, routinely consider a foreseeable future threshold of roughly 100 years, particularly when addressing climate change considerations. For example, the agencies jointly stated in the second revision of their recovery plan for the Northwest Atlantic population of loggerhead sea turtles:

> Research has identified sea level rise as one of the most important potential impacts of global climate change. The best available science indicates that *by 2100* South Florida seas will be approximately 20 inches higher than they were in 1990 (IPCC 2001). An increase of this magnitude would drastically alter the coastline, changing the extent, quality, and location of sandy beaches available for loggerhead nesting. In the short term, even small changes in sea level could be expected to exacerbate beach erosion and increase artificial beach/dune alterations meant to protect coastal properties. (NMFS and USFWS 2008 at II-53 (emphasis added)).

Furthermore, following a recent workshop on reclassification criteria for endangered large whale species, NMFS has adopted a policy guideline that "[a] large cetacean species shall no longer be considered endangered when, given current and projected conditions, the probability of quasi-extinction is less than 1% in 100 years" (NMFS 2005 at III-1, Recovery Plan for the North Atlantic Right Whale).

Perhaps most importantly, the time period NMFS uses in its listing decision must be long enough so that actions can be taken to ameliorate the threats to the petitioned species and prevent extinction. *See Defenders of Wildlife v. Norton*, 258 F.3d 1136, 1142 (9th Cir. 2001) (quoting legislative history noting that the purpose of the ESA is "not only to protect the last remaining members of [a listed] species but to take steps to insure that species which are likely to be threatened with extinction never reach the state of being presently endangered"). Slowing and reversing impacts from anthropogenic greenhouse gas emissions, a primary threat to all of the petitioned fish species, will be a long-term process for a number of reasons, including the long-lived nature of carbon dioxide and other greenhouse gases and the lag time between emissions and climate changes. NMFS must include these considerations in its listing decision.

For all these reasons, the use of less than the year 2100 as the "foreseeable future" in this rulemaking would clearly be unreasonable, frustrate the intent of Congress to have imperiled species protected promptly and proactively, and fail to give the benefit of the doubt to the species as required by law.

As detailed throughout, neither anthropogenic greenhouse gas emissions nor any of the other threats to the petitioned pomacentrid reef fish species are speculative or too far in the future to understand or address. These new and modern threats are already here, and the impacts are already manifesting in coral populations. Urgent action, including listing under the ESA and dramatic cuts in greenhouse gas emissions levels, is needed now to ensure that these species do not become extinct in the foreseeable future. As described below, each of the petitioned fish species qualifies for listing under the ESA.

II. Each of the Petitioned Fish Species Is Threatened or Endangered Based on the Five Endangered Species Act Listing Factors

This petition requests the protection of the orange clownfish (*Amphiprion percula*) and seven damselfish occurring in U.S. waters that are highly threatened by ocean warming and ocean acidification that degrade and destroy their coral reef and anemone habitat, and pose direct threats to the fish by impairing their sensory capabilities, behavior, aerobic capacity, swimming ability, and reproduction. As summarized by coral reef fish scientists, "[c]limate change is rapidly emerging as the single greatest threat to coral-reef fishes" (Pratchett et al. 2008). Some of the petitioned species also face threats from global marine aquarium trade. Each of these threats is described under the five ESA listing factors below.

A. The Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range (Listing Factor A): Coral Reef Habitat Loss Due to Anthropogenic Greenhouse Gas Pollution Resulting in Ocean Warming and Ocean Acidification Threatens the Petitioned Reef Fish

The petitioned pomacentrid reef fish are habitat specialists that directly depend on live corals for survival, including shelter, reproduction, recruitment, and food. The continued survival of these fish species is severely threatened by the loss and degradation of their coral reef and anemone habitat due to ocean warming and ocean acidification resulting from anthropogenic greenhouse gas emissions. While overfishing, disease, coastal development, and pollution also pose significant local threats to the petitioned species by degrading and destroying their coral reef habitat in many regions (Burke et al. 2011, Brainard et al. 2012), ocean warming and ocean acidification pose global threats across the range of the petitioned pomacentrid species and are predicted to lead to the collapse of coral reef habitats within this century (Hoegh-Guldberg et al. 2007, Veron et al. 2009).

As detailed below, the coral reefs on which the petitioned reef fish rely are in crisis. Nearly 20% of the world's coral reefs have already been lost, and approximately one-third of all zooxanthellate reef-building coral species as at risk of extinction according to the IUCN (Carpenter et al. 2008, Veron et al. 2009). Adding to historic threats, coral reef ecosystems are now experiencing significant harms from ocean warming and ocean acidification that are resulting in mass bleaching events and reduced calcification of coral reefs. The impacts of ocean warming and ocean acidification to coral reefs will worsen within this century as greenhouse gas pollution continues to rise, threatening coral reef ecosystems with collapse (Veron et al. 2009). Studies projecting the impacts of ocean warming on corals indicate that the majority of the world's corals will be subjected to recurring mass bleaching events at frequencies from which they will be unable to recover by the 2020s or 2030s, in the absence of significant thermal adaptations by corals and their symbionts (Hoegh-Guldberg 1999, Sheppard 2003, Donner et al. 2005, 2007, Donner 2009, Burke et al. 2011). Due to the synergistic impacts of ocean acidification, mass bleaching, and other impacts, reefs are projected to experience "rapid and terminal" declines worldwide at an atmospheric CO_2 concentration of 450 ppm, which is expected to occur before mid-century (Veron et al. 2009). As summarized by coral reef scientists, "reefs are likely to be the first major planetary-scale ecosystem to collapse in the face of climate changes now in progress" (Veron et al. 2009: 1433). Indeed, NMFS in its 2011 *Status Review Report of 82 Candidate Coral Species Petitioned Under the U.S. Endangered Species Act* concluded that ocean warming and ocean acidification pose an extinction risk for many coral species, concluding that 56 of the 82 corals reviewed are "likely" or "more likely than not" to fall below a critical risk threshold for extinction by 2100 due primarily to threats from ocean warming, ocean acidification, and disease (Brainard et al. 2012).

Among coral-dependent reef fishes, damselfish are considered particularly vulnerable to declines in coral cover due to their widespread use of bleaching-prone, branching corals such as acroporid and pocilloporid corals as habitat and settlement sites (Pratchett et al. 2008, Bonin 2012). Numerous studies have documented significant declines in damselfish and anemonefish populations following coral depletion due to bleaching and other disturbances (Pratchett et al. 2008, Wilson et al. 2006). For example, a meta-analysis that examined the effects of disturbance-mediated coral loss on reef fishes found that 11 of 14 (79%) pomacentrid species considered declined in abundance following coral depletion (Wilson et al. 2006: Figure 2). The effects of coral habitat loss are most severe for habitat specialists that shelter and recruit to live coral and diet specialists that feed on coral because they are not able to adjust their habitat use and diet as preferred corals become scarcer (Wilson et al. 2006, 2008, Pratchett et al. 2008, Graham 2011, Bonin 2012). The seven petitioned damselfish are habitat specialists that rely on bleaching-prone branching corals as juveniles and/or adults for recruitment, reproduction, shelter, and/or food, while the orange clownfish depends on coral reef-habitat and anemones that are also affected by bleaching. The high coral-reef dependency of the petitioned fish species places them at significant risk of extinction due to the accelerating loss and degradation of their essential coral reef habitat due to ocean warming and ocean acidification, in combination with local threats. As summarized by Feary et al. (2007a), "[f]ish species with a strong preference for, or an obligate association with, live coral are likely to decline in both abundance and diversity in response to increased mortality of their host corals."

The following section summarizes the best-available science on the threats posed by ocean warming and ocean acidification to the petitioned pomacentrid reef fish, including: (1) a brief overview of climate change and current greenhouse gas emission trends, (2) the threats posed by ocean warming to the petitioned species' coral reef and anemone habitat, and (3) the threats posed by ocean acidification to the petitioned species' coral reef habitat.

1. Overview of climate change and current greenhouse gas emission trends

There is a strong, international scientific consensus that climate change is occurring, is primarily human-induced, and threatens human society and natural systems. The Intergovernmental Panel on Climate Change (IPCC) in its 2007 Fourth Assessment Report expressed in the strongest language possible its finding that global warming is occurring: "Warming of the climate system is unequivocal, as is now evident from observations of increases in global average air and ocean temperatures, widespread melting of snow and ice, and rising global average sea level" (IPCC 2007). The IPCC concluded that most of the recent warming observed has been caused by human activities(IPCC 2007). In the United States, the U.S. Global Change Research Program in its 2009 report *Climate Change Impacts in the United States* stated that "global warming is unequivocal and primarily human-induced" and "widespread climate-related impacts are occurring now and are expected to increase," (USGCRP 2009) and the U.S. National Research Council concluded that "[c]limate change is occurring, is caused largely by human activities, and poses significant risks for—and in many cases is already affecting—a broad range of human and natural systems" (NRC 2010). Based on observed and expected harms from climate change, in 2009 the U.S. Environmental Protection Agency concluded that greenhouse gas pollution endangers the health and welfare of current and future generations (Federal Register 74: 66496-66546).

Due to U.S. and international failures to adequately address climate change, greenhouse gas emissions are increasing at an accelerating pace. The atmospheric concentration of CO_2 (the dominant greenhouse gas driving the observed changes in the Earth's climate) reached an annual average of ~392 parts per million (ppm) in 2011[1] compared to the pre-industrial concentration of ~280 ppm. The current CO_2 concentration has not been exceeded during the past 800,000 years and likely not during the past 15 to 20 million years (Denman et al. 2007, Tripati et al. 2009). Atmospheric CO_2 emissions have risen particularly rapidly since the 2000s (Raupach et al. 2007, Friedlingstein et al. 2010, Global Carbon Project 2010). The global fossil fuel CO_2 emissions growth rate was 1.1% per year during 1990-1999 compared with 3.1% during 2000-2010, and since 2000, this growth rate has largely tracked or exceeded the most fossil-fuel intensive emissions scenario projected by the IPCC (i.e., A1FI) (Figure 12) (Raupach et al. 2007, McMullen and Jabbour 2009, Richardson et al. 2009, Global Carbon Project 2010, 2011). The CO_2 emissions growth rate fell slightly in 2009 due largely to the global financial and economic crisis; however, the decrease was less than half of what was expected and was short-lived (Friedlingstein et al. 2010). Global CO_2 emissions increased by 5.9% in 2010 resulting in a record 33 billion tons of CO_2 emitted (Global Carbon Project 2011, Olivier et al. 2011), and CO_2 emissions reached another record high in 2011.[2] As summarized by NMFS: "While many international, national, and local initiatives have sought to reduce the growth in greenhouse gas emissions, recent emissions growth and an apparent lack of significant international political

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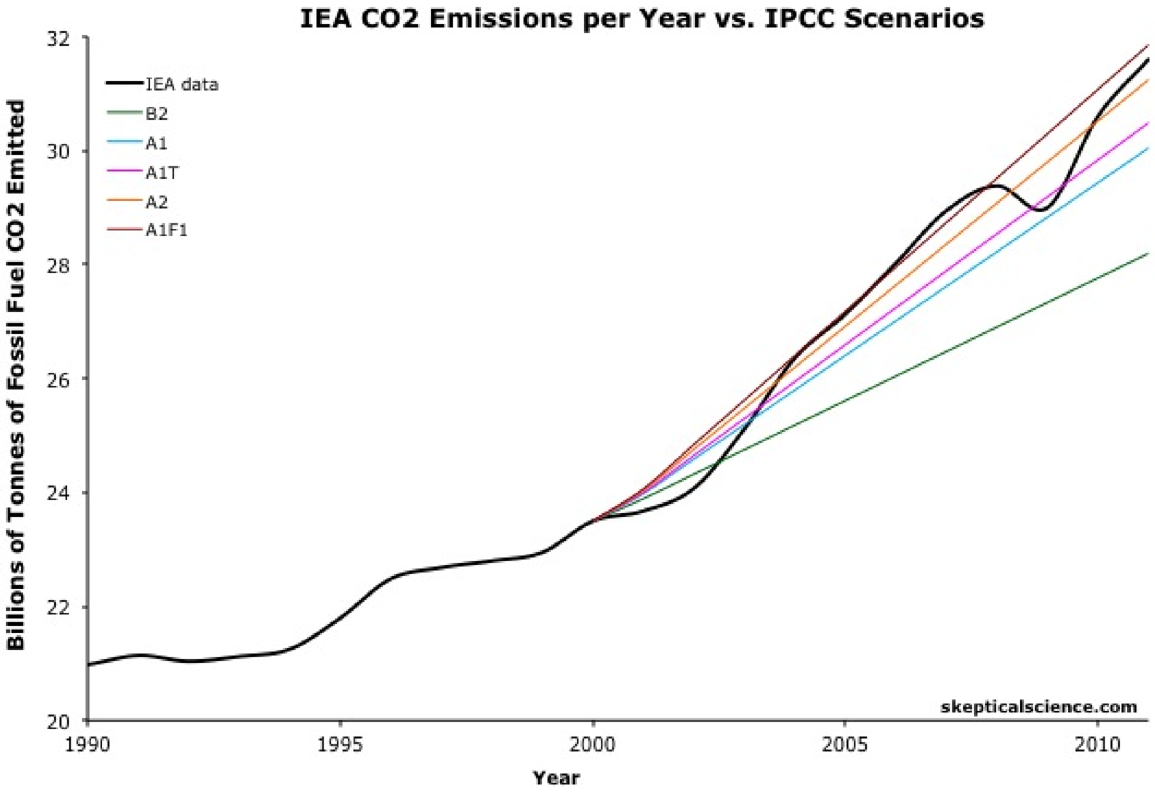
[1] *See* National Oceanic and Atmospheric Administration, *Trends in Atmospheric Carbon Dioxide*, www.esrl.noaa.gov/gmd/ccgg/trends/global.html (last visited June 5, 2012).

[2] *See* International Energy Agency, *Global carbon-dioxide Emissions Increase by 1.0 Gt in 2011 to Record High,* http://www.iea.org/newsroomandevents/news/2012/may (last visited June 5, 2012).

action to control emissions to date have resulted in an acceleration of CO_2 emissions at or above the worst-case scenario used in the IPCC's Third and Fourth Assessment Reports" (Brainard et al. 2012: 25).

Figure 12. Observed CO_2 emissions relative to the IPCC SRES emission scenarios. Source: http://www.skepticalscience.com/iea-co2-emissions-update-2011.html



Current atmospheric concentrations of greenhouse gases are already resulting in severe and significant climate change impacts that are projected to worsen as emissions rise (USGCRP 2009). Key changes include warming temperatures, the increasing frequency of extreme weather events, rapidly melting glaciers, ice sheets, and sea ice, rising sea levels, and a thirty percent increase in surface ocean acidity (USGCRP 2009). Many climate change risks are substantially greater than assessed by the IPCC in 2007 (Fussel 2009, Smith et al. 2009), and the rates of many negative changes are tracking the worst case scenarios projected by the IPCC (Rogers and Laffoley 2011). As summarized by Fussel (2009), "many risks are now assessed as stronger than in the AR4 [IPCC Fourth Assessment Report], including the risk of large sea-level rise already in the current century, the amplification of global warming due to biological and geological carbon-cycle feedbacks, a large magnitude of 'committed warming' currently concealed by a strong aerosol mask, substantial increases in climate variability and extreme weather events, and the risks to marine ecosystems from climate change and ocean acidification."

2. Ocean warming threatens the coral reef and anemone habitat of the petitioned pomacentrid reef fish

Ocean warming threatens the petitioned reef fish by degrading and destroying their coral reef habitat. As detailed below, (a) tropical ocean temperatures are warming significantly in the range of the petitioned damselfish and anemonefish, (b) ocean warming is degrading and destroying the coral reef and anemone habitat of the petitioned reef fish due to the increasing frequency and intensity of mass bleaching events, and (c) the loss and degradation of habitat resulting from bleaching has adverse impacts on damselfish and anemonefish populations that include population declines, reduced recruitment, increased susceptibility to predation, lower fish growth rates, and increased competition for shrinking coral habitat.

a. Current and predicted trends in ocean warming

Global average surface temperatures increased by ~0.2°C per decade over the past 30 years, with most of this added energy absorbed by the world's oceans (Hoegh-Guldberg and Bruno 2010). As a result, all major ocean basins have warmed over the last 50 years at nearly every latitude from the surface to the deep ocean to at least 2000 m (Levitus et al. 2012). The heat content of the upper 700 m of the global ocean increased by 14 × 1022 J since 1975, and the average temperature in the upper layers of the ocean rose by ~0.6°C over the past 100 years (Hoegh-Guldberg and Bruno 2010).

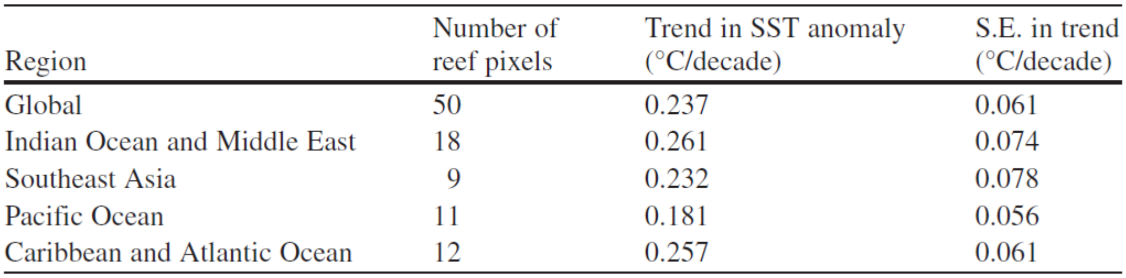
In its 2011 *Status Review Report* for 82 candidate coral species, NMFS summarized the science on ocean warming and concluded that ocean temperatures have increased up to 0.2°C–0.4°C per decade in waters around many coral reefs, the frequency of warm-season temperature extremes has increased during the previous two decades, and a further temperature increase in waters around coral reefs of 2.8°C–3.6°C is expected during this century, depending on the ocean basin, under a business-as-usual emissions scenario (Brainard et al. 2012):

> It is well documented that the Earth's temperature has increased during the previous century to levels that had not been reached in over 1,000 years (Chapman and Davis, 2010; IPCC, 2007b). As a result of rising atmospheric greenhouse gases, average global surface air temperatures have already warmed by ~ 0.74°C during the century from 1906 to 2005 (IPCC, 2007b). The rate of warming has also increased from 0.07°C per decade over the past 100 years to 0.13°C per decade for the most recent 50 years (IPCC, 2007b), including increases of up to 0.2°C–0.4°C per decade in waters around many coral reefs (Strong et al., 2008). In particular, the decades of the 1980s and 1990s exhibited a rapid temperature rise to levels above the average for the previous millennium. This average value is the mean of many local measurements, some of which of course are much higher than the average. The global trend in average temperature is reflected in a number of long-term records of sea surface temperature (SST). More important than the global average temperature from a coral perspective, the frequency of warm-season temperature extremes increased during the previous two decades and is inducing more frequent episodes of mass coral bleaching and associated mortality (Eakin et al., 2009).

> As rapid as the warming in the previous century has been, the warming in the 21st century is predicted to be greater, even if emissions of anthropogenic greenhouse gases were to cease today (IPCC, 2007b). This “committed” warming is greater than 1.0°C globally averaged temperature (IPCC, 2007b) and greater than 0.5°C in most ocean waters around coral reefs (Table 3.2.1; Donner, 2009). Of course, CO_2 emissions continue to rise, currently at or exceeding the worstcase scenarios used in the IPCC AR4 assessment (Fig. 3.2.3), and the stabilization of atmospheric CO_2 levels is considered unlikely for several decades at least. This worst-case, fossil-fuel-intensive A1FI scenario assumes no substantial changes in emission policies or technologies. At that rate of CO_2 emissions, a further temperature increase in waters around coral reefs of 2.8°C–3.6°C is expected during this century, depending on the ocean basin. At a minimum, ocean temperatures around coral reefs will rise more than 1°C this century, but this would require drastic changes in greenhouse gas emissions across the globe. While significant CO_2 emission reduction would decrease both the ultimate amount and rate of global warming and effects on corals, thus far little movement toward reducing emissions has occurred through international agreements or U.S. legislation. Therefore, reductions are considered unlikely in the short term. Even the most aggressive actions to reduce emissions will only slow ocean warming, not prevent it. Natural forces put into place by anthropogenic climate change will continue to influence coral reefs for at least 100 years (Solomon et al. 2009). (Brainard et al. 2012: 28-29)

Consistent with these findings, Eakin et al. (2009) reported that regional sea surface temperature trends averaged 0.24°C per decade during 1985 to 2006 at a number of reef sites across the globe (Table 1). The IPCC Fourth Assessment Report found that warming rates in the Indian Ocean increased significantly between 1970 and 1999, exceeding 0.2°C per decade in some regions (Bindoff et al. 2007). Chollett et al. (2012) reported that average sea surface temperature over the Caribbean Sea and the southeastern Gulf of Mexico increased by 0.29ºC per decade during 1985 to 2009, with the greatest warming in the tropical Atlantic, eastern Caribbean Sea, central Gulf of Mexico, and Loop Current region.

