Ecological Impacts and Practices of the Coral Reef Wildlife Trade

istockphoto.com

By Daniel J. Thornhill
Defenders of Wildlife
Updated December 12, 2012

Table of Contents

Acknowledgments: This report would not have been possible without the help and support of colleagues at Defenders of Wildlife, Environmental Defense Fund, the Humane Society International and the Humane Society of the United States, and the World Wildlife Fund. Thank you to Cara Cooper, J. Chris Haney, Ted Morton, America M. Pintabutr, and Teresa Telecky for their helpful contributions to this report.

If you have questions, noticed errors, or wish to suggest revisions to the document, please contact Dr. Daniel Thornhill, by email at dthornhill@defenders.org.

Executive Summary

It is estimated that 14–30 million fish, 1.5 million live stony corals, 4 million pounds of coral skeleton, 65–110 thousand pounds of red and black coral, and 9–10 million other invertebrates are removed each year from ecosystems across the world to supply the aquarium, curio/home décor, and coral jewelry industries. Together, these three industries are known as the global trade in ornamental coral reef wildlife. This trade has a collective annual value estimated in the hundreds of millions of dollars and is an extensive industry that involves over 45 source countries. Although collection of coral reef wildlife primarily occurs in Southeast Asia and the Caribbean Sea, the majority (>60%) of collected animals are exported to the United States. Available evidence also suggests that trade has grown over the past several decades, with the possible exception of the recent global recession. For example, the importation of live corals to the U.S. increased by 600% from 1988 to 2007. However, precise quantification of the size and value of the ornamental trade in coral reef wildlife is stymied by a lack of monitoring and regulation, underreporting of landings, illegal harvesting practices, including poaching and cyanide fishing, and a sizeable black market for reef-dwelling organisms.

The purpose of this review is to examine the ecological impacts and practices associated with the ornamental trade in coral reef wildlife. Out of the thousands of ornamental species collected across the globe, trade impacts have only been assessed for a handful of species and locations. Notwithstanding this, collection for trade has had negative population and ecosystem impacts and, in a number of cases, these impacts have been scientifically documented. The first section of this report examines seven "case studies" in considerable detail. The focus of each case study is on the supply chain practices and ecological impacts associated with trade in that species or group of species. The case studies also provide background detail on the biology of that species or group as well as a brief review of the efficacy of select conservation and management practices.

The first case study (Chapter 2) examines a popular aquarium fish, yellow tang, on the Kona coastline of Hawaii. Yellow tang populations declined as a result of collection to supply the aquarium trade and these declines raise concerns about the sustainability of collection. However, the yellow tang example also highlights the role management can play in protecting coral reef species. The state of Hawaii established a series of fisheries reserve areas along the Kona coast to protect ornamental fish from collection. Yellow tang and other ornamental aquarium fish are now recovering in these reserves and young yellow tang recruits are dispersing into areas open to collection. With additional conservation measures, including harvesting quotas, the yellow tang fishery could be further improved.

Chapter 3 discusses the Banggai cardinalfish, a species that was 'rediscovered' in the mid-1990s and rapidly became popular in the aquarium trade. This species is highly susceptible to over-exploitation due to its limited range, specific habitat requirements, low reproductive output, and extreme inability to disperse. Every year, approximately 1 million Banggai cardinalfish are collected for trade out of a total population of 2.4 million individuals. This level of exploitation resulted in population declines exceeding 90% in certain areas and the extinction of some local populations. Attempts to achieve protections from international commercial trade through the Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES) have been unsuccessful.

However, captive breeding and regional conservation efforts offer some hope for the future.

The brilliantly-colored mandarinfish is the topic of Chapter 4. Males of this species are highly prized in the aquarium trade for their elaborate fins. Mandarinfish are quite reclusive, which has led collectors to develop a spear-fishing method for their capture. Spearing these tiny fish can result in injury, paralysis, or even death. Hobbyists' preference for large male fish also raises concerns about disruption of the mandarinfish mating system. Female mandarinfish may refuse to mate with smaller males; when large males are removed by collectors, the reproduction of this species is impaired. Finally, this species has a specialized diet of live zooplankton and meiofauna. This diet is difficult to replicate in captivity and, as a result, wild caught mandarinfish often refuse to eat. Because of this, wild caught mandarinfish often starve to death within a few weeks of purchase. Death in captivity increases demand for new fish to replace the ones that were lost, thereby driving additional collection and damage to wild mandarinfish populations. However, captive bred mandarinfish have recently become available and these captive bred fish can be conditioned to consume a prepared diet. Such efforts may reduce the demand for wild caught mandarinfish, thereby reducing impacts on wild populations.

Giant anemones and their symbiotic anemonefish are the topic of Chapter 5. These animals form a tight symbiosis; anemones grow faster and live longer when harboring symbiotic fish whereas anemonefish cannot survive without the protection of the anemone's stinging tentacles. Their attractive appearance and interesting biology makes anemones and anemonefish popular reef aquarium species. Both anemones and anemonefish can be bred in captivity; however, most of the animals in trade are still collected from the wild. Collection has caused significant declines in anemone and anemonefish populations in the Philippines, Australia, Singapore, and elsewhere.

Seahorses, the subject of Chapter 6, are collected in the bycatch of shrimp or demersal fish trawls as well as directly targeted in artisanal fisheries throughout the world. These unusual and enigmatic animals are used in traditional medicine, dried and sold as curios, or used as aquarium pets. Catch rates of seahorses, known as catch per unit effort, are in decline throughout most of Southeast Asia and in the Caribbean Sea. Collectors also report that it is increasingly difficult to find and harvest seahorses. Declining catch suggests that seahorse populations have been over-exploited and are in need of additional protections.

Chapter 7 addresses the impacts of trade on giant clams. Giant clams, or tridacnids, are the largest bivalve mollusks in the world. Their huge size, colorful appearance, fluted shells, and flavorful meat has led to the overfishing of many giant clam populations. Tridacnids are popular as food and as aquarium pets. Their shells are also used in home décor. Populations of giant clams have been depleted throughout much of the world; one survey found that tridacnids were absent from over 90% of the reefs where they should naturally occur.

Scleractinian or stony corals (Chapter 8) form the structural and trophic framework of coral reef ecosystems. These animals build an elaborate calcareous skeleton; together corals and other calcifying organisms accrete the reef structure over time. Stony corals are collected for use in home aquariums and their attractive skeletons are also popular in home décor. Several scientific studies have documented overfishing of corals in the Philippines, Indonesia, and other countries. Because collection of corals is collection of the reef itself, over-harvesting has the potential for far-reaching consequences for coral reef wildlife.

In addition to directly causing declines in species and biodiversity in marine ecosystems, the coral reef wildlife trade has had several broader ecological impacts. Part II of the report examines the consequence of these larger scale impacts on coral reef ecosystems and wildlife. Chapter 9 delves into the injury and death of coral reef wildlife in the supply chain. Supply chain mortality rates range from less than 5% to greater than 90% of the animals collected from the wild. Rough handling of wildlife, low quality holding facilities, long transit times, and other careless practices cause unnecessary injury and death to wildlife in trade. Losses due to injury and death result in more collection of coral reef organisms from the wild, thereby exacerbating the negative ecological consequences of trade.

One of the leading causes of supply chain mortality is the use of destructive fishing methods, including fishing with cyanide and other poisons (Chapter 10). Cyanide is dispensed onto the reef by divers with squirt bottles. The poison rapidly stuns fish, rendering the animals easier to capture. In addition to being an effective anesthetic, cyanide is also a potent poison. As a result, cyanide fishing poisons and often kills ornamental fish and non-target organisms like invertebrates, non-ornamental reef fish, and habitat-forming corals. Cyanide fishing is currently one of the leading threats to coral reefs in Southeast Asia and other locations that supply the coral reef wildlife trade.

One indirect consequence of the coral reef wildlife trade is the introduction of invasive species to coral reefs (Chapter 11). The best documented example is the introduction of Pacific lionfish to the waters off of south Florida, putatively via the ornamental trade. Since their introduction in the early 1990s, lionfish numbers have grown exponentially. Lionfish invaders have spread as far as Long Island to the north and throughout the Caribbean Sea to the south. Lionfish are voracious predators that consume several fish per hour. Their effects on native reef fish populations have been significant. One study in the Bahamas demonstrated that a single lionfish reduced the recruitment of native fish species by an average of 79% on experimental patch reefs relative to lionfish-free control reefs.

Over-collection of coral reef wildlife can potentially cause far-reaching consequences for coral reef ecosystems (Chapter 12). Collection disrupts trophic webs and removes important functional groups from the reef framework of corals and live rock to sharks and other top predators. Key functional groups that are taken by trade include parasite cleaners, corallivores, and algae gazers. Herbivores keep macroalgae in check and thereby protect corals from competition and algal overgrowth. The overfishing of herbivores for food fisheries has been shown to contribute to ecosystem decline and similar problems might occur through the ornamental trade. The combined effects of collection could reduce the resistance and resilience of coral reefs to larger threats, like climate change and ocean acidification, that imperil these ecosystems globally.

Chapter 1
Introduction to Coral Reefs and the Coral Reef Wildlife Trade

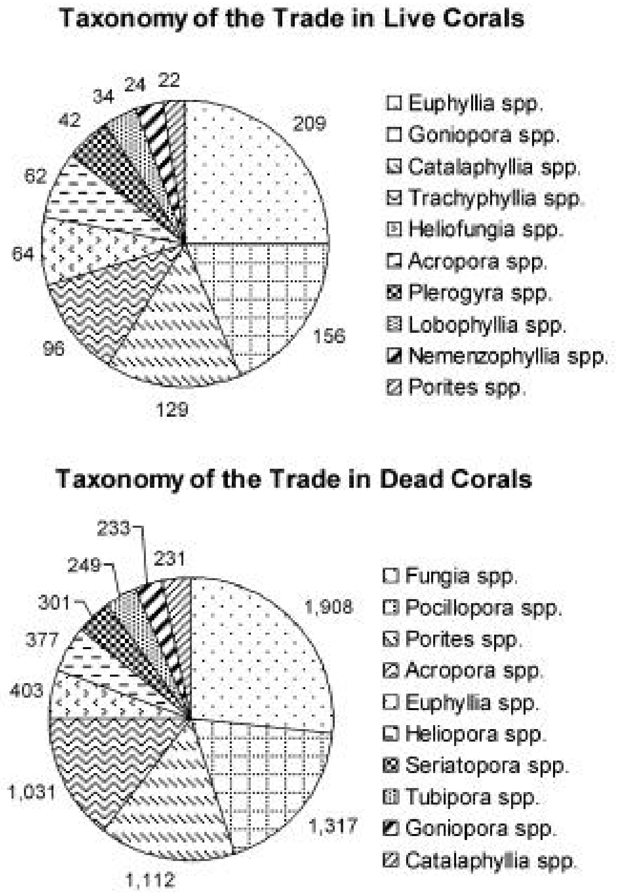
Coral reefs are highly productive and diverse marine ecosystems found throughout the world's tropical and sub-tropical oceans. This ecosystem is based around reef-building corals, a symbiotic association between cnidarian animals (corals), endosymbiotic dinoflagellate algae (*Symbiodinium* spp.), and various other microbial taxa (bacteria, archaea, endolithic algae, apicomplexans, fungi, etc.). Reef-building corals and other marine calcifiers build calcium carbonate skeletons for structural support and protection. Together, the collective skeletal deposition of reef-building organisms forms a raised structure known as a coral reef. These structures can reach enormous sizes; in the case of the Great Barrier Reef, the reef structure is visible from outer space. Coral reefs provide the home as well as feeding and nursery grounds to a tremendous diversity of fish, reptiles, invertebrate animals, and microbes. Although they represent less than 1% of the benthic habitat in the oceans, they provide habitat to over 25% of marine species.

Over 100 countries are home to coral reefs, many of which are developing nations with limited resources (Moore and Best 2001). About 275 million people currently live within 30 km of coral reefs (Burke et al. 2011), where they receive various indirect and direct benefits from these ecosystems. Reefs provide an estimated $375 billion U.S. in economic benefits and ecosystems services each year (Moore and Best 2001). Coral reefs provide the nursery grounds and homes to many fish species that are used in commercial, subsistence, and recreational fisheries. Reefs supply food to over 1 billion people in Asia (Smith et al. 2008). In developing countries, one quarter of the total fish catch is harvested from coral reefs (Smith et al. 2008). These ecosystems are also valuable drivers of recreation and tourism, and therefore economic development, in over 100 countries (Smith et al. 2008, United Nations World Tourism Organization 2010). Reefs are natural, self-building, and self-repairing buffers that protect 150,000 km of shorelines from waves and storms (Smith et al. 2008, Burke et al. 2011). They also provide considerable educational and scientific value, a source of new pharmaceuticals, and support the livelihoods of millions of people (U.S. Commission on Ocean Policy 2004, Glaser and Mayer 2009).

Despite the importance of coral reefs, these ecosystems are imperiled throughout the world. A recent report found that 19% of coral reefs are already lost, 15% are in jeopardy of loss within 10–20 years, and 20% are in danger of loss within 20–40 years (Wilkinson 2008). Similarly, Burke et al. (2011) estimated that 75% of remaining coral reefs are currently threatened. Even some of the most remote and pristine reefs have experienced species loss from over fishing and climate change (Hodgson 1999). The complexity of reef ecosystems and the high gross, but low net, productivity renders these environments especially vulnerable to over-exploitation (Birkeland 2001, Lieberman and Field 2001). Increased sea-surface temperatures associated with climate change are a primary global threat to reef ecosystems (Hoegh-Guldberg 1999). At local and regional scales, coral reefs have been degraded by many different stressors, including nutrient inputs, overfishing, destructive fishing practices, hurricanes and storms, outbreaks of predatory starfish, exotic species introductions, sedimentation from poor land use practices, diseases, and pollution (Gardner et al. 2003, Lesser 2004). Recently, ocean acidification has emerged as another potentially serious threat to the long-term sustainability of reefs (Hoegh-Guldberg et al. 2007).

Another potentially serious – but understudied – threat to coral reef ecosystems is the collection and trade in ornamental coral reef wildlife. This trade includes collection of coral reef organisms for the aquarium, jewelry, curio, and home décor industries. Examples of this trade include the removal of live corals, reef fish, and invertebrates for the aquarium trade, harvesting of precious corals for use in jewelry and sculptures, and use of coral skeletons, giant clam shells, and dried seahorses as decorative or curiosity items. This trade removes coral reef organisms at nearly every trophic level and, as a result, it is in many respects the trade in an entire ecosystem (McManus 2001).

Figure 1: The taxonomy of corals involved in the live and dead ornamental coral reef wildlife trade. Figure taken from Green and Shirley (1999). Numbers represent the tons of corals in trade from 1985 through 1997.



A tremendous diversity and volume of wildlife are involved in the ornamental coral reef wildlife trade (Rhyne et al. 2012). Every year, approximately 14–30 million fish, 1.5 million live stony corals, 4 million pounds of coral skeleton, 65–110 thousand pounds of red and black coral, and 9–10 million other invertebrates are removed from coral reef ecosystems across the world (Wood 2001a,b, Wabnitz et al. 2003, Bruckner 2005, Tsounis et al. 2010, Murray et al. 2012, but see Rhyne et al. 2012 who assert that the volume of marine fish has been overestimated). There is a tremendous diversity of species in this wildlife trade, including at least 1,802 species of fish, more than 140 species of corals, and more than 500 species of non-coral invertebrates (Wabnitz et al. 2003, Rhyne et al. 2012). Coral reef wildlife is removed from nature to serve as pets, jewelry, curiosities, and decorative items and comprises a substantial portion of the overall widlife trade. A total of 90.3% of wildlife specimens in trade are fish (both marine and freshwater; Smith et al. 2009), many of which are taken from coral reefs. Furthermore, 33.5% of wildlife shipments are cnidarians (e.g., corals and anemones) and

25.9% of shipments are fish (again including marine and freshwater species; Smith et al. 2009).

The species in trade are typically selected based on their attractiveness or unusual appearance. For example, fish targeted in the aquarium trade are often juveniles and males, which are preferred for their small size and bright colors (Wabnitz et al. 2003). Juvenile fish are advantageous in that they are less expensive to transport and are appropriately sized for a home aquarium (Wood 2001b). However, juvenile fish are more easily stressed and susceptible to death in captivity which may exacerbate the impacts of trade (Wood 2001a,b). For corals, species with attractive growth forms and large polyps are considered especially desirable (Moore and Best 2001). Rare species are especially preferred and these tend to fetch high prices among collectors (Moore and Best 2001, Rhyne et al. 2012).

According to an analysis of one year of U.S. import records, damselfish (Pomacentridae) constitute over 50% of the volume of fish in trade (Rhyne et al. 2012). This is followed by wrasses (Labridae), angelfish (Pomacanthidae), gobies (Gobiidae), surgeonfishes and tangs (Acanthuridae), cardinalfishes (Apogonidae), wormfishes (Microdesmidae), butterflyfish (Chaetodontidae), dragonets (Callionymidae), and sea basses and groupers (Serranidae) as the top 10 familes of marine aquarium fish imported into the U.S. (Rhyne et al. 2012).

Table 1 lists the top 20 marine aquarium fish species imported into the United States according to the analysis of 2004–2005 import data by Rhyne et al. (2012). Demand for coral reef wildlife is known to change through time. As a result of this shifting demand and different sources that can be used to quantify species volumes in trade, there are various lists of the highest volume species in trade. Table 2 provides two alternative lists based on importer and exporter data from the Global Marine Aquarium Database (Wabnitz et al. 2003).

Table 1: The top 20 coral reef fish species imported into the United States. The species are listed in rank order according to import volume from highest to lowest. Table based on Rhyne et al. (2012).

Scientific Name	**Common Name**
Chromis viridis	Blue/Green Chromis
Chrysiptera cyanea	Blue Damsel
Dascyllus trimaculatus	Three-spot Dascyllus
Dascyllus aruanus	Whitetail Dascyllus
Amphiprion ocellaris/percula	False Percula Clownfish/Orange Clownfish
Chrysiptera parasema	Yellowtail Damsel
Dascyllus melanurus	Four stripe Damselfish
Chrysiptera hemicyanea	Azure Damselfish
Nemateleotris magnifica	Firefish
Pteropogon kauderni	Banggai Cardinalfish
Synchiropus splendidus	Mandarinfish
Paracanthurus hepatus	Hippo Tang or Blue Tang
Labroides dimidiatus	Bluestreak Cleaner Wrasse
Centropyge loricula	Flame Angelfish
Premnas biaculeatus	Maroon Clownfish

Centropyge bispinosus	Coral Beauty Angelfish
Pseudocheilinus hexataenia	Sixline Cleaner Wrasse
Amphiprion frenatus	Tomato Clownfish
Gramma loreto	Royal Gramma
Sphaeramia nematoptera	Pajama Cardinalfish

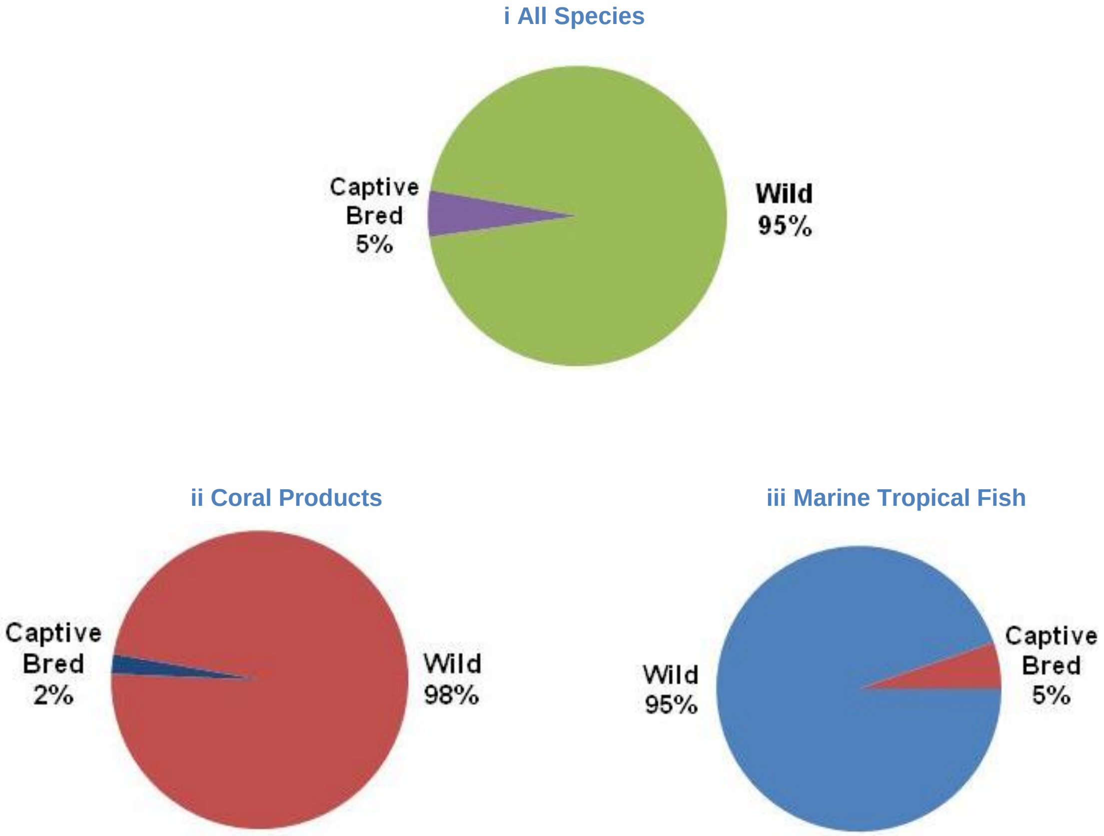
Table 2: Top 10 most traded species of coral reef aquarium fish according to the Global Marine Aquarium Database records from 1997 to 2002. Table adapted from Wabnitz et al. (2003).

Exporter Data		**Importer Data**	
Scientific Name	**Common Name**	**Scientific Name**	**Common Name**
Amphiprion ocellaris	False Percula Clownfish	*Chromis viridis*	Blue/Green Chromis
Chrysiptera cyanea	Blue damsel	*Zebrasoma flavescens*	Yellow Tang
Dascyllus aruanus	Whitetail Dascyllus	*Amphiprion ocellaris*	False Percula Clownfish
Amphiprion percula	Orange Clownfish	*Dascyllus aruanus*	Whitetail Dascyllus
Chromis viridis	Blue/Green Chromis	*Pomacentrus australis*	Australian damsel
Abudefduf spp.	Sergeant Majors	*Chrysiptera parasema*	Yellowtail damsel
Dascyllus trimaculatus	Three-spot Dascyllus	*Chrysiptera cyanea*	Blue Damsel
Paracanthurus hepatus	Hippo Tang	*Dascyllus* spp.	Dascyllus
Dascyllus albisella	White-spotted Damsel	*Dascyllus trimaculatus*	Three-spot Dascyllus
Chrysiptera hemicyanea	Azure Damselfish	*Labroides dimidiatus*	Bluestreak Cleaner Wrasse

Figure 1 examines the primary coral genera taken for the curio (dead corals) and aquarium trade (live corals). Very few species of coral reef wildlife are bred and raised in captivity; instead, the vast majority (approximately 95%) are taken from the wild (Wood 2001b, Wabnitz et al. 2003, Bruckner 2005, Craig et al. in press, Figure 2). However, CITES import records indicate a recent increase in the amount of aquacultured corals in trade (Wood et al. 2012).

Coral reef species are collected for the ornamental wildlife trade in at least 45 different countries around the world (Wood 2001a,b, Smith et al. 2008, Rhyne et al. 2012). Indonesia and the Philippines are the two largest exporters of coral reef wildlife (Wood 2001b, Wabnitz et al. 2003, Rhyne et al. 2012). More than 60% of globally traded wildlife is imported into the U.S. (Wood 2001b, Wabnitz et al. 2003, Smith 2008, Craig et al. in press). The nations of the European Union and Japan are also major importers (Wood 2001b, Wabnitz et al. 2003, Smith 2008, Craig et al. in press). Trade has grown considerably since the 1980s (Moore and Best 2001). For example, from 1988 to 2007 the importation of live corals to the U.S. increased by 600% and global imports of live corals grew by 1500% (Tissot et al. 2010). Figure 3 demonstrates the growth in imports of coral products into the U.S. from 1999 to 2009 (see also Craig et al. in press). However, recent evidence suggests a decline in trade volumes for certain taxa or commody groups, including Florida invertebrates and sand dollars corresponding to the global recession and live rock and sand corresponding to the development of small home reef aquaria ("nano-reef aquaria"; Rhyne and Tlusty 2012).

Figure 2: The percentage of ornamental coral reef wildlife species captured from the wild vs. bred and raised in captivity based on U.S. import data. Data and figure taken from Craig et al. (in press).

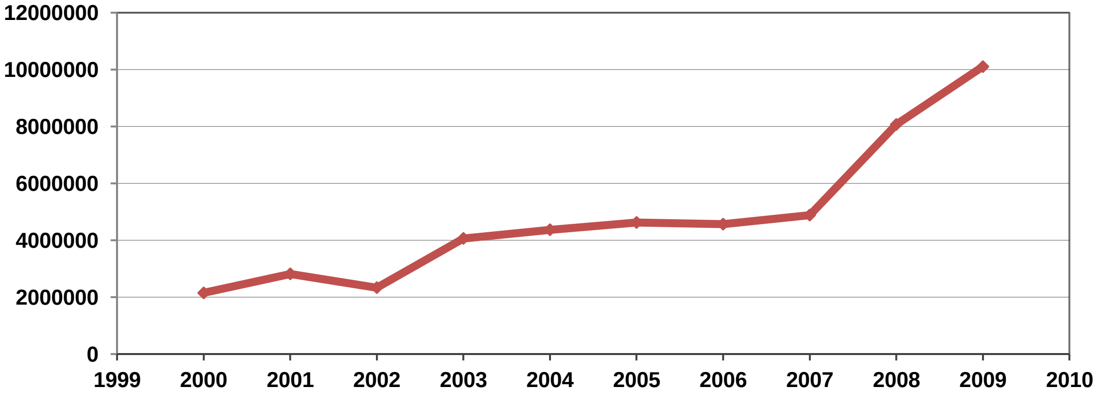


Source: TRAFFIC analysis of USFWS LEMIS data

The total value of the trade is unknown. The coral reef aquarium trade is estimated to be globally worth \$200-330 million U.S. annually (Wabnitz et al. 2003, but see Smith et al. 2008 for a larger estimated value). Coral jewelry is valued at well over \$300 million U.S. annually (Tsounis et al. 2010). There are no estimates available for the value of curio/home décor species in this trade. A precise quantification of the size and value of the ornamental trade in coral reef wildlife is stymied by a lack of monitoring and regulation, underreporting of landings, illegal harvesting practices including poaching and cyanide fishing, and a sizeable black market for reef-dwelling organisms. Therefore, the numbers presented here must be considered very rough approximations and may be underestimates of the actual extent of the coral reef curio, aquarium, and jewelry industries.

Previous reports and studies have examined the structure and organization of trade as well as the volume and identity of species involved (e.g., Wood 2001b, Wabnitz et al. 2003, Rhyne et al. 2012, Craig et al. in press). Despite this, the negative consequences of trade on organisms, populations, and coral reef ecosystems remain poorly understood. In order to address this lack of knowledge, this report reviews and synthesizes the available scientific evidence on the ecological and humane consequences of the coral reef wildlife trade. Because trade spans the globe and involves over 2,000 coral reef species, it is currently impossible to exhaustively document trades' negative consequences on coral reef wildlife and ecosystems. Unfortunately, there is a severe lack of data documenting the impacts of this global industry for the majority of traded coral reef species. Information, when it is available, is often haphazardly collected, out of date, or confounded by other problems. Nevertheless, several excellent, peer-reviewed scientific studies and reports have been conducted on a subset of species or locations and this work provides a window into the negative effects of the aquarium, home décor, and jewelry trades.

Figure 3: Growth in the number of corals and coral products imported into the U.S. from 2000 to 2009. Data and figure taken from Craig et al. (in press).



The first section of this report provides seven detailed case studies on the ecological consequences of the coral reef wildlife trade (Chapters 2-9), beginning with several groups of colorful reef fish and concluding with an examination of the corals and other invertebrates that are responsible for building the reef itself. The organisms covered include yellow tang (Chapter 2), Banggai cardinalfish (Chapter 3), mandarinfish (Chapter 4), giant anemones and anemonefish (Chapter 5), seahorses (Chapter 6), giant clams (Chapter 7), and stony corals (Chapter 8). In order to provide background information and the context for understanding the effects of collection, each case study begins with an introduction to the basic biology of the species examined. This is followed by a detailed overview of trade in this species and its (often negative) effects on wildlife populations and the reef ecosystem. Each case study concludes with a brief description of several conservation measures that have been attempted and the efficacy of those efforts.

Part two of this report examines the wider consequences of trade on communities of organisms and reef ecosystems. This discussion begins with destructive fishing practices and waste in the supply chain (Chapter 9), with a special chapter highlighting the impacts of fishing with cyanide and other poisons (Chapter 10). It then examines an indirect consequence of global trade – introductions of exotic and invasive species (Chapter 11). Finally, the report will review the limited evidence for and discuss concerns about the ecosystem-level consequences of the coral reef wildlife trade (Chapter 12).

Part I:
Case Studies

Chapter 2
Yellow Tang

Introduction to yellow tang biology:

Yellow tang (*Zebrasoma flavescens*) is a species of charismatic algae-grazing fish found across the tropical northern Pacific Ocean. They occur as far west as Japan and Guam, but are most abundant in the waters surrounding the Hawaiian Islands (Eble et al. 2009). Within Hawaii, they are most plentiful on the west coast of the big island of Hawaii where they are commonly collected as aquarium fish (reviewed in Walsh et al. 2004).

To native Hawaiians, yellow tang are known as Lau-Ī-Pala, meaning yellow ti-leaf. The name derives from their bright yellow coloration and oval or ti-leaf body shape. There is some sexual dimorphism in this species, with males being slightly larger than females (Claisse et al. 2009a). This size disparity is a product of faster juvenile growth rates in males and a difference in the timing of reproductive maturity between sexes (Claisse at al. 2009a). Yellow tang also have sharp white tail spines that are used in defense. The tail spines are a defining characteristic of their family, the Acanthuridae, and earned this fish family its common name: the surgeonfish. Although yellow tang are generally bright yellow, genetic evidence indicates that the brown tang (*Z. scopas*) could actually be a color morph of the yellow tang (Steinke et al. 2009, see also Barlow 1974). Alternatively, the similar DNA sequences derived from the two “species” could be a case of incomplete lineage sorting, wherein the evolution of a particular gene has not caught up with the splitting of two species from their common ancestor (Steinke et al. 2009).

Yellow tang dwell on and around the coral reefs of Hawaii. As new recruits and juveniles, yellow tang occupy mid-depth stands of branching corals, particularly the finger coral, *Porites compressa*, and deep aggregates of coral rubble and sand (Walsh 1984, 1985, Ortiz and Tissot 2008). Adults relocate to more varied habitats, but are most abundant on shallow turf-rich boulder habitats on the reef flat, an area commonly described as the “pavement zone” (Walsh 1984, 1985, Ortiz and Tissot 2008, Claisse et al. 2009a). Adult fish aggregate into large schools (Hoover 1993). Yellow tangs tend to have small daytime ranges but may move around more at night, roaming as far as 800 m at a time (J.T. Claisse unpublished observation cited in Williams et al. 2009).

Yellow tang have long, narrow mouths that are specialized for feeding on soft filamentous algae (Hoover 1993). Like other surgeonfish, they are important grazers of algae on coral reefs that contribute to regulating the balance of competition for space between corals and various types of algae. They are even reported to serve as cleaning fish that remove algae from the carapaces of green sea turtles (*Chelonia mydas*) (Losey et al. 1994, Zamzow 1998). Because yellow tang are specialized algae feeders, they sometimes experience health problems in captivity when they are fed an inappropriate diet based on animal protein (Michael 2005).

Yellow tang reproduce multiple times per year on a lunar cycle (Bushnell et al. 2010). Egg production is highest during the full moon and during the late spring and summer, with the lowest egg production recorded from November to February (Bushnell et al. 2010). Females produce 44 to more than 24,000 eggs per spawning, with females larger than 12.0 cm producing the most eggs (Bushnell et al. 2010). Successful spawning results in pelagic larvae that disperse for 55 to 60 days before settling on a reef and maturing into juvenile fish (estimated in Eble et al. 2009). Larval settlement peaks between May and August each year (Walsh 1987).

The larval dispersal distances of coral reef fishes are important considerations when determining if depleted populations can be replenished from afar. For yellow tang, larval fish begin life as passive dispersers whose movements are determined entirely by currents (reviewed in Christe et al. 2010). Over time, the larvae develop the ability to adjust their depth and eventually become strong swimmers (reviewed in Christe et al. 2010). Christe et al. (2010) conducted a novel parentage analysis that matched post-settlement juveniles with their parents. This study demonstrated that larval yellow tangs disperse from 15 to 184 km from the place they were spawned (Christe et al. 2010). Over longer (evolutionary) time scales, yellow tang populations show signs of genetic connectivity across thousands of kilometers of ocean (Eble et al. 2009). Despite this, Elbe et al. (2009) found moderately restricted gene flow along the Hawaiian archipelago. Such a pattern suggests limited dispersal across the archipelago at time scales that are relevant for population recovery from over-collection.

Life is dangerous for larval and juvenile marine fish. For yellow tang, only about 1% of juvenile recruits survive to adulthood (Claisse et al. 2009b). Because of high mortality and other factors, there is a great deal of variability in the number of new recruits to Hawaiian reefs from year to year. Williams et al. (2009) found annual recruitment rates to range from as few as approx. two individual yellow tang per 100 m^2 of reef to as many as 10-17 individual fish per 100 m^2. Walsh et al. (2004) also reported inter-annual variability in recruitment strength. Such variability could cause traditional management techniques, such as bag limits and total allowable catches, to be unsuccessful if limits are based on high recruitment years. However, survival of recruits increases in areas with suitable habitat and few adult fish (i.e., reduced competition) (Claisse et al. 2009b), suggesting that high fecundity and robust source populations (such as in marine protected areas [MPAs]) could restock areas that had been depleted due to overfishing.

Upon settlement, yellow tang grow to reproductive size over 4–6 years, after which growth slows to a halt (Choat and Axe 1996, Williams et al. 2009). For those fish that do survive to adulthood, mortality rates decrease considerably. Yellow tang can be very long lived and are able to reach 41 years of age (Claisse et al. 2009a), but it is unlikely that most individuals reach this age. Adult yellow tang have the capacity to reproduce for several decades (Williams et al. 2009).

Collection and overharvesting of Hawaiian yellow tang:

Yellow tang is the most commonly collected aquarium species in Hawaii (Tissot et al. 2004), making them one of Hawaii's most prominent exports (Hoover 1993). In 1995, for example, yellow tang accounted for 52% of total aquarium-species collections in Hawaii (Miyasaka 1997). Since then collection of yellow tang has only increased. Today they represent the most common fish species by volume (approximately 80% of collections) and value (approximately 70% of value) for aquarium fish landings in Hawaii.

The Hawaiian ornamental reef fish industry has a total estimated export value of approximately $1.06 million U.S. (Walsh et al. 2003). The total aquarium collection industry value is likely much higher, but it is challenging to accurately estimate due to underreporting of catch data (Walsh et al. 2004). Among inshore fisheries, aquarium collection is second only to akule (Bigeye Scad) hook and line fishing (Walsh et al. 2004). Various reports have examined the value of this industry, with estimates of gross sales ranging from $3.2 to $4.9 million U.S. (Cesar et al. 2002, Hawaii Tropical Fish Association report 1993, as reviewed in Walsh et al. 2004). Regardless of the actual number, yellow tang are an especially valuable species, with their dollar value increasing on a per-specimen basis in recent years (Walsh et al. 2004).

As of 2007, there were 67 permit-holding fishers, of which 37 were actively engaged in ornamental aquarium fish collection (Williams et al. 2009, Stevenson et al. 2011). The average collector is a 47 year old male that has been collecting ornamental aquarium fish for 16 years (Stevenson et al. 2011). Collectors generally enjoy their work and thereby derive considerable non-monetary benefits from this occupation (Stevenson et al. 2011). Seventy-one percent of collectors, especially those involved in the business for many years, said they would not change occupations even if better economic opportunities were available elsewhere (Stevenson et al. 2011). Collectors work in groups of one to three people and they collect fish about three days per week (Stevenson et al. 2011). Most collectors work in relatively shallow water via scuba, which limits their collection activity due to physiological limits of bottom time (Stevenson et al. 2011). However, recent technological advances have increased fisher's ability to maximize effort; these include NITROX gas mixtures that increase bottom time when diving, underwater scooters that increase the searchable area during a dive, and GPS devices that enable desirable locations to be pinpointed (Stevenson et al. 2011).

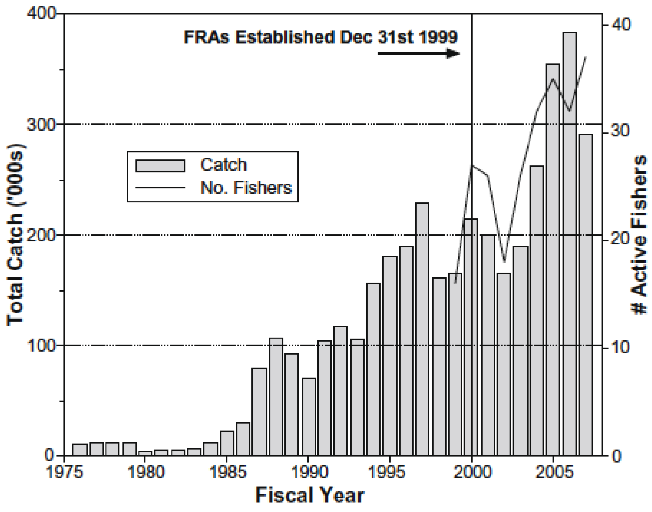
Collectors typically harvest yellow tang and other Hawaiian ornamental aquarium fish using mesh nets and fences (Walsh et al. 2004). Two types of mesh nets are most common, a V-shaped cross net and a multiple-net design that involves a moveable hook net (described in Stevenson et al. 2011). Fishermen often herd fish into nets using 1.3 cm diameter fiberglass sticks, known as "tickle sticks" (Stevenson et al. 2011). High-value species that take refuge in branching corals are often collected with small hand nets and tickle sticks (Stevenson et al. 2011). The fish are extracted by hand or using a hand net and placed in a live-well basket to be surfaced (Stevenson et al. 2011). Once surfaced, fish commonly experience excess pressure in their swim bladder. This pressure is typically relieved by venting the bladder with a hypodermic needle (Stevenson et al. 2011, see the review of "venting" in Chapter 10 of this report).