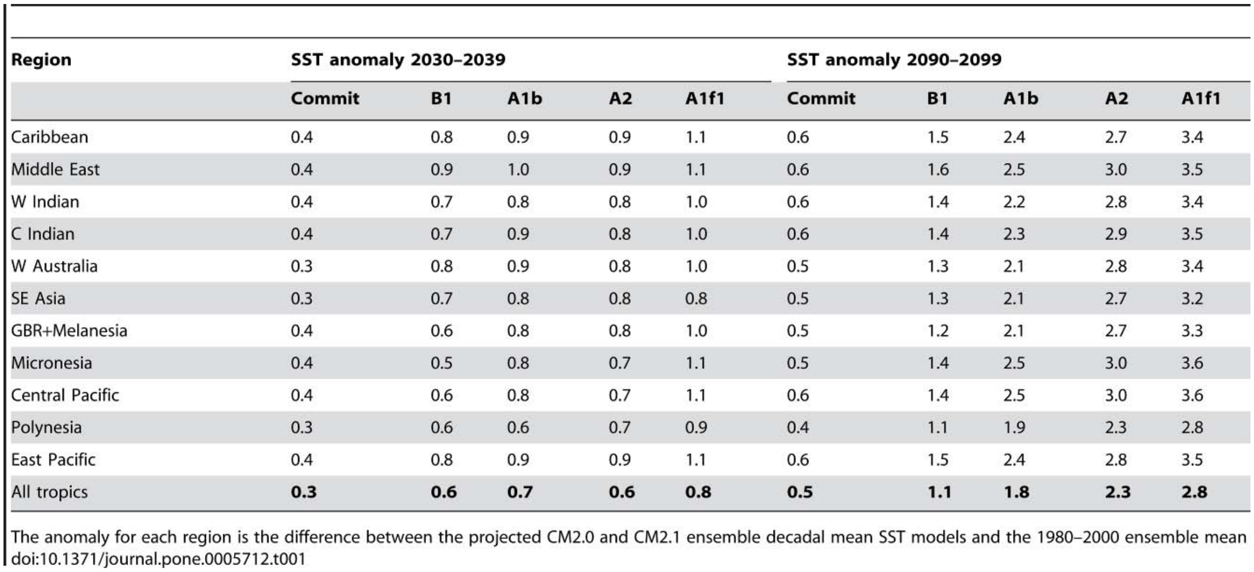
Table 1. Trends in SST anomalies across five geographic regions from the Pathfinder reanalysis of the 22-year satellite record, 1985–2006. The SST anomaly values are averaged across specific reef pixels within each region and for each year.
Source: Eakin et al. (2009): Table 4.1.



Region	Number of reef pixels	Trend in SST anomaly (°C/decade)	S.E. in trend (°C/decade)
Global	50	0.237	0.061
Indian Ocean and Middle East	18	0.261	0.074
Southeast Asia	9	0.232	0.078
Pacific Ocean	11	0.181	0.056
Caribbean and Atlantic Ocean	12	0.257	0.061

Sea surface temperature will increase significantly in this century, with warming up to 3.6°C under a high emissions scenario (Donner 2009). Specifically, under a low B1 emissions scenario, sea surface temperature is predicted to increase by 0.5 to 0.9°C by 2030-2039 and by 1.1 to 1.6°C by 2090-2099 (Donner 2009). Under a high A1FI emissions scenario similar to what the world has been tracking, sea surface temperature is predicted to increase by 0.8 to 1.1°C by 2030-2039 and by 2.8 to 3.6°C by 2090-2099 (Donner 2009). Sea surface temperature will increase by 0.4 to 0.6°C by 2090–2099 just due to the physical warming commitment from greenhouse gas accumulated through the year 2000, meaning the even if all emissions ceased in 2000 that sea surface temperature would still rise significantly (Table 2) (Donner 2009).

Table 2. Annual mean SST anomaly (°C) averaged across each ocean province where the columns show the warming to which the Earth is already committed and the warming expected under different SRES emissions scenarios.
Source: Donner (2009): Table 1.



Region	SST anomaly 2030–2039					SST anomaly 2090–2099				
	Commit	B1	A1b	A2	A1f1	Commit	B1	A1b	A2	A1f1
Caribbean	0.4	0.8	0.9	0.9	1.1	0.6	1.5	2.4	2.7	3.4
Middle East	0.4	0.9	1.0	0.9	1.1	0.6	1.6	2.5	3.0	3.5
W Indian	0.4	0.7	0.8	0.8	1.0	0.6	1.4	2.2	2.8	3.4
C Indian	0.4	0.7	0.9	0.8	1.0	0.6	1.4	2.3	2.9	3.5
W Australia	0.3	0.8	0.9	0.8	1.0	0.5	1.3	2.1	2.8	3.4
SE Asia	0.3	0.7	0.8	0.8	0.8	0.5	1.3	2.1	2.7	3.2
GBR+Melanesia	0.4	0.6	0.8	0.8	1.0	0.5	1.2	2.1	2.7	3.3
Micronesia	0.4	0.5	0.8	0.7	1.1	0.5	1.4	2.5	3.0	3.6
Central Pacific	0.4	0.6	0.8	0.7	1.1	0.6	1.4	2.5	3.0	3.6
Polynesia	0.3	0.6	0.6	0.7	0.9	0.4	1.1	1.9	2.3	2.8
East Pacific	0.4	0.8	0.9	0.9	1.1	0.6	1.5	2.4	2.8	3.5
All tropics	**0.3**	**0.6**	**0.7**	**0.6**	**0.8**	**0.5**	**1.1**	**1.8**	**2.3**	**2.8**

The anomaly for each region is the difference between the projected CM2.0 and CM2.1 ensemble decadal mean SST models and the 1980–2000 ensemble mean
doi:10.1371/journal.pone.0005712.t001

b. Adverse impacts of ocean warming on the petitioned species' coral reef and anemone habitat

Ocean warming is increasing the severity of frequency of mass coral bleaching events that degrade and destroy coral reef habitat. As summarized by Pratchett et al. (2008), "climate-induced coral bleaching has caused massive devastation to coral-reef habitats and is predicted to become more frequent and more severe in coming decades." Similarly, NMFS in its *Status Review Report of 82 Candidate Coral Species Petitioned Under the U.S. Endangered Species Act* concluded that "ocean warming and related impacts of climate change have already created a clear and present threat to many corals, that this will continue in the future" (Brainard et al. 2012). NMFS further highlighted the growing threats to corals around the globe from ocean warming: "A recent independent global analysis of threats to coral reefs found that thermal stress, while regionally variable, has indeed influenced corals in all reef regions around the globe. Bleaching and mortality of adult coral colonies have been the most visible signs of the effects of climate change, but it is also likely that ocean warming will have detrimental effects on virtually every life history stage of reef corals as impaired fertilization, developmental abnormalities, mortality, and impaired settlement success of larval phases have all been documented" (internal citations omitted) (Brainard et al. 2012: 29). The section below briefly summarizes recent scientific studies on the impacts of bleaching on coral reef ecosystems.

i. Overview of coral bleaching

Warming of the tropical oceans has raised the baseline sea surface temperature to levels where coral reefs live much closer to their upper thermal limits and are more vulnerable to thermal stress and bleaching (Eakin et al. 2009). Coral bleaching occurs when ocean temperatures exceed summer maxima by 1° to 2°C for 3 to 4 weeks, causing zooxanthellate corals expel their endosymbiotic dinoflagellates (*Symbiodinium* spp.) which they rely on for energy and growth (Hoegh-Guldberg et al. 2007). Coral bleaching and mortality become progressively worse as thermal anomalies intensify and lengthen (Hoegh-Guldberg et al. 2007).

Coral bleaching affects corals and coral reefs by causing direct mortality; lowering reproductive capacity; reducing growth, calcification rates, and repair capabilities following bleaching; making corals more susceptible to disease and other stressors; and altering community structure (Hoegh-Guldberg 1999, Rosenberg and Ben-Haim 2002, Fischlin et al. 2007, Baker et al. 2008). One of the most direct effects of bleaching is that affected corals tend to die at greater rates, where the mortality of corals following a bleaching event is generally proportional to the length and extent to which temperatures rise above summer maxima for any locality (Hoegh-Guldberg 1999). Increased temperatures and bleaching can also reduce coral reproductive capacity by inhibiting spawning and lowering the number of reproductive propagules after bleaching events. *Id.* This impairment of reproductive capacity can slow the rate at which coral populations can re-establish themselves by lowering the number of available recruits. *Id.* Numerous studies have found that reef-building corals that undergo bleaching have reduced growth, calcification and repair capabilities following bleaching (Hoegh-Guldberg 1999, Fischlin et al. 2007). Bleaching can also make corals more vulnerable to other stressors, leading to increases in coral diseases and the breakdown of the reef framework by bioeroders (Baker et al. 2008). Overall, corals that survive and recover their dinoflagellate symbionts after mild thermal stress typically show reduced growth, calcification, and fecundity and may experience greater incidences of coral disease (Hoegh-Guldberg et al. 2007). Mass bleaching events can also catalyze fundamental phase shifts in coral communities, wherein algae or other non-coral taxa become dominant and corals are unable to reestablish themselves (Baker et al. 2008). In many cases where reef communities have "recovered" from bleaching events, significant declines in diversity and shifts in the relative abundances of zooxanthellate corals have occurred, as those more susceptible to disturbance (e.g., highly imperiled, framework-building *Acropora* and *Montastraea* species) are replaced with more resilient species. *Id.*

ii. Observed impacts to corals from ocean warming: mass bleaching events

Mass bleaching of corals, defined as bleaching of multiple coral species on an ecologically significant scale, was first recorded in 1978/79 when atmospheric CO_2 was 336 ppm (Veron et al. 2009). Until the late 1970s, the bleaching of corals had been reported for small-scale events (i.e., meters to hundreds of meters) in response to a range of localized stresses: low salinity conditions (such as from the inundation of rain onto exposed reefs), pollution, or unusually high or low water temperatures (such as warm water flowing from the water cooling exhaust of a power plant) (Hoegh-Guldberg 2005). The role of elevated sea temperatures in triggering mass coral bleaching has been extensively supported by field and laboratory studies (Hoegh-Guldberg 2005). For example, McWilliams et al. (2005) examined the relationships between yearly temperature anomalies and the geographic extent and intensity of coral bleaching in the Caribbean between 1983 and 2000, and found exponential increases in the geographical extent and intensity of coral bleaching in the Caribbean with increasing SST anomalies. A rise in regional SST of 0.18°C resulted in a 35% increase in geographic extent of coral bleaching and a 42% increase in intensity of bleaching. *Id.*

Severn major world-wide bleaching events have occurred since 1978/79 (Veron et al. 2009) "with a pattern of increasing frequency and intensity" (Hoegh-Guldberg 1999). The mass coral bleaching event of 1997/98 affected every geographic coral-reef realm in the world (Hoegh-Guldberg 1999), occurring in over 70 countries (Figure 13), and killed up to 90% of affected corals and 16% of coral communities globally (Veron et al. 2009, Pratchett et al. 2011). In the Western Indian Ocean, an estimated 46% of corals disappeared by the end of the event (Hoegh-Guldberg 2005). The 1997/98 event marked "the start of a decline from which there has been no significant long-term recovery" (Veron et al. 2009: 1430). The 2002 event had particularly severe impacts on Asia and the Great Barrier Reef (Veron et al. 2009), while the 2005 event severely impacted the Caribbean and tropical Atlantic (Donner et al. 2007), leading to "a new phase of decline characterized by diminishing habitat complexity in reefs of the Caribbean and a deterioration of species diversity" (Veron et al. 2009). With the 2005 event, coral cover surveys detected bleaching of 90% of coral cover in the British Virgin Islands, 80% in the U.S. Virgin Islands, 66% in Trinidad and Tobago, 52% in the French West Indies, and 85% in the Netherlands Antilles (Donner et al. 2007). Anthropogenic warming may have increased the probability of thermal stress events for corals in this region by an order of magnitude (Donner et al. 2007).

iii. Branching corals are particularly susceptible to bleaching

The branching corals that the petitioned damselfish species rely on are particularly susceptible to bleaching (Marshall and Baird 2000, Loya et al. 2001, McClanahan et al. 2004, 2012, Bonin 2012). Bleaching susceptibility varies among coral taxa, and branching corals such as acroporids and pocilloporids are more prone to bleaching than many other species (Marshall and Baird 2000, Loya et al. 2001). As a result, coral reefs are predicted to shift from communities dominated by branching corals to those dominated by less structurally complex massive and encrusting corals (Marshall and Baird 2000, Loya et al. 2001), which would result in large-scale habitat loss for the petitioned damselfish. The loss of branching *Acropora* is of particular concern because these structurally complex corals typically dominate coral assemblages, and are also vulnerable to other disturbances such as tropical storms, *Acanthaster planci* outbreaks and coral diseases which can jeopardize their recovery (Bonin 2012). Already more than 50% of all *Acropora* species already have an elevated risk of extinction according to IUCN Red List criteria (Carpenter et al. 2008), highlighting the vulnerability of branching corals.

Figure 13. Number of countries affected by coral bleaching events 1980 to 2010. Source: Burke et al. (2011): Figure 3.1.

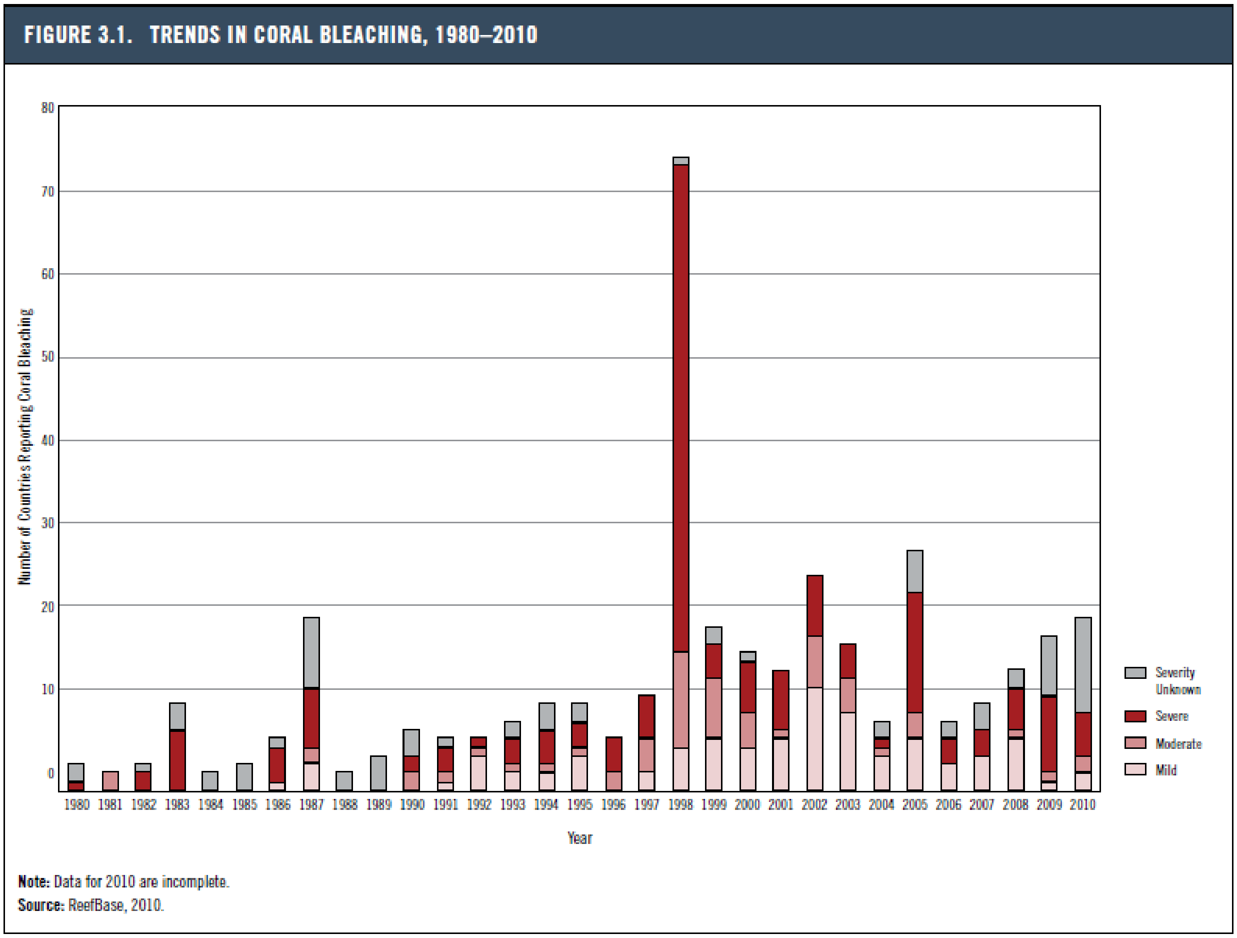


FIGURE 3.1. TRENDS IN CORAL BLEACHING, 1980–2010

Note: Data for 2010 are incomplete.
Source: ReefBase, 2010.

iv. Loss of coral reefs due to ocean warming will accelerate in this century

Studies projecting the impacts of ocean warming on corals indicate that the majority of the world's corals will be subjected to recurring mass bleaching events at frequencies from which they will be unable to recover by the 2020s or 2030s, in the absence of thermal adaptations by corals and their symbionts (Hoegh-Guldberg 1999, Sheppard 2003, Donner et al. 2005, 2007, Donner 2009, Burke et al. 2011).

Hoegh-Guldberg (1999) was the first major study to project the impacts of climate warming on coral bleaching frequency. Hoegh-Guldberg (1999) predicted the occurrence of coral bleaching at sites in French Polynesia, Jamaica, Rarotonga, Thailand, and at three sites on the Great Barrier Reef under the IPCC mid-level IS92a scenario, using coral temperature thresholds based on historical observations of bleaching and mortality at each site. This study found that most regions would experience mass bleaching at the level experienced in the 1997-1998 bleaching event biannually within 20 to 40 years and annually within 30 to 50 years.

Sheppard (2003) predicted the occurrence of coral bleaching for 33 Indian Ocean coral reefs using temperature thresholds based on observations during the 1998 coral bleaching event and a minimum recovery period of five years. The study found that most coral reefs south of the Equator would experience mass bleaching at least every five years by 2010 to 2030, although not until the latter half of the century for some coral reefs north of the equator.

In a comprehensive global assessment of coral bleaching, Donner et al. (2005) found that under IPCC A2 and B2 scenarios, severe bleaching would occur every 3 to 5 years at the majority of the world's reefs in the 2030s and would become a biannual event by the 2050s. Severe bleaching would be an annual or biannual event at 80 to 100% of the reefs worldwide by the 2080s in each model under each scenario. In addition, Donner et al. (2005) estimated the rate of temperature adaptation or acclimatization required to avoid surpassing the coral bleaching thresholds in future decades. The majority of the world's coral reefs would require adaptation of at least 0.2–0.3°C per decade to ensure that low-intensity bleaching events (degree heating month > 1°C month) would not occur more than once or twice a decade by the 2030s to 2050s.

Donner et al. (2007) projected impacts to the Eastern Caribbean region, and found that ocean warming under the IPCC B1 and A1B scenarios would lead to mass bleaching conditions (i.e. degree heating month >2°C per month) at least biannually to annually by the 2020s or 2030s. If corals were able to adapt by increasing their thermal tolerance level by 1–1.5°C, mass coral bleaching events might be postponed by 30–50 years.

More recent research by Donner (2009) projected that 80% of the world's reefs, including corals in the regions inhabited by the petitioned pomacentrid species, would experience bleaching at five-year intervals by 2030 under the lowest IPCC emission scenario (B1), with the exception being reefs in the Middle East. Under the higher A1B and A1FI scenarios, the majority of the world's corals, including corals in the regions inhabited by the petitioned pomacentrid species, would be subjected to mass bleaching at unsustainable (< 5 year) intervals by 2020. A 1.5°C increase in the thermal tolerance of corals and their symbionts would postpone the A1B severe bleaching forecast by 50–80 years for most of the world's coral reefs. Moreover, this study found that committed warming from greenhouse gases in the atmosphere in the year 2000 is projected to cause over half of the world's coral reefs to experience harmfully frequent bleaching at 5-year intervals by 2080. The most susceptible reefs occur in the East Pacific, Polynesia, Central Pacific, Micronesia, Southeast Asia, Western Australia, and the Indian Ocean (Table 3).

Finally, Burke et al. (2011) projected that under an A1B emissions scenario that approximately 50% of the world's reefs will experience thermal stress sufficient to induce severe bleaching in at least five out of ten years during the 2030s. In the Coral Triangle Region, more than 80% of reefs are projected to reach this level of thermal stress during the 2030s. During the 2050s, this percentage is expected to grow to more than 95% for both the Coral Triangle Region and the world. These projections assume that greenhouse gas emissions continue on current trajectories and local threats are not addressed. Although coral reefs can recover from infrequent and mild bleaching, this degree of high, regular stress presents a significant risk of irreversible damage.