Several common collection practices have the potential to injure or kill fish as well as damage corals and reef habitat. For yellow tang and other ornamental fish, handling of the animals during collection and transport can cause injury and death (Stevenson et al. 2011). Despite this, Hawaii's ornamental aquarium fishery has low collection mortality compared to ornamental fisheries in other parts of the world (see the review of supply chain mortality later in Chapters 10 and 11). For example, Stevenson et al. (2011) observed 33 hours of ornamental aquarium fish collection. They found fish mortality and discarded fish were rare during collection, comprising less than 1% of the total catch (216 fish were discarded and 14 fish died due to collection) (Stevenson et al. 2011).

During collection corals and reef habitat are sometimes damaged from abrasive contact with tickle sticks or sand-mimicking tarps that are placed over the coral to prevent the fish from taking refuge (Stevenson et al. 2001). Although the extent of this damage has not been well documented, Tissot and Hallacher (2003) did not find widespread coral or habitat damage associated with this fishery. Fishing with poison and other destructive fishing methods is illegal in Hawaii and these practices are not commonly employed (Walsh et al. 2004). However, there are a number of anecdotal reports of collectors using bleach to stun fish and even breaking coral apart to access hiding animals in Hawaii (W. Walsh personal communication cited in Tissot and Hallacher 2003; also described as anecdotal reports in Tissot 1999).

From a collector's perspective, the ideal animal for collection is a juvenile of about 5–10 cm in size (T. C. Stevenson, personal communication cited in Williams et al. 2009). Desirable individuals are juveniles that are over 3 months old (Stevenson et al. 2011). Juvenile yellow tang are reported to have a high mortality rate in captivity (Williams et al. 2009, Stevenson et al. 2011), particularly in the first few weeks following collection. According to collectors, juveniles are more susceptible to chemicals and parasites in holding tanks (Stevenson et al. 2011). This increases the risk of death in captivity and could thereby increase ecological impacts on the reef as additional fish may be collected to offset losses related to mortality. Larger adult fish (i.e., those above 13 cm in length) are not targeted by aquarium collectors, because they are too large for most tanks, and generally are not harvested by other fisheries (Williams et al. 2009).

Figure 4: The total catch of Hawaiian ornamentals by year. Figure taken from Williams et al. (2009).

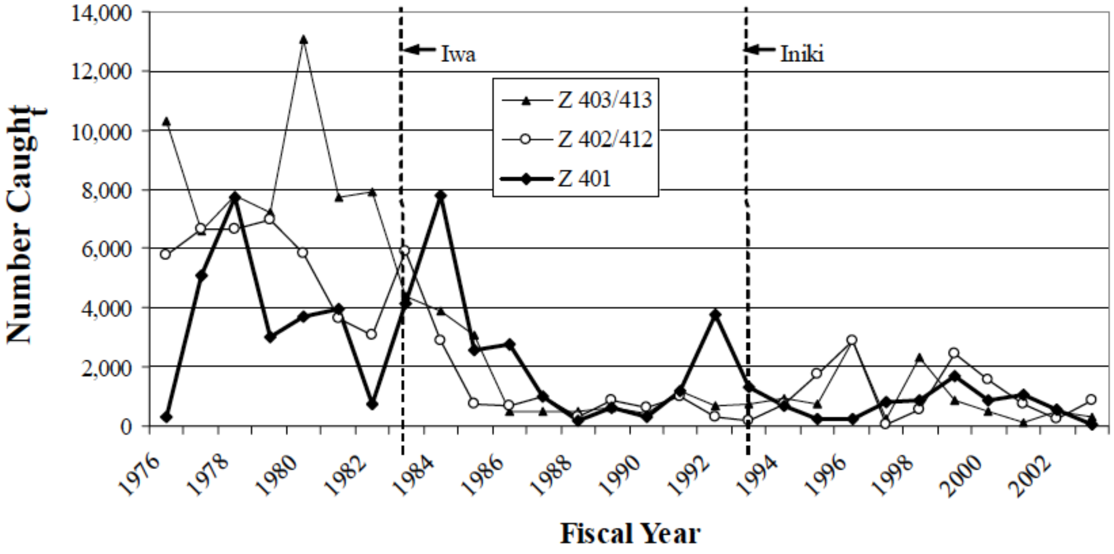


Ornamental aquarium fish collection has been ongoing for at least 50 years in Hawaii, with collection steadily increasing over time (Walsh et al. 2004). Collection began in Oahu as a small-scale industry and, with the availability of commercial air travel and improvements in diving technology, the industry went commercial in the later 1960s and early 1970s (Walsh et al. 2004). In 1973, approximately 90,000 fish ornamental aquarium fish (including yellow tang and other species) were collected from Hawaiian coral reefs (valued at ~$50,000 U.S.; Katekaru 1978). By 1995, that total had grown to 422,823 ornamental aquarium fish (valued at ~$844,843 U.S.; Miyasaka 1997). The growth in this fishery has continued since then. From 1999 to 2007, the volume of fish taken by this trade doubled (Williams et al. 2009, Figure 4).

Based on an examination of records up to 2007, yellow tang catch peaked at 382,921 in 2006 and has since ranged between 300,000 and 400,000 yellow tang per year (Williams et al. 2009). It is estimated that a full one-third (Dierking 2007) to one-half (Zeller et al. 2005) of the catch is unreported (see also Walsh et al. 2004). One study found that only 14% of the permit holders involved in the aquarium trade consistently filed the required month collection reports (Walsh et al. 2004). As a result of poor data reporting practices, collection volumes must be considered minimum estimates of the number of fish taken.

The large number of fishes removed from Hawaiian coral reefs raises important questions about the population-level impacts of the ornamental trade. Is collection having a negative effect on reef fish populations and the reef ecosystem? Does the high reproductive capacity of yellow tang replace the hundreds of thousands of fish removed each year? This topic was first examined by Nolan (1978). Nolan concluded that as of 1974 the aquarium trade did not adversely affect Hawaiian reef fish populations. Unfortunately, Nolan's work was based on a problematic and uncontrolled experimental design, calling into question his conclusions (Tissot and Hallacher 2003). Furthermore, the Nolan study occurred during a time when there was significantly fewer ornamental fish taken from Hawaii's coral reefs than there are today. A more modern reexamination of these questions is clearly required.

Figure 5: Declining catch of yellow tang in Oahu based on catch report data. Figure taken from Walsh et al. (2004).



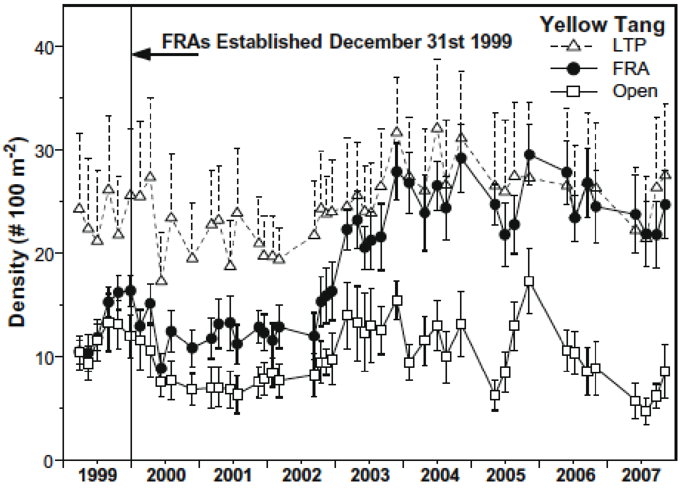
During the 1970s and 1980s, collection of yellow tang primarily occurred around the island of Oahu (Walsh et al. 2004, Tissot et al. 2009). This ended in the 1980s with the total collapse of the Oahu yellow tang fishery. The collapse was brought about by two factors, hurricanes and over-collection (Walsh et al. 2004). Specifically, two hurricanes injured or killed many fish and damaged *P. compressa* corals that serve as essential habitat for juvenile yellow tang (Walsh et al. 2004). Over-harvesting of yellow tang compounded these losses, resulting in substantial declines in yellow tang populations (Walsh et al. 2004, Figure 5). As a result, the fishery collapsed and collectors in Oahu have shifted to other species, particularly invertebrates, or moved to the Kona coast of the big island of Hawaii where most (>94%) yellow tang collection occurs today (Walsh et al. 2004, Tissot et al. 2009). It is important to note that the yellow tang catch declines in Oahu cannot be attributed to a lack of effort. This fish species remains a popular aquarium species and the price per fish has actually increased over time in inflation adjusted dollars.

Since the 1980s, the vast majority of ornamental fish collection has occurred along the Kona coast of Hawaii. In order to determine the impacts of collection in this region, Tissot and Hallacher (2003) conducted an experiment comparing reef fish communities at collection sites with areas where collection did not occur. Sites were paired by habitat type to insure that comparisons were ecologically meaningful. Tissot and Hallacher (2003) selected ten common aquarium species, including yellow tang, to serve as benchmarks for the impacts of the ornamental trade as well as nine species that were not involved in trade to serve as controls. They found that collection caused statistically-significant population declines for 7 out of 10 aquarium species examined, including yellow tang (Tissot and Hallacher 2003). Mean fish density of aquarium fishes was lower at collection sites; the differences in fish abundance at collection vs. non-collection sites ranged between -38% to -75% depending on fish species examined (Tissot and Hallacher 2003). Seven out of nine control species (those species not involved in the aquarium trade) showed no difference between sites (Tissot and Hallacher 2003). Based on these results, the authors concluded that aquarium collectors have "significant effects on the abundance of targeted fishes on the Kona coast of Hawaii" (Tissot and Hallacher 2003).

Tissot and Hallacher's (2003) work was followed up with a more extensive study comparing collection and non-collection sites (Tissot et al. 2004). That more recent study found that aquarium fish species, including yellow tang, were 14–97% less abundant (the mean decline was 26%) in collection areas compared to areas where collection did not occur (Tissot et al. 2004). For yellow tang specifically, the population decrease due to collection was between -43% (Tissot et al. 2004) and -47% (Tissot and Hallacher 2003). In both studies the population declines were statistically significant. (For another popular aquarium fish, the Four Spot Butterflyfish, *Chaetodon quadrimaculatus*, the statistically significant decreases were -97% in Tissot et al. [2004] and -42% in Tissot and Hallacher [2003].)

Concern about the impacts of the aquarium trade led to the creation of a series of fisheries replenishment areas (FRAs) along the Kona coast (see the discussion of yellow tang conservation measures below). The establishment of protected areas allowed for further examination of the impacts of the aquarium trade, including the most exhaustive monitoring study of yellow tang populations to date (Williams et al. 2009). From 1999 to 2007, Williams et al. (2009) recorded a significant decrease of yellow tang density within collection areas (-45%) (Williams et al. 2009). By comparison, populations in protected areas were either increasing or stable indicating that aquarium collectors were the cause of the declines (Williams et al. 2009, Figure 6). Densities of yellow tang of target size (juveniles) were five times higher at protected sites compared to collection areas (Williams et al. 2009). Adult fish populations were also much larger at protected sites (Williams et al. 2009). Taken together, the data demonstrate a strong and negative impact of collection on yellow tang populations.

Figure 6: Density of yellow tang over time in three different areas on the Kona coastline: open areas (Open), fish reserve areas (FRA), and long-term protected sites (LTP). Figure taken from Williams et al. (2009).



For yellow tang living in the FRAs and for the reefs within the FRA system, the protected areas have been a clear success (Williams et al. 2009). Ironically, the establishment of protected areas may be simultaneously contributing to yellow tang decline in other locations. Since establishment of the FRAs, yellow tang populations have continued decline in collection zones (Williams et al. 2009), likely as a consequence of concentrating fishers onto a smaller area without decreasing fishing effort (Stevenson et al. 2011). In order to avoid further declines, additional measures are needed to control fishing effort (Stevenson et al. 2011).

Beyond population-level declines in yellow tang and other aquarium fish species, there is additional cause for concern about the ecological impacts of the ornamental trade in Hawaii. In recent years, aquarium collectors have increasingly targeted herbivorous surgeonfish (Sevenson et al. 2011). Based on observations aboard collection boats, surgeonfish comprised 89% of the total catch, with yellow tang alone making up 69% of fish collected (Stevenson et al. 2011). Even if yellow tang are excluded from the catch data, herbivorous surgeonfish make up 65% of the remaining total catch (including *Ctenochaetus strigosus*, *Naso lituratus*, and *Acanthurus nigrofuscus*; Stevenson et al. 2011). Surgeonfish are important grazers of algae that protect corals from competition and remove algae that would otherwise overgrow the reef (Aronson and Precht 2001). In Hawaii the removal of algae grazers has the potential to lead to algal overgrowth of reefs because (1) herbivores naturally occur at lower abundance on Hawaiian reefs compared to other reef systems, (2) the ornamental trade has reduced the abundance of yellow tang and other herbivorous surgeonfishes (see above), and (3) anthropogenic nutrient inputs (which stimulate algal growth) are increasing in the near-shore waters around Hawaii (Stevenson et al. 2011).

In addition to their work on aquarium reef fish populations, Tissot and Hallacher (2003) also examined the indirect effects of aquarium collection on coral cover, macroalgae cover, and on coral bleaching. They did not observe any indirect effects on these factors related to aquarium fish collection (Tissot and Hallacher 2003). However, the study did not examine the role of nutrients or the presence of other grazers, such as sea urchins on coral vs. macroalgae cover (Tissot and Hallacher 2003). Even if no immediate impact was observed, the loss of functional redundancy among algae grazers on reefs can have significant effects on reef ecosystems. For example, the removal of surgeonfish and other algae grazers from Jamaica due to overfishing, combined with a devastating sea urchin disease and nutrient pollution, led to the loss of the Jamaican reef ecosystem in the 1980s (reviewed in Aronson and Precht 2001). Despite conservation efforts, the Jamaican reef system has failed to recover over the past 30 years (Aronson and Precht 2001), serving as a warning of what can happen when grazers are overfished. Furthermore, the macroalgae work of Tissot and Hallacher (2003) did not measure filamentous algae. Filamentous algae are the preferred food for yellow tang and would be the best algal group to examine to look for a grazing effect. Therefore, additional studies are still needed to determine the ecosystem-level effects of the aquarium trade in Hawaii.

Ornamental reef fish conservation and management in Hawaii:

Hawaii has a long history of efforts to protect coral reefs including coral reef fish. The first law regulating the trade was enacted in 1953 by the territorial government of Hawaii (Walsh et al. 2004). Act 154 enabled the Board of Agriculture to issue permits for the collection of fish for the aquarium trade using fine-mesh nets and traps (Walsh et al. 2004). In the ensuing years an active ornamental fishery has developed in this state. Although there is still need for additional protections and improvement of existing management practices, Hawaiian ornamental reef fish management has achieved notable improvements. This section will review Hawaii's efforts to manage yellow tang and other Hawaii ornamentals with an eye towards the efficacy of various management approaches.

Today, Hawaiian marine resources are managed primarily by the Division of Aquatic Resources (DAR) within the Department of Land and Natural Resources (DLNR) (Tissot et al. 2009). The DAR has a number of management tools at its disposal to regulate fisheries through its administrative rulemaking authority (Tissot et al. 2009). Common fisheries management practices include species-specific size and seasonal limits, catch quotas, gear restrictions, aquaculture-base stock enhancement, and a variety of MPAs, however, most of these measures are not utilized in the aquarium fishery (Tissot et al. 2009). Enforcement of DLNR administrative rules is delegated to the Hawaii Division of Conservation and Resources Enforcement (DOCARE) (Tissot et al. 2009). Unfortunately DOCARE is chronically underfunded and oftentimes lacks political will, which weakens marine resource management in Hawaii (Tissot et al. 2009).

Since as early as 1970, there have been publically-expressed concerns about the long-term sustainability of ornamental reef fish collection in Hawaii (Walsh 1978, Tissot and Hallacher 2003, Capitini et al. 2004). However, for many years DAR was hesitant to take action due to the lack of definitive data on the negative effects of collection (Tissot 1999, Tissot and Hallacher 2003). As will be seen throughout this report, a lack of data is a common impediment to implementing better management of the ornamental coral reef wildlife trade. In the early 1970s, concern about the negative effects of aquarium fish collection led to the (largely unenforced) requirement of monthly collection reports from fishers (Tissot et al. 2004). There were additional public calls for research into the impacts of trade and the establishment of sanctuary areas (Walsh et al. 2004). Public concern was sufficiently strong for the Hawaii Division of Fish and Game (a DAR precursor) to declare a moratorium on harvesting aquarium fish in 1973 (Walsh et al.

2004). However, this measure was rescinded on June 29, 1973, two days before it was scheduled to take effect (Walsh et al. 2004). The fishery went largely unmanaged for the subsequent 25 years despite a large increase in both collection permits issued and number of fish collected (Tissot et al. 2004).

The decline of colorful reef fish populations led to conflict between collectors and dive tour operators in West Hawaii (Capitini et al. 2004, Tissot et al. 2004). In 1987 an informal “Gentlepersons’ Agreement” was arranged among collectors and other user groups wherein collection would not occur in certain areas (Walsh et al. 2004). In 1991 these areas became formalized no-collection zones, known as the Kona Coast Fishery Management Areas, comprising 4 miles of coastline (Walsh et al. 2004). In 1992 an additional 1.3 miles was reserved near the Old Kona Airport, designated as a Marine Life Conservation District (MLCD) (Walsh et al. 2004). Combined, these reserves comprised 7.4% of the Kona coastline.

In May 1996 the Hawaii House of Representatives passed resolution HCR 184 designating a working group to develop a comprehensive management plan for regulating aquarium fish collecting in West Hawaii (Capitini et al. 2004). The working group developed recommendations, but the effort was stalled by interests in the aquarium industry (Capitini et al. 2004). An environmental advocacy group, the LOST FISH Coalition, responded with a 4,000-signature petition asking the legislature to ban aquarium collection in West Hawaii (Capitini et al. 2004). In response, the House introduced HB 3457 to set up a Regional Fisheries Management Area and designate 50% of the Kona coastline as marine protected areas (Capitini et al. 2004). A compromise was reached that reduced the non-extractive protected area to 30% of the Kona coast (Capitini et al. 2004).

In 1998 Hawaii’s State Legislature passed Act 306 creating the West Hawaii Regional Fishery Management Area. One of the mandates of this Act was the reservation of at least 30% of the West Hawaii coastline as Fisheries Replenishment Areas (FRAs). The Act also required substantial involvement by local community members in resource management decisions. In response to Act 306, nine Fish Replenishment Areas (FRAs) were established in West Hawaii in 2000 (Tissot et al. 2004). These FRAs are entirely closed to aquarium collectors. The FRAs encompass 35.2% of the coastline of West Hawaii, including the previous 7.4% of reserved coast plus new areas that added 27.8% of the Kona coastline (Tissot et al. 2004, Williams et al. 2009).

The process of designating the West Hawaii FRAs involved a diverse group of 24 stakeholders and received overwhelming public support (including greater than 93% positive responses at a DAR public hearing with record breaking attendance; Capitini et al. 2004). Despite this, the meetings designating the FRAs were contentious, including conflicts between collectors and other stakeholders involved in the process (Capitini et al. 2004). Collectors felt little incentive to participate in the process and resented the role of managers as the facilitators of the council (Capitini et al. 2004). Although consensus was reached, according to Capitini et al. (2004) “certain community interests reasserted themselves through actions in the state legislative/administrative arena that significantly weakened previously agreed-on regulations.” The outcome of the process was the creation of nine FRAs; however, restrictions on collection equipment were stripped from the provisions at the eleventh hour (Capitini et al. 2004). The result was a weaker FRA and ornamental reef fish management system than was initially intended.

Following the establishment of the FRAs, monitoring studies revealed significant increases in the abundance of aquarium fish within the protected areas (Walsh et al. 2004). For example, Tissot et al. (2004) collected baseline data and monitored fish populations inside and outside of reserves. From 2000 to 2002, two out of ten aquarium fish species, including yellow tang, exhibited population increases in the no-collection zones (Tissot et al. 2004). The overall density of aquarium fishes increased by 26% and the mean density of ornamental fish in FRAs increased by 50% relative to reference areas (Tissot et al. 2004). Yellow tang populations increased by 74% in FRAs and this increase was statistically significant (Tissot et al. 2004). Control species (non-aquarium species) did not show any change resulting from the FRAs. The effectiveness of the FRAs varied from one area to another. Some of this inter-site variation is attributable to differences in suitable habitat abundance within a site (Ortiz and Tissot 2008). Many of the sites examined by Tissot et al. (2004) were largely mid-depth coral habitat, areas dominated by juveniles. Therefore, the population changes observed by Tissot et al. (2004) were likely changes in the abundance of juvenile yellow tang (that are targeted by collectors) and not necessarily of non-targeted adult fish.

Williams et al. (2009) conducted a similar study to Tissot et al. (2004) that compared collection areas to FRAs to long-term protected areas (LTPs; areas outside the FRA system where fishing for the aquarium trade does not occur and has not occurred historically). Their focus was exclusively on yellow tang. Starting in 2003, Williams et al. (2009) detected a major increase of yellow tang within the FRAs; yellow tang populations increased to the levels found in LTPs and remained consistently above collection areas. From 1999 to 2007, yellow tang densities were stable in LTPs, increased by 72% in FRAs, and declined by 45% in collection areas (Williams et al. 2009). The changes in FRAs and collection zones were all statistically significant (Williams et al. 2009). The densities of juvenile yellow tang were five times higher at protected sites compared to collection areas and adult tang populations were also much larger at protected sites (Williams et al. 2009). The changes in yellow tang populations were pronounced and can only be attributable to the effect of the closure of the FRAs to collection.

Based on these findings, Williams et al. concluded that over-exploitation had occurred in the yellow tang fishery. However, the FRAs had prevented the most severe over-exploitation from taking place (Williams et al. 2009). Furthermore, the FRAs enabled recovery in previously overfished areas both in the protected areas and in areas adjacent to the FRAs. Locations adjacent to the marine protected areas had higher densities of fish (41% greater) when compared to sites some distance away, indicating that the FRAs were seeding adjacent areas with fish (Williams et al. 2009). Subsequent studies have demonstrated that the FRAs are providing new recruits to unprotected sites that are separated by considerable distances (up to 184 km; Christe et al. 2010).

In spite of the successes of the FRA system, a note of caution is required. Excluding fishers from much of the Kona coastline has concentrated fishing effort into the remaining habitat, where populations of yellow tang and certain other aquarium fish species continue to decline (Williams et al. 2009, DAR 2010, Stevenson et al. 2011). Additional management measures are clearly necessary to prevent further population depletions. To address this need, Williams et al. (2009) recommended establishing a limited-entry fishery and protecting reproductive-age fish from harvest. The authors stopped short of recommending bag limits or total allowable catches for yellow tang due to the considerable recruitment variability in this species (Williams et al. 2009, see above). My personal assessment is that limits on entry and total allowable catch for aquarium species would significantly improve the sustainability of the West Hawaii ornamental fishery.

Outside of the Kona coast of Hawaii, other management measures have been attempted to protect ornamental reef fish. For instance, the Waikiki-Diamondhead Fisheries Management Area in Oahu employed periodic closures (i.e., rotational management) to protect ornamental coral reef fish from over-harvesting (Williams et al. 2006). Rotational closures led to population-level increases for a wide diversity of species, including yellow tang and other acanthurids, but the increases were overwhelmed by dramatic population declines during the open collection periods (Williams et al. 2006). Therefore, this measure appears to be largely ineffective in sustainably managing Hawaiian reef fish (Williams et al. 2006).

Finally, the pressure put on wild populations of yellow tang could be reduced if this species could be cultured successfully throughout its complete life cycle. Yellow tang have spawned successfully in captivity, for instance at the Wakiki Aquarium (Hall and Warmolts 2003), but they have not yet been fully reared to adulthood. Like many coral reef fishes, yellow tang have a pelagic-larval life stage with specialized-feeding requirements that make aquaculture difficult for this species (Claisse et al. 2009b). However, programs to collect fish shortly after settlement and raise them to adults for commercial sale, known as tank-raised fish, have seen recent success (e.g., http://www.advancedaquarist.com/2002/3/media, http://www.coralmagazine-us.com/content/tank-raised-tangs-triggers-become-reality). Juvenile yellow tang have high natural rates of mortality (Claisse et al. 2009b) and as a result, programs that target very young fish for collection have the *potential* to be more sustainable than current collection practices.

Chapter 3
Banggai Cardinalfish

Introduction to Banggai cardinalfish biology:

The Banggai cardinalfish, *Pterapogon kauderni*, is a rare species of cardinalfish (family Apogonidae) that is popular in the ornamental aquarium trade (Michael 1996, Kolm and Berglund 2003). The species was originally discovered in 1920 (Koumans 1933), but it went forgotten for many years until it was rediscovered in the mid-1990s (Allen and Steene 1995). Upon rediscovery, Banggai cardinalfish became a popular aquarium fish, largely due to its attractive appearance, rarity, limited distribution, and interesting biology.

Banggai cardinalfish are endemic to the Banggai archipelago on the eastern coast of Sulawesi, Indonesia (Allen and Steene 1995). This species naturally occupies 31 out of the 55 islands comprising the Banggai archipelago (Vagelli and Erdmann 2002, Vagelli et al. 2009). Their total range comprises approximately 6,000 km^2; however, when the specific habitat requirements are considered Banggai cadinals only occupy a total area of about 30 km^2 (Vagelli and Erdmann 2002, 2007 CITES Appendix II listing proposal, Vagelli et al. 2009).

Like most coral reef ornamentals, Banggai cardinalfish are attractively-colored fishes. They are marked with alternating black and light-colored bars with white spots. Males and females are similar in appearance (i.e., no sexual dimorphism) and have an even sex ratio (Vagelli and Volpedo 2004). However, males can be recognized during breeding by their enlarged oral cavity. Banggai cardinalfish grow to a maximum length of approximately 65–75 mm (Vagelli 2008, Michael 2005).

Banggai cardinalfish live on shallow-water coral reef and seagrass habitats, ranging from 0.5 to 6 m in depth (Allen 2000, Vagelli 2008). They reside around anemones, corals, and urchins and use these hosts' stinging nematocysts or sharp spines for protection (Allen 2000). Banggai cardinalfish associate with different animal hosts throughout their life stages. Newly recruited juveniles associate with large anemones that dwell among sea grass beds (Allen and Steel 1995, Allen 2000, Vagelli and Erdmann 2002, Vagelli 2004a). Adult fish live directly on the reef in association with either the sea urchin *Diadema setosum* or branching corals (Allen and Steel 1995, Allen 2000, Vagelli and Erdmann 2002, Vagelli 2004a).

Banggai cardinalfish are unusual among apogonids in that they are active during the daytime (Vagelli 2008). They feed on microcrustaceans, teleost fishes, and mollusks (Vagelli and Erdmann 2002). Banggai cardinalfish notably prey on the larval stages of several coral reef fish parasites and therefore may have an important ecosystem role in controlling parasite loads in other reef fish (Vagelli 2008). In home aquariums, their success varies from doing very well to wasting away and starving to death (Michael 2005). They need to be fed meaty foods that simulate their natural diet twice per day (Michael 2005).

Spawning in Banggai cardinalfish occurs several times per year on a lunar cycle (Vagelli and Volpedo 2004). Females exhibit courtship behaviors, including "twitch" and "rush" displays, which convey information about their proximity to spawning and fecundity (Kolm 2004). Females produce a clutch of approximately forty 3 mm-diameter eggs that are bound together by filaments (Vagelli 1999). Their maximum recorded clutch size is 90 eggs (Allen and Steen 1995, Vagelli 1999). The sex roles of Banggai cardinfish are largely reversed compared to many vertebrates, with males providing significant parental care. Males incubate the eggs within a buccal pouch within their mouths for about 20 days, a phenomenon known as "mouthbrooding" (Vagelli 1999). Males do not eat during this process, which means they are only able to reproduce several times per year and produce relatively few offspring per adult (Vagelli 2008). Reproductive output is linked to body size, especially in males (Kolm 2002, Kolm and Olsen 2003). After hatching, offspring remain in the male's mouth for another 9 to 10 days, after which they do not return to the male for protection (Vagelli 1999). Banggai cardinalfish lack a planktonic larval stage (Vagelli 1999), which leads to a highly-limited dispersal capacity. The lack of a dispersal phase is unique among apogonids (Vagelli 1999). They reach maturity in 9 to 11 months (Vagelli 1999). Members of this species have a short life span: they live to a maximum of about 4 years under ideal conditions in captivity or about 1–2 years in the wild (2007 CITES Appendix II listing proposal).

As adults, Banggai cardinals are gregarious fish that form stable social groups of 2 to 200 individuals. These groups are not familial or kin groups; instead groups are comprised of a mix of related and unrelated individuals. If an individual Banggai cardinalfish is removed from its group, it exhibits strong homing behavior and returns to its home (Kolm et al. 2005). Thus, adult fish appear to have very low dispersal ability. Their unusual reproductive biology (see above) further restricts the species' ability to disperse to new locations. As a result, Banggai cardinalfish exhibit extremely high population structure for a marine fish (Vagelli 2008). Genetic evidence based on mitochondrial DNA and microsatellite markers shows population genetic structure at multiple spatial scales, including a strong phylogeographic break between the southern island of Bangkulu and other areas of their range (Bernardi and Vagelli 2004), significant population genetic structure at scales of just 2 to 5 km (Hoffman et al. 2005; Vagelli et al. 2009), and some evidence for isolation by distance (Hoffman et al. 2005). Statistical assignment tests corroborate this high population structure, with 10 out of 12 populations on Bangkulu Island being genetically differentiated from one another (Vagelli et al. 2009). Genetic evidence also suggests that this isolation has been longstanding, with time for genetic mutations to evolve within isolated populations (Hoffman et al. 2005). As a result, Banggai cardinalfish have very limited natural ability to re-colonize an area if the populations are severely depleted by over-collection. The extreme isolation of populations also suggests that each population should be managed independently (Vagelli 2008).

Collection and overharvesting of Banggai cardinalfish:

Since the species was rediscovered in the mid-1990s, Banggai cardinalfish have become popular aquarium fishes (Michael 1996, Kolm and Berglund 2003). The limited range, rarity, low reproductive capacity, ease of capture, and restricted dispersal ability of Banggai cardinalfish make them easily vulnerable to depletion from overharvesting (Lunn and Moreau 2004). Ironically, the species is popular for many of the same reasons making it vulnerable: the Banggai cardinalfish is a rare, unusual, and biologically-interesting species that can do well in confined conditions.

According to Lunn and Moreau (2004), trade in Banggai cardinalfish began in 1992 with traders from outside of the Sulawesi region coming in to collect. Changes to regional fishing regulations have required outside fishermen to hold collection permits since 1995 and this change enabled local collectors to become more involved in the fishery (Lunn and Moreau 2004). Collectors in the Banggai region are poor and collection of Banggai cardinalfish generates income for several hundred people (Lilley 2008). Nevertheless, only a very small percentage of the local population (≤0.1%) participates in the Banggai cardinalfish trade (Indrawan and Suseno 2008, Vagelli 2008). Rates of illegal poaching by non-locals appear to be high, but poaching is difficult to document by its very nature (Lilley 2008). For example, boats from Bali are not permitted to collect in the Banggai archipeligo, yet this activity has happened since at least 2001 (Lunn and Moreau 2004, Vagelli 2008).

Banggai cardinalfish are not the sole source of income for most collectors. Fishers often make a living by combining Banggai cardinalfish harvest with fishing for other species for the aquarium trade, catching fish for human consumption, and/or harvesting seaweed (Lunn and Moreau 2004). The prices paid to collectors for individual Banggai cardinfish are very low. As of 2008, collectors were paid about 250 rupiah (IDR) per fish, the equivalent of approximately $0.02 U.S. (Lilley 2008). Despite these low prices, ease of capture makes this fish desirable over more expensive but difficult to harvest alternatives (Lunn and Moreau 2004). Local buyers sell the fish to exporters for roughly 1500 rupiah ($0.16 U.S.) and exporters sell the fish at approximately $2–5 U.S. (Lilley 2008). Importers in turn sell the fish for about $9.55 U.S. and retailers sell the fish for approximately $20 U.S. (Lilley 2008).

In the wild, Banggai cardinalfish are collected using methods that do not involve highly destructive poisons or coral smashing (Lunn and Monreau 2004). However, these destructive fishing practices are used to collect other fish species on the coral reefs where Banggai cardinalfish occur (Lunn and Moreau 2004). Most Banggai cardinalfish collection methods involve simple nets or containers. One common technique is to herd urchins associated with Banggai cardinalfish into containers. The fish willingly follow their urchin hosts and are ultimately trapped (Lunn and Moreau 2004). Banggai cardinalfish are also captured using coarse nets, such as a funnel-shaped net known as a "cang" (Lilley 2008). Because of their gregarious nature, hundreds of fish can be captured at a time using a single cang. Unfortunately, crowding and course net material often results in severe damage to fishes' scales, fins, and eyes (Lilley 2008). This method results in high rates of mortality and rejection of visibly-injured fish by buyers (Lilley 2008).

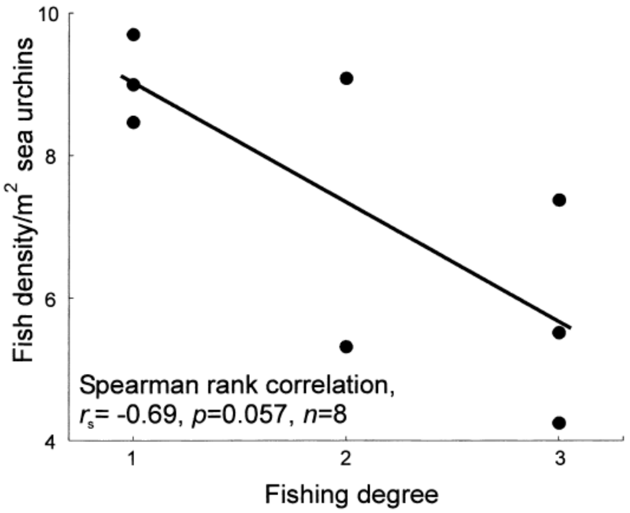
After collection, Banggai cardinalfish are sorted and transferred into Styrofoam boxes aboard a canoe or boat. Collectors report that between 25–50% of fish are thrown back at this stage because they were killed or too severely injured during collection (Lilley 2008). The fishes are then moved to shallow water holding pens near the collectors' homes. Interviews with collectors suggest that mortality rates during holding and transportation are high (Lunn and Moreau 2004). Collectors estimated that about 50% of fish in holding pens die during this stage (Lilley 2008). In total, only about one out of every four fish that are initially collected makes it to the buyer for export (Lilley 2008).

Once in the hands of the middlemen, Banggai cardinalfish mortality and illness remain high. Banggai cardinalfish are often packed at densities of 30 to 50 fish per container and given infrequent water changes, leading to increased mortality (Lilley 2008). Vagelli (2008) places mortality at this stage at an average of 25–30% (occasionally as high as 50%) with an additional 15% rejected by buyers due to injury and damage to the specimens. In captivity, Banggai cardinalfish commonly die from epidemics of iridoviruses (*Megalocytivirus*) (Weber et al. 2009). Captured Banggai cardinalfish sold in the U.S. experience high infection levels of this virus (Weber et al. 2009). Infection occurs post-capture at either export or import centers (Weber et al. 2009). Susceptibility to this iridovirus disease is a result of the combined stress of capture, handling, crowding, and long-distance transportation (Weber et al. 2009). The high rate of injury, disease, and death creates a positive feedback loop driving more and more collection to compensate for supply-chain losses. The unfortunate result is additional population declines.

In recent years Banggai cardinalfish have become a staple of the ornamental coral reef fish trade (Rhyne et al. 2012). Based on import data, this species recently ranked as one of the ten most valuable marine aquarium fish imported into the U.S. (Balboa 2003). Despite this, trade remains unmonitored and largely undocumented (Lilley 2008). The species' popularity and the intensity of collection have raised concerns about the sustainability of the Banggai cardinalfish fishery. Concern about over-exploitation of this species began in November 1998 when a team from Conservation International witnessed more than 5,000 fish being held for aquarium fish exporters in one small village (Allen 2000, Allen and Werner 2002). The inference from this observation was that heavy collection occurred throughout the Banggai archipelago.

As of 2002, an estimated 50,000 to 60,000 specimens were being collected from the wild each month (Vagelli and Erdmann 2002). By 2004 the number being exported increased to at least 118,000 fish per month (Lunn and Moreau 2004). The export figure should not be confused with the actual number of fish collected; the Lunn and Moreau (2004) estimate did not consider pre-sale mortality in fishers' holding cages, collection from all regions, or poaching by outsiders (Lunn and Moreau 2004). The

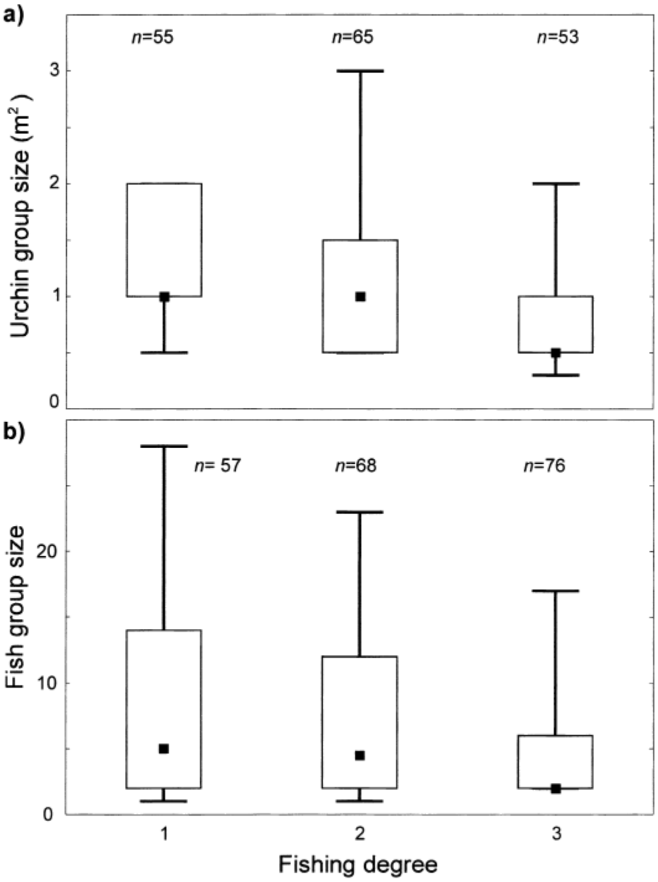
Figure 7: The relationship between fishing pressure and Banggai cardinalfish density. Higher fishing degree values indicate heavier levels of exploitation. Figure taken from Kolm and Berglund (2003).



number of Banggai cardinalfish exported annually has increased from 600,000–700,000 fish in 2001 to 700,000–900,000 fish in 2004 (Vagelli 2005) to a total of 1,000,000 fish by 2007 (Vagelli 2008). In contrast, U.S. import data suggested that approximately 150,000–200,000 Banggai cardinalfish were imported into the U.S. for a one year period, 2004–2005 (Rhyne et al. 2012). Considering that the total population size was estimated to be 2.4 million individuals (Vagelli and Erdmann 2002), there is the potential for overharvesting at current collection rates.