Table 3. The year that reefs experience degree-heating-months ≥2°C per month at a probability that exceeds once every five years.
Source: Donner (2009): Table 2

	Commitment		**SRES B1**		**SRES A1b**	
	Base	**+1.5 K**	**Base**	**+1.5 K**	**Base**	**+1.5 K**
Caribbean	n/a	n/a	2016	n/a	2018	2074
Middle East	n/a	n/a	2033	n/a	2023	2070
W Indian Ocean	2063	n/a	2024	n/a	2013	2088
C Indian Ocean	2063	n/a	2019	n/a	2012	2082
Western Australia	2071	n/a	2024	n/a	2016	2086
SE Asia	2061	n/a	2021	n/a	2012	2074
GBR+Melanesia	2095	n/a	2028	n/a	2017	2092
Micronesia	2009	n/a	2010	n/a	2005	2065
Central Pacific	2005	n/a	2005	2062	2005	2051
Polynesia	2072	n/a	2016	n/a	2015	2094
East Pacific	2048	n/a	2014	n/a	2012	2073

In summary, under a mid-level A1B emissions scenario, half or more of the world's coral reefs are likely to experience harmfully frequent bleaching by the 2030s or 2030s absent adaptations by corals and their symbionts. Even if all greenhouse gases were to cease immediately, the warming commitment from greenhouse gases accumulated until 2000 would cause over half of the world's coral reefs to experience harmfully frequent bleaching by 2080. These studies provide strong evidence that ocean warming poses an imminent, high-magnitude, and accelerating threat to the petitioned pomacentrid species through loss and degradation of their coral reef habitat.

v. The anemone habitat of the orange clownfish is vulnerable to bleaching

The orange clownfish is threatened by bleaching and subsequent loss of anemone habitat resulting from ocean warming. The ten species of anemone that provide habitat for 26 species of anemonefish are susceptible to bleaching because they form a symbiosis with *Symbiodinium* species (Hill and Scott 2012). Host sea anemone bleaching has been documented at reef locations including Okinawa, Japan; Port Moresby, Papua New Guinea; and the Great Barrier Reef, Australia (Hill and Scott 2012). Bleaching causes the anemones to expel their algal symbionts and lose photosynthetic pigments from within symbionts, which can put them under physiological stress due to the loss of nutrition derived from photosynthesis (Saenz-Agudelo et al. 2011, Hill and Scott 2012). Several studies have shown that anemone bleaching can lead to reductions in anemone abundance and size (Hattori 2002, Jones et al. 2008, Frisch and Hobbs 2009, Saenz-Agudelo et al. 2011, Hill and Scott 2012).

For example, Hill and Scott (2012) examined the tolerance of the common host anemone *Entacmaea quadricolor* to thermal stress and stress from irradiance at the North Solitary Island, Australia. At temperatures 1ºC above summer average (27ºC), anemones expelled their symbionts, while at temperatures 3ºC above the summer average, anemones experienced severe bleaching which included the additional loss of photosynthetic pigments from within symbionts, and in some cases, mortality. The researchers noted that ocean temperatures in this region are expected to rise between 2 to 6 ºC over the next 100 to 200 years, meaning that the thermal threshold at which bleaching will occur is expected to be reached and exceeded more frequently in the future. Furthermore, the study found that the anemones harbor a single clade of *Symbiodinium* symbionts, and thus do not appear to have the flexibility to shift to potentially more heat-resistant symbiont types. The study concluded that "[j]ust like their relatives, the scleractinian corals, these anemones are showing bleaching thresholds that are approximately 1ºC above the current summer maximum. Once temperatures deviate outside the normal range and reach or exceed 27ºC, extensive anemone bleaching is expected to occur, which will have adverse impacts on both the anemones and their resident fish."

vi. Atmospheric CO_2 must be reduced to less than 350 parts per million to protect the coral reef habitat of the petitioned pomacentrids

Because the temperature-related effects of global warming on coral reefs have been extensively documented, relationships between rising CO_2 levels, rising ocean temperature, and effects on coral reefs provide a well-grounded basis for predicting how reefs will be affected by future levels of warming and CO_2 (Veron et al. 2009). Numerous studies have documented detrimental effects to the petitioned coral species at our current atmospheric concentration of ~392 ppm CO_2, and many studies indicate that an atmospheric CO_2 concentration of less than 350 ppm is needed to protect corals.

Veron et al. (2009) found that temperature-induced mass bleaching events causing widespread coral mortality began to when atmospheric CO_2 levels exceeded ~320 ppm, and outlined evidence for the need to reach an atmospheric CO_2 concentration of less than 350 ppm CO_2 to protect corals:

> Temperature-induced mass coral bleaching causing mortality on a wide geographic scale started when atmospheric CO_2 levels exceeded ~320 ppm. When CO_2 levels reached ~340 ppm, sporadic but highly destructive mass bleaching occurred in most reefs world-wide, often associated with El Niño events. Recovery was dependent on the vulnerability of individual reef areas and on the reef's previous history and resilience. At today's level of ~387 ppm, allowing a lag-time of 10 years for sea temperatures to respond, most reefs world-wide are committed to an irreversible decline. Mass bleaching will in future become annual, departing from the 4 to 7 years return-time of El Niño events. Bleaching will be exacerbated by the effects of degraded water-quality and increased severe weather events. In addition, the progressive onset of ocean acidification will cause reduction of coral growth and retardation of the growth of high magnesium calcite-secreting coralline algae. If CO_2 levels are allowed to reach 450 ppm (due to occur by 2030–2040 at the current rates), reefs will be in rapid and terminal

> decline world-wide from multiple synergies arising from mass bleaching, ocean acidification, and other environmental impacts. Damage to shallow reef communities will become extensive with consequent reduction of biodiversity followed by extinctions. Reefs will cease to be large-scale nursery grounds for fish and will cease to have most of their current value to humanity. There will be knock-on effects to ecosystems associated with reefs, and to other pelagic and benthic ecosystems. Should CO_2 levels reach 600 ppm reefs will be eroding geological structures with populations of surviving biota restricted to refuges. Domino effects will follow, affecting many other marine ecosystems. This is likely to have been the path of great mass extinctions of the past, adding to the case that anthropogenic CO_2 emissions could trigger the Earth's sixth mass extinction. (Veron et al. 2009: 1428).

Similarly, Hansen et al. (2008) concluded that a ~385 ppm CO_2 concentration is already deleterious for corals and that a concentration of 300 to 350 ppm CO_2 would significantly alleviate stresses from ocean warming and ocean acidification:

> Coral reefs are suffering from multiple stresses, with ocean acidification and ocean warming principal among them. Given additional warming 'in-the-pipeline', 385 ppm CO_2 is already deleterious. A 300-350 ppm CO_2 target would significantly relieve both of these stresses. (Hansen et al. 2008: 226).

The United Nations Environment Programme's 2009 Climate Science Compendium found that we are already committed to ocean acidification that will damage or destroy coral reefs, indicating that current CO_2 concentration is too high:

> Already we are committed to ocean acidification that will damage or destroy coral reefs and the many species of marine life that inhabit or depend upon the ecosystem services of the reefs. (McMullen and Jabbour 2009: 7).

Finally, Donner (2009) found that that atmospheric greenhouse gas concentrations in 2000 have committed half of the world's coral reefs to harmfully frequent bleaching at 5-year intervals by 2080, meaning that today's level of ~387 ppm CO_2 is unsustainable for corals. Donner (2009) concluded that "Without any thermal adaptation, atmospheric CO_2 concentrations may need to be stabilized below current levels to avoid the degradation of coral reef ecosystems from frequent thermal stress events" (p. 1).

c. Adverse impacts on the petitioned reef fish due to coral and anemone bleaching

Numerous studies have documented significant declines in pomacentrid populations following coral habitat loss due to bleaching and other disturbances, highlighting the vulnerability of the petitioned species to coral loss and degradation (Pratchett et al. 2008, Wilson et al. 2006). The most severe effects occur to coral-dwelling and coral-eating specialists such as the petitioned pomacentrid species (Wilson et al. 2006: Figure 2). Coral bleaching harms coral-dwelling damselfish and other reef fish because reductions in live coral cover and reef topographic complexity lower the availability of recruitment habitat, shelter, and food (Pratchett et al. 2008). Declines in juveniles resulting from lost recruitment habitat can have serious consequences for future adult abundance, while reductions in adult survival and reproductive output resulting from loss of coral habitat can also result in population declines (Pratchett et al. 2008).

Studies that have examined the responses of damselfish and anemonefish to habitat resulting from bleaching and other disturbances have found that fish populations are negatively affected due to decreased recruitment, higher vulnerability to predation on bleached and dead coral hosts, lower fish growth on bleached and dead corals, and higher competition for shrinking habitat.

i. Declines in damselfish abundance following coral habitat loss

Booth and Beretta (2002) found that the densities of three damselfish species that typically associate with live coral--*Pomacentrus wardi*, *P. moluccensis*, and *Chrysiptera rollandi*--declined at bleached sites but not at unbleached sites ten months after the 1997-1998 bleaching event at One Tree Island lagoon in the southern Great Barrier Reef. The study concluded that their results "suggest deleterious effects of above-normal water temperature on damselfish assemblages through changes in live coral cover."

Graham et al.(2007) found that the abundance of coral-dwelling planktivorous damselfish experienced a long-term decline after the 1998 bleaching event in the Seychelles in both fished areas and marine protected areas (MPAs). The planktivores, which were principally coral-dwelling damselfish, declined markedly in abundance in the MPAs from a mean of 279.8 fish per site to 11.2 fish per site, and in fished areas from a mean of 90.3 fish per site to 44.4 fish per site. The significant declines in damselfish abundance, particularly in MPAs, were associated with a similar pattern of decline in preferred coral species nearly seven years after the bleaching event. The larger declines in the MPAs were likely due to the greater cover of complex coral species prior to the bleaching event. In addition, the decline in smaller fish (<30 cm) and an increase in larger fish (>45 cm) in both fished areas and MPAs was thought to be a time-lagged response to the reduction in reef structural complexity, where fish that were being lost by mortality and fishing were not being replaced by juveniles.

In an expanded study of the impacts of the 1998 mass bleaching event across the Indian Ocean, Graham et al. (2008) found that the abundance of damselfish planktivores declined following the loss of live-coral cover throughout the western Indian Ocean. Specifically, total reef-fish species richness and the abundance of damselfish planktivores, obligate corallivores, and fishes less than 20 cm declined with loss of live coral cover throughout an area spanning seven countries, 66 sites, and 26 degrees of latitude. Planktivores (comprised principally of damselfish) and corallivores showed the strongest response to declining coral cover, and the researchers concluded that they "are likely to be the groups most threatened from the predicted ongoing decline in global reef health."

ii. Mechanisms underlying damselfish declines following coral loss: lower recruitment, higher predation risk, lower growth, more competition for space

Studies that have examined the responses of coral-dwelling damselfish to coral loss have found profound effects on damselfish recruitment, including lower recruitment on bleached and dead corals compared with live corals and declines in recruit numbers following reductions in live corals or death of corals after bleaching. Booth and Beretta (2002) reported that damselfish species that typically associate with live corals had lower recruitment on bleached corals compared to the same sites pre-bleaching or to recovered sites, and as a result, species diversity declined and community structure changed ten months after the 1997-1998 bleaching event at One Tree Island lagoon in the southern Great Barrier Reef. Similarly, in a field experiment where live coral cover was reduced by 95% to 100%, Feary et al. (2007) detected rapid reductions in the abundance of new recruits, particularly for the live-coral-associated damselfish *Pomacentrus moluccensis*, and a shift from a coral-associated fish community to an algal-associated fish community. For the damselfish *Pomacentrus moluccensis* in Papua New Guinea, Bonin et al. (2009) found that the recruits settled on both healthy and bleached *Acropora* corals; however, the number of recruits remaining on dead corals declined significantly after four weeks, indicating that coral specialists quickly disappear when corals die from bleaching, even prior to structural erosion of the dead corals. In another study, settlement-stage larvae of three damselfish species, with different degrees of association with live coral, all preferred live coral habitat over bleached or dead coral habitat, and larval fish used visual and olfactory cues to distinguish healthy corals from bleached and dead corals, although the relative importance of these cues varied among species (McCormick et al. 2010). Overall, patches of healthy coral received nearly ten times the number of recruits as colonies that were sub-lethally bleached, highlighting the importance of healthy coral condition to recruitment. Specifically, the damselfish *Pomacentrus moluccensis* (which associates with live coral) and *P. amboinensis* (which inhabits rubble, dead coral, and live coral patches at the base of reefs) exhibited a strong preference for live coral and avoided bleached coral in the habitat choice trials, while *Dischistodus sp*., a habitat generalist (mostly *D. perspicillatus* which inhabits sand and rubble patches adjacent to the reef edge), exhibited more even use of the alternative habitats but still preferred live coral over bleached and dead coral. The researchers highlighted that there are likely significant benefits for associating with live coral and avoiding bleached coral, as suggested by studies that have found that settlement to sub-optimal habitats may affect growth, body condition, and survival of juvenile reef fishes. The study concluded that "[t]hese results suggests that coral bleaching will affect settlement patterns and species distributions by influencing the visual and olfactory cues that reef fish larvae use to make settlement choices" (McCormick et al. 2010). On a final note, Bonin (2012) cautioned that habitat specialists, such as the damselfish *Chrysiptera parasema*, *Pomacentrus moluccensis*, *Dascyllus melanurus* and *Chromis retrofasciata* that use only branching hard corals, will increasingly be forced to use alternative recruitment microhabitats as coral reef habitat degradation continues, and this is likely to reduce population replenishment.

In addition to lowering recruitment, coral bleaching makes damselfish more vulnerable to predation on bleached and dead coral hosts. Coker et al. (2009) showed that the coral-dwelling damselfishes *Pomacentrus moluccensis* and *Dascyllus aruanus* are more susceptible to predation when they are associated with bleached and dead coral hosts than when they are associated with healthy coral colonies. The predator *Pseudochromis fuscus* was almost twice as likely to strike at damselfish prey associated with the stark white colonies of bleached corals, suggesting that damselfish are much more conspicuous against the bleached-white background. The study concluded that increased susceptibility to predation may contribute to declines in abundance of coral-dwelling fishes following host-coral bleaching. The researchers also noted that coral-dwelling fishes that escape the immediate effects of coral mortality by moving to alternative coral habitats may experience lower growth, survival, or reproductive rates in less preferred coral habitats, especially as the increasing severity and geographic extent of coral bleaching events decreases the likelihood of finding alternate coral habitats.

Coral bleaching can also reduce the growth rates of coral-associated damselfish. A study of two planktivorous damselfishes, *Chrysiptera parasema* and *Dascyllus melanurus*, in Papua New Guinea found that the growth rates of both species were directly related to the percentage live coral cover, with highest growth in areas of 100% live coral and the slowest growth in individuals associated with dead corals (Feary et al. 2009). For *Chrysiptera parasema*, fishes on the 0% coral treatment showed one-quarter of the growth as those in the 50% and 100% coral treatments after four weeks. The researchers predicted that the slower growth of fish on dead or degraded coral will have a number of negative fitness consequences including delayed maturity, reduced lifetime fecundity, and increased vulnerability to gape-limited predation.

Finally, coral bleaching appears to increase competition for space by reducing suitable coral habitat. Coker et al. (2012) examined the response of the damselfish *Dascyllus aruanus* to bleaching and mortality of host coral colonies with respect to the condition of the host colony, availability of preferred habitat, and the presence of conspecifics on alternative coral habitat. The study found that *D. aruanus* moved to alternate habitats once the host coral died, although one-third of fish could not immediately relocate due to competition from aggressive resident fish on neighboring healthy colonies. Fish selected new habitat based mostly on the presence of conspecifics. The study noted that the potential for relocation after bleaching will decline as disturbances become more severe and widespread, thereby increasing the distance to suitable habitats and reducing the availability of preferred habitat types. Furthermore, increased densities within habitat patches may increase competitive effects, with potential adverse consequences for growth, survival and reproductive success.

iii. Declines in anemonefish density, reproduction, and recruitment following anemone bleaching

Two recent studies have found that anemone bleaching can reduce the densities, reproduction, and recruitment of anemonefish (Jones et al. 2008, Saenz-Agudelo et al. 2011). Jones et al. (2008) reported that densities of anemonefishes were lower on reefs in reef areas in the Keppel Islands that had been impacted by bleaching than on reefs in Far North Queensland that had been less impacted. Importantly, no anemonefishes or anemones were found on or near bleached corals in the Keppel Islands. Furthermore, the highest densities of fishes were found on reefs that were closed to fishing and aquarium collecting in both the Keppel Islands and Far North Queensland, which suggests that collecting is compounding the effects of bleaching. The researchers cautioned that bleaching can have long-lasting population-level effects on anemonefish given the slow rate of regeneration of anemones and anemonefish after bleaching events: “the slow reproductive rate of anemones, the minimal migration of adult and juvenile anemonefishes and the absolute dependence of the fishes on suitable anemone habitat means that regeneration on bleached reefs that have lost anemones and fishes may take decades."

Saenz-Agudelo et al. (2011) found that bleaching has adverse effects on the reproduction and recruitment of the panda anemonefish *Amphiprion polymnus*. At a site near Port Moresby, Papua New Guinea, the study found that a high-temperature anomaly in 2009 caused the bleaching of ~35% of anemones in waters 6 meters or less, and that bleached anemones suffered a ~34% reduction in body size. Following the bleaching event, female panda anemonefish living in bleached anemones experienced reduced egg production (~38% lower than the prior year), and total recruitment was significantly lower (~54% less) than the prior year. Densities of anemonefish on anemones did not change after bleaching, and recruiting larvae did not appear to avoid bleached anemones at settlement, suggesting that habitat limitation via anemone saturation may also influence anemonefish density and recruitment. The study concluded that "these results provide the first field evidence of detrimental effects of climate-induced bleaching and habitat degradation on reproduction and recruitment of anemonefish."

3. Ocean acidification threatens the coral reef habitat of the petitioned pomacentrid reef fish

Ocean acidification poses an imminent, high-magnitude threat to the petitioned reef fish by degrading their coral reef habitat. Corals are already experiencing lower calcification rates that have been linked to ocean acidification in the Indo-Pacific and Caribbean regions inhabited by the petitioned reef fish species (Cooper et al. 2008, Gledhill et al. 2008, Bak et al. 2009, De'ath et al. 2009, Bates et al. 2010, Fabricius et al. 2011, Friedrich et al. 2012). Studies projecting the combined impacts of ocean acidification and ocean warming on corals predict that coral erosion will exceed calcification rates at atmospheric CO_2 concentrations of 450 to 500 ppm which are expected by or before mid-century (Hoegh-Guldberg et al. 2007), and that all coral reefs will begin dissolve at carbon dioxide concentrations of 560 ppm (Silverman et al. 2009). In the Caribbean, a recent study concluded that "coral reef communities are likely to be essentially gone from substantial parts of the Southeast Caribbean by the year 2035" (Buddemeier et al. 2011). Due to the synergistic impacts of ocean acidification, mass bleaching, and other impacts, reefs are projected to experience "rapid and terminal" declines worldwide at atmospheric carbon dioxide concentrations 450 ppm, which are expected to occur before mid-century:

> If CO_2 levels are allowed to reach 450 ppm (due to occur by 2030-2040 at the current rates), reefs will be in rapid and terminal decline world-wide from multiple synergies arising from mass bleaching, ocean acidification, and other environmental impacts. Damage to shallow reef communities will become extensive with consequent reduction of biodiversity followed by extinction (Veron et al. 2009: 1428).

In short, ocean acidification will act synergistically with ocean warming to further threaten the petitioned reef fish species with extinction within this century.

The following section provides a summary of the best-available science on (a) current and predicted trends in ocean acidification and its ecological effects; and (b) current and predicted impacts on the coral reef habitat of the petitioned pomacentrid reef fish.

a. Current and predicted trends in ocean acidification and ecological effects

Ocean acidification poses a global threat to marine ecosystems. The oceans have absorbed about 30% of the carbon dioxide released into the atmosphere by humans from fossil fuel burning, cement production, and land-use changes (Friedrich et al. 2012). Currently the global oceans are absorbing about 22 million tons of carbon dioxide each day (Feely et al. 2008). This uptake of carbon dioxide is fundamentally changing the chemistry of the ocean, causing the ocean's slightly alkaline waters to become more acidic and lowering the availability of the carbonate ions, calcite and aragonite (Caldeira and Wickett 2005, Orr et al. 2005, Fabry et al. 2008).

Since the industrial revolution began, surface ocean pH has declined by 0.1 units on average, decreasing from 8.16 in 1800 to 8.05 today, corresponding to a 30% increase in acidity (Caldeira and Wickett 2005, Orr et al. 2005). The current anthropogenic trend in ocean acidification exceeds the trend in natural variability significantly, up to 30 times in some regions (Friedrich et al. 2012). The rate of change in ocean acidity is unprecedented in the past 300 million years, a period that includes four mass extinctions (Zeebe 2012, Honisch et al. 2012). For example, the current change in seawater chemistry is an order of magnitude faster than what occurred 55 million years ago during Paleocene-Eocene Thermal Maximum, which is considered to be the closest analogue to the present, when 96% of marine species went extinct (Zeebe 2012, Honisch et al. 2012).

The surface concentration of carbonate ions has also decreased by more than 10% since the pre-industrial era (Caldeira and Wickett 2003, Orr et al. 2005, Feely et al. 2008), and decreasing aragonite and calcite saturation states have been observed in the Pacific and Atlantic. In the Pacific Ocean, Feely et al. (2012) reported a decrease of the saturation state of surface seawater with respect to aragonite and calcite as well as an upward shoaling of the saturation horizon by about 1 to 2 meters per year on average, based on observations from cruise surveys. This study found that aragonite saturation state has declined 16% since the industrial revolution due, in large part, to anthropogenic CO_2 (*Id.*, see also Ishii et al. 2011). The authors concluded with a warning for coral reef systems in the Pacific:

> If CO_2 emissions continue as projected out to the end this century, the resulting changes in the marine carbonate system would mean that many coral reef systems in the Pacific would probably no longer be able to maintain the necessary rate of calcification required to sustain their vitality. (Id.: GB3001).

Friedrich et al. (2012) reported declining trends in the surface saturation state for aragonite at four sites across the Atlantic and Pacific—the Canary Islands, Bermuda, Hawaii, and Caribbean--estimated at -0.09, -0.04, -0.08, and -0.09 units per decade, respectively. The study found that these present-day, anthropogenic rates of change in surface aragonite saturation state are one to two orders of magnitude larger than estimated for the last glacial termination

If emissions continue unabated, ocean acidity is predicted to increase up to 150% by the end of the century. Specifically, under a low emission scenario (B1), the atmospheric CO_2 concentration would approach 560 ppm by 2100, 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). Under a business-as-usual scenario (similar to A1B or A2) where the CO_2 concentration reaches 788 ppm, ocean pH would drop 0.3 or 0.4 units amounting to a 100 to 150% change in acidity, respectively, and tropical ocean surface concentrations of carbonate would decline by 45% (Orr et al. 2005, Meehl et al. 2007). A pH change of this magnitude has not occurred for more than 20 million years (Feely et al. 2004).

The ecological consequences of ocean acidification are predicted to be overwhelmingly negative. A meta-analysis of studies on the biological responses to ocean acidification found a significant negative effect across marine organisms on survival, calcification, growth and reproduction (Kroeker et al. 2010). One of the major impacts of ocean acidification is that it impairs the ability of marine organisms like corals to build protective calcium carbonate shells, liths, and skeletons because carbonate minerals, calcite and aragonite, become less available (Feely et al. 2004, Orr et al. 2005, Fabry et al. 2008). Nearly all calcifying organisms studied, including species from the major marine calcifying groups and plankton at the base of the marine food web, have shown an adverse response of reduced calcification in response to elevated carbon dioxide in laboratory experiments (Kleypas et al. 2006, Fabry et al. 2008, Kroeker et al. 2010). According to the U.S. Environmental Protection Agency:

> As more CO_2 dissolves in the ocean, it reduces ocean pH, which changes the chemistry of water. These changes present potential risks across a broad spectrum of marine ecosystems…For instance, ocean acidification related reductions in pH is forecast to reduce calcification rates in corals and may affect economically important shellfish species including oysters, scallops, mussels, clams, sea urchins, and lobsters…Impacts to shellfish and other calcifying organisms that represent the base of the food web may have implications for larger organisms that depend on shellfish and other calcifying organisms for prey.
> (74 Fed. Reg. 17485)

Ocean acidification also disrupts metabolism and other biological functions in marine life. Changes in the ocean's carbon dioxide concentration result in accumulation of carbon dioxide in the tissues and fluids of fish and other marine animals, called hypercapnia, and increased acidity in the body fluids, called acidosis. These impacts can cause a variety of problems for marine animals including difficulty with acid-base regulation, metabolic activity, respiration, and ion exchange, leading to impairment of growth and higher mortality rates (Ishimatsu et al. 2004, Royal Society 2005, Fabry et al. 2008).

The adverse effects of ocean acidification are already being observed for many species. Reduced calcification is being detected in marine organisms in the wild, including reduced coral calcification rates (Cooper et al. 2008, Gledhill et al. 2008, De'ath et al. 2009, Bates et al. 2010), and reduced shell weights of modern foraminifera in the Southern Ocean (Moy et al. 2009). Since 2006 both natural and hatchery oysters in the Pacific Northwest have experienced severe collapses of oyster production with some shellfish hatcheries experiencing up to 80% mortality of oyster larvae in some years; this die-off of oyster larvae has been definitively linked to ocean acidification (Barton et al. 2012).

b. Current and predicted impacts of ocean acidification on the coral reef habitat of the petitioned pomacentrid reef fish

Ocean acidification threatens the tropical coral reef habitat of the petitioned reef fish species by impairing coral calcification and making reefs more vulnerable to degradation by erosion, storms, predation, and other disturbances. NMFS in its *Status Review Report of 82 Candidate Coral Species Petitioned Under the U.S. Endangered Species Act* concluded that ocean acidification threatens many corals with extinction within this century: "[a]fter extensive review of available scientific information, the BRT [biological review team] considers ocean warming, disease, and ocean acidification to be the most influential threats in posing extinction risks to the 82 candidate coral species between now and the year 2100" (Brainard et al. 2012: xxxiii).

Ocean acidification negatively impacts tropical corals by reducing the availability of carbonate ions essential for building calcium carbonate skeletons, thereby impairing coral calcification rates and skeletal formation (Kleypas et al. 1999). The full impacts of ocean acidification on corals include the slowing of carbonate accumulation, reduction of growth rates, weakening of coral skeletons, reduction of cementation, and destabilization of reef structures (Kleypas et al. 2001, Guinotte and Fabry 2008). Reduced calcification that slows coral growth can make corals less able to compete for space and can weaken coral skeletons increasing their vulnerability to erosion, storm damage, and predation (Guinotte et al. 2003, Gledhill et al. 2008). As a result, coral abundance and reef-building capabilities are expected to largely diminish over this century (Hoegh-Guldberg 2005).

Reef-building corals may exhibit several responses to reduced calcification, all of which have deleterious consequences for reef ecosystems. First, coral may exhibit a decreased linear extension rate (Hoegh-Guldberg et al. 2007). For example, the significant decline in calcification rate of *Porites* corals throughout the Great Barrier Reef since 1990 was principally due to a decline in linear extinction rate of 13.3% (De'ath et al. 2009). Secondly, corals may reduce skeletal density in order to maintain their physical extension or growth rates, which in turn can increase coral erosion (Hoegh-Guldberg et al. 2007). Brittle coral skeletons are more vulnerable to storm damage, and coral grazers such as parrotfish prefer to remove carbonates from lower-density substrates (Hoegh-Guldberg et al. 2007). As noted by Hoegh-Guldberg et al. (2007), erosion rates that outpace calcification rates would reduce the structural complexity, habitat quality, and habitat diversity of corals, and would impact the ability of reefs to absorb wave energy. Third, corals might invest greater energy in calcification in order to maintain skeletal growth and density, which would divert resources from essential activities such as reproduction and potentially reduce the recolonization ability of corals (Hoegh-Guldberg et al. 2007).

Corals are already experiencing significantly lower calcification rates that have been linked to ocean acidification in the Indo-Pacific and Caribbean regions inhabited by the petitioned reef fish species. In the Indo-Pacific, researchers reported unexpectedly low calcification rates and found that corals have declined by 14–30% over the past 20 years in large geographic regions around the world, with ocean warming and acidification considered the most likely causes (Fabricius et al. 2011). Reduced calcification rates have been observed in the Pacific, including the Great Barrier Reef where calcification rates of some corals have declined 14% since 1990 (De'ath et al. 2009; Cooper et al. 2008). In the Caribbean, the rapidly declining aragonite saturation state appears to be resulting in reduced calcification of some coral species (Bak et al. 2009, Bates et al. 2010; Gledhill et al. 2008). Calcification rates at coral reef locations in the Caribbean may have already dropped by ~15% with respect to their pre-industrial values (Friedrich et al. 2012). On a global scale, modeling by Silverman et al. (2009) suggests 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.

Optimal coral growth occurs when aragonite saturation state (Ω) in surface waters 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 reef growth, below which reef accretion shifts to dissolution (Hoegh-Guldberg et al. 2007). Cao and Caldeira (2008) found that before the industrial revolution, 98.4% of coral reefs were found near open ocean waters with an aragonite saturation state above 3.5. If atmospheric CO_2 were to be stabilized at 380 ppm, only 38% coral reefs would be surrounded by waters with a saturation state above 3.5; at a CO_2 stabilization of 450 ppm, this would fall to 8%; and at a stabilization level of 550 ppm, no existing coral reefs would near waters that are conducive for reef growth.

Studies that have examined the combined effects of ocean acidification and ocean warming on a global scale (Hoegh-Guldberg et al. 2007, Silverman et al. 2009, Meissner et al. 2012) and in regions inhabited by the petitioned pomacentrids--the Caribbean (Buddemeier et al. 2011) and Hawaiian Islands (Hoeke et al. 2011)—provide evidence that coral reef habitats will be largely degraded and destroyed within this century, absent significant reductions in greenhouse gas emissions.

Hoegh-Gulberg et al. (2007) projected that at an atmospheric CO_2 concentration stabilized at 380 ppm, coral reefs would remain coral-dominated and carbonate-accreting in most areas of their current distribution. However, at atmospheric CO_2 concentrations of 450 to 500 ppm, reef erosion will exceed calcification because coral reef accretion stops and erosion begins at aragonite saturation values < 3.3 which is projected to occur when CO_2 concentrations approaches 480 ppm and carbonate ion concentrations drop below 200 mmol kg^{-1} (aragonite saturation < 3.3) in most of the global ocean (Hoegh-Guldberg et al. 2007). In this scenario, the density and diversity of corals will decline, habitat complexity and reef biodiversity will diminish, disease incidence will likely increase, coralline algae will decline leading to reduced availability of settlement substrate for corals, macroalgae will likely form stable communities that are resistant to coral settlement, and corals will become even more sensitive to local stressors (Hoegh-Guldberg et al. 2007). At atmospheric CO_2 concentrations greater than 500 ppm, carbonate-ion concentrations would fall well below 200 mmol kg^{-1} (aragonite saturation < 3.3) and ocean temperatures would rise above 2°C relative to current values (Hoegh-Guldberg et al. 2007). According to Hoegh-Guldberg et al. (2007), "[t]hese changes will reduce coral reef ecosystems to crumbling frameworks with few calcareous corals."

Silverman et al. (2009) provided global estimates of the decline in calcification of corals taking into account the synergistic effects of elevated sea surface temperatures and ocean acidification at different levels of atmospheric CO_2. At 450 ppm CO_2, all corals are expected to decrease calcification by 60-80% relative to pre-industrial rates. Once atmospheric carbon concentrations reach 560 ppm, all corals are expected to decrease calcification by 80%, at which point they will cease to grow and start to dissolve (Silverman et al. 2009). While these researchers accounted for the synergistic effects of elevated sea surface temperature, bleaching, and ocean acidification, they note that their projections are likely conservative given the unexamined additional negative impacts due to pollution, predation, and disease.

Meissner et al. (2012) examined the effects of ocean warming and ocean acidification on coral reefs under three new IPCC emissions pathways--representative concentration pathways RCP 3PD, RCP 4.5, and RCP 8.5--over the next 400 years. This study found that that the aragonite saturation state threshold (3.3 Ω) at which accretion stops and erosion begins would be crossed before 2050 under the RCP 4.5 and 8.5 scenarios. By year 2030, 66 to 85% of the reef locations considered in the study would experience severe bleaching events at least once every 10 years. Even under the most optimistic emissions pathway, virtually every reef considered in this study (97%) would experience severe thermal stress by year 2050. In all their simulations, changes in surface seawater aragonite saturation lead changes in temperatures.

On a regional scale, Buddemeier et al. (2011) estimated future coral cover in areas of the eastern Caribbean that were impacted by a massive coral bleaching event in 2005, under warming sea surface temperatures, ocean acidification, and episodic bleaching events. Under the SRES B1, A1B, and A1FI emissions scenarios, coral cover on most Caribbean reefs is projected to drop below 5% by the year 2035. When corals are allowed to gain 1–1.5ºC of heat tolerance by changing their symbionts, coral cover dropped below 5% in 2065. The study noted that adding other stressors such as storms and anthropogenic damage would result in more severe declines. The study concluded that "coral reef communities are likely to be essentially gone from substantial parts of the Southeast Caribbean by the year 2035, given the current low cover values following the 2005 event."

Hoeke et al. (2011) modeled changes in shallow-water scleractinian coral cover in the Hawaiian Archipelago for years 2000–2099 using an A1B mid-level emissions scenario, factoring in rising sea surface temperature and ocean acidification. Assuming low resiliency of corals to ocean warming and acidification (i.e., no ability of corals to adapt to increasing temperature and a linear decrease in coral calcification rate of approximately 30% between 1999 and 2099), coral cover had a high probability of declining by more than 50% by 2050 at Johnston Atoll, Oahu, and Midway, while total loss of viable coral cover by 2099 was certain at Johnston Atoll and Midway and very high (80% probability) at Oahu. Under a high resiliency scenario (i.e., the episodic heat mortality threshold was allowed to increase by 1ºC over the century and decreasing aragonite saturation state did not have a significant effect on calcification rates), the probability of declines in coral cover greater than 50% by 2099 ranged between 47 to 66% depending on region. The authors noted that they used a simplified model of ocean acidification and that impacts may be much greater. For example, they noted that "susceptibility of crustose coralline algae calcification rates (a major component of Hawaiian reefs) and reef matrix cementation (relative to corals) may significantly impact coral populations by altering recruitment success, competition for space, and increased bio- and physical erosion of reefs."

There is also much to be learned from areas that have naturally low pH and aragonite saturation states. Recent in situ studies of areas that have low saturation, such as volcanic vents, provide a glimpse into the future for coral reefs and other ecosystems. These studies have reported significantly reduced coral cover, lacking any corals that build reef frameworks, and up to 30% reduction in overall diversity in conditions predicted by the end of the century (Fabricius et al. 2011, Crook et al. 2011).

B. Other Natural or Manmade Factors Affecting The Continued Existence of the Petitioned Pomacentrid Reef Fish (Listing Factor E): Direct Impacts of Ocean Acidification and Ocean Warming on Pomacentrid Reef Fish

In addition to causing habitat loss, ocean acidification and ocean warming directly threaten the survival of the petitioned clownfish and damselfish through a wide array of adverse impacts that are predicted to lead to negative fitness consequences and population declines. Ocean acidification impairs the sensory capacity and behavior of larval clownfish and damselfish. Ocean warming reduces reproductive functions, swimming performance, and aerobic capacity of damselfish. As discussed below, these direct effects of greenhouse gas emissions threaten the survival of the petitioned fish species given the expected dramatic increase in both temperature and acidity within this century.

1. Direct impacts of ocean acidification to the petitioned clownfish and damselfish

Recent laboratory studies have found that ocean acidification at levels predicted later this century impairs the sensory capabilities and behavior of larval clownfish and damselfish, which leads to higher mortality and is likely to result in population declines. At dissolved CO_2 levels expected within this century, damselfish and clownfish suffer impairment of hearing, smell, and vision, and important behaviors are altered such as the ability to detect and avoid predators and find suitable settlement sites. Research indicates that many of these effects result from impaired function of an important neurotransmitter, and thus are likely to affect a wide range of species (Nilsson et al. 2012). As summarized by Nilsson et al. (2012), rising CO_2 levels will have far-reaching consequences for reef fish populations and ecosystem function:

> Predicted future CO_2 levels have been found to alter sensory responses and behaviour of marine fishes. Changes include increased boldness and activity, loss of behavioural lateralization, altered auditory preferences and impaired olfactory function. Impaired olfactory function makes larval fish attracted to odours they normally avoid, including ones from predators and unfavourable habitats. These behavioural alterations have significant effects on mortality that may have far-reaching implications for population replenishment, community structure and ecosystem function. (Nilsson et al. 2012).

a. Ocean acidification impairs larval orange clownfish smell and hearing, harming their ability to find settlement sites and avoid predators

Ocean acidification at levels expected to occur within this century (i.e., 700 to 1000 ppm CO_2) impairs larval orange clownfish smell, hearing, and behavior, which makes it more difficult for them to locate settlement sites on reef habitat and avoid predators (Munday et al. 2009, 2010, Dixson et al. 2010, Nilsson et al. 2012). In fact, elevated CO_2 makes larval clownfish attracted to odors from predators and unfavorable habitat (Munday et al. 2010, Dixson et al. 2010). Ocean acidification also impairs the hearing capacity of larval clownfish, which is likely to have negative effects on settlement success and survival (Simpson et al. 2011).

i. Effects on smell

Larval clownfish use olfactory cues, such as odors from rainforest vegetation and anemones, to locate suitable reef habitat for settlement (Munday et al. 2009). Munday et al. (2009) found that larval orange clownfish reared at levels of ocean pH expected to occur at the end of this century under a business-as-usual CO_2 emission trajectory (pH 7.8) had an impaired ability to discriminate among odors important for settlement. Specifically, larval clownfish were strongly attracted to olfactory stimuli that were not associated with reefs and that they normally avoided. Moreover, when reared under pH levels that might be attained later next century under business-as-usual (pH 7.6), larval clownfish no longer responded to any olfactory cues and could not distinguish between chemical cues of preferred versus non-preferred habitat. The researchers warned that the impairment of the olfactory system at higher CO_2 levels and lower pH would disrupt larval settlement and jeopardize population replenishment with likely population-level consequences and species declines:

> Olfactory cues appear to be a key mechanism by which the pelagic larvae of many coastal marine species identify and navigate toward adult habitat and then select suitable settlement sites. Disruption to this process would have significant consequences for the replenishment of adult populations and could lead to the decline of many coastal species. (Munday et al. 2009: 1850).

Ocean acidification also impairs the ability of larval orange clownfish to avoid predators (Dixson et al. 2010, Munday et al. 2010). Newly hatched orange clownfish larvae innately detect predators using olfactory cues, and they retain this ability through settlement (Dixson et al. 2010). However, when clownfish eggs and larvae were exposed to seawater with pH 7.8 and 1000 ppm CO_2, settlement-stage larval clownfish became strongly *attracted* to the smell of predators and lost the ability to discriminate between predators and non-predators (Dixson et al. 2010). Munday et al. (2010) conducted similar experiments using seawater simulating different levels of ocean acidification. Larval orange clownfish reared under CO_2 environments at ~390 ppm (current-day) and 550 ppm maintained a strong avoidance of the predator cue at all times. However, many individuals reared at 700 ppm CO_2 were attracted to the smell of predators after four days. Larval clownfish reared at 850 ppm exhibited a strong attraction to the predator cue just two days after hatching, and after eight days, larvae spent over 94% of their time in the water stream containing the predator cue.

ii. Effects on hearing

Ocean acidification also interferes with the auditory perception of juvenile orange clownfish. Auditory cues from coral reefs are important for guiding larval fish during settlement. Nocturnal reef sounds promote settlement and daytime predator-rich noises discourage settlement (Simpson et al. 2011). In an experiment to test the effects of CO_2-enriched water on hearing, juvenile clownfish reared in ambient CO_2-conditions avoided playbacks of daytime reef noise, an adaptive response. However, juveniles raised at 600, 700 and 900 µatm *p*CO2 predicted to occur within this century under differing emissions scenarios did not avoid daytime reef noise, a maladaptive behavior. The researchers concluded that "[t]his study provides, to our knowledge, the first evidence that ocean acidification affects the auditory response of fishes, with potentially detrimental impacts on early survival." The researchers further warned that "ocean acidification could compromise auditory behaviours crucial for survival" within this century:

> On the current CO_2 emissions trajectory, the average concentration of CO_2 in the atmosphere and surface waters of the ocean is predicted to exceed 500 matm by mid-century and could approach 1000 matm by 2100 [2,3], suggesting that ocean acidification could compromise auditory behaviours crucial for survival…. (Simpson et al. 2011: 919).

iii. Effects of rising temperature and ocean acidification on juvenile growth and survival

Ocean acidification and ocean warming are also likely to have negative, interactive effects on juvenile clownfish growth and survival. A study by Nowicki et al. (2012) on the related species, the cinnamon clownfish *Amphiprion melanopus*, found that food consumption and foraging activity of juvenile clownfish increased under the highest temperature and CO_2 levels, putting this species under increased physiological stress. The study used three temperatures (28.5, 30, and 31.5 °C) and three pCO_2 levels (420, 530, and 960 µatm) consistent with current-day ocean conditions and predictions for mid-century and late-century. Rearing at high temperature (31.5°C) and control (420 µatm) or moderate (530 µatm) CO_2 resulted in a reduction of food consumption and foraging activity, while rearing at high temperature and high CO_2 (960 µatm) resulted in an elevation in these behaviors. The study concluded that "the interactive effects of increased [sea surface temperature] and CO_2 could have significant effects on the growth and survival of juvenile reef fishes by late century."

b. Ocean acidification impairs larval damselfish smell, vision, learning, anti-predator behavior, behavioral lateralization, and neurotransmitter function, harming their ability to find settlement sites and avoid predators

Research on the effects of ocean acidification on six species of larval damselfish found that elevated CO_2 levels expected within this century impair damselfish smell, vision, learning, behavior, and brain function, leading to higher risk of mortality. Studies have found that exposure to elevated CO_2 threatens damselfish in the following ways: (1) larval damselfish became attracted to predator odors and displayed much riskier behaviors, making them more prone to predation; for example, Ward's damsel and whitetail damsel larvae suffered a five-fold to nine-fold increase in predation rate at CO_2 levels of 700 to 850 ppm (Munday et al. 2010, Ferrari et al. 2011a); (2) larval damselfish could not discriminate between habitat odors, making it more difficult to locate appropriate settlement habitat (Devine et al. 2011); (3) timing of settlement was dramatically altered, and larval damselfish settled during a dangerous time—the full moon rather than new moon—when they were more vulnerable to predation (Devine et al. 2011); (4) larval damselfish could not visually recognize important predator species or evade predators (Ferrari et al. 2012b); (5) larval damselfish could not learn to respond appropriately to a common predator by watching other fish react or smelling injured fish (Ferrari et al. 2012a); (6) the function of an important neurotransmitter was impaired, which likely led to the sensory and behavioral impairment observed in acidified conditions (Nilsson et al. 2012).