Several researchers have examined the population impacts of overfishing on Banggai cardinalfish populations. As early as 2000, collection for the ornamental trade had reduced fish population density and group size (Kolm and Berglund 2003). Kolm and Berglund (2003) reported that the density of Banggai cardinalfish on Indonesian reefs was inversely related to fishing pressure. In other words, higher rates of fishing resulted in lower Banggai cardinalfish population sizes (Figure 7). Collection also reduced the group size of the urchins that are used by Banggai cardinalfish for protective habitat (Kolm and Berglund 2003, Figure 8). Taken together, fishing pressure had negatively impacted Banggai cardinalfish populations.

Figure 8: Mean urchin (a) and Banggai cardinalfish (b) group size as related to fishing pressure. Higher fishing pressure values indicate heavier levels of exploitation. Figure taken from Kolm and Berglund (2003).



More anecdotal reports also support the impacts of trade on Banggai cardinalfish populations. For example, informal surveys of Indonesian reefs confirmed that fishing activity was correlated with fish population size. Areas where collection takes place reportedly had fewer Banggai cardinals, as compared to areas without recent collection (Lilley 2008). Based on interviews of collectors involved in trade, there was wide-spread acknowledgement among fishers that harvested populations were over-exploited (Lilley 2008).

Field surveys of populations fished from 2001 to 2004 documented population declines exceeding 90% (CITES 2007). Specifically, populations from Masoni Island were reduced to just 37 fish in the 4,800 m^2 survey area, with just 150 fish detected on the entire island as of 2007 (Vagelli 2008). Similarly, only 27 fish were found at Peleng Island (Vagelli 2008). At Bakakan Island the population size dropped from 6,000 individuals in 2001 to just 350 fish in the most recent surveys (Vagelli 2008). Limbo Island has possibly experienced the most severe declines. In 2001, only 0.02 fish per m^2 could be located at Limbo Island (Vagelli 2008). Almost no fish remained at Limbo Island by 2004 and the population has not recovered since then (Vagelli 2008). According to Vagelli (2008), Banggai cardinalfish populations had been reduced in abundance by about 90% across the survey area (2008). This rate of decline is predicted to drive the species to extinction within a decade (CITES 2007).

Despite the population declines seen throughout the Banggai cardinalfish's native range, there have been several successful exotic-species introductions of Banggai cardinalfish into other areas. For instance, in 2000 Banggai cardinalfish were found in Lembeh Strait, an area approx. 400 km from the natural Banggai cardinalfish range (Erdmann and Vagelli 2001). The apparent source of the introduction was from the escape or release of Banggai cardinalfish from the holding facility of a nearby aquarium fish exporter (Erdmann and Vagelli 2001). Genetic testing of the Lembeh Strait population provided further evidence that was consistent with an introduction (Vagelli et al. 2009). Banggai cardinalfish have been introduced elsewhere outside of their endemic range including Luwuk (Vagelli and Erdmann 2002), Tumbak, and Palu Bay (Moore and Ndobe 2007). It is ironic that a species so heavily exploited in its natural range can apparently be easily introduced to other areas. However, the extremely low dispersal capacity and low reproductive output of Banggai cardinalfish (see above) have prevented these introductions from causing any widespread ecological problems (i.e., becoming invasive species). The introductions also suggest that reintroduction programs could successfully restore Banggai cardinals to areas where they had been severely depleted (provided that the genetics of the source population was sufficiently considered).

In addition to the threats posed by overfishing, Banggai cardinalfish have experienced population declines from several of the other problems imperiling Indonesia's coral reefs. Although Banggai cardinalfish are not targeted for collection by destructive fishing practices, their habitat is commonly degraded by dynamite and cyanide fishing of other fish species (Indrawan 1999, Lilley 2008). Heavy exploitation by aquarium fish collectors in combination with habitat destruction caused by destructive fishing practices (i.e., explosives, cyanide, and coral destruction while netting fish) have all contributed to population declines (Allen 2000). Careless boat handling (e.g., anchor damage), sedimentation from poor land use practices, nutrient pollution from fertilizer and sewage, and high volumes of plastics, Styrofoam, and other solid waste on Indonesian coral reefs further threaten this species (Indrawan 1999, Lilley 2008).

Efforts to conserve and protect Banggai cardinalfish:

Like most coral reef ornamental fishes, there are currently virtually no regulations on the collection and trade in Banggai cardinalfish (Lunn and Moreau 2004, CITES 2007). As of 2008, there were no official no-take zones for Banggai cardinalfish (Lilley 2008), but efforts were underway to establish several at that time (Ndobe and Moore 2008). The one existing requirement regulating Banggai cardinalfish collection in Indonesia is that non-local collectors must obtain collection permits in order to harvest this species; however, this appears not to be enforced. Beyond this regulation, there are several impediments to improving management of this species under the structure of fisheries management law in Indonesia. Most notably, Indonesia’s regional autonomy laws governing natural resource management (Laws 22/1999, 25/1999, and 32/2004 as well as Govt. Regulations 25/2000) designate authority for the management of marine ornamental fish to regional governments (USAID DRSP 2006). This decentralized governance structure makes it challenging to enforce any potential national or international regulations (Indrawan and Suseno 2008). The lack of resources for monitoring and enforcement further weakens the capacity for protection and management of this (and many other) species.

Since 1997 there have been various initiatives to conserve Banggai cardinalfish and to establish a captive breeding program (Vagelli 2004b). Aquaculture of most marine ornamental fish has been stymied by the feeding and habitat requirements of pelagic larval fish. However, Banggai cardinalfish lack a larval stage (see above), making this a much easier species to breed in captivity. There are widespread reports of captive breeding successes by hobbyists, commercial breeders, scientists, and public zoos and aquariums (e.g., Hall and Warmolts 2003, Moe 2003). The fish reproduce readily in captivity and juveniles will eat common aquarium feeds such as brine shrimp (Vagelli 2004b). Still, captive breeding is not without challenges. In captivity, juvenile Banggai cardinalfish commonly experience ‘shock syndrome,’ characterized by “rapid, short and jerky bursts of motion, brief spiral swims and falling to the bottom”, as well as rapid ventilation (Vagelli 2004b). Most individuals raised on a conventional brine shrimp diet died before reaching adulthood (mean mortality was 80.7%; Vagelli 2004b). Fortunately, advances in aquaculture can reduce problems like shock syndrome. Vagelli (2004b) demonstrated that shock-syndrome mortality was substantially reduced by feeding fish a highly unsaturated fatty acid-enriched diet (reducing mean mortality to 5.3%). In spite of such promising advances, the low reproductive output and concomitant high cost to benefit ratio of Banggai cardinalfish aquaculture has hindered expansion of aquaculture efforts. As long as inexpensive, wild-harvested fish are available, it will be difficult for captive breeding programs to outcompete wild-caught fisheries.

Because of the population declines described in the previous section, the Banggai cardinalfish was proposed to be listed on Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) in 2007 (Indrawan and Suseno 2008, Vagelli 2008). Export permits for Appendix II specimens can be issued by the exporting country only when the following conditions are met (CITES Treaty, Article IV): (a) “[a] Scientific Authority of the State of export has advised that such export will not be detrimental to the survival of that species”; (b) “[a] Scientific Authority in each Party shall monitor both the export permits granted by that State for specimens of species included in Appendix II and the actual exports of such specimens. Whenever a Scientific Authority determines that the export of specimens of any such species should be limited in order to maintain that species throughout its range at a level consistent with its role in the ecosystems in which it occurs and well above the level at which that species might become eligible for inclusion in Appendix I, the Scientific Authority shall advise the appropriate Management Authority of suitable measures to be taken to limit the grant of export permits for specimens of that species”; and (c) “a Scientific Authority of the State of introduction advises that the introduction will not be detrimental to the survival of the species involved”. The proposal was recommended to the U.S. CITES Scientific Authority by Alejandro Vagelli, a scientist and expert on Banggai cardinalfish (Vagelli 2008). The U.S. evaluated Vagelli’s proposal favorably and invited Indonesia to co-sponsor it at the 14th meeting of the Conference of the Parties to CITES (Vagelli 2008). The proposal also received support from the CITES Secretariat, International Union for Conservation of Nature (IUCN), the European Community, and several conservation organizations (Vagelli 2008).

The Indonesian central government and Ministry of Fisheries and Marine Affairs left the decision about whether or not to support the proposal to provincial and regional governments (Indrawan and Suseno 2008). However, they expressed concern about the increased CITES paperwork burden with listing (Indrawan and Suseno 2008). The provincial government of Central Sulawesi indicated that listing would disrupt local livelihoods, whereas the regional government of Banggai Island hesitantly supported the proposal because they thought listing would improve recognition of the regions unique biodiversity (Indrawan and Suseno 2008). In the end Indonesia declined to co-sponsor the listing proposal on grounds that it would be detrimental to people’s livelihoods people and that government-led conservation efforts were ongoing (Vagelli 2008).

Vagelli (2008) argues that the reasons given by the CITES delegates from Indonesia as to why they did not support the proposal were misleading; few people are involved in the collection of this species and no such conservation programs existed at that time. Vagelli (2008) also indicates that Indonesian CITES authorities underreported the actual declines in Banggai cardinalfish abundance. According to Vagelli (2008), misinformation about the status of this fish was disseminated, purportedly by representatives of the ornamental aquarium industry. In addition, the United Nations Food and Agriculture Organization (FAO) opposed the proposal on the basis of Indonesia’s objections and the misperception that the Banggai cardinalfish was a high-productivity species (Vagelli 2008). Several countries voiced opposition to the proposal based on Indonesia’s position and, as opposition to listing increased, the U.S. delegation withdrew the proposal (Vagelli 2008).

While listing of Banggai cardinalfish under CITES Appendix II failed, other national and international organizations have recognized the threats faced by this species. The population declines described in the preceding section led the International Union for Conservation of Nature (IUCN) to recognize *P. kauderni* as an ‘‘Endangered’’ species in the Red List in 2007 (Allen and Donaldson 2007). Furthermore, the (currently inactive) Marine Aquarium Council (MAC) helped develop voluntary “best practices” for collection of various marine species including the Banggai cardinalfish (Lilley 2008). MAC developed guidelines while the Yayasan Alam Indonesia Lestari (LINI or the Indonesian Nature Foundation) took on responsibility for training collectors and government officials (Lilley 2008). LINI assisted Indonesia’s Department of Marine Affairs and Fisheries in creating a management plan for the species (Lilley 2008). LINI offered suggestions including: formation of a fisher’s association that can collectively bargain for fish prices, implementation of a long-term monitoring program, establishment of no-take zones, improving waste disposal and public awareness of the damage caused by trash, use of better quality nets and fishing gear to avoid injuring fish, improved training of all individuals involved in harvesting and export, and involvement of stakeholders in conservation efforts (Lilley 2008). Furthermore, the New Jersey State

Aquarium and Zoological Society of London both had Banggai cardinalfish programs aimed at elucidating the species' biology and the impacts of trade in order to aid conservation (Hall and Warmolts 2003).

Finally, after the failure of the proposal to list the Banggai cardinalfish under CITES Appendix II, a national meeting convened in Palu, Central Sulawesi to develop a plan of action (Ndobe and Moore 2008). The meeting was attended by a diverse coalition, including local, regional, and national government and management officials, fishers, non-government organizations, and members of academia (Ndobe and Moore 2008). A multi-year sustainable management plan was developed at this meeting resulting in the establishment of the Banggai Cardinalfish Centre and a Marine Protected Area (Ndobe and Moore 2008). In late 2008, efforts were underway to:

(1) Develop additional marine protected areas
(2) Develop a ministerial decree for management of the fishery
(3) Further develop the Banggai Cardinalfish Centre
(4) Engage local scientists in Banggai cardinalfish research
(5) Develop a captive breeding program
(6) Conduct monitoring of the trade by the Fisheries Resources Directorate and District Fisheries Service
(7) Survey and monitor wild Banggai cardinalfish populations
(8) Train fishermen to comply with MAC standards and
(9) Distribute a children's book to improve awareness of this unique fish (Ndobe and Moore 2008).

So far the success and current status of these efforts have not been widely reported. However these efforts offer hopeful possibilities for the future of Banggai cardinalfish.

Chapter 4
Mandarinfish

Introduction to mandarinfish biology:

Another popular fish in the coral reef aquarium trade is the mandarinfish, *Synchiropus splendidus* (formerly *Callionymus splendidus, Neosynchiropus splendidus,* or *Pterosynchiropus splendidus*). Mandarinfish occur throughout the Indo-Pacific, ranging approximately from the Ryukyu Islands of southern Japan, south to Australia, west to Indonesia and the Philippines and east to New Caledonia (Myers 1999). Throughout this range, mandarinfish are known by many different names which can lead to considerable confusion about their common identity. Synonymous common names include: mandarin dragonet, green mandarin, striped mandarinfish, striped dragonet, green dragonet, mandarin asli, mandarin goby, and sometimes psychedelic mandarinfish. An unrelated species with a similar name, the mandarin fish or Chinese perch (*Siniperca chuatsi*), adds further confusion to the situation.

Whatever one names them, mandarinfish are a gorgeous species of dragonets (family Callionymidae) with blue and orange markings that form vibrant circles, stripes, swirls, and dots. In fact the name "mandarinfish" derives from the colorful silk robes of a 19th century Chinese mandarin (Miles 2004). Mandarinfish are small in size, 60–90 mm in length, with males being about 10 mm larger than females (Michael 2005, Sadovy et al. 2005, Rasotto et al. 2010). Although there is no sexual dimorphism in mandarinfish coloration, males have an extended dorsal spine/first dorsal fin that makes them highly desirable to aquarium hobbyists (Myers 1999, Sadovy et al. 2001, Rasotto et al. 2010). This male dorsal spine is displayed to ward off other males and to attract females during mating (Rasotto et al. 2010). The fish's pectoral fins are also brightly colored and they commonly perch atop these fins while sitting on the sea floor.

One unique aspect of mandarinfish biology is their truly-blue pigmentation. More precisely, these fish have blue cellular pigment organelles (cyanosomes) within chromatophores (dendritic cells; Goda and Fujii 1995). In most animals, blue coloration does not derive from actual blue pigment, but instead arises from structures (e.g., crystals) that reflect blue light and incoherently scatter other wavelengths (Goda and Fujii 1995, Bagnara et al. 2007). Mandarinfish are unusual in that they were the first animal ever reported to have blue pigments and are one of only two vertebrate species known to have chromatophores containing a truly-blue pigment (the second is the closely-related psychedelic fish, *S. picturatus*; Goda and Fujii 1995, Bagnara et al. 2007).

Mandarinfish's bright markings are very conspicuous to other coral reef wildlife and these markings putatively serve as a warning to potential predators (Sadovy et al. 2005). Mandarinfish have foul-smelling and bitter-tasting mucus that likely includes toxic chemicals to deter predators (Paxton and Eschmeyer 1998, Gonzales and Savaris 2005, Sadovy et al. 2005). When speared and injured by collectors (see below), mandarinfish release large quantities of this mucus and these secretions will poison other fish species (Gonzales and Savaris 2005). The mucus is produced by two cell types, mucus cells (i.e., globlet cells) and an unusual set of secretory cells that are believed to produce toxic and repellent compounds (Sadovy et al. 2005). In the field, predatory threadfin breams (a nemipterid fish) have been observed attempting to eat and then forcefully rejecting mandarinfish (Sadovy et al. 2005). It is likely that the bright coloration warns most predators that a mandarinfish makes a distasteful and potentially toxic meal (Sadovy et al. 2005). Mucus also protects mandarinfish from common skin infections in home aquaria (Michael 2005) and possibly in the wild.

In addition to its anti-predator and anti-infection properties, the mucus produced by mandarinfish may serve as a protective layer against skin abrasion (Sadovy et al. 2005). Mandarinfish skin lacks scales and yet these bottom-dwelling fish live in close contact with abrasive substrates like corals. Mucus cells are concentrated on their lower (i.e., ventral) side, likely for protection from chaffing against such substrates (Sadovy et al. 2005).

As benthic coral reef fish, mandarinfish tend to stay directly on the reef bottom. They commonly take refuge in the branches of *Porites* spp. corals, but can also be found in coral rubble and silted areas of the reef. Mandarinfish occur at depths from 1 to 18 m (Randall et al. 1990, Lieske and Myers 1994, Myers 1999) and adult fish range over many square meters of the reef (Sadovy et al. 2001). Mandarinfish are most active at dusk and dawn or during overcast times when light levels are reduced (Sadovy et al. 2001, Gonsales and Savaris 2005). The rest of their time, including during the day and when sleeping at night, is spent hiding within crevices or coral branches (Gonsales and Savaris 2005). This reclusiveness makes capturing mandarinfish difficult and has led collectors to develop a spear-fishing method to capture them (described by Gonsales and Savaris 2005, see below).

Mandarinfish feed along the bottom on small crustaceans (e.g., amphipods and copepods) and other small invertebrate meiofauna, especially those caught on coral substrates (Sadovy et al. 2001). Gut content analysis from seven mandarinfish revealed a number of prey items, including harpacticoid copepods, polychaete worms, small gastropods, gammaridean amphipods, fish eggs, and ostracods (Sadovy et al. 2001, see also Sano et al. 1984). Their food requirements are quite specific and, as a result, wild-caught fish do poorly in captivity (Sadovy et al. 2001, see below).

Just after sunset every day, groups of three to five female mandarinfish gather at specific locations on the reef (Sadovy 2001). Males arrive shortly thereafter and display to females through a combination of dorsal fin displays and “agitation” of the entire body, described by Rasotto et al. (2010) as “a distinct movement of head-to-caudal-fin shaking”. Although the adult sex ratio is naturally even between males and females (Sadovy et al. 2001), not all males get the opportunity to mate. Male mandarinfish are able to mate multiple times at every spawning, but females only spawn once per night (or even once every several days; Rasotto et al. 2010). This situation leads to intense competition for female mates (Sadovy 2001). Larger males actively chase off smaller males that attempt to mate and prevent interruption of courtship by other males (Sadovy 2001, Rasotto et al. 2010). Females also prefer larger males; in mate choice experiments, females spent much more time in front of large males and attempted to pair exclusively with larger mates (Rasotto et al. 2010). This preference was also observed in the field where, despite courtship by males of various sizes, females mated almost exclusively with large males (Rasotto et al. 2010). Taken together, the combination of male vs. male competition and female choice creates a size-based dominance hierarchy where the biggest males mate with the greatest number of females (Sadovy 2001).

Once a mate is selected, the mandarinfish pair aligns while swimming about a meter into the water column and releases eggs and sperm for external fertilization (Sadovy 2001, Rasotto et al. 2010). The male's anal fin forms a funnel during mating that completely surrounds the female genital opening (Rasotto et al. 2010). Smaller males cannot form a full funnel when paired with larger females and this situation limits the number of eggs that small males can fertilize (Rasotto et al. 2010). Furthermore, large males produce significantly more sperm than small males (larger females also produce the most eggs; Rasotto et al. 2010). Mandarinfish egg fertilization rate is directly linked to the number of sperm produced; larger females may not have all their eggs fertilized when mating with a smaller male (Rasotto et al. 2010). Therefore, there is a fecundity risk for females that mate with smaller males, which likely underlies female mandarinfish's strong preference for larger mates.

Mandarinfish are one of the smallest pelagic-spawning fish known (Sadovy et al. 2001). During spawning, females produce 12–205 small (0.7–0.8mm diameter) eggs (Sadovy et al. 2001). The eggs are neutral to slightly positively buoyant and lightly adhere to one another (Sadovy et al. 2001). The fertilized eggs drift around as a pelagic mass that splits into smaller and smaller egg groupings over time (Sadovy et al. 2001), which is a common characteristic of all dragonets (Takita 1983). Within 12.5 to 16 hours, embryos hatch from the egg (Sadovy et al. 2001). Similar to other dragonets, mandarinfish larvae are among the smallest fish ever recorded at hatching (Leis and Rennis 1983, Houde 1984, Sadovy et al. 2001). After 36 hours larvae are able to swim in very short bursts and the larvae actively feed at the water surface within 6 to 7 days (Sadovy et al. 2001). All dragonets, including mandarinfish, have a short interval from hatching to settlement (Takai and Yoshioka 1979, Eda et al. 1994a,b, 1997). Within 8 to 9 days of hatching, juvenile mandarinfish settle on the reef (Sadovy et al. 2001). Settled mandarinfish remain within a small area for one month or more and take several months to reach adulthood (Sadovy et al. 2001). Although no genetic studies have been conducted on mandarinfish dispersal and genetic connectivity, their short time as pelagic larvae suggests that dispersal distances are somewhat limited. The fish have few natural predators and may live for 10–15 years in the wild.