These studies have important implications for the petitioned damselfish. These studies found that ocean acidification had negative effects across all six damselfish species tested, spanning two genera--*Pomacentrus* and *Neopomacentrus*--where the magnitude of the effects varied by species (all species were harmed, just to different degrees). The mechanism for the sensory and behavioral impairment is thought to be the disruption of an important neurotransmitter (the GABA-A receptor) which is found in all pomacentrids. Thus, the negative effects documented for the six damselfish study species are almost certain to apply across damselfish species, with differences in the magnitude of the effects varying by species.

i. Effects on smell

Like orange clownfish, larval damselfish exposed to elevated CO_2 become attracted to the odor of predators, and two species tested experienced a five-fold to nine-fold increase in predation rate as a result. Munday et al. (2010) examined the influence of CO_2-enriched water on the olfaction, behavior, and mortality rates of wild-caught larvae of Ward's damsel *Pomacentrus wardi*. About one-half of the fish kept at 700 ppm CO_2 were strongly attracted to the predator odor after four days, while all larvae kept at 850 ppm developed a strong attraction to the predator cue after two days. Furthermore, settlement-stage larvae raised at elevated CO_2 suffered high mortality when transplanted to coral-reef habitat. They were more active, ventured farther away from shelter, were less responsive to threats (i.e., were behaviorally bolder), and generally displayed riskier behavior than controls. As a result, larvae kept at 700 ppm suffered a five-fold increase in mortality from predation while larvae kept at 850 ppm experienced a nine-fold increase in mortality from predation. The researchers concluded that "additional CO_2 absorbed into the ocean will reduce recruitment success and have far-reaching consequences for the sustainability of fish populations":

> Our results show that CO_2 concentrations predicted to occur in the ocean by the end of this century may have dramatic effects on the behavior of fish larvae, with highly significant consequences for population replenishment and sustainability. Average CO_2 concentrations in the atmosphere and shallow ocean could reach 850 ppm by the end of the century (3, 4). Levels of dissolved CO_2 in this range (700–850 ppm) impaired the ability of larvae to respond to predator odors and caused them to exhibit riskier behavior in natural coral-reef habitat. Altered behavior was associated with dramatically higher mortality during a life-history stage when individuals are inherently vulnerable to predation, with the effects 50% greater at 850 ppm compared with 700 ppm CO_2. Increased recruitment

> mortality, of the scale detected here, would seriously threaten population sustainability. If other marine species exhibit similar behavioral responses, with corresponding impacts on survival, the effects of rising atmospheric CO_2 on marine biodiversity will be profound.

Another study tested the sensitivity of four sympatric damselfish species to elevated CO_2, and found that the anti-predator responses of all four were negatively affected, although their responses varied. The four species tested were the whitetail damsel *Pomacentrus chrysurus*, lemon damsel *Pomacentrus moluccensis*, Ambon damsel *Pomacentrus amboinensis* and Nagasaki damsel *Pomacentrus nagasakiensis* (Ferrari et al. 2011a). At 700 ppm, the Ambon damsel experienced the highest CO_2-induced loss of response to predation risk (95% loss), followed by the whitetail and lemon damsel at ~55% loss, and the Nagasaki damsel at ~30% loss (Ferrari et al. 2011a: Figure 1). For larval damselfish exposed to 850 ppm, three species experienced a complete loss of anti-predator response, exhibiting a maladaptive response to predator risk, and the Nagasaki damsel experienced a 40% reduction in response to predation risk. In another experiment focused on the whitetail damsel, whitetail damsel larvae raised under 700 ppm (predicted by ~2070) and 850 ppm (predicted by ~2100) suffered a five-fold to seven-fold increase in predation-related mortality, respectively, in the first few hours of settlement in the field. These larvae were more active, moved higher and further from the reef, had higher feeding rates, and were bolder than control fish (Ferrari et al. 2011a).

In a related experiment, Ferrari et al. (2011b) found that elevated CO_2 affects predator-prey dynamics. On coral reefs, there is intense predation of recruits, with at least 60% of newly settling fish being killed by predation in a single night; therefore, changes in predator-prey interactions can have significant consequences for damselfish populations. This study used four damselfish species (*Pomacentrus moluccensis*, *P. amboinensis*, *P. nagasakiensis* and *P. chrysurus*) including both small and large juveniles, one predator (the dottyback *Pseudochromis fuscus*), and two CO_2 levels (440 and 700µatm CO_2). In these experiments, the small juveniles of all four damselfish species suffered significantly higher predation under 700 µatm compared with 440 µatm, although there was no difference for the larger juveniles. The dottyback switched its preferred prey from P. *nagasakiensis* and *P. chrysurus* under 440 µatm to *P. moluccensis* and *P. amboinensis* under 700 µatm. The researchers hypothesized that the ability of *P. moluccensis* and *P. amboinensis* to detect and avoid predators was more heavily impaired under elevated CO_2 conditions, as suggested by Ferrari et al. (2011a), making them more vulnerable to the dottyback. Thus, some damselfish species may be more likely to be negatively affected by predation than others.

ii. Effects on timing of settlement

Devine et al. (2011) found that exposure of larval damselfish to CO_2-enriched water at levels expected within this century dramatically alters the timing of settlement "with potential consequences for larval survival and population replenishment." This study first examined the effects of elevated CO_2 on olfaction and habitat selection at settlement for three damselfish species that have differing habitat preferences: a habitat generalist (Ambon damsel *Pomacentrus amboinensis*), a rubble specialist (whitetail damsel *Pomacentrus chrysurus*) and a live coral specialist (lemon damsel *Pomacentrus moluccensis*). Exposure to elevated CO_2 (700, 850 ppm) disrupted the ability of settlement-stage larvae to discriminate between habitat odors, although larvae were able to select appropriate settlement habitat at the small spatial scale of aquaria when other sensory information was available. *P. chrysurus* exposed to CO_2-enriched water at 850 ppm exhibited peak settlement around the full moon, whereas control fish exhibited the highest settlement rates around the new moon. For many fish, peak settlement occurs during the new moon as a means to reduce mortality since lunar illumination is minimal. The authors concluded that rising CO_2 could cause larvae to settle at unfavorable times with potential fitness consequences for reef fish.

iii. Effects on vision

An experiment with the Ambon damsel *Pomacentrus amboinensis* found that juveniles exposed to elevated levels of CO_2 expected by the end of the century were not able to visually recognize on important predator species or execute evasive behavior normally seen in other juvenile damselfish (Ferrari et al. 2012b). Juvenile damselfish exposed to 440, 550 and 700 µatm CO_2 did not differ in their response to sight of the predator, the spiny chromis *Acanthochromis polyancanthus*. However, fish exposed to 850 µatm showed a significantly weaker anti-predator response by maintaining a higher foraging level, activity rate, and area use compared to other fish. Moreover, they moved closer to the chromis and lacked the bobbing behavior typically displayed by juvenile damselfish in threatening situations.

iv. Effects on learning

Ferrari et al. (2012a) found that higher concentrations of CO_2 may prevent damselfish from learning to avoid predators using visual and chemical information. Fish can learn to recognize and avoid predators through conditioning with odor cues from injured conspecifics (chemical cues) and through cultural learning by watching more experienced conspecifics react to predators (visual cues). Pre-settlement Ambon damselfish exposed to 850 µatm CO_2 failed to learn to respond appropriately to a common predator, the dottyback *Pseudochromis fuscus*, through either method of learning, although controls were able to use both types of learning. The researchers concluded that "[i]f our treatments represent future oceanic conditions on coral reefs, then evidence suggests that new recruit fishes will have a much reduced ability to assess predation risk and will as a consequence have much lower survival" (Ferrari et al. 2012a: 5). The researchers highlighted the importance of learning in allowing individuals to identify new habitats and mates, food sources, new threats, and adjust their behavior and phenology in response to environmental change. They warned that "[i]f CO_2 exposure is altering the cognitive ability of species, by either preventing them from learning or by altering the interpretation of environmental cues, the ecological consequences of ocean acidification will be far reaching, and may impinge on any conservation efforts to mitigate the ecological effects of ocean acidification" (Ferrari et al. 2012a: 8).

v. Effects on behavioral lateralization

Domenici et al. (2012) demonstrated that elevated CO_2 concentration (880 µatm) disrupted behavioral lateralization--an expression of brain function—in the larval damselfish yellowtail demoiselle *Neopomacentrus azysron*. Behavioral lateralization, the tendency to favor the right or left side in behavioral activities, has adaptive significance by enhancing performance in cognitive tasks and anti-predator behavior. "These results provide compelling evidence that elevated CO_2 directly affects brain function in larval fishes. Given that lateralization enhances performance in a number of cognitive tasks and anti-predator behaviours, it is possible that a loss of lateralization could increase the vulnerability of larval fishes to predation in a future high-CO_2 ocean" (p. 78).

vi. Effects on brain function

A study by Nilsson et al. (2012) indicates that altered behavior of larval and juvenile damselfish following exposure to elevated CO_2 is likely caused by impairment at the neurological level. Nilsson et al. (2012) found that the abnormal olfactory preferences and loss of behavioral lateralization in larval orange clownfish *Amphiprion percula* and the larval damselfish, yellowtail demoiselle *Neopomacentrus azysron,* exposed to high CO_2 is likely caused by interference with brain neurotransmitter function. Specifically, exposure to elevated CO_2 causes disrupted internal acid-base balance in marine fish; fish regulate their acid-base balance to avoid acidosis by accumulating HCO_3 and releasing Cl and H to the water. This change in the gradient of HCO_3 and Cl leads to the reversal of the function of a major neurotransmitter receptor in the brain, the GABA-A receptor, from being inhibited to being excited. The disruption in the neuronal pathway mediates a wide range of functions including olfactory discrimination, activity levels, and risk perception. As noted above, Nilsson et al. (2012) concluded that rising CO_2 levels are likely to harm a "wide range of marine species":

> Thus, our results indicate that high CO_2 interferes with neurotransmitter function, a hitherto unrecognized threat to marine populations and ecosystems. Given the ubiquity and conserved function of GABA-A receptors, we predict that rising CO_2 levels could cause sensory and behavioural impairment in a wide range of marine species… (Nilsson et al. 2012).

2. Ocean warming adversely affects damselfish reproduction and aerobic performance

Reef fishes such as damselfish that live in tropical oceanic environments are thought to be especially susceptible to temperature increases because they have evolved to live in a relatively thermostatic environment (Nilsson et al. 2009). As discussed above, average sea-surface temperatures in the vicinity of coral reef regions inhabited by the petitioned species are projected to increase by at least several degrees Celsius within this century. Changes of a few degrees Celsius can influence the physiological condition, developmental rate, growth rate, early life history traits, and reproductive performance of coral reef fishes, all of which can affect their population dynamics, community structure, and geographical distributions (Nilsson et al. 2009, Rankin et al. 2009). As reviewed below, studies on damselfish provide evidence that elevated ocean temperatures will have negative effects on reproduction, swimming performance, and aerobic capacity, which can lead to adverse population-level effects.

a. Negative effects on damselfish reproduction

Reproduction in tropical reef fishes is highly sensitive to temperature fluctuations (Munday et al. 2008). Temperature provides a cue for spawning and affects egg survival, embryonic duration, size at hatching, developmental rate, pelagic larval duration, and larval survival (Pankhurst and Munday 2011). The early life history stages of reef fish are more sensitive than adults to environmental fluctuations and are likely to be particularly vulnerable to increases in ocean temperature (Munday et al. 2008, Pankhurst and Munday 2011).

Several studies indicate that even relatively small increases in temperature can negatively affect the breeding rate, egg size, sperm production, and embryonic survival of damselfish. Donelson et al. (2010) found that breeding rate, egg size, and sperm production of the spiny chromis damselfish *Acanthochromis polyacanthus* were negatively affected by increases in temperature, and that these impacts were exacerbated by lower food availability. Fewer pairs bred at elevated water temperatures predicted to become close to the average for the study region over the next 50 to 100 years (30.0°C and 31.5°C) compared to the current-day average summer temperature for the collection location (28.5°C), and no pairs reproduced at either of the higher temperatures on a low quantity diet. Moreover, at higher temperatures, eggs were smaller and sperm production was reduced. The study concluded that "reduced breeding rate at warmer temperatures combined with reduced sperm production indicates the potential for significant declines in *A. polyacanthus* populations as the ocean warms." This study also suggests that the adverse effects of rising ocean temperatures on damselfish reproduction will be exacerbated if there is a concurrent negative impact on food availability (Pankhurst and Munday 2011).

Rapid rates of ocean warming are also likely to have negative effects on damselfish larval supply and development. For example, Lo-Yat et al. (2010) found that the larval supply of coral reef fishes in the nearshore waters around Rangiroa Atoll (French Polynesia) declined 51% below the mean value during a strong warm-water El Niño event when the sea surface temperature anomaly index rose 3.5ºC above average, the strength of the westward surface current toward the reef decreased, and concentrations of chlorophyll a (a measure of productivity) declined. The study concluded that "our results suggest that warming temperatures in the world's oceans will have negative effects on the reproduction of reef fishes and survival of their larvae within the plankton, ultimately impacting on the replenishment of benthic populations." Another study found that larval Ambon damselfish *Pomacentrus amboinensis* survived for a shorter period of time on their yolk-sac resources at higher temperatures (i.e., 31ºC compared with 29ºC) (Gagliano et al. 2007 cited in Munday et al. 2008). Warmer ocean temperatures have also been linked with increased mortality of recently settled juveniles of the humbug damselfish *Dascyllus aruanus*, as well as a decrease in allelic richness as the mortality rate rose, indicating that warmer waters impaired the genetic diversity of the population (Pini et al. 2011). On the other hand, some studies have shown that larval growth rate and recruitment of some reef fish species increase with warmer conditions that lie within the temperature range currently experienced by reef fish (Munday et al. 2008). However, the benefits associated with these modest increases in temperature may be unlikely to occur under the much larger temperature increases predicted under climate change, and may be counteracted by the negative effects of increased temperature on other developmental stages, such as reduced egg production and increased egg mortality, which can reduce the number of larvae entering the pelagic phase (Munday et al. 2008). A faster rate of growth and development would also increase the energy and food requirements for larval fish which could place them under higher physiological stress (Pankhurst and Munday 2011).

On a final note, seasonal changes in temperature are important cue for spawning in reef fish, and rising temperatures due to climate change have the potential to alter the timing and duration of spawning, truncate seasonal spawning, or completely inhibit spawning (Pankhurst and Munday 2011). For example, increasing temperatures stimulate reproduction in spring-spawning species while decreasing temperatures cue reproductive development in autumn-spawning species (Pankhurst and Munday 2011). Thus, the magnitude of temperature increases and the timing of increases with respect to the natural seasonal timing of spawning will be important factors in determining the impacts on reef fish species (Pankhurst and Munday 2011). Overall, the effects of increased temperature on reproduction of the petitioned species are likely to include many adverse impacts.

b. Negative effects on damselfish aerobic capacity and swimming performance

Several studies indicate that the aerobic capacity and swimming performance of damselfish is significantly diminished under temperature conditions expected within this century. Warming temperatures can negatively affect damselfish by increasing their basal metabolic rate and associated oxygen demands (Nilsson et al. 2009). Because the circulatory and ventilatory systems of reef fish have a limited ability to keep pace with increased oxygen demands, aerobic performance (e.g. aerobic scope) is expected to decrease with rising ocean temperature, which would result in less energy going towards critical functions such as feeding, growth, and reproduction (Nilsson et al. 2009). Lower aerobic performance at higher temperatures would affect all aspects of a fish's individual performance and ultimately affect population viability (Nilsson et al. 2009). For example, reduced performance at higher temperatures lowers fishes' ability to cope with otherwise normal fluctuations in habitat conditions such as flow variations due to tidal changes, currents and storms; leads to a diminished capacity to conduct ecological tasks such as foraging, which are fundamental to growth, survival and reproduction; and affects the ability of a species to maintain cellular processes whilst continuing physical activity (Johansen and Jones 2011), thus posing a significant threat at the individual and population levels.

Nilsson et al. (2009) detected a decline in the aerobic capacity of three different species of damselfish at water temperatures that were two or three degrees Celsius warmer than current conditions, including the petitioned species, black-axil chromis *Chromis atripectoralis*, as well as *Dascyllus aruanus*, and *Acanthochromis polyacanthus*. At an increase of 4ºC, these damselfish only retained half their aerobic scope. Similarly, Johansen and Jones (2011) found that warmer water temperatures at 3ºC over ambient levels reduced the aerobic performance of ten coral reef damselfishes, including several evolutionary lineages, and also greatly reduced the swimming performance of certain species. The study tested ten damselfish species from two subfamilies and four genera including two petitioned species: black-axil chromis *Chromis atripectoralis*, ternate chromis *Chromis ternatensis*, whitetail dascyllus *Dascyllus aruanus*, reticulated damsel *Dascyllus reticulatus*, yellowtail demoiselle *Neopomacentrus azysron*, Chinese demoiselle *Neopomacentrus bankieri*, regal demoiselle *Neopomacentrus cyanomos*, neon damsel *Pomacentrus coelestis*, scaly damsel *Pomacentrus lepidogenys*, and lemon damsel *Pomacentrus moluccensis*. Increased temperature had a significant negative effect across all five measures of aerobic performance: maximum swimming speed, gait-transition speed, maximum aerobic metabolic rate, resting aerobic metabolic rate, and aerobic scope. Five species spanning three genera (*Dascyllus*, *Neopomacentrus* and *Pomacentrus*) showed severe reductions in swimming performance, and five species spanning all four genera showed significant reductions in metabolic performance with aerobic scope reduced by 24.3–64.9%. The authors concluded that "significant loss of species may occur" at ocean warming levels expected within this century, absent adaptation:

> [C]omparisons of remaining performance capacities with field conditions indicate that 32° C water temperatures will leave multiple species with less swimming capacity than required to overcome the water flows commonly found in their respective coral reef habitats. Consequently, unless adaptation is possible, significant loss of species may occur if ocean warming of ≥3° C arises. (Johansen and Jones 2011).

Similarly, Donelson et al. (2011) found that the damselfish *Acanthochromis polyacanthus* experienced decreases in aerobic capacity of 15% and 30% under elevated temperatures of 1.5ºC and 3.0ºC, respectively. However, the study found that this species may have some ability to acclimate to elevated temperatures. When both parents and offspring were reared throughout their lives at elevated temperatures, they were able to regain their aerobic capacity. While the study offers some hope that damselfish may have some capacity for acclimation, the study also provided several caveats to this conclusion. First, genetic diversity of the acclimated damselfish was reduced at the warmest temperatures due to differential breeding success among familial lineages. The researchers warned that reduced diversity could compromise disease resistance and reduce plasticity to further environmental changes in these populations. Second, the researchers noted that the physiological modifications involved in acclimation and adaptation may come at a cost to other attributes, such as growth. Finally, Johansen and Jones (2011) cautioned that the potential for acclimatization, developmental plasticity, and genetic adaptation may not be adequate to compensate for the negative effects of rapid warming:

> Although acclimation, developmental plasticity and genetic adaptation may alleviate some physical and physiological limitations (e.g. Donelson et al., 2011), such adaptations may be inadequate to fully compensate for the predicted rate of global warming, leaving reef communities with severely reduced resilience to any further external stressors. For instance, although prolonged acclimation periods (of 42 weeks) have previously been show to increase the thermal tolerance of reef fishes (Eme & Bennett, 2009), severe reductions in physiological performance was still evident in this study after longer acclimation periods. In addition, the effect of increased temperature on the physiological performance of tropical coral reef damselfishes varied greatly among species and did not seem related to phylogenetic distance nor ecological adaptations, leaving the thermal resilience of many coral reef fish species unclear. However, significant reductions in swimming ability, some in species that showed no reduction in aerobic scope (e.g. D. aruanus), demonstrate the benefit of studying temperature's effect on coral reef

> fishes from multiple angles and highlight that performance capacities of individual species must be seen in context of species-specific distribution, ecology and habitat usage. (Johansen and Jones 2011).

As summarized by Pankhurst et al. (2011), the potential for genetic adaptation to rapid climate change depends on a range of factors, including the amount of adaptive genetic variation present, effective population sizes, generation time, and connectivity between populations that can aid in the spread of tolerant genotypes, and this information is little known for damselfish. In short, the best-available science indicates that the petitioned damselfish species will experience reduced aerobic capacity that is likely to result in negative population-level effects that exacerbate the large set of other stressors from ocean warming and ocean acidification.

C. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes (Listing Factor C): Over-Harvest for the Global Marine Aquarium Trade

Damselfish and anemonefish are the most commonly harvested species in the global trade of marine aquarium fish. Studies suggest that several of the petitioned species, including the orange clownfish, black-axil chromis, blue-green damselfish, and Hawaiian damselfish, are threatened by overexploitation in the global marine aquarium trade in some regions. As summarized by Jones et al. (2008), “localized depletion is a recognized threat to heavily targeted species such as the anemonefishes.” The widespread and growing trade in coral reef fish and corals adds to the cumulative stresses that the petitioned pomacentrids face from ocean warming and ocean acidification.

1. Overview of the global trade in marine aquarium fish

The global trade in ornamental marine fish is a major, multi-million dollar industry, valued at US $200-330 annually, that is estimated to remove up to 30 million marine fish each year in addition to large quantities of coral reef habitat (Shuman et al. 2005, Tissot et al. 2010, Rhyne et al. 2012). A recent study found that the United States, during a one-year period, imported more than 11 million marine ornamental fish of more than 1,800 species from 125 families (Rhyne et al. 2012). In addition, the global trade in corals is estimated to involve 1.5 million live stony corals and more than two million kilograms of dead corals each year from hundreds of coral species (Tissot et al. 2010, Rhyne et al. 2012).

The majority of marine ornamental species are collected from wild populations as juveniles or adults from coral reef ecosystems (Shuman et al. 2005, Stevenson et al. 2011). Overall, only about one percent of the marine fish species in commercial trade are captive bred (Bruckner 2005). Unfortunately, for some reef fish species, the marine aquarium trade has caused the virtual elimination of local populations, major changes in age structure, and the promotion of collection practices that destroy reef habitats (Tissot et al. 2010).

The United States is the world’s largest importer of marine ornamental species, with imports representing 80% of global trade, followed by Europe and Japan (Shuman et al. 2005, Tissot et al. 2010, Rhyne et al. 2012). Approximately 40 countries supply fish to the marine aquarium trade in the United States (Rhyne et al. 2012). The Philippines and Indonesia are the largest global suppliers, accounting for ~87% of exports (55% and 31% of individuals, respectively), followed by Brazil, Maldives, Vietnam, Sri Lanka and Hawai'i (Bruckner 2005, Rhyne et al. 2012). The diversity and volume of species traded is substantial and growing (Shuman et al. 2005, Tissot et al. 2010). For example, between 1988 and 2007, the imports of live corals taken directly from reefs to the United States increased by 600%, while the global volume of live coral imports increased nearly 1500% (Tissot et al. 2010).

Despite the diversity of fish species involved in the aquarium trade, a large portion of global trade is concentrated on a few families and species, with damselfish and anemonefish (Pomacentridae) dominating global trade (Zajicek et al. 2009, Rhyne et al. 2012). An analysis of data from the Global Marine Aquarium Database during 1997 to 2002 found that Pomacentridae accounted for 43% of all fish traded (Zajicek et al. 2009). More recently, Rhyne et al. (2012) analyzed marine ornamental fish imported into the United States between May 2004 and May 2005. This study found that more than half of the marine aquarium fish imported into the United States were damselfish and anemonefish (Figure 14). Twenty species represented 52% of the total number of individuals imported, and ten of these top 20 species were damselfish or anemonefish, which accounted for 76% of the individuals in this 52% (Rhyne 2012). The most commonly imported species, accounting for ~9% of imports and more than 900,000 individuals each year, was the petitioned blue-green damselfish *Chromis viridis*, while the orange clownfish *Amphiprion percula* and its sister species *A. ocellaris* were the fifth most imported species with over 400,000 individuals imported each year (Figure 15) (Rhyne et al. 2012).

Figure 14. Composition by family of marine aquarium fish imported into the United States. Data for the top 20 families are provided, with the remainder grouped as 'other'. The number proceeding the family name in the legend signifies the number of species imported within each family.
Source: Rhyne et al. (2012): Figure 1.

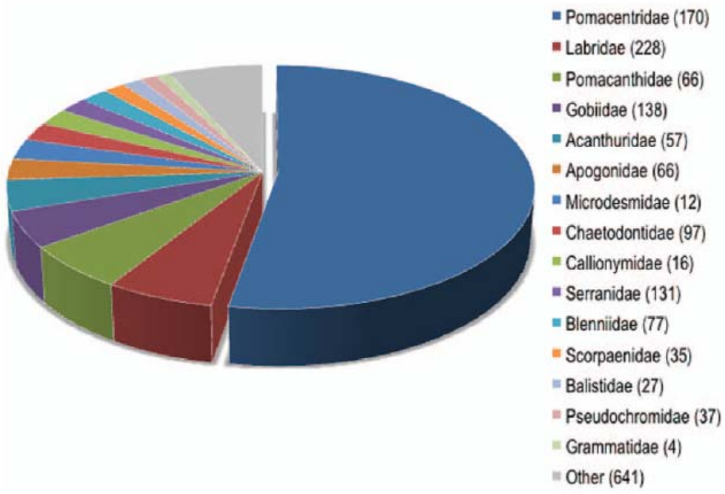
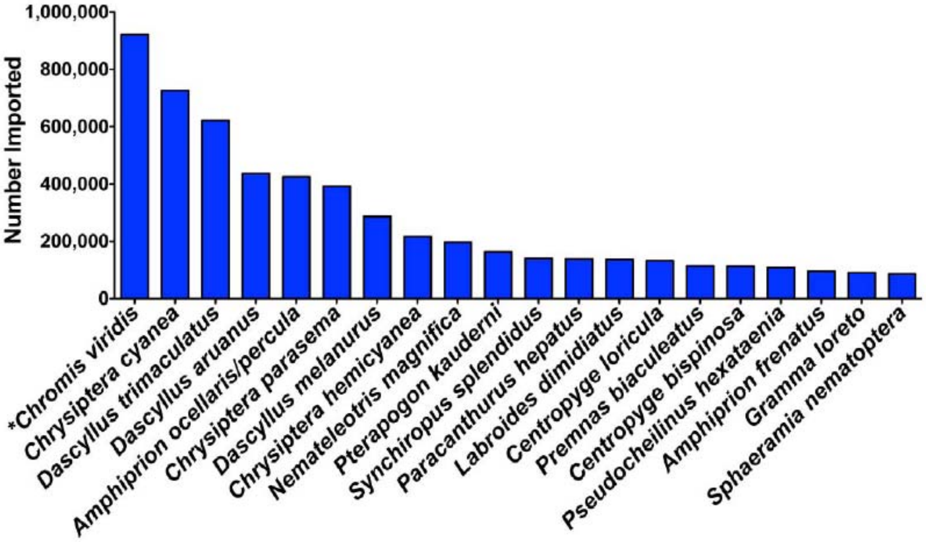


Figure 15. Top 20 marine aquarium fish imported into the United States. * indicate species complexes, which could represent more than one species which are all traded under the same name.
Source: Rhyne et al. (2012): Figure 2.



2. Evidence of impacts of the marine aquarium trade on petitioned species

Several studies indicate that the orange clownfish is threatened by overexploitation in some regions due to the marine aquarium trade. The orange clownfish has several characteristics that make it vulnerable to localized population decline by the marine aquarium trade: (1) it is a habitat specialist that is restricted to host anemones which are themselves subject to harvesting and susceptible to overexploitation, (2) it has life history traits which make it slow to recover from population declines, including long lifespan (i.e. 30 years for females), restricted movement (i.e. they rarely migrate further than a few hundred meters), and a short larval lifespan, high larval mortality rate, and low recruitment success of larval fish (Jones et al. 2008). In addition, anemones are long-lived, slow growing, and have relatively low reproductive rates (i.e. they spawn infrequently, have low spawning success, and have short larval lifespan) which makes anemones vulnerable to depletion (Shuman et al. 2005, Jones et al. 2008).

Two studies have found evidence for population-level declines of anemonefish due to the marine aquarium trade. A study in the Central Visayas region of the Philippines determined that aquarium fishing activities negatively impacted the populations of both anemones and anemonefish in the study region (Shuman et al. 2005). Analysis of catch records obtained from collectors over a four-month period showed that anemonefish and anemones comprised 57% of the total catch. Underwater surveys found that both anemone and anemonefish densities were significantly lower in exploited areas than in protected areas. The low density of anemones on exploited reefs accounted for over 80% of the reduced density of anemonefish at those sites. The average price for anemones was estimated to be US $0.35, while anemonefish ranged from US $0.03 to 0.12 per fish, making both anemones and anemonefish profitable in this region. A study on the Great Barrier Reef by Jones et al. (2008) found that the highest densities of anemonefishes were found on reefs that were closed to fishing and aquarium collecting in both the Keppel Islands and Far North Queensland. The study concluded that collecting is compounding the negative effects of bleaching in this region.

High profile exposure from the Pixar movie "Finding Nemo" increased the global demand and trade of the orange clownfish (Osterhoudt 2004, Prosek 2010, Rhyne et al. 2012). As discussed above, the orange clownfish and its sister species *A. ocellaris* were the fifth most imported species in the United States in 2004-2005, accounting for more than 400,000 individuals imported (Figure 15) (Rhyne et al. 2012).

Analyses of the aquarium fishery in Hawaii, the Philippines, and Florida also indicate that damselfish, including two petitioned species *Dascyllus albisella* and *Chromis atripectoralis*, may face threats from overharvest. A study of Hawaii's aquarium fishery examined catches from the west coast of the Big Island of Hawaii where most of the fish catch value is generated (Stevenson et al. 2011). The study noted that Hawaii's aquarium fishery developed rapidly in the early 1970s to become the state's most lucrative nearshore fishery by the 2000s (Stevenson et al. 2011). The most commonly targeted fish---the yellow tang--declined by 45% in areas open to aquarium fishing between 1999 and 2007, likely due to increased fishing pressure in permitted areas after marine protected areas were established in 1999 (Stevenson et al. 2011); these declines raise cause for concern for the sustainability of the aquarium fishery. In addition, Pomacentridae was the second most commonly targeted fish family after Acanthuridae (Stevenson et al. 2011). The Hawaiian damselfish (*Dascyllus albisella*) was one of the top three most preferred fish by aquarium fishers, reflected by its high selectivity index (i.e. its high importance in fishers' catches relative to its lower abundance on the reef) (Stevenson et al. 2012), which also raises cause for concern over the potential for overharvest of this species.

A study in the central Visayan region of the Philippines, which historically had the highest concentration of coral reef fishes of any large-marine area in the world, found evidence that the marine aquarium fishery is lowering the overall species richness and abundance of coral reef fish in the region (Nañola et al. 2010). The study found that this region now has unusually low counts of species exploited in fisheries and the aquarium trade in this region, and documented numerous reports of intense fishing, habitat degradation, and subsequent species declines at local scales. Four species of damselfishes that are subject to the aquarium trade were markedly absent in this region, including one of the petitioned species, the black-axil chromis *Chromis atripectoralis.* Finally, an analysis of Florida's marine aquarium fishery that damselfish were the third most economically important species group in the fishery (Adams et al. 2001), which raises cause for concern over the sustainability of damselfish harvest.

D. Existing Regulatory Mechanisms Are Inadequate to Address Threats to the Petitioned Pomacentrid Species From Greenhouse Gas Pollution, Degradation of their Coral Reef Habitat, and the Marine Aquarium Trade (Listing Factor D)

1. Regulatory mechanisms addressing greenhouse gas pollution are inadequate to protect the petitioned pomacentrid species

As acknowledged by NMFS in its *Status Review Report of 82 Candidate Coral Species* and accompanying *Management Report*, national and international regulatory mechanisms have been ineffective in reducing emissions to levels that do not jeopardize coral reef habitats. No countries are reducing emissions enough to keep the increase in global temperature below 2ºC (the international target for limiting global temperature rise, but which is insufficient to protect the world's coral reefs), and the top ten emitters accounting for over 60% of the global emissions, including the United States, are performing poorly or very poorly at meeting needed greenhouse gas reductions (NMFS 2012: 43). As summarized by NMFS in the *Status Review Report*: "While many international, national, and local initiatives have sought to reduce the growth in greenhouse gas emissions, recent emissions growth and an apparent lack of significant international political action to control emissions to date have resulted in an acceleration of CO_2 emissions at or above the worst-case scenario used in the IPCC's Third and Fourth Assessment Reports (Brainard et al. 2012: 25). The continued failure of the U.S. government and the international community to implement effective and comprehensive greenhouse gas reduction jeopardizes the petitioned reef fish and many of the corals on which they depend with extinction.

a. U.S. regulatory mechanisms to reduce greenhouse gas emissions are insufficient

While existing domestic 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 United States, these agencies are either failing to implement or only partially implementing these laws for greenhouse gases. For example, the Environmental Protection Agency (EPA) has issued a rulemaking regulating greenhouse gas emissions from automobiles that will reduce greenhouse emissions emitted per vehicle mile traveled by passenger vehicles in the future, but because the improvements are modest and more vehicles are projected to be driven more miles in the future, the rule will not reduce emissions from this sector overall but will only slow the rate of increase somewhat compared to what it would be without the rule. EPA, Light-Duty Vehicle Greenhouse Gas Emission Standards and Corporate Average Fuel Economy Standards; Final Rule, 75 Fed. Reg. 25,324 (May 7, 2010). Meanwhile even the government concedes that "these reductions in emissions are not sufficient by themselves to reduce total HD vehicle emissions below their 2005 levels by 2020." NHTSA, *Medium- and Heavy-Duty Fuel Efficiency Improvement Program – Final Environmental Impact Statement* (June 2011). The EPA has also to date issued only a single proposed rule under the new source pollution standard program for stationary sources of pollution, for electric generating units (power plants). While there is enormous potential to reduce emissions through this program overall and through the power plants rule in particular, the EPA has instead proposed a weak and flawed rule that it admits will not reduce emissions from these sources between now and 2020 compared to what would be expected without the rule. EPA, Standards of Performance for Greenhouse Gas Emissions for New Stationary Sources: Electric Utility Generating Units, 77 Fed. Reg. 22392, 22430-33 (April 13, 2012). Indeed, in the rulemaking the EPA conceded that new power plant rule on greenhouse gas emissions "will not have direct impact on U.S. emissions of greenhouse gases under expected economic conditions." Id. at 22401. While full implementation of our flagship environmental laws, particularly the Clean Air Act, would provide an effective and comprehensive greenhouse gas reduction strategy, due to their nonimplementation, existing domestic regulatory mechanisms must be considered inadequate to protect the petitioned reef fish species from climate change and ocean acidification.

b. International measures to reduce greenhouse gas emissions are inadequate

International initiatives are also currently inadequate to effectively address climate change. The United Nations Framework Convention on Climate Change, negotiated in 1992 at Rio de Janeiro, Brazil, provides the forum for the international negotiations. In the Framework Convention, signed and ratified by the United States, the world agreed to take the actions necessary to avoid dangerous climate change. Parties to the Convention also agreed as a matter of fairness that the world's rich, developed countries, having caused the vast majority of emissions responsible for the problem, would take the lead in solving it. It was not until the 1997 meeting in Kyoto, Japan, that the first concrete, legally binding agreement for reducing emissions was signed: the Kyoto Protocol. The Protocol requires the world's richest countries to reduce emissions an average of 5 percent below 1990 levels by 2012, while developing nations also take steps to reduce emissions without being subject to binding emissions targets as they continue to raise their standard of living. The United States has been a major barrier to progress in the international negotiations. After the Clinton administration extracted many concessions from the rest of the world in exchange for the United States signing on in Kyoto, the Senate rejected the equity principles behind the Convention, saying the United States shouldn't agree to reduce its own emissions unless all other countries — regardless of their responsibility or ability — were similarly bound. Citing the same excuses, President George W. Bush repudiated the Kyoto Protocol entirely. Thus the United States is the only industrialized country in the world that has yet to ratify the Kyoto Protocol. The United States negotiating team under both the George W. Bush and the Obama administrations has pursued two primary objectives in the international talks: to refuse any legally binding emissions reduction commitments until all other countries— but particularly China and India — do so, and to push back the date for a new agreement. Not surprisingly, the United States had failed to meet its (never ratified) Kyoto pledge to reduce emissions to 7.2% below 1990 levels by 2012; to the contrary, U.S. emissions have increased by 10.5% since 1990 (EPA 2012).

Moreover, Kyoto Protocol's first commitment period only sets targets for action through 2012, and there is still no binding international agreement governing greenhouse gas emissions in the years beyond 2012. While the 2009 U.N. Climate Change Conference in Copenhagen called on countries to hold the increase in global temperature below 2°C (an inadequate target for avoiding dangerous climate change and protecting corals), the *non-binding* "Copenhagen Accord" that emerged from the conference, and the subsequent "Cancún Accords" of 2010 and "Durban Platform" of 2011, failed to enact binding regulations that limit emissions to reach this goal.[3] Even if countries were to meet their Copenhagen and Cancún pledges, analyses have

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[3] The non-legally binding Copenhagen Accord of 2009 and Cancún Accords of 2010 recognize the objective of limiting warming to 2°C above pre-industrial, but do not enact binding regulations to achieve this goal (http://cancun.unfccc.int/cancun-agreements/main-objectives-of-the-agreements/#c33; unfccc.int/resource/docs/2009/cop15/eng/11a01.pdf). According to the Durban Platform, developed and developing nations agreed to a process to develop a "new protocol, another legal instrument, or agreed outcome with legal force that will be applicable to all Parties to the UN climate convention"; this legal instrument must be developed as of 2015 and will not take effect until 2020 (unfccc.int/resource/docs/2011/cop17/eng/l10.pdf).

found that collective national pledges to cut greenhouse gas emissions are inadequate to achieve the 2°C target, and instead suggest emission scenarios leading to 2.5°C to 5°C warming (Rogelj et al. 2010, UNEP 2010, UNEP 2011). As of May 2012, many governments were not implementing the policies needed to meet their 2020 emission reduction pledges, making it more difficult to keep global temperature rise to 2°C (Höhne et al. 2012). As noted in the NMFS *Management Report*, the U.S. has yet to issue regulations to limit greenhouse gas emissions in accordance with the U.S. pledge under the Copenhagen Accord (NMFS 2012: 14). Due to these failures, the planet is heading for a temperature rise of at least 3.5°C, and higher if the 2020 pledges are not met (Höhne et al. 2012), which would be devastating for the petitioned reef fish and their coral reef habitat.

2. Regulatory mechanisms for protecting coral reef habitat are inadequate

Despite international and domestic efforts to reduce threats to coral reefs, more than 60% of the world's coral reefs are immediately and directly threatened by local threats including overfishing, coastal development and pollution according to a comprehensive 2011 review of the status and threats to the world's coral reefs (Burke et al. 2011):

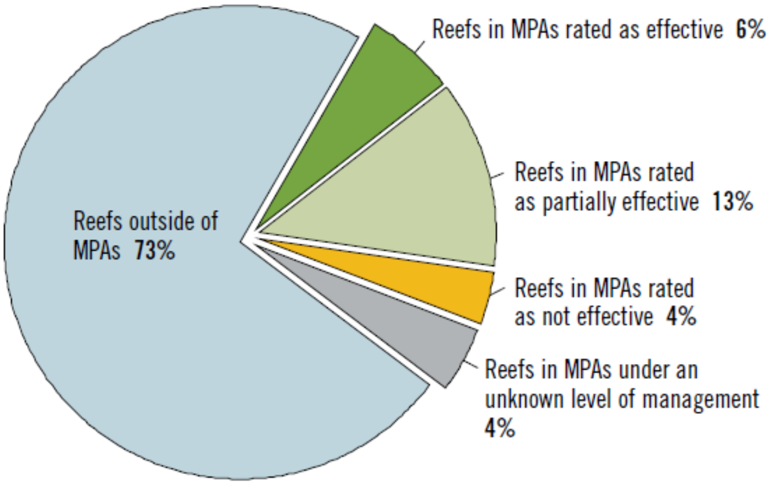
> More than 60 percent of the world's reefs are under immediate and direct threat from one or more local sources —such as overfishing and destructive fishing, coastal development, watershed-based pollution, or marine-based pollution and damage.
>
> Of local pressures on coral reefs, overfishing—including destructive fishing—is the most pervasive immediate threat, affecting more than 55 percent of the world's reefs. Coastal development and watershed-based pollution each threaten about 25 percent of reefs. Marinebased pollution and damage from ships is widely dispersed, threatening about 10 percent of reefs. (Burke et al. 2011: 6).

This high level of threat clearly indicates that existing regulatory mechanisms are inadequate to protect the coral reefs on which the petitioned pomacentrids depend.

Moreover, marine protected areas (MPAs), while potentially beneficial, are insufficient to conserve the coral reef habitat of the petitioned pomacentrids. Even with one-quarter of the world's coral reefs within protected areas, many are ineffective or only offer partial protection (Figure 16). Only 6% of coral reefs are in effectively managed MPAs (Burke et al. 2011). The success of marine reserves to recover coral reefs has also been brought into question.

Figure 16. Percentage of coral reefs inside MPAs and the effectiveness of MPAs.
Source: Burke et al. (2011): Figure ES-4.

FIGURE ES-4. CORAL REEFS BY MARINE PROTECTED AREA COVERAGE AND EFFECTIVENESS LEVEL



Note: The global area of coral reefs is 250,000 sq km (which represents 100% on this chart), of which 67,350 sq km (27%) is inside MPAs.

Even with laws that aim to conserve coral reef ecosystems, many countries lack adequate capacity or political will to enforce laws. Moreover, there are currently no effective regulatory mechanisms to address the growing stresses on coral reef ecosystems from increasing human populations and poverty.

3. Regulatory mechanisms for the marine aquarium trade are inadequate

United States and international regulations are inadequate to protect the petitioned pomacentrids from threats from the global marine aquarium trade. As summarized by Tissot et al. (2010), management and regulation of collection and international trade in marine ornamental species are inadequate due to (1) insufficient legal and regulatory tools and enforcement in the United States (the world's leading importer) governing the import of coral reef species combined with (2) insufficient legal and regulatory tools and enforcement for collection, export, and management of source fish populations in the source countries. Tissot et al. (2010) highlighted problems of weak governance capacity in major source countries such as Indonesia and the Philippines; high international demand, particularly from the United States, which provide few incentives to strengthen trade policies or management practices; roving collectors that can undermine local management efforts; and inadequate enforcement of the few existing laws, allowing collectors to use illegal and harmful collection methods such as sodium cyanide. Jones et al. (2008) reported that the marine aquarium industry is almost entirely self-regulating. Management authorities use fisheries logbook information on catch per unit effort of harvested fish to assess stock depletion of target species, but this information is "notoriously unreliable" for assessing populations, which can lead to overharvest of heavily targeted species (Jones et al. 2008).