Collection and overharvesting of mandarinfish:

Because of its gorgeous markings, elaborate fins, and small body size, the mandarinfish has become a very popular species among hobbyists that maintain reef-based aquariums. The species is heavily collected in the Philippines and Indonesia (Gonsales and Savaris 2005, Reksodihardjo-Lilley and Lilley 2007), with Batasan Island being one major source location in the Philippines (Sadovy et al. 2001). Compounded by its popularity, the collection methods used to harvest mandarinfish and the very poor success rate of this species in captivity raise serious doubts about whether current practices are ecologically sustainable or humane.

The reclusive nature of mandarinfish has required collectors to use specialized collection techniques. About three times per week, mandarinfish collectors venture out during dawn and dusk to collect animals (Sadovy 2001, Gonzales and Savaris 2005). Even during the most active periods of the day, mandarinfish are still quick to hide. Common collection methods like netting or cyanide fishing oftentimes fail (Sadovy

2001). As a result, fishers developed a spear to stab and capture mandarinfish (Sadovy et al. 2001). This spear is typically a 65 cm long bamboo stick tipped with one to two needles that is forcefully propelled using an elastic sling (Sadovy et al. 2001). Fish are typically stabbed in the abdomen by scuba divers, snorkelers, or air compressor divers (Sadovy et al. 2001). A detailed account of the method was provided by Gonzales and Savaris (2005).

Remarkably, mandarinfish can survive being speared. Despite this, stabbed mandarinfish exhibit symptoms of injury and stress, including copious mucus secretion (Gonzales and Savaris 2005). There are many anecdotal reports of fish missing eyes, being paralyzed, or dying as a result of injuries from spearing (for example: http://en.microcosmaquariumexplorer.com/wiki/Mandarin_Harvest_Realities). Buyers and middlemen also note the prevalence of mandarinfish injured by spearing (Sadovy et al. 2001). Buyers and middlemen prefer fish collected with one needle because severe injury and mortality is higher with the two-needle spears (Sadovy et al. 2001). Because of the injury and death associated with this collection technique, mandarinfish spearing has been called inhumane by some experts (Walster 2008).

Sadovy et al. (2001) interviewed eleven aquarium fish collectors from the Batasan Islands, the Philippines. According to these interviews, mandarinfish were heavily targeted between 1987 and 1995, when they comprised a large portion of the interviewees' income (Sadovy et al. 2001). Despite mandarinfish's popularity, data on this fishery and on the biology of this species is limited (Sadovy et al. 2001, Wabnitz et al. 2003). Available evidence indicates that over-collection has led to population declines. When fishing pressure was high in the late 1980s, for example, two compressor divers would bring in more than 1,000 fish over 3 hours of work (Sadovy et al. 2001). By 2000, mandarinfish populations had substantially declined (Sadovy et al. 2001). At that time, free divers brought in only 2–23 fish per diver after two hours of collection, averaging just 10–15 fish per day (Sadovy et al. 2001). Mandarinfish collectors have acknowledged that the fishery was depleted (Sadovy 2001, Sadovy et al. 2001). In order to find mandarinfish, some collectors must travel great distances from home to remote reefs, requiring from five days to three weeks at sea (Gonsales and Savaris 2005, Reksodihardjo-Lilley and Lilley 2007). Collectors say that they must travel great distances because populations of aquarium species on the reefs near their villages are depleted (Reksodihardjo-Lilley and Lilley 2007). Hobbyists have also noted that this species was driven to extinction in some locations and experienced population depletion in others (http://en.microcosmaquariumexplorer.com/wiki/Mandarin_Harvest_Realities).

In addition to reducing population sizes, heavy collection caused major declines in mandarinfish size. Collectors pursue the largest available mandarinfish and shift to any size class of fish as the population declines (Sadovy et al. 2001). In the 1980s, mean mandarinfish length was 60 mm, but by 2000 the average fish length declined to just 30 mm (Sadovy et al. 2001, Wabnitz et al. 2003). Notably, 30 mm is around the size of reproductive maturity for mandarinfish (Sadovy et al. 2001). Decreasing the average size of mandarinfish could decrease the minimum reproductive size and prevent reproduction, thereby destabilizing mandarinfish populations.

Another threat to mandarinfish is hobbyists' preference for males with elaborate dorsal fins (Sadovy et al. 2001, Wabnitz et al. 2003, Rasotto et al. 2010). The fishery targets large males and, as a result, more than 70% of mandarinfish in the supply chain are males (Chan and Sadovy 1998, Sadovy et al. 2001). The removal of so many male mandarinfish has the potential to diminish reproduction (Vincent and Sadovy 1998, Sadovy et al. 2001). Sadovy (2001) highlighted two effects of experimental removal of large males: (1) it resulted in a female biased sex ratio and (2) it caused females to become hesitant to mate with the remaining males, which were smaller in size than those that had been removed. This led to decreased mating success and more time spent searching for mates, thereby increasing the risk of predation (Sadovy 2001, Rasotto et al. 2010). Field researchers examining female-biased mandarinfish populations observed a number of predations, something that is otherwise rarely observed due to the reclusiveness and toxic mucus of this species (Sadovy 2001). Similarly, Rasotto et al. (2010) found that smaller males could not fertilize all of their mates' eggs due to low sperm production and an inability to form a complete anal-fin funnel over the female genital opening. Thus, collection of larger males could reduce female fecundity and future recruitment throughout mandarinfish populations (Rasotto et al. 2010).

Once captured and brought aboard the collection vessel, mandarinfish are packed at high volumes in polyethylene bags (Gonsales and Savaris 2005, Reksodihardjo-Lilley and Lilley 2007). Mandarinfish are not mixed with other species in order to prevent their toxic mucus from poisoning the catch (Gonsales and Savaris 2005). Oftentimes, collectors have few bags, bags are often of the wrong size, and collectors often reuse bags (Reksodihardjo-Lilley and Lilley 2007). Therefore, collectors pack many fish into each bag (regardless of the species collected), causing high rates of fish stress and injury (Reksodihardjo-Lilley and Lilley 2007). Mandarinfish and other species collected for the aquarium trade are often killed when their holding bags burst aboard the collection boat (Gonsales and Savaris 2005). The weight of accumulated bags filled with coral reef fish, as well as nails and splinters in the boat, cause this to happen (Reksodihardjo-Lilley and Lilley 2007). At this stage of the supply chain, mortality and rejection of all collected ornamental reef fish, including mandarinfish, is approximately 10% (Gonsales and Savaris 2005).

Collectors sell fish to middlemen and traders for several dollars per fish (e.g., prices were around $7 U.S. per mandarinfish from 2002 to 2004 at Batasan Island; Gonzales and Savaris 2005). Traders hold mandarinfish in shallow bowls or plastic bags at high densities (i.e., 50 fish in 5 liters of water; Sadovy et al. 2001). The fish appeared stressed under these conditions and traders confirmed that mortalities from capture to shipment were high (Sadovy et al. 2001). Gonzales and Savaris (2005) estimate that aquarium fish mortality for all species, including mandarinfish, is approximately 30% at this stage. They attributed deaths to stress and injury from spearing and transportation, ammonium accumulation in holding tanks, and salinity or temperature fluctuations (Gonzales and Savaris 2005).Traders from Batasan Island exported from 1,800 to 2,400 mandarinfish per month (Sadovy et al. 2001). Gonzales and Savaris (2005) estimated overall aquarium fish mortality (including mandarinfish) along the supply chain to be 90%.

Wild-caught mandarinfish do not acclimate well to home aquaria. Their specialized habitat and diet requirements cause mandarinfish to commonly starve to death in captivity (Wilkerson 1996, Wabnitz et al. 2003, Michael 2005). Mandarinfish require either a specialized diet of live micro-crustaceans or sufficient habitat to support their prey (Michael 2000, 2005). Mandarinfish are also poor competitors for food and hobbyists cannot keep species with similar diets in the same tank as mandarinfish (Michael 2000, 2005). Mandarinfish also require plenty of substrate and hiding places to succeed in captivity (Wabnitz et al. 2003). Hobbyists report that mandarinfish have one of the worst captive survival rates among marine fish in home aquariums (http://en.microcosmaquariumexplorer.com/wiki/Breeding_the_Green_Mandarin). Sadovy et al. (2001) noted the difficulties of keeping mandarinfish and only recommends them for experienced aquarists. A high rate of death in captivity increases demand for more fish to be collected from coral reefs of the Indo-Pacific. This results in additional ecological impacts and mandarinfish deaths along the supply chain.

Mandarinfish conservation and aquaculture:

There has been very little research or reporting on mandarinfish conservation or the efficacy of different management measures for this species. However, recent aquaculture advances suggest that over-collection could be reduced substantially by the availability of captive-bred mandarinfish. For many years mandarinfish were considered too difficult for captive production, yet they remained the subject of active aquaculture research (e.g., Gopakumar 2005). Dragonets are among the most successful fish with a pelagic larval phase to be raised in captivity (Sadovy et al. 2001). In fact, a dragonet was one of the first species to be raised in captivity (Holt 1898) and captive breeding has been successful for many species in this family (Takita 1980, Takita and Okamoto 1979, Takai and Yoshioka 1979, Eda et al. 1994a, 1997, Gonzales et al. 1996).

Mandarinfish and other dragonets have several characteristics that make them a promising species for commercial aquaculture. Most notably, mandarinfish have robust, low-mortality larvae that mature quickly and settle rapidly (Sadovy et al. 2001). The larvae will feed on rotifers, copepods, and crustacean nauplii in captivity (Wilerson 1996, Mai 2000). Scientists and public aquariums report successful captive breeding programs for mandarinfish (Hall and Warmolts 2003, Moe 2003). Recently, these breeding efforts have occurred at a commercial scale. The company Oceans, Reefs, and Aquariums (ORA) offered the first batches of captive-bred mandarinfish available for sale to the general public (http://www.orafarm.com/products/fish/dragonets.html). The most promising development is that ORA’s captive-bred and -raised mandarinfish can be trained to eat a prepared diet, thus overcoming the specific feeding requirements and high starvation rates that confound hobbyists who purchase wild-caught mandarinfish. If mandarinfish aquaculture continues to succeed, and if captive-bred mandarinfish can be sold at a competitive price compared to their wild-caught counterparts, aquaculture would dramatically reduce trade’s negative impacts on mandarinfish.

Author’s note: Until this point, the case studies in this report have examined trade’s impacts on individual species. In the chapters that follow, the report examines how trade affects groups of species, genera, and higher taxonomic ranks of organisms, concluding with an examination of damage to entire coral reef ecosystems. We begin this transition by examining symbiotic anemonefish and their host anemones.

Chapter 5
Giant Anemones and Anemonefish

Introduction to anemonefish biology:

Anemonefish (also known as clownfish) are a group of attractive coral reef fish in the family Pomacentridae (damselfish and sergeants). These charismatic fish are named for their obligate and intimate associations with giant anemones. Anemonefish are well-known to the general public, including starring roles in the 2003 Pixar film *Finding Nemo*. The anemonefish lineage includes 29 nominal species in the genus *Amphiprion*, as well as the maroon anemonefish, *Premnas biaculeatus*. All 30 described species of anemonefish share a common ancestry and no non-anemonefish species are a part of this group (i.e., anemonefish are monophyletic; Quenouille et al. 2004, Santini and Polacco 2006, Cooper et al. 2009). Table 3 lists each species' common and scientific names, as well as its involvement in the coral reef wildlife trade. Their colorful appearance, interesting biology, and success in captivity have made anemonefish extremely popular as reef-aquarium fish.

Anemonefish live on and around coral reefs at depths of 3–20 m (e.g., Mariscal 1970a, Chadwick and Arvedlund 2005, Hattori 2006). They are found throughout the tropical Indo-Pacific, from the east coast of Africa and the Red Sea through the Indian Ocean to the Pacific Islands (Table 3). They range as far north as Tokyo, Japan and south to southeastern Australia. They do not occur in the Atlantic Ocean or Caribbean Sea.

Table 3: Species of anemonefish and symbiotic anemones, their distribution, and occurence in the coral reef wildlife trade. Unless otherwise noted, information based on Fautin and Allen (1992), Shimek (2004), Michael (2005), and Allen et al. (2008, 2010).

Species	Common Name(s)	Distribution	Occur in Trade?
Anemonefish[1]	-	-	-
Amphiprion akallopisos[2]	Skunk anemonefish, Indian Ocean skunk clownfish	Widespread in Indian Ocean, including Madagascar, Comoro Islands, Seychelles, Andaman Islands, west coast of Thailand, and western and southern coasts of Sumatra and Java. It also occurs in the Java Sea.	Yes
A. akindynos	Barrier reef anemonefish	Great Barrier Reef of Australia and adjacent Coral Sea to New Caledonia and the Loyalty Islands.	Yes
A. allardi	Allard's anemonefish	East Africa between Kenya and Durban.	Yes
A. barberi	Barberi clownfish	Central Pacific: Fiji, Tonga and American Samoa	Yes

A. bicinctus	Two-band anemonefish	Red Sea, Gulf of Aden, and Chagos Archipelago	Yes
A. chagosensis	Chagos anemonefish	Chagos Archipelago in the western Indian Ocean	Very Rare
A. chrysogaster	Mauritian anemonefish	Mauritius (western Indian Ocean) and probably Reunion	Very Rare, possibly unavailable
A. chrysopterus	Orange-fin anemonefish	Widespread in the western Pacific including New Guinea, Coral Sea, New Britain, Solomon Islands, Vanuatu, Fiji, Caroline Islands, Mariana Islands, Gilbert Islands, Samoa, Society Islands, and Tuamotu Islands.	Yes
A. clarkii	Clark's anemonefish, yellowtail clownfish	The most widely distributed anemonefish, ranging from the islands of Micronesia and Melanesia in the western Pacific to the Persian Gulf, and from Australia to Japan	Yes
A. ephippium	Red saddleback anemonefish	Andaman and Nicobar Islands, Thailand, Malaysia, Sumatra, and Java	Yes
A. frenatus	Tomato anemonefish	South China Sea and immediately adjacent areas, northwards to Japan	Yes
A. fuscocaudatus	Seychelles anemonefish	Seychelles Islands and Aldabra in the western Indian Ocean	No
A. latezonatus	Wide-band anemonefish	Lord Howe Island off eastern Australia and rocky mainland reefs near the Queensland - New South Wales border	Rare
A. latifasciatus	Madagascar anemonefish	Madagascar and the Comoro Islands in the western Indian Ocean.	Very Rare
A. leucokranos	White-bonnet anemonefish	Northern Papua New Guinea, including Manus Island and New Britain, and the Solomon Islands	Yes
A. mccllochi	McCulloh's anemonefish	Lord Howe Island off New South Wales, Australia, and nearby Norfolk Island	Yes
A. melanopus	Cinnamon clownfish, red and black	Indonesia (Bali westward), Melanesia, Micronesia, southeastern Polynesia, and	Yes

	anemonefish	Great Barrier Reef - Coral Sea	
A. nigripes	Maldives anemonefish	Maldive Islands and Sri Lanka in the central Indian Ocean	Yes
A. ocellaris	False clown anemonefish, ocellaris clownfish	Andaman and Nicobar Islands (Andaman Sea), Indo-Malayan Archipelago, Philippines, northwestern Australia; coast of Southeast Asia northwards to the Ryukyu Islands	Yes
A. omanensis	Oman anemonefish	Oman, Arabian Peninsula	Very rare
A. pacificus	Pacific anemonefish	Wallis Island, Tonga, Fiji and Samoa	Possibly unavailable
A. percula[1]	Clown anemonefish, percula clownfish, orange clownfish	Cocos (Keeling) Islands and Christmas Island in the eastern Indian Ocean, Indo-Australian Archipelago northwards to the Ryukyu Islands, Fiji and Micronesia	Yes
A. perideraion[2]	Pink skunk anemonefish	Cocos (Keeling) Islands and Christmas Island in the eastern Indian Ocean, Indo-Australian Archipelago northwards to the Ryukyu Islands, Fiji and Micronesia	Yes
A. polymnus	Saddleback anemonefish	Indo-Malayan Archipelago northwards to the Ryukyu Islands; also reported from the Northern Territory, Australia	Yes
A. rubrocinctus	Australian anemonefish	Northwestern Australia	Yes
A. sandaracinos[2]	Orange skunk anemonefish, orange anemonefish	Christmas Island and Western Australia in the eastern Indian Ocean, Indonesia, Melanesia, Philippines, and northwards to the Ryukyu Islands	Yes
A. sebae	Sebae anemonefish	Northern Indian Ocean including Java, Sumatra, Andaman Islands, India, Sri Lanka, Maldive Islands, and southern Arabian Peninsula	Yes
A. thiellei	Thielle's anemonefish	Western Central Pacific: described from two aquarium dealer specimen believed to have originated in the vicinity of Cebu, Philippines	Very rare

A. tricinctus	Three-band anemonefish	Marshall Islands in the central-western Pacific Ocean	Yes
Premnas biaculeatus[3]	Maroon anemonefish, Spine-cheek anemonefish	Indo-Malayan Archipelago to northern Queensland.	Yes
Anemones	-	-	-
Cryptodendrum adhaesivum	Adhesive sea anemone	Australia to southern Japan and Polynesia, Micronesia, and Melanesia westward to Thailand, Maldives, and the Red Sea	Rarely
Entacmaea quadricolor	Bubbletip sea anemone, Bubble-tentacle sea anemone	Micronesia and Melanesia to East Africa and the Red Sea, and from Australia to Japan	Yes
Heteractis aurora	Beaded sea anemone	Micronesia and Melanesia to East Africa and the Red Sea, and Australia to the Ryukyu Islands	Yes
H. crispa	Leathery sea anemone	French Polynesia, Micronesia, and Melanesia to the Red Sea, and Australia to Japan	Yes
H. magnifica	Magnificent sea anemone	French Polynesia to East Africa, and Australia to the Ryukyu Islands	Yes
Heteractis malu	Delicate sea anemone	Scattered localities from the Hawaiian Islands to Australia and northwards to Japan	Yes
Macrodactyla doreensis	Corkscrew tentacle sea anemone	Japan south to New Guinea and northern Australia	Yes
Stichodactyla gigantea	Gigantic sea anemone	Micronesia to the Red Sea, and Australia to the Ryukyu Islands	Yes
S. haddoni	Haddon's sea anemone	Fiji Islands to Mauritius, and Australia to the Ryukyu Islands	Yes
S. mertensii	Mertens' sea anemone	Micronesia and Melanesia to East Africa, and Australia to the Ryukyu Islands	Rarely

[1]: Timm et al. (2008) found a possible cryptic species in *A. percula*.