Tissot et al. (2010) identified several ways in which legal mechanisms to regulate trade in the United States are insufficient: (1) Although the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) has the capacity to regulate trade to prevent collection that is detrimental to species survival, few coral reef species subject to international trade are listed under CITES, and it is very difficult to list deserving species under CITES as illustrated by the failure to list red and pink coral under Appendix II in 2007 and 2010. (2) Although the Lacey Act prohibits wildlife imports obtained in violation of international laws such as the collection of reef fish by cyanide, the United States lack the enforcement capacity to detect whether imported fish species were collected illegally, such as through cyanide detection tests. Furthermore, while the Lacey Act provides for the humane and healthful transport of live animals into the United States, there are currently no mandatory or enforced regulations in place for human and healthful transport. (3) The U.S. Fish and Wildlife Service, which is tasked with inspecting shipments of imported wildlife to ensure that these imports comply with the law, has insufficient capacity at present for enforcement; additional resources, training, and better tracking systems are needed. Based on the current state of weak regulation and enforcement of the marine aquarium trade in the United States and internationally, regulatory mechanisms must be deemed inadequate.

Critical Habitat Designation

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); *see also id.* at § 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.

Id. at § 1532(5)(A).

The Center for Biological Diversity expects that NMFS will comply with this unambiguous mandate and designate critical habitat concurrently with the listing of the petitioned pomacentrid reef fish species that occur in U.S. waters. We believe that all current and historic areas utilized by these species meet the criteria for designation as critical habitat and must therefore be designated as such.

Conclusion

As demonstrated in this Petition, each of the eight petitioned pomacentrid reef fish species faces high-magnitude and growing threats to its continued existence. NMFS must promptly make a positive 90-day finding on this Petition, initiate a status review, and expeditiously proceed toward listing and protecting these species. We look forward to the official response as required by the ESA.

References Cited

The pdfs of all references cited in this petition will be submitted to NMFS on a compact disk accompanying the petition, with the exception of books which will not be submitted.

Adams, C. M., S. L. Larkin, and D. J. Lee. 2001. Volume and value of marine ornamentals collected in Florida , 1990 – 98. Aquarium Sciences and Conservation 3:25-36.

Allen, G. R. 1991. Damselfishes of the World. Page 271. Hans A. Baensch, Melle, Germany.

Asoh, K. 2003. Reproductive parameters of female Hawaiian damselfish Dascyllus albisella with comparison to other tropical and subtropical damselfishes. Marine Biology 143:803-810.

Asoh, K., and T. Yoshikawa. 2002. The role of temperature and embryo development time in the diel timing of spawning in a coral-reef damselfish with high-frequency spawning synchrony. Environmental Biology of Fishes 64:379-392.

Australian Institute of Marine Science. 2012. Hoskyn Island's Reef: intensive surveys--trends in fish abundance. Retrieved August 15, 2012, from http://data.aims.gov.au/reefpage2/rpdetail.jsp?fullReefID=23080S&sampleType=FISH.

Bak, M., G. Nieuwland, and E. H. Meesters. 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, A. C., P. W. Glynn, and B. Riegl. 2008. Climate change and coral reef bleaching: An ecological assessment of long-term impacts, recovery trends and future outlook. Estuarine, Coastal and Shelf Science 80:435-471.

Barton, A., B. Hales, G. Waldbusser, C. Langdon, and R. A. Feely. 2012. The Pacific oyster, Crassostrea gigas, shows negative correlation to naturally elevated carbon dioxide levels: Implications for near-term ocean acidification effects. Limnol. Oceanography.

Bates, N., A. Amat, and A. Andersson. 2010. Feedbacks and responses of coral calcification on the Bermuda reef system to seasonal changes in biological processes and ocean acidification. Biogeosciences 7:2509–2530.

Ben-Tzvi, O., A. Abelson, O. Polak, and M. Kiflawi. 2008. Habitat selection and the colonization of new territories by Chromis viridis. Journal of Fish Biology 73:1005-1018.

Bindoff, N. L., J. Willebrand, V. Artale, A. Cazenave, J. Gregory, S. Gulev, K. Hanawa, C. Le Quéré, S. Levitus, Y. Nojiri, C. K. Shum, L. D. Talley, and A. Unnikrishnan. 2007. 2007: Observations: Oceanic Climate Change and Sea Level. Page *in* S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, and H. L. Miller, editors. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth

Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Bohlke, J. E., and C. C. Chaplin. 1993. Fishes of the Bahamas and Adjacent Tropical Waters. University of Texas Press, Austin.

Bonin, M. C. 2012. Specializing on vulnerable habitat: Acropora selectivity among damselfish recruits and the risk of bleaching-induced habitat loss. Coral Reefs 31:287-297.

Bonin, M., P. Munday, M. McCormick, M. Srinivasan, and G. Jones. 2009. Coral-dwelling fishes resistant to bleaching but not to mortality of host corals. Marine Ecology Progress Series 394:215-222.

Booth, D. J. 1992. Larval settlement patterns and preferences by domino damselfish Dascyllus albisella Gill. Journal of Experimental Marine Biology and Ecology 155:85-104.

Booth, D. J. 2002. Distribution changes after settlement in six species of damselfish (Pomacentridae) in One Tree Island lagoon, Great Barrier Reef. Marine Ecology Progress Series 226:157-164.

Booth, D. J., and G. A. Beretta. 2002. Changes in a fish assemblage after a coral bleaching event. Marine Ecology Progress Series 245:205-212.

Brainard, R. E., C. Birkeland, C. M. Eakin, P. Mcelhany, M. W. Miller, M. Patterson, and G. A. Piniak. 2012. Status Review Report of 82 Candidate Coral Species Petitioned Under the U.S. Endangered Species Act. U.S. Dep. Commerce, NOAA Tech. Memo., NOAA-TM-NMFS-PIFSC-27, 530 p. + 1 Appendix. Page Russell The Journal Of The Bertrand Russell Archives.

Bruckner, A. W. 2005. The importance of the marine ornamental reef fish trade in the wider Caribbean. Revista de Biologia Tropical 53.

Buddemeier, R. W., D. R. Lane, and J. A. Martinich. 2011. Modeling regional coral reef responses to global warming and changes in ocean chemistry: Caribbean case study. Climatic Change 109:375-397.

Burke, L., K. Reytar, M. Spalding, and A. Perry. 2011. Reefs at Risk Revisited. World Resources Institute, Washington, D.C.

Buston, P. M., G. P. Jones, S. Planes, and S. R. Thorrold. 2012. Probability of successful larval dispersal declines fivefold over 1 km in a coral reef fish. Proceedings. Biological Sciences / The Royal Society 279:1883-8.

Buston, P. M., and J. Elith. 2011. Determinants of reproductive success in dominant pairs of clownfish: a boosted regression tree analysis. The Journal of Animal Ecology 80:528-38.

Buston, P. M., and M. B. García. 2007. An extraordinary life span estimate for the clown anemonefish Amphiprion percula. Journal of Fish Biology 70:1710-1719.

Caldeira, K., and M. E. Wickett. 2005. Ocean model predictions of chemistry changes from carbon dioxide emissions to the atmosphere and ocean. Journal of Geophysical Research-Oceans 110.

Cameron, M., J. L. Bengtson, P. Boveng, J. K. Jansen, B. P. Kelly, S. P. Dahle, E. A. Logerwell, J. E. Overland, C. L. Sabine, G. T. Waring, and J. Wilder. 2010. Status review of the bearded seal (Erignathus barbatus). U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-211, 246 p.

Cao, L., and K. Caldeira. 2008. Atmospheric CO_2 stabilization and ocean acidification. Geophysical Research Letters 35, L19609.

Carpenter, K. E., M. Abrar, G. Aeby, R. B. Aronson, S. Banks, A. Bruckner, A. Chiriboga, J. Cortés, J. C. Delbeek, L. Devantier, G. J. Edgar, A. J. Edwards, D. Fenner, H. M. Guzmán,

B. W. Hoeksema, G. Hodgson, O. Johan, W. Y. Licuanan, S. R. Livingstone, E. R. Lovell, J. a Moore, D. O. Obura, D. Ochavillo, B. A. Polidoro, W. F. Precht, M. C. Quibilan, C. Reboton, Z. T. Richards, A. D. Rogers, J. Sanciangco, A. Sheppard, C. Sheppard, J. Smith, S. Stuart, E. Turak, J. E. N. Veron, C. Wallace, E. Weil, and E. Wood. 2008. One-third of reef-building corals face elevated extinction risk from climate change and local impacts. Science (New York, N.Y.) 321:560-3.

Chollett, I., F. E. Müller-Karger, S. F. Heron, W. Skirving, and P. J. Mumby. 2012. Seasonal and spatial heterogeneity of recent sea surface temperature trends in the Caribbean Sea and southeast Gulf of Mexico. Marine Pollution Bulletin 64:956-65.

Coker, D. J., M. S. Pratchett, and P. L. Munday. 2009. Coral bleaching and habitat degradation increase susceptibility to predation for coral-dwelling fishes. Behavioral Ecology 20:1204-1210.

Coker, D. J., M. S. Pratchett, and P. L. Munday. 2012. Influence of coral bleaching, coral mortality and conspecific aggression on movement and distribution of coral-dwelling fish. Journal of Experimental Marine Biology and Ecology 414-415:62-68.

Cole, A. J., M. S. Pratchett, and G. P. Jones. 2008. Diversity and functional importance of coral-feeding fishes on tropical coral reefs. Fish and Fisheries 9:286-307.

Cooper, T. F., G. De'Ath, K. E. Fabricius, and J. M. Lough. 2008. Declining coral calcification in massive Porites in two nearshore regions of the northern Great Barrier Reef. Global Change Biology 14:529–538.

Crook, E. D., D. Potts, M. Rebolledo-Vieyra, L. Hernandez, and a. Paytan. 2011. Calcifying coral abundance near low-pH springs: implications for future ocean acidification. Coral Reefs 31:239-245.

Danilowicz, B. S. 1995a. Spatial patterns of spawning in the coral-reef damselfish Dascyllus albisella. Marine Biology 122:145-155.

Danilowicz, B. S. 1995b. The role of temperature in spawning of the damselfish Dascyllus albisella. Bulletin of Marine Science 57:624-636.

DeMartini, E. E., T. W. Anderson, J. C. Kenyon, J. P. Beets, and A. M. Friedlander. 2010. Management implications of juvenile reef fish habitat preferences and coral susceptibility to stressors. Marine and Freshwater Research 61:532.

Deloach, N. 1999. Reef Fish Behavior. New World Publications, Verona, Italy.

Denman, K. L., G. Brasseur, A. Chidthaisong, P. Ciais, P. M. Cox, R. E. Dickinson, D. Hauglustaine, C. Heinze, E. Holland, D. Jacob, U. Lohmann, S. Ramachandran, P. L. da Silva Dias, S. C. Wofsy, and X. Zhang. 2007. 2007: Couplings Between Changes in the Climate System and Biogeochemistry. Page *in* S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, and H. L. Miller, editors. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom, and New York, NY, USA.

Devine, B. M., P. L. Munday, and G. P. Jones. 2011. Rising CO2 concentrations affect settlement behaviour of larval damselfishes. Coral Reefs 31:229-238.

De'ath, G., J. M. Lough, and K. E. Fabricius. 2009. Declining coral calcification on the Great Barrier Reef. Science 323:116-119.

Dixson, D. L., P. L. Munday, M. Pratchett, and G. P. Jones. 2011. Ontogenetic changes in responses to settlement cues by Anemonefish. Coral Reefs 30:903-910.

Dixson, D. L., P. L. Munday, and G. P. Jones. 2010. Ocean acidification disrupts the innate ability of fish to detect predator olfactory cues. Ecology Letters 13:68-75.

Domenici, P., B. Allan, M. I. McCormick, and P. L. Munday. 2012. Elevated carbon dioxide affects behavioural lateralization in a coral reef fish. Biology Letters 8:78-81.

Donelson, J. M., P. L. Munday, M. I. McCormick, and C. R. Pitcher. 2011. Rapid transgenerational acclimation of a tropical reef fish to climate change. Nature Climate Change 2:30-32.

Donelson, J., P. Munday, M. McCormick, N. Pankhurst, and P. Pankhurst. 2010. Effects of elevated water temperature and food availability on the reproductive performance of a coral reef fish. Marine Ecology Progress Series 401:233-243.

Donner, S. D. 2009. Coping with commitment: projected thermal stress on coral reefs under different future scenarios. PLoS One 4:e5712.

Donner, S. D., T. R. Knutson, and M. Oppenheimer. 2007. Model-based assessment of the role of human-induced climate change in the 2005 Caribbean coral bleaching event. Proceedings of the National Academy of Sciences of the United States of America 104:5483-5488.

Donner, S. D., W. J. Skirving, C. M. Little, M. Oppenheimer, and O. Hoegh-Guldberg. 2005. Global assessment of coral bleaching and required rates of adaptation under climate change. Global Change Biology 11:2251-2265.

EPA. 2012. Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2010.

Eakin, C. M., J. M. Lough, and S. F. Heron. 2009. Climate Variability and Change: Monitoring Data and Evidence for Increased Coral Bleaching Stress. Pages 41-67 *in* Springer-Verlag, editor. Coral Bleaching. Berlin, Heidelberg.

Eakin, C. M., J. a Morgan, S. F. Heron, T. B. Smith, G. Liu, et al. 2010. Caribbean corals in crisis: record thermal stress, bleaching, and mortality in 2005. PloS One 5:e13969.

Elliott, J. K., and R. N. Mariscal. 2001. Coexistence of nine anemonefish species: differential host and habitat utilization, size and recruitment. Marine Biology 138:23-36.

Emslie, M. J., M. Logan, D. M. Ceccarelli, A. J. Cheal, A. S. Hoey, I. Miller, and H. P. A. Sweatman. 2012. Regional-scale variation in the distribution and abundance of farming damselfishes on Australia's Great Barrier Reef. Marine Biology 159:1293-1304.

Fabricius, K. E., C. Langdon, S. Uthicke, C. Humphrey, S. Noonan, G. De'ath, R. Okazaki, N. Muehllehner, M. S. Glas, and J. M. Lough. 2011. Losers and winners in coral reefs acclimatized to elevated carbon dioxide concentrations. Nature Climate Change 1:165-169.

Fabry, V. J., B. A. Seibel, R. A. Feely, and J. C. Orr. 2008. Impacts of ocean acidification on marine fauna and ecosystem processes. ICES Journal of Marine Sciences 65:414-432.

Feary, D. A., G. R. Almany, G. P. Jones, and M. I. Mccormick. 2007a. Coral degradation and the structure of tropical reef fish communities. Marine Ecology Progress Series 333:243-248.

Feary, D. A., G. R. Almany, M. I. McCormick, and G. P. Jones. 2007b. Habitat choice, recruitment and the response of coral reef fishes to coral degradation. Oecologia 153:727-37.

Feary, D. A., M. I. McCormick, and G. P. Jones. 2009. Growth of reef fishes in response to live coral cover. Journal of Experimental Marine Biology and Ecology 373:45-49.

Feely, R. A., C. L. Sabine, J. M. Hernandez-Ayon, D. Ianson, and B. Hales. 2008. Evidence for upwelling of corrosive "acidified" water onto the continental shelf. Science 320:1490-1492.

Feely, R. A., C. L. Sabine, K. Lee, W. Berelson, J. Kleypas, V. J. Fabry, and F. J. Millero. 2004. Impact of anthropogenic CO2 on the CaCO3 system in the oceans. Science 305:362-366.

Feely, R. A., C. L. Sabine, R. H. Byrne, F. J. Millero, A. G. Dickson, R. Wanninkhof, A. Murata, L. a. Miller, and D. Greeley. 2012. Decadal changes in the aragonite and calcite saturation state of the Pacific Ocean. Global Biogeochemical Cycles 26:1-15.

Ferrari, M. C. O., D. L. Dixson, P. L. Munday, M. I. McCORMICK, M. G. Meekan, A. Sih, and D. P. Chivers. 2011a. Intrageneric variation in antipredator responses of coral reef fishes affected by ocean acidification: implications for climate change projections on marine communities. Global Change Biology 17:2980-2986.

Ferrari, M. C. O., R. P. Manassa, D. L. Dixson, P. L. Munday, M. I. McCormick, M. G. Meekan, A. Sih, and D. P. Chivers. 2012a. Effects of ocean acidification on learning in coral reef fishes. PloS One 7:e31478.

Ferrari, M. C. O., M. I. McCormick, P. L. Munday, M. G. Meekan, D. L. Dixson, Ö. Lonnstedt, and D. P. Chivers. 2011b. Putting prey and predator into the CO2 equation--qualitative and quantitative effects of ocean acidification on predator-prey interactions. Ecology Letters 14:1143-8.

Ferrari, M. C. O., M. I. McCormick, P. L. Munday, M. G. Meekan, D. L. Dixson, O. Lönnstedt, and D. P. Chivers. 2012b. Effects of ocean acidification on visual risk assessment in coral reef fishes. Functional Ecology 26:553-558.

Fischlin, A., G. F. MIdgley, J. T. Price, R. Leemans, B. Gopal, C. Turley, M. D. A. Rounsevell, O. P. Dube, J. Tarazona, and A. A. Velichko. 2007. 2007:Ecosystems, their properties, goods, and services. Pages 211-272 *in* M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden, and C. E. Hanson, editors. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge.

Florida Museum of Natural History. 2011. Orange clownfish. , www.flmnh.ufl.edu/fish/gallery/descript/orangeclanemone/orangeclanemone.html.

Friedlingstein, P., R. A. Houghton, G. Marland, J. Hackler, T. A. Boden, T. J. Conway, J. G. Canadell, M. R. Raupach, P. Clais, and C. Le Quéré. 2010. Update on CO_2 emissions. Nature Geoscience 3:811-812.

Friedrich, T., A. Timmermann, A. Abe-Ouchi, N. R. Bates, M. O. Chikamoto, M. J. Church, J. E. Dore, D. K. Gledhill, M. González-Dávila, M. Heinemann, T. Ilyina, J. H. Jungclaus, E. McLeod, A. Mouchet, and J. M. Santana-Casiano. 2012. Detecting regional anthropogenic trends in ocean acidification against natural variability. Nature Climate Change 2:1-5.

Frisch, A. J., and J. Hobbs. 2009. Rapid assessment of anemone and anemonefish populations at the Keppel Islands: a report to the Great Barrier Reef Marine Park Authority. Research publication No. 94. Great Barrier Reef Marine Park Authority, Townsville.

Froukh, T., and M. Kochzius. 2008. Species boundaries and evolutionary lineages in the blue green damselfishes Chromis viridis and Chromis atripectoralis (Pomacentridae). Journal of Fish Biology 72:451-457.

Fussel, H.-M. 2009. An updated assessment of the risks from climate change based on research published since the IPCC Fourth Assessment Report. Climatic Change 97:469-482.

Gledhill, D. K., R. Wanninkhof, F. J. Millero, and M. Eakin. 2008. Ocean acidification of the greater Caribbean region 1996–2006. Journal of Geophysical Research 113:C10031.

Global Carbon Project. 2010. Carbon Budget 2009, report available at http://www.globalcarbonproject.org/index.htm.

Global Carbon Project. 2011. Carbon Budget 2010, report available at http://www.globalcarbonproject.org/index.htm.

Graham, N., T. R. McClanahan, M. A. MacNeil, S. K. Wilson, N. V. C. Polunin, S. Jennings, P. Chabanet, S. Clark, M. D. Spalding, Y. Letourneur, L. Bigot, R. Galzin, M. C. Ohman, K. C. Garpe, A. J. Edwards, and C. R. C. Sheppard. 2008. Climate warming, marine protected areas and the ocean-scale integrity of coral reef ecosystems. PloS One 3:e3039.

Graham, N., S. K. Wilson, S. Jennings, N. V. C. Polunin, J. Robinson, J. P. Bijoux, and T. M. Daw. 2007. Lag effects in the impacts of mass coral bleaching on coral reef fish, fisheries, and ecosystems. Conservation Biology 21:1291-300.

Guinotte, J. M., R. W. Buddemeier, and J. a. Kleypas. 2003. Future coral reef habitat marginality: temporal and spatial effects of climate change in the Pacific basin. Coral Reefs 22:551-558.

Guinotte, J. M., and V. J. Fabry. 2008. Ocean acidification and its potential effects on marine ecosystems. Annals of the New York Academy of Sciences 1134:320-342.

Gutierrez, L. 1998. Habitat selection by recruits establishes local patterns of adult distribution in two species of damselfishes: Stegastes dorsopunicans and S. planifrons. Oecologia 115:268-277.

Hansen, J., M. Sato, P. Kharecha, D. Beerling, V. Masson-Delmotte, M. Pagani, M. Raymo, D. L. Royer, and J. C. Zachos. 2008. Target atmospheric CO2: Where should humanity aim? Open Atmospheric Science Journal 2:217-231.

Hattori, A. 2002. Small and large anemonefishes can coexist using the same patchy resources on a coral reef, before habitat destruction. Journal of Animal Ecology 71:824-831.

Hill, R., and A. Scott. 2012. The influence of irradiance on the severity of thermal bleaching in sea anemones that host anemonefish. Coral Reefs 31:273-284.