[2]: Steinke et al. (2009) sampled 13 species of *Amphiprion* and found that three species in the subgenus *Phalerebus*, including *A. akallopisos*, *A. perideraion*, and *A. sandaracinos*, shared COI mtDNA sequences. These results indicate that either (1) these nominal species are color morphs of a single variable species or (2) that frequent hybridization

occurs between these three “species.” However, Timm et al. (2008) sampled the same three species and found genetic distinctiveness between the same three *Phalerebus* species based on cyt b and mitochondrial control region DNA. Timm et al. note that the sequence divergence levels were low, reflecting the close relatedness of the three species.
[3]: According to Cooper et al. (2009) the genus *Premnas* will likely be synonymized with *Amphiprion*

The wide distribution and similar appearance of many anemonefish species have led to ambiguous boundaries between some *Amphiprion* species. One problem is the misidentification of regional color morphs of a single species as several different species (e.g., Steinke et al. 2009). The opposite situation also exists; distinct anemonefish species are sometimes identical in appearance (Drew et al. 2008, Timm et al. 2008). Hybridization between taxa adds further challenges (Fautin and Allen 1992, Santini and Polacco 2006, but see Ollerton et al. 2007). Clearly, anemonefish are a diverse and complex group that requires additional study. Taken together, this situation is confusing for any scientist, manager, or customs official that depends on accurate species identifications to do their job successfully.

Despite the complexity of anemonefish evolution and taxonomy, it is still possible to make certain generalizations about their biology. As adults, anemonefish are entirely dependent upon giant anemones for their survival (Cleveland et al. 2011). The fish are relatively defenseless and would be eaten by predators without the refuge and protection provided by an anemone host (Fricke and Fricke 1977, Fautin 1991). Anemone tentacles and epidermis contain stinging cells, known as nematocysts, which effectively protect anemonefish from predators. Remarkably, anemonefish are not harmed by the host’s nematocysts (Elliott and Mariscal 1997a,b). The fish are protected by a coating of mucus that acts as a “chemical camouflage” and prevents the anemone from stinging (Mebs 1994, 2009). Because of obligate dependence on anemones, the abundance of anemonefish on the reef is limited by anemone availability (reviewed in Pinsky et al. 2010). The specificity of the relationship also varies, with some anemonefish able to associate with any giant anemone, whereas others are restricted to very few species of host (Fautin and Allen 1992, Table 4).

Although anemonefish require a host anemone for survival, giant anemones and zooxanthellae can live without ectosymbiotic fish (Mebs 2009). Nevertheless, hosting anemonefish provides an anemone with nutrients, protection, and cleaning, which increase growth and survival. The crystal-clear, low-nutrient waters of coral reefs are often nitrogen limited (Muscatine and Porter 1977). When anemonefish defecate near their giant anemone hosts, it transfers carbon, nitrogen, and a small amount of phosphorous to the anemone (Roopin et al. 2008, 2011, Godinot and Chadwick 2009, Cleveland et al. 2011). This recycling of nutrients between symbiotic partners enables anemones to grow larger, regenerate faster, and produce more biomass than would otherwise be possible on nutrient-limited reefs (Porat and Chadwick-Furman 2005, Roopin and Chadwick 2009, Cleveland et al. 2011). Anemonefish extend other benefits to giant anemones by rigorously defending their hosts from butterflyfish and other predators (Fautin 1991, Godwin and Fautin 1992, Porat and Chadwick-Furman 2004).

Table 4: Host anemone and ectosymbiont anemonefish symbiotic associations. Table based on Fautin and Allen (1992), Allen et al. (2008, 2010).

Species (Anemone [column] vs. Fish [row])	*C. adhaesivum*	*E. quadricolor*	*H. aurora*	*H. crispa*	*H. magnifica*	*H. malu*	*M. doreensis*	*S. gigantea*	*S. haddoni*	*S. mertensii*
A. akallopisos					X					X
A. akindynos		X	X	X	X			X[1]	X	X
A. allardi		X	X							X
A. barberi		X		X						
A. bicinctus		X	X	X	X			X		
A. chagosensis[2]										
A. chrysogaster			X		X		X		X	X
A. chrysopterus		X	X	X	X				X	X
A. clarkii	X	X	X	X	X	X	X	X	X	X
A. ephippium		X		X						
A. frenatus		X								
A. fuscocaudatus										X
A. latezonatus				X						
A. latifasciatus										X
A. leucokranos				X	X					X
A. mccllochi		X								
A. melanopus		X[3]		X[4]	X[5]					

A. nigripes					X					
A. ocellaris					X			X		X
A. omanensis		X		X						
A. pacificus					X					
A. percula				X	X			X		
A. perideraion				X	X[3]		X	X		
A. polymnus				X					X	
A. rubrocinctus		X[3]						X		
A. sandaracinos				X						X
A. sebae									X	
A. thiellei[2]	?	?	?	?	?	?	?	?	?	?
A. tricinctus		X	X	X						X
P. biaculeatus		X								

[1]: Conflicting reports in Fautin and Allen (1992) about whether or not this host-ectosymbiont association occurs
[2]: Host-anemone affiliation unknown
[3]: Typical association
[4]: Occasional association
[5]: Rare association

Finally, the fish mix water around their anemones and remove sediments, mucus, and necrotic tissue (Liberman et al. 1995, Goldshmid et al. 2004, Arvedlund et al. 2006, Stewart et al. 2006). All of these services improve the anemone's tentacle extension, survivorship, growth, and reproduction (Schmitt and Holbrook 2003, Porat and Chadwick-Furman 2004, 2005, Holbrook and Schmitt 2005).

Anemonefish are omnivorous fish that eat planktonic algae, copepods, eggs, larvaceans, tunicates, isopods and other small crustaceans, fish scales, and mollusks (Galetto and Bellwood 1994, Frédérich et al. 2009). Anemonefish also opportunistically consume the remnants of whatever food their host anemone has captured. During the day these fish actively feed on plankton, returning to their host anemones at night. The percentage of time spent away from the host varies by species (reviewed in Cleveland et al. 2011), but they rarely migrate further than several hundred meters as adults (Hattori 2005).

Anemonefish live in small groups of unrelated individuals that typically associate with a single anemone (Buston et al. 2007). Members of the group communicate with each other by producing sounds that include "pops" and "clicks" (Parmenteir et al. 2007). These vocalizations are used in agonistic communication and sharing information about group members' reproductive status (Parmenteir et al. 2007, Colleye et al. 2009). At least one species, *A. akallopisos*, has developed different dialects of vocalizations in different regions of the world (Parmenteir et al. 2005).

An anemonefish group consists of a male and female breeding pair and up to four subordinate non-breeders (Buston et al. 2007). A reproductive pair of anemonefish is monogamous until one partner dies or departs for another anemone (reviewed in Whiteman and Côté 2004). The group size is correlated with the size of the anemone host. If the group grows too large, the current residents will forcefully eject new recruits (Buston 2003b). Non-breeding anemonefish do not directly assist the breeders in reproduction (Buston 2004a), but the presence of subordinates enhances anemone growth and survival, which indirectly increases the breeding fishes' reproductive success (Fricke 1979, Buston 2002, 2004a).

One of the most remarkable features of anemonefish is that individuals change sex over the course of their adult lives, a phenomenon known as sequential or protandrous hermaphrodism. An individual's age and social rank in the group determines its sex; the oldest, largest, and highest-ranking member of the group is the reproductive female, the second-ranked individual is the breeding male, and the younger, smaller, and subordinate individuals are non-reproductive (Buston and Cant 2006, Iwata et al. 2008, 2010). After their larval stage, juveniles begin as non-reproductive subordinate fish (Godwin 1994). If a breeding male fish departs or dies, a subordinate anemonefish will transform into a reproductive adult male (Godwin 1994). If the adult female is removed, the breeding adult male changes into the reproductive female and a non-breeder becomes the reproductive male (Godwin 1994).

Protandrous hermaphrodism likely evolved in response to limited habitat and mate availability (Fishelson 1998, Whiteman and Côté 2004). The ability to change sex ensures the presence of a suitable mate in an isolated group with low recruitment (Whiteman and Côté 2004). Although subordinate individuals must wait to reproduce, they avoid the risks of injury or death from dispersing to another site or antagonistically contesting their position in the group (Buston 2004b). Subordinates also have guaranteed future reproductive opportunities if they survive long enough to replace one of the breeders (Buston 2004b).

Anemonefish spawn a few times each year on a lunar cycle (Allen 1975, Ross 1978, Thresher 1984, Fautin and Allen 1992, Richardson et al. 1997a). The group's male prepares a nest site adjacent to the anemone where the female lays several hundred to several thousand demersal eggs (the precise number varies by species; Allen 1980, Thresher et al. 1989, Richardson et al. 1997a, Yasir and Qin 2007). The male fish actively tends and defends eggs, fanning them to keep them well-oxygenated and removing dead eggs with his mouth (Tyler 1995, Green and McCormick 2005a,b). After about 6 to 7 days, anemonefish eggs hatch and release pelagic larvae (Thresher et al. 1989, Yasir and Qin 2007).

Adult anemonefish are very site attached, but larval anemonefish have high dispersal ability (reviewed in Timm and Kochzius 2008). Larvae disperse for 7 to 22 days, with the duration depending on environmental conditions and the population or species of anemonefish (Thresher et al. 1989, Wellington and Victor 1989, Bay et al. 2006, Almany et al. 2007). Anemonefish larvae can swim immediately upon hatching and their swimming ability continues to improve with age (Fisher et al. 2000). Swimming distance is enhanced by feeding (Fisher and Bellwood 2001). Anemonefish larvae are born with well-developed, acute binocular vision for prey location and the ability to capture and ingest prey by suction feeding (Coughlin 1993, 1994). The larvae can swim for long periods of time and influence the trajectory of their dispersal (Fisher and Bellwood 2002). In one example the maximum swimming distance of *A. melanopus* larvae was estimated at 28.7 km (Fisher and Bellwood 2001).

Anemonefish larvae use a number of homing cues to locate a suitable coral reef. The larvae are born with an innate ability to locate a reef by following chemical cues from anemones, rainforest vegetation, and reef water (Murata et al. 1986, Dixson et al. 2008). The fish also have sensitive hearing and juveniles likely swim towards sounds made by adults in order to find a home on the reef (Parmentier et al. 2009). The fish may also imprint on environmental cues that they perceived while developing in the egg or right after hatching (Arvedlund et al. 1999, 2000a,b, Simpson et al. 2005).

Mortality is high in larval anemonefish, with most individuals perishing before recruiting to a reef (Thresher et al. 1989, Yasir and Qin 2007). For example, *A. akindynos* in the Great Barrier Reef had just 0.37 ± 0.27 (mean ± SD) recruits per $100m^{-2}$ of reef (Sale et al. 1986). Of the surviving larvae, many (i.e., 30–60% of recruits) return to settle on the reef where they were born (Jones et al. 2005, Almany et al. 2007, Planes et al. 2009, Saenz-Agudelo et al. 2009). The high local recruitment presumably results from the imprinting and homing abilities described above.

Several studies tracked the dispersal distance and recruitment potential of anemonefish larvae in the context of species management and marine protected areas. For instance, Pinsky et al. (2010) measured the dispersal of *A. clarkii* across the central islands of the Philippines. They found that, on average, larvae dispersed 11 km per generation, indicating that a dense network of closely-connected small marine protected areas, with each marine protected area less than 10 km from the next site, was necessary to sustain populations (Pinsky et al. 2010). As a result, additional marine protected area sites would be needed to manage *A. clarkii*. In contrast, Planes et al. (2009) found that although most (33–43%) *A. percula* larvae from Papua New Guinea returned to their natal reef, up to 10% were long-distance migrants that moved up to 35 km per generation. Planes et al. compared the *A. percula* dispersal distances with the spatial arrangement of a marine protected area network and concluded that the marine protected area system provided both replenishment (self-recruitment) and dispersal of individuals to other locations in the network (Planes et al. 2009). Finally, Timm and Kochzius (2008) examined *A. ocellaris* across the Indo-Malay Archipelago in the Coral Triangle. They found population breaks by region, with the 1) Indian Ocean and Java Sea, 2) Central Indo-Malay Archipelago, 3) southwestern coast of New Guinea, and 4) Batam, Indonesia each constituting a separate genetic group (Timm and Kochzius 2008).Therefore, larval dispersal was not sufficiently connecting the four genetic groups of *A. ocellaris* and each population should therefore be managed separately (Timm and Kochzius 2008). In all of three cases, knowing the relevant level(s) of dispersal and genetic connectivity are important considerations for marine protected area design.

Anemonefish live up to 30 years in the wild (Buston and Garcia 2006). As adults, anemonefish have relatively low morality (~14% per year in *A. percula*), with the lower social rank fish facing the greatest mortality risk (Buston 2003a).

Introduction to giant anemone biology:

Although anemonefish have been the subject of considerable scientific research, less is known about the population dynamics or biology of giant anemones. All across the Indo-Pacific, anemonefish associate with at least ten species of large anemones (Table 3). These anemone hosts are not part of one closely-related or monophyletic group; instead the ten species are scattered across three unrelated families in the order Actinaria (Cnidaria: Anthozoa; Dunn 1981, Fautin and Allen 1992, Elliott et al. 1999). For each species, the name, geographic distribution, and involvement in trade are described in Table 3.

Giant anemones occur on shallow coral reefs and associated habitats. They are commonly found in sea grass beds, rocky areas, and coral reefs (e.g., Hattori and Kobayashi 2009), doing best in areas with hard substrates and moderate wave action (Mariscal 1970b, Richardson et al. 1997b). Giant anemones are typically poor competitors with corals and as a result, their abundance is highest in rocky areas adjacent to reefs (reviewed in Scott et al. 2011). They attach to hard substrates for prolonged periods, but are also capable of changing location over time. For example, the anemone *S. gigantea* moves around and takes its ectosymbiont fish, *A. ocellaris*, with it as it moves (Mitchell 2003).

Giant anemones commonly form facultative, mutualistic symbioses with anemonefish and other ectosymbionts (e.g., symbiotic shrimp). As reviewed above, harboring fish or crustaceans enhances the growth and reproductive success for giant anemones while providing protection and habitat for the ectosymbiont. However, giant anemone symbioses are more complex than simple fish/crustacean and anemone relationships. These actinarians also harbor intracellular dinoflagellates (genus *Symbiodinium*) within their gastrovascular tissues (Ollerton et al. 2007, Cleveland et al. 2011). *Symbidiodinium* (sometimes known as zooxanthellae) are a diverse group of single-celled, photosynthetic protists (Coffroth and Santos 2005). These dinoflagellates supply their hosts with photosynthetically fixed carbon and energy that supplies the host's metabolic needs (e.g., Muscatine et al. 1981). In return, the hosts provide a stable home and a steady supply of nutrients (which are enhanced by the presence of ectosymbiotic fish).

Giant anemones are slow growing and are presumed to be very long lived (Fautin 1991). As such, these anemones are negatively affected by disturbance and do better in areas protected from storm swell (Richardson et al. 1997b). Like corals, giant anemones are sensitive to high temperature stress. High temperatures lead to a dissociation of the *Symbiodinium* from the anemone, causing starvation and possibly the death of the anemone and associated anemonefish (Jones et al. 2008). For example, bleaching was strongly indicated as a cause of anemone and anemonefish decline in the Keppel Islands on the southern Great Barrier Reef (Jones et al. 2008).

Like all cnidarians, giant anemones possess stinging cells known as nematocysts (reviewed in Mebs 2009). The cells are concentrated on the tentacles of the anemone where they aid in both defense and feeding. Nematocysts function like microscopic harpoons or needles that mechanically sting prey and then inject toxins into it (reviewed in Mebs 2009). Nematocyst toxins are approximately 20 kDa peptides that cause pain, loss of muscular coordination, paralysis, and tissue damage (Mebs 1994, 2009, Ravindran et al. 2010). Giant anemones also have a mucus coating on their bodies that contains cytolytic poisons, compounds which are lethal at dilute concentrations to most fish (Mebs 1994, 2009). Anemone cytolytic poisons strip the mucus coating of fish gills and perforate tissue, thereby ruining the proper function of the gills (Mebs 2009). Remarkably, anemonefish have evolved a mechanism to avoid this damage and to prevent nematocysts from firing (Mebs 1994).

Like most aspects of their biology, relatively little is known about reproduction in giant anemones. For at least two species, *Entacmaea quadricolor* and *Heteractis crispa*, male and female anemones synchronously broadcast spawn sperm and eggs into the water column a few nights each year (Scott and Harrison 2007a, 2009). *Entacmaea quadricolor* also reproduces asexually through longitudinal fission (Dunn 1981, Fautin 1986). Nothing is known about the reproductive mode of other giant anemone species, but it is likely similar to that of *E. quardicolor* and *H. crispa*. After spawning, sperm and eggs fuse to form a ciliated planula larva which becomes motile within 36 hours (Scott and Harrison 2007b). The larvae disperse for 4 to 12 days (Scott and Harrison 2007b, 2008), with relatively few larvae surviving to reach adulthood (Fautin 1991). Dispersal distances and mortality rates have not been examined for any giant anemone species.

Collection and overharvesting of anemonefish and giant anemones:

Anemonefish are extremely popular in the ornamental aquarium trade and their popularity has led to early efforts at captive breeding (Dawes 2003, Green 2003). As a result, anemonefish were among the first coral reef fishes raised in captivity throughout their entire life cycle and now represent one of the most well-known and well-developed captive breeding programs for marine fishes (Dawes 2003). Despite the successes of anemonefish aquaculture, these fish and their giant anemone hosts are still primarily collected from the wild (Wabnitz et al. 2003). For example, in the western Pacific Ocean, anemonefish constitute two out of the top ten exported aquarium fish (including *A. ocellaris* at 4.9% of exports and *A. percula* at 3.0% of exports; Green 2003). *Amphiprion* *ocellaris*, *A. percula*, and *P. baculeatus* are also among the top 20 marine aquarium fish imported into the U.S. (Rhyne et al. 2012). For giant anemones, *H. malau* is the second most exported invertebrate species in the West Pacific (representing 9.3% of invertebrate exports; Green 2003). In some areas anemones and anemonefish dominate the trade. For example, anemones and anemonefish comprise 60% of the aquarium organisms collected in Cebu, the Philippines (Shuman et al. 2005).

Collection of anemonefish and giant anemones involves many different species and collection locations scattered across myriad nations throughout the Indo-Pacific (Table 3). Therefore, it is challenging to characterize the specific collection practices and source locations without over-generalizing. However, Edwards and Shepherd (1992) and Saleem and Islam (2008) provided comprehensive overviews of the coral reef wildlife trade in the Maldives (including information on anemone and anemonefish collection); that fishery will be reviewed here as an example of the structure of the anemonefish trade.

Trade in ornamental coral reef fishes began in the Maldives in 1980 (Edwards and Shepard 1992). Early trade was located around the capital island Malé (Edwards and Shepherd 1992) until the construction of regional airports made collection possible in other areas (Saleem and Islam 2008). Collectors were initially brought in from Sri Lanka, but today, collection is conducted primarily by locals (Saleem and Islam 2008). The industry has steadily grown over the years, from two exporting companies and 25 employees in 1988 to seven companies and approximately 90 people by 2007 (Edwards and Shepherd 1992, Saleem and Islam 2008). As of 2007, the trade involved the export of 358,378 fish per year, earning $590,530 U.S. in total revenue (Saleem and Islam 2008). A total of 140 fish species and 5 invertebrate species are traded, with no trade in stony corals allowed (Saleem and Islam 2008). Anemones and anemonefish comprise a significant portion of trade, with three species of giant anemones being among the most commonly exported invertebrates (Edwards and Shepherd 1992). Fish and invertebrates are exported primarily to Europe, as well as Sri Lanka and the U.S. (Edwards and Shepherd 1992, Saleem and Islam 2008).

In the Maldives, collectors harvest fish using small hand-nets; cyanide, moxy nets that damage corals, or other destructive fishing practices are illegal (Edwards and Shepherd 1992, Saleem and Adam 2004). Furthermore, collectors and exporters use relatively sanitary holding facilities, as well as acclimation procedures and practices that minimize mortality (Edwards and Shepherd 1992). Conflicts with the fishing industry are minimal because there is little overlap between aquarium species and fish species harvested as food (Edwards and Shepherd 1992). However, there are conflicts between dive tour operators and the aquarium trade (Saleem and Adam 2004).