Hoegh-Guldberg, O. 1999. Climate change, coral bleaching and the future of the world’s coral reefs. Marine and Freshwater Research 50:839-66.

Hoegh-Guldberg, O. 2005. Low coral cover in a high-CO2 world. Journal of Geophysical Research 110:1-11.

Hoegh-Guldberg, O., P. J. Mumby, A. J. Hooten, R. S. Steneck, P. Greenfield, E. Gomez, C. D. Harvell, P. F. Sale, A. J. Edwards, K. Caldeira, N. Knowlton, C. M. Eakin, R. Iglesias-Prieto, N. Muthiga, R. H. Bradbury, A. Dubi, and M. E. Hatziolos. 2007. Coral reefs under rapid climate change and ocean acidification. Science 318:1737-1742.

Hoegh-Guldberg, O., and J. F. Bruno. 2010. The impact of climate change on the world’s marine ecosystems. Science (New York, N.Y.) 328:1523-8.

Hoeke, R. K., P. L. Jokiel, R. W. Buddemeier, and R. E. Brainard. 2011. Projected changes to growth and mortality of Hawaiian corals over the next 100 years. PloS one 6:e18038.

Honisch, B., a. Ridgwell, D. N. Schmidt, E. Thomas, S. J. Gibbs, a. Sluijs, R. Zeebe, L. Kump, R. C. Martindale, S. E. Greene, W. Kiessling, J. Ries, J. C. Zachos, D. L. Royer, S. Barker, T. M. Marchitto, R. Moyer, C. Pelejero, P. Ziveri, G. L. Foster, and B. Williams. 2012. The Geological Record of Ocean Acidification. Science 335:1058-1063.

Höhne, N., B. Hare, M. Vieweg, M. Schaeffer, C. Chen, M. Rocha, and H. Fekete. 2012. Reality gap : Some countries progress in national polices , but many risk failing to meet pledges. Climate Action Tracker Update, 24 May 2012. Climate Analytics, Ecofys, and Potsdam Institute for Climate Impacts Research.

IPCC. 2007. Climate Change 2007: Synthesis Report. An Assessment of the Intergovernmental Panel on Climate Change. Available at www.ipcc.ch.

Ishii, M., N. Kosugi, D. Sasano, S. Saito, T. Midorikawa, and H. Y. Inoue. 2011. Ocean acidification off the south coast of Japan: A result from time series observations of CO_2 parameters from 1994 to 2008. Journal of Geophysical Research 116:1-9.

Ishimatsu, A., T. Kikkawa, M. Hayashi, K. S. Lee, and J. Kita. 2004. Effects of CO2 on marine fish: Larvae and adults. Journal of Oceanography 60:731-741.

Jackson, J., K. Cramer, M. Donovan, A. Friedlander, A. Hooten, and V. Lam. 2012. Tropical Americas Coral Reef Resilience Workshop; IUCN Report.

Johansen, J. L., and G. P. Jones. 2011. Increasing ocean temperature reduces the metabolic performance and swimming ability of coral reef damselfishes. Global Change Biology 17:2971-2979.

Jones, A., S. Gardner, and W. Sinclair. 2008. Losing "Nemo": bleaching and collection appear to reduce inshore populations of anemonefishes. Journal of Fish Biology 73:753-761.

Jones, G. P., L. Santana, L. J. McCook, and M. I. McCormick. 2006. Resource use and impact of three herbivorous damselfishes on coral reef communities. Marine Ecology Progress Series 328:215-224.

Kelly, B. P., J. L. Bengtson, P. L. Boveng, M. F. Cameron, S. P. Dahle, J. K. Jansen, E. A. Logerwell, J. E. Overland, C. L. Sabine, G. T. Waring, and J. M. Wilder. 2010. Status review of the ringed seal (Phoca hispida). U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-212, 250 p.

Kleypas, J. A., R. W. Buddemeier, D. Archer, J. P. Gattuso, C. Langdon, and B. N. Opdyke. 1999. Geochemical Consequences of Increased Atmospheric Carbon Dioxide on Coral Reefs. Science 284:118-120.

Kleypas, J. A., R. A. Feely, V. J. Fabry, C. Langdon, C. L. Sabine, and L. L. Robbins. 2006. Impacts of ocean acidification on coral reefs and other marine calcifiers: A guide to future research. Report of a workshop held 18–20 April 2005, St. Petersburg, FL, sponsored by NSF, NOAA, and the U.S. Geological Survey, 88 pp.

Kleypas, J., R. Buddemeier, and J.-P. Gattuso. 2001. The future of coral reefs in an age of global change. International Journal of Earth Sciences 90:426-437.

Kroeker, K. J., R. L. Kordas, R. N. Crim, and G. G. Singh. 2010. Meta-analysis reveals negative yet variable effects of ocean acidification on marine organisms. Ecology Letters.

Lecchini, D., J. Shima, B. Banaigs, and R. Galzin. 2005. Larval sensory abilities and mechanisms of habitat selection of a coral reef fish during settlement. Oecologia 143:326-34.

Levitus, S., J. I. Antonov, T. P. Boyer, O. K. Baranova, H. E. Garcia, R. a. Locarnini, a. V. Mishonov, J. R. Reagan, D. Seidov, E. S. Yarosh, and M. M. Zweng. 2012. World ocean heat content and thermosteric sea level change (0–2000 m), 1955–2010. Geophysical Research Letters 39:1-5.

Lewis, A. R. 1998. Effects of experimental coral disturbance on the population dynamics of fishes on large patch reefs. Journal of Experimental Marine Biology and Ecology 230:91-110.

Lo-Yat, A., S. D. Simpson, M. Meekan, D. Lecchini, E. Martinez, and R. Galzin. 2010. Extreme climatic events reduce ocean productivity and larval supply in a tropical reef ecosystem. Global Change Biology 17:1695-1702.

Loya, Y., K. Sakai, Y. Nakano, and R. V. Woesik. 2001. Coral bleaching: the winners and the losers. Ecology Letters 4:122-131.

Mann, D. A., and G. Sancho. 2007. Feeding Ecology of The Domino Damselfish, Dascyllus Albisella. Copeia 3:566-576.

Marshall, P. A., and A. H. Baird. 2000. Bleaching of corals on the Great Barrier Reef: differential susceptibilities among taxa. Coral Reefs 19:155-163.

McClanahan, T. R., A H. Baird, P. A Marshall, and M. A. Toscano. 2004. Comparing bleaching and mortality responses of hard corals between southern Kenya and the Great Barrier Reef, Australia. Marine Pollution Bulletin 48:327-35.

McClanahan, T. R., S. D. Donner, J. a. Maynard, M. A. MacNeil, N. a. J. Graham, J. Maina, A. C. Baker, J. B. Alemu I., M. Beger, S. J. Campbell, E. S. Darling, C. M. Eakin, S. F. Heron, S. D. Jupiter, C. J. Lundquist, E. McLeod, P. J. Mumby, M. J. Paddack, E. R. Selig, and R. van Woesik. 2012. Prioritizing Key Resilience Indicators to Support Coral Reef Management in a Changing Climate. PLoS ONE 7:e42884.

McCormick, M. I., J. a. Y. Moore, and P. L. Munday. 2010. Influence of habitat degradation on fish replenishment. Coral Reefs 29:537-546.

McMullen, C. P., and J. Jabbour. 2009. Climate Change Science Compendium 2009. United Nations Environment Porgramme, Nairobi, EarthPrint, available at http://www.unep.org/compendium2009/.

McWilliams, J. P., I. M. Cote, J. A. Gill, W. J. Sutherland, and A. R. Watkinson. 2005. Accelerating impacts of temperature-induced coral bleaching in the Caribbean. Ecology 86:2055-2060.

Meehl, G. A., T. F. Stocker, W. D. Collins, P. Friedlingstein, A. T. Gaye, J. M. Gregory, A. Kitoh, R. Knutti, J. M. Murphy, A. Noda, S. C. B. Raper, I. G. Watterson, A. J. Weaver, and Z.-C. Zhao. 2007. 2007: Global Climate Projections. Page *in* S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, and G. H. Miller, editors. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge University Press, Cambridge, UK, and New York, NY, USA. Available at www.ipcc.ch.

Meissner, K. J., T. Lippmann, and A. Sen Gupta. 2012. Large-scale stress factors affecting coral reefs: open ocean sea surface temperature and surface seawater aragonite saturation over the next 400 years. Coral Reefs.

Moy, A. D., W. R. Howard, S. G. Bray, and T. W. Trull. 2009. Reduced calcification in modern Southern Ocean planktonic foraminifera. Nature Geoscience.

Munday, P. L., D. L. Dixson, J. M. Donelson, G. P. Jones, M. S. Pratchett, G. V. Devitsina, and K. B. Doving. 2009. Ocean acidification impairs olfactory discrimination and homing ability of a marine fish. Proceedings of the National Academy of Sciences of the United States of America 106:1848-1852.

Munday, P. L., D. L. Dixson, M. I. Mccormick, M. Meekan, M. C. O. Ferrari, and D. P. Chivers. 2010. Replenishment of fish populations is threatened by ocean acidification. PNAS 107:12930-12934.

Munday, P. L., G. P. Jones, M. S. Pratchett, and A. J. Williams. 2008. Climate change and the future for coral reef fishes. Fish and Fisheries 9:261-285.

NMFS. 2012. Management Report for 82 Corals Status Review under the Endangered Species Act : Existing Regulatory Mechanisms and Conservation Efforts.

NRC. 2010. Advancing the Science of Climate Change, National Research Council, available at www.nap.edu.

Nañola, C. L., P. M. Aliño, and K. E. Carpenter. 2010. Exploitation-related reef fish species richness depletion in the epicenter of marine biodiversity. Environmental Biology of Fishes 90:405-420.

Nilsson, G. E., N. Crawley, I. G. Lunde, and P. L. Munday. 2009. Elevated temperature reduces the respiratory scope of coral reef fishes. Global Change Biology 15:1405-1412.

Nilsson, G. E., D. L. Dixson, P. Domenici, M. I. McCormick, C. Sørensen, S.-A. Watson, and P. L. Munday. 2012. Near-future carbon dioxide levels alter fish behaviour by interfering with neurotransmitter function. Nature Climate Change 2:201-204.

Nowicki, J. P., G. M. Miller, and P. L. Munday. 2012. Interactive effects of elevated temperature and CO_2 on foraging behavior of juvenile coral reef fish. Journal of Experimental Marine Biology and Ecology 412:46-51.

Olivier, J. G., G. Janssens-Maenhout, J. A. H. Peters, and J. Wilson. 2011. Long-term trend in global CO2 emissions. 2011 report, The Hague: PBL/JRC. Available at http://www.pbl.nl/en/publications/2011/long-term-trend-in-global-co2-emissions-2011-report.

Ollerton, J., D. McCollin, D. G. Fautin, and G. R. Allen. 2007. Finding NEMO: nestedness engendered by mutualistic organization in anemonefish and their hosts. Proceedings of the Royal Society B: Biological Sciences 274:591-598..

Orr, J. C., V. J. Fabry, O. Aumont, L. Bopp, S. C. Doney, R. A. Feely, A. Gnanadesikan, N. Gruber, A. Ishida, F. Joos, R. M. Key, K. Lindsay, E. Maier-Reimer, R. Matear, P. Monfray, A. Mouchet, R. G. Najjar, G. K. Plattner, K. B. Rodgers, C. L. Sabine, J. L. Sarmiento, R. Schlitzer, R. D. Slater, I. J. Totterdell, M. F. Weirig, Y. Yamanaka, and A. Yool. 2005. Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. Nature 437:681-686.

Osterhoudt, S. 2004. Buying Nemo. http://www.emagazine.com/magazine-archive/buying-nemo.

Pankhurst, N. W., and P. L. Munday. 2011. Effects of climate change on fish reproduction and early life history stages. Marine and Freshwater Research 62:1015.

Pini, J., S. Planes, E. Rochel, D. Lecchini, and C. Fauvelot. 2011. Genetic diversity loss associated to high mortality and environmental stress during the recruitment stage of a coral reef fish. Coral Reefs 30:399-404.

Pitkin, L. 2001. Coral Fish. Smithsonian Institution Press, Washington, D.C.

Pratchett, M. S., L. K. Bay, P. C. Gehrke, J. D. Koehn, K. Osborne, R. L. Pressey, H. P. a. Sweatman, and D. Wachenfeld. 2011. Contribution of climate change to degradation and loss of critical fish habitats in Australian marine and freshwater environments. Marine and Freshwater Research 62:1062.

Pratchett, M., P. Munday, K. Wilson, G. P. Jones, and T. R. Mcclanahan. 2008. Effects of climate-induced coral bleaching on coral-reef fishes--ecological and economic consequences. Oceanography and Marine Biology - An Annual Review 46:251-296.

Pressley, P. H. 1980. Lunar periodicity in the spawning of yellowtail damselfish, Microspathodon chrysurus. Environmental Biology of Fishes 5:153-159.

Prosek, J. 2010. Beautiful friendship. National Geographic Magazine, http://ngm.nationalgeographic.com/print/2010/01/clownfish/prosek-text.

Randall, J. E. 1985. Guide to Hawaiian Reef Fishes. Harrowood Books, Newton Square, PA.

Randall, J. E. 2005. Reef and Shore Fishes of the South Pacific. University of Hawaii Press, Honolulu, HI.

Raupach, M. R., G. Marland, P. Ciais, C. Le Quéré, J. G. Canadell, G. Klepper, and C. B. Field. 2007. Global and regional drivers of accelerating CO2 emissions. Proceedings of the National Academy of Sciences of the United States of America 104:10288-10293.

Rhyne, A. L., M. F. Tlusty, P. J. Schofield, L. Kaufman, J. a Morris, and A. W. Bruckner. 2012. Revealing the appetite of the marine aquarium fish trade: the volume and biodiversity of fish imported into the United States. PloS One 7:e35808.

Richardson, K., W. Steffen, H. J. Schellnhuber, J. Alcamo, T. Barker, R. Leemans, D. Liverman, M. Munasinghe, B. Osman-Elasha, N. Stern, and O. Waever. 2009. Synthesis Report from Climate Change: Global Risks, Challenges and Decisions, Copenhagen 2009, 10-12 March, available at www.climatecongresss.ku.dk.

Rogers, A. D., and D. Laffoley. 2011. International Earth system expert workshop on ocean stresses and impacts. Summary Report. IPSO Oxford, 18 pp.

Rosenberg, E., and Y. Ben-Haim. 2002. Microbial diseases of corals and global warming. Environmental Microbiology 4:318-326.

Royal Society. 2005. Ocean acidification due to increasing atmospheric carbon dioxide. Policy document 12/05. Available at www.royalsoc.ac.uk. The Royal Society, London, UK.

Saenz-Agudelo, P., G. P. Jones, S. R. Thorrold, and S. Planes. 2011. Detrimental effects of host anemone bleaching on anemonefish populations. Coral Reefs 30:497-506.

Sheppard, C. R. C. 2003. Predicted recurrences of mass coral mortality in the Indian Ocean. Nature 425:294-7.

Shuman, C. S., G. Hodgson, and R. F. Ambrose. 2005. Population impacts of collecting sea anemones and anemonefish for the marine aquarium trade in the Philippines. Coral Reefs 24:564-573.

Sikkel, P. C., and D. L. Kramer. 2006. Territory revisits reduce intrusion during spawning trips by female yellowtail damselfish, Microspathodon chrysurus. Animal Behaviour 71:71-78.

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

Simpson, S. D., P. L. Munday, M. L. Wittenrich, R. Manassa, D. L. Dixson, M. Gagliano, and H. Y. Yan. 2011. Ocean acidification erodes crucial auditory behaviour in a marine fish. Biology letters 7:917-20.

Smith, J. B., S. H. Schneider, M. Oppenheimer, G. W. Yohe, W. Hare, M. D. Mastrandrea, A. Patwardhan, I. Burton, J. Corfee-Morlot, C. H. D. Magadza, H.-M. Fussel, A. B. Pittock, A. Rahman, A. Suarez, and J.-P. van Ypersele. 2009. Assessing dangerous climate change through an update of the Intergovernmental Panel on Climate Change (IPCC) "reasons for concern". Proceedings of the National Academy of Sciences of the United States of America 106:4133-4137.

Stevenson, R. A. 1963a. Life History and Behavior of Dascyllus albisella Gill, a Pomacentrid Reef Fish; PhD thesis. University of Hawaii.

Stevenson, R. A. 1963b. Behavior of the Pomacentrid Reef Fish Dascyllus albisella Gill in Relation to the Anemone Marcanthia cookei. Copeia 1963:612-614.

Stevenson, T. C., B. N. Tissot, and J. Dierking. 2011. Fisher behaviour influences catch productivity and selectivity in West Hawaii's aquarium fishery. ICES Journal of Marine Science 68:813-822.

Sweatman, H. P. A. 1983. Influence of conspecifics on choice of settlement sites by larvae of two pomacentrid fishes (Dascyilus aruanus and D. reticulatus) on coral reefs. Marine Biology 75:225-229.

Taylor, M. F. J., K. F. Suckling, and J. J. Rachlinski. 2005. The effectiveness of the Endangered Species Act: a quantitative analysis. Bioscience 55:360-367.

Thresher, R. E., and E. B. Brothers. 1989. Evidence of intra- and inter-oceanic regional differences in the early life history of reef- associated fishes. Marine Ecology Progress Series 57:187-205.

Timm, J., M. Figiel, and M. Kochzius. 2008. Contrasting patterns in species boundaries and evolution of anemonefishes (Amphiprioninae, Pomacentridae) in the centre of marine biodiversity. Molecular Phylogenetics and Evolution 49:268-76.

Tissot, B. N., B. a. Best, E. H. Borneman, A. W. Bruckner, C. H. Cooper, H. D'Agnes, T. P. Fitzgerald, A. Leland, S. Lieberman, A. Mathews Amos, R. Sumaila, T. M. Telecky, F. McGilvray, B. J. Plankis, A. L. Rhyne, G. G. Roberts, B. Starkhouse, and T. C. Stevenson. 2010. How U.S. ocean policy and market power can reform the coral reef wildlife trade. Marine Policy 34:1385-1388.

Tolimieri, N. 1998. Contrasting effects of microhabitat use on large-scale adult abundance in two families of Caribbean reef fishes. Marine Ecology Progress Series 167:227-239.

Tripati, A. K., C. D. Roberts, and R. A. Eagle. 2009. Coupling of CO2 and ice sheet stability over major climate transitions of the last 20 million years. Science 326:1394-1397.

USGCRP. 2009. Global Climate Change Impacts in the United States. U.S. Global Change Research Program. Thomas R. Karl, Jerry M. Melillo, and Thomas C. Peterson, (eds.). Cambridge University Press, 2009.

Veron, J., O. Hoegh-Guldberg, T. Lenton, J. Lough, D. Obura, P. Pearce-Kelly, and C. Sheppard. 2009. The coral reef crisis: The critical importance of 350ppm CO2. Marine Pollution Bulletin 58:1428–1436.

Vuuren, D. P., J. Edmonds, M. Kainuma, K. Riahi, A. Thomson, K. Hibbard, G. C. Hurtt, T. Kram, V. Krey, J.-F. Lamarque, T. Masui, M. Meinshausen, N. Nakicenovic, S. J. Smith, and S. K. Rose. 2011. The representative concentration pathways: an overview. Climatic Change 109:5-31.

Walsh, S. M., S. L. Hamilton, B. I. Ruttenberg, M. K. Donovan, and S. a Sandin. 2012. Fishing top predators indirectly affects condition and reproduction in a reef-fish community. Journal of Fish Biology 80:519-37.

Wilkes, A. A., M. M. Cook, L. Anthony, J. Eme, J. M. Grim, B. C. Hohmann, L. Conner, C. J. Mcgill, C. M. Pomory, and A. Wayne. 2008. A Comparison of Damselfish Densities on Live Staghorn Coral (Acropora cervicornis) and Coral Rubble in Dry Tortugas National Park. Southeastern Naturalist 7:483-492.

Wilkinson, C., and D. Souter. 2008. Status of Caribbean Coral Reefs After Bleaching and Hurricanes in 2005. Global Coral Reef Monitoring Network, and Reef and Rainforest Research Centre, Townsville, 152 p.

Wilson, S. K., S. C. Burgess, A. J. Cheal, M. Emslie, R. Fisher, I. Miller, N. V. C. Polunin, P. Hugh, and A. Sweatman. 2008a. Habitat utilization by coral reef fish: implications for specialists vs. generalists in a changing environment. Journal of Animal Ecology 77:220-228.

Wilson, S. K., R. Fisher, M. S. Pratchett, N. a. J. Graham, N. K. Dulvy, R. a. Turner, a. Cakacaka, N. V. C. Polunin, and S. P. Rushton. 2008b. Exploitation and habitat degradation as agents of change within coral reef fish communities. Global Change Biology 14:2796-2809.

Wilson, S. K., N. a. J. Graham, M. S. Pratchett, G. P. Jones, and N. V. C. Polunin. 2006. Multiple disturbances and the global degradation of coral reefs: are reef fishes at risk or resilient? Global Change Biology 12:2220-2234.

Zajicek, P., S. Hardin, and C. Watson. 2009. A Florida Marine Ornamental Pathway Risk Analysis. Reviews in Fisheries Science 17:156-169.

Zeebe, R. E. 2012. History of Seawater Carbonate Chemistry, Atmospheric CO_2, and Ocean Acidification. Annual Review of Earth and Planetary Sciences:141-165.