In the early years, aquarium fish and invertebrate species were managed under a general quota that allowed 100,000 animals of any species to be exported each year (Edwards and Shepherd 1992). However, concerns about the sustainability of coral reef wildlife collection (Edwards and Shepherd 1992) led to a three-tiered system of management (Saleem and Islam 2008). Tier A includes 17 species where harvesting is prohibited (Saleem and Islam 2008). The second category, Tier B, includes 66 species managed under a species-specific export quota (Saleem and Islam 2008). Clownfish (*Amphiprion*) species are listed under Tier B. The remaining 71 species are included in Tier C, which are not managed on a species by species basis (Saleem and Islam 2008).

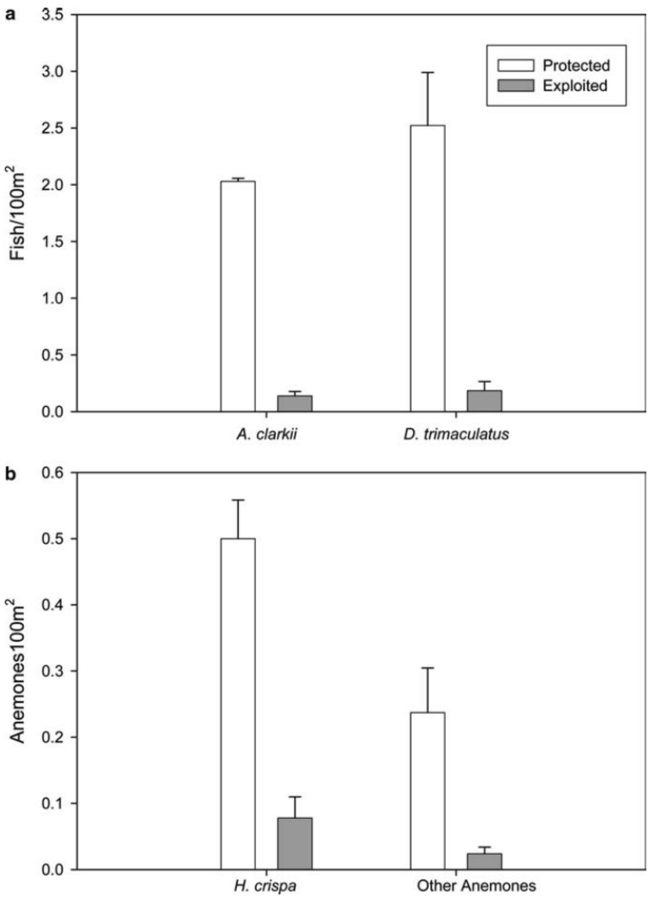
Instead, collection is done under an aggregate quota system, with a combined total of 300,000 fish and invertebrates allowed to be exported each year (Saleem and Islam 2008).

Although the aquarium fish trade in the Maldives avoids many of the worst practices associated with trade, there are still causes for concern about its ecological sustainability. As part of their assessment, Edwards and Shepherd (1992) examined export data, investigated practices at an exporters holding facility, and conducted field surveys to measure the abundance of reef fish species in the Maldives. Based on this information, they made very rough calculations of maximum sustainable yield of 65 ornamental fish species, including three anemonefish. Their analysis identified 27 species of concern as being potentially over-exploited, with 12 species showing evidence of over-exploitation. Two anemonefish species, *A. clarkii* and *A. nigripes*, were identified as species of special concern (Edwards and Shepherd 1992). These two anemonefish are heavily exploited, but occur at low abundance (Edwards and Shepherd 1992). Furthermore, their dependence on anemones makes anemonefish easy to find and harvest (Edwards and Shepherd 1992). The authors highlighted how the unique biology of anemonefish likely renders them unusually susceptible to over-collection (Edwards and Shepherd 1992).

Beyond anemonefish and anemones, there was further cause for concern about the sustainability of the Maldives' coral reef wildlife trade. About 20% of collected species comprised 70% of the volume of marine ornamental exports, indicating a failure of the quota system to limit collection for the most popular species (Edwards and Shepherd 1992, Saleem and Adam 2004). Furthermore, certain species, including the poison goby and long nose filefish, have disappeared entirely from the coral reefs of the Maldives due to bleaching events and heavy collection (Saleem and Islam 2008). There is also evidence that collectors harvest an area heavily until stocks decline, at which time they move on to a new collection site (Saleem and Islam 2008). Taken together, this suggests that over-collection has significantly impacted the coral reefs of the Maldives.

Investigations of the trade's impacts from other regions of the world also found significant population declines in giant anemones and anemonefish. On the reefs of Cebu, the Philippines, Shuman et al. (2005) analyzed catch records and conducted field surveys that compared collection sites to protected areas in order to examine the impact of trade on giant anemones and anemonefish. The fish examined include *Amphiprion* spp. and the three-spot Dascyllus, *Dascyllus trimaculatus* (a species of damselfish that behaves somewhat like anemonefish by associating with anemones during their juvenile life stage; Fautin and Allen 1992). Shuman et al. (2005) encountered significant and dramatic declines in both anemones and anemonefish in exploited areas when compared to protected zones (Figure 9). Furthermore, *A. clarkii* and giant anemones were significantly larger in the protected area compared to the exploited sites (Shuman et al. 2005, Figures 10–11). The number of fish per anemone was significantly higher in the protected zone (Shuman et al. 2005). Shuman et al. (2005) attributed the declines in fish and anemone numbers, size, and numbers of fish per anemone in the exploited areas directly to collection for the ornamental trade. Other than fishing of large *D. trimaculatus*, there was no other extraction of these organisms and no other ecological problems that could be linked to the decline (Shuman et al. 2005).

Figure 9: Declines in anemonefish (a) and giant anemone density in exploited areas, as compared to control protected sites. Figure taken from Shuman et al. 2005.

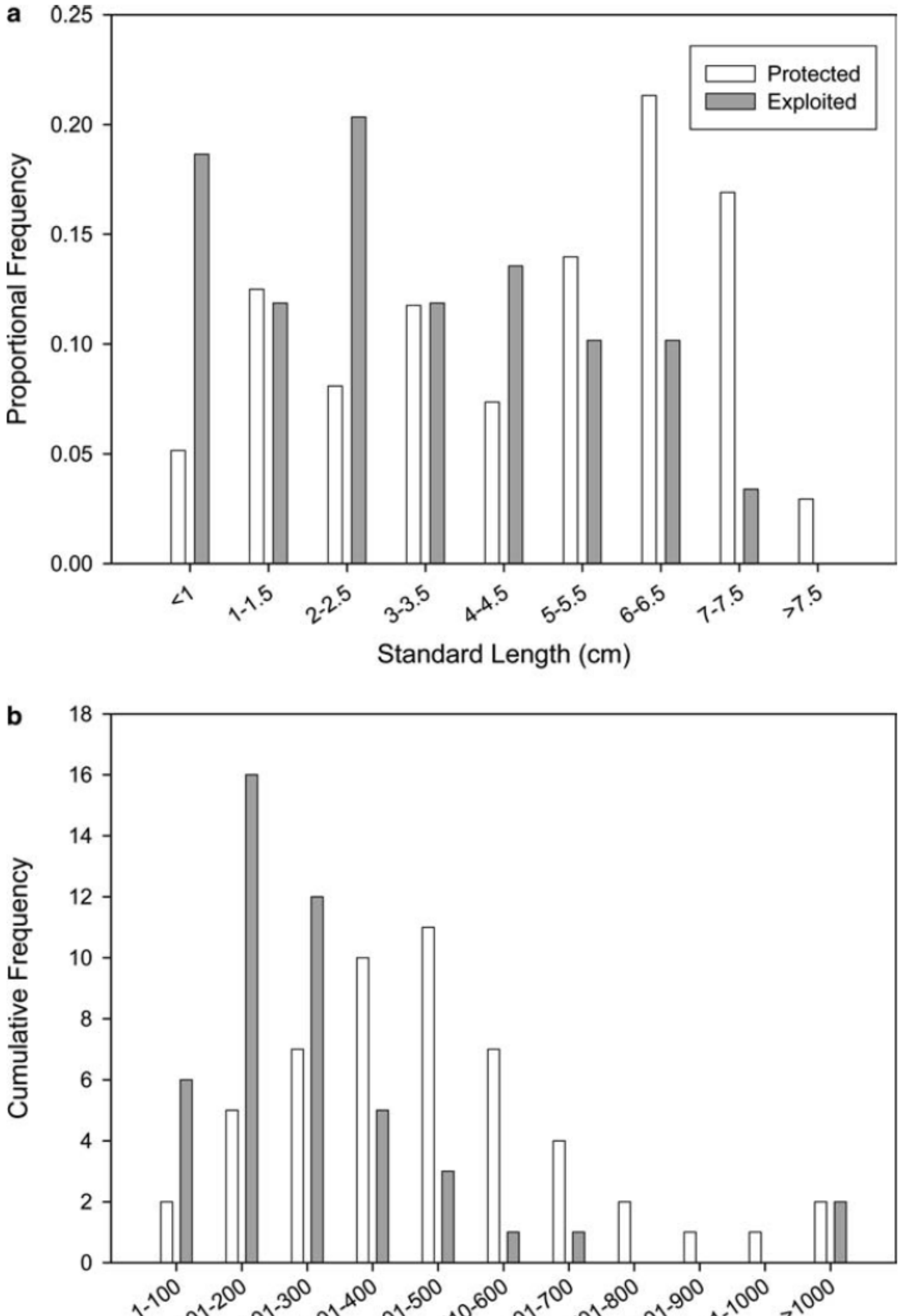


In peninsular Malaysia and Singapore, Sin et al. (1994) noted severely depleted populations of anemonefish. The declines were so severe that some populations may be so depleted that they are unrecoverable (known as an “Allee effect”; Sin et al. 1994). A comparison between surveys conducted by Sin et al. with the work of de Beaufort (1940) identified ten species of damselfishes that were now locally extinct, possibly as a result of collection for the ornamental industry. The authors noted the rapid expansion of the ornamental aquarium market in Singapore from 1968 to 1979 as a possible cause of the change in fish abundance (Sin et al. 1994).

Several thousand miles away, trade has also impacted anemone and anemonefish populations on the Great Barrier Reef. In the Keppel Islands region of the southern Great Barrier Reef, Australia, anemonefish catch per unit effort declined by almost 50% from 2000 to 2004 (Jones et al. 2008 citing Department of Primary Industries and Fisheries, Queensland, unpublished data). Few fisheries data are available for anemones, but the catch of anemones and corallimorphs (another cnidarian group) declined from 407 specimens per vessel in 2004 to 96 specimens per vessel in 2006 (Jones et al. 2008). Taken together these data suggest that the abundance of anemones and anemonefish had declined in the Keppel Islands.

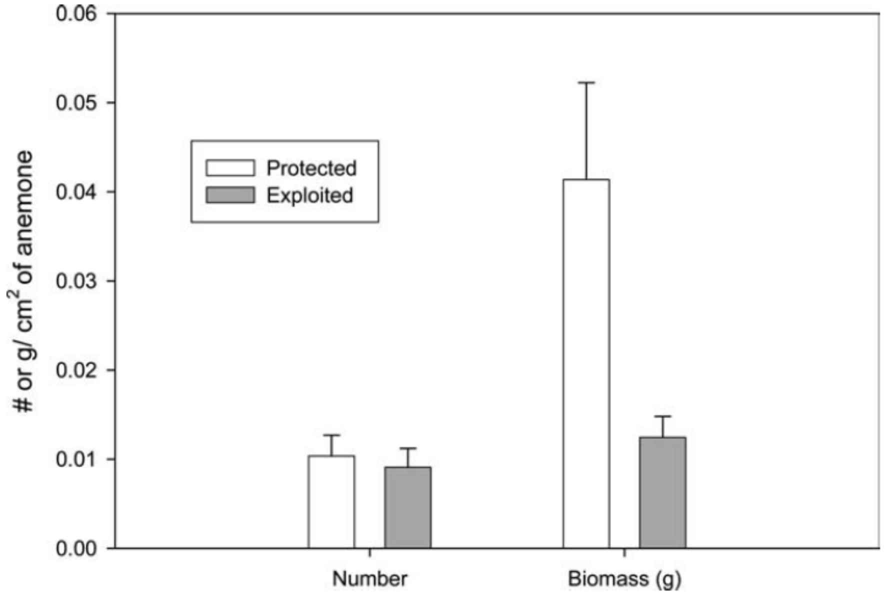
Jones et al. (2008) tested whether collection had caused anemone/anemonefish population declines by comparing population abundances of animals to the management status of different areas (i.e., open vs. closed to collection) and bleaching history of that area. The species examined included the anemonefish *A. melanopus* and *D. aruanus* (two sightings; another *Dascyllus* species which can associate with anemones as young fish) as well as the anemones *E. quadricolor* and *H. crispa* (one recording). Importantly, no anemones or anemonefish were found on reefs in the Keppel Islands that were both bleached and subjected to collection for the aquarium trade (Jones et al. 2008). By contrast, anemones and anemonefish were still present on bleached reefs subject to protection, indicating that collection played a significant role in population declines in the Keppel Islands region (Jones et al. 2008).

Figure 10: The size distribution of *A. clarkii* (a) and all anemonefish (b) in protected and exploited sites. Figure taken from Shuman et al. (2005).



Jones et al. (2008) also examined anemonefish and anemone populations in far North Queensland, another region of the Great Barrier Reef. Here the diversity of animals was higher, including the anemonefish *A. akindynos*, *A. melanopus*, *A. perideraion*, *P. biaculeatus,* and *D. trimaculatus* as well as the anemones *E. quadricolor*, *H. crispa*, and *S. mertensii*. For both anemones and anemonefish, protected sites contained higher densities of animals (Jones et al. 2008). Most anemonefish (86%) were found on unbleached sites and a slim majority (51%) occurred in sites closed to fishing (Jones et al. 2008). With the exception of one reef open to collection, the highest densities of anemonefish and anemones were found on reefs protected from harvesting by aquarium collectors (Jones et al. 2008). Taken together, the results from the Keppel Islands and far North Queensland suggest that collection for the aquarium trade had caused population declines in anemones and anemonefish as well as compounded the impacts of coral bleaching (Jones et al. 2008).

Figure 11: Number and biomass of the giant anemone, *H. crispa*, inside exploited and protected areas. Figure taken from Shuman et al. (2005).



In Australia, breeding adult anemonefish are targeted by collectors, with some sub-adults left on the reef (Jones et al. 2008). Unfortunately, this may be exactly the wrong strategy; targeting the fecund and long-lived adults that insure future recruitment and leaving the individuals with naturally higher mortality on the reef can exacerbate population declines and stymie recover (Jones et al. 2008). Consistent with this idea, there is at least one report of anemonefish removal halting future recruitment (Sale et al. 1986).

Giant anemones and anemonefish have several biological characteristics that render these species particularly susceptible to over-collection. Anemonefish have highly-specialized habitat requirements, limited availability of anemones, long life spans, slow growth rates, and low recruitment ability, all of which are poor characteristics for a commercially harvested fish (Ollerton et al. 2007). Based on the information available, it appears that giant anemones also have low reproductive outputs and slow growth rates, which could slow recovery from collection (Scott et al. 2007a). Jones et al. (2008) described how the slow reproductive rate of anemones and anemonefish, combined with the symbiosis, reduces the ability of these species to recover from population declines (Jones et al. 2008). Symbiosis makes both partners vulnerable; when fish are removed from anemones, the anemone may be eaten by predators within several hours (Godwin and Fautin 1992) and anemonefish cannot survive without an anemone host. Population declines in one symbiotic partner can lead to population declines in the other partner, potentially creating a positive feedback cycle leading to populations dropping below the minimum necessary density for successful reproduction (i.e., an Allee effect) and ultimately leading to localized extinctions (Jones et al. 2008).

The ecological impacts of over-exploitation on the reef are exacerbated by high supply-chain mortality in anemonefish. When fish die between collection and arrival in a home aquarium, additional fish must be removed from the wild in order to replace those that were lost and to satisfy demand. The supply-chain mortality rates of anemonefish have been investigated in several studies. Chow et al. (1994) examined the physiological response of the false clown anemonefish, *A. ocellaris*, to transportation conditions. Within two days of collection, 40% of the anemonefish were dead (Chow et al. 1994). The authors examined the tolerance of anemonefish to various conditions within the shipping container: temperature, dissolved oxygen concentration, pH, dissolved carbon dioxide concentration, and ammonia concentration (Chow et al. 1994). False clown anemonefish were sensitive to large and sudden changes in temperature; the fish had to be maintained between 24 and 32 °C or they would become stressed and possibly die (Chow et al. 1994). Sudden temperature fluctuations are a common cause of death during transportation in this and many other aquarium fish species (Chow et al. 1994). The transportation bags containing anemonefish remained well oxygenated over the two day period, but as water quality diminished, carbon dioxide, ammonia, and hydrogen ions accumulated in the bag over time (Chow et al. 1994). None of the chemicals measured reached concentrations exceeding the species' median tolerance limits (the point at which 50% of the fish would die). However, the combined stress of changing chemical and temperature conditions in the bags caused 40% of the anemonefish to perish (Chow et al. 1994). This high rate of mortality during collection and shipping contrasts sharply with an approximately 14% annual mortality rate in the wild for *A. percula* (Buston 2003a).

Condylactis gigantea

Condylactis gigantean is another species of large anemone that is commonly collected for the aquarium trade in the Atlantic Ocean and Caribbean Sea. It occurs from 1 to 30 meters deep on hard bottom environments around coral reefs (Chiappone et al. 2001).This species does not harbor anemonefish in the wild, but it does naturally associate with crustacean ectosymbionts (e.g., *Mithraculus sculptu*, *Periclimenes* spp.) and will accept anemonefish in captivity (Silbiger and Childress 2008, Mebs 2009). As with giant anemones, *C. gigantea* forms symbiotic relationships with *Symbiodinium* (Loram et al. 2007). *Condylactis gigantea* are probably long lived, but they have low fecundity, spawn only sporadically, and produce planktonic larvae with low survival (Jennison 1981, Chiappone et al. 2001). These factors make them susceptible to over-collection (Jennison 1981, Chiappone et al. 2001).

Gasparini et al. (2005) documented the ornamental trade in Brazil where *C. gigantea* is under heavy collection. For two decades, *C. gigantea* was collected from the Arraial do Cabo region near Rio de Janeiro (Gasparini et al. 2005). Before 1990, the species occurred at densities of 1–2 anemones per 10–15 m^2 (Gasparini et al. 2005). Harvest of this species peaked in the early 1990s, with 100 anemones collected per day (Gasparini et al. 2005). The fishery then collapsed and most collectors moved on to Espírito Santo State (Gasparini et al. 2005). The last *C. gigantea* individual was collected in 2003 and no recovery has been reported since. Not a single *C. gigantea* could be found at Arraial do Cabo (Gasparini et al. 2005).

Condylactis gigantea is also harvested for the aquarium trade in the Florida Keys, U.S. Chiappone et al. (2001) compared catch records from the 1990s and to field survey data. The catch record data indicated a trend of increasing landings and volume over time, peaking at 11.8 million anemones landed from 1997–1999 (Chiappone et al. 2001). However, surveys of 134 sites in Florida Keys spread over 250 km found a total of 15 anemones (Chiappone et al. 2001). No anemones were found at 92% of the sites surveyed and the maximum density was just 0.038 anemones per square meter. Although the data were insufficient to attribute these low numbers to any particular cause, the authors did suggest that the history of heavy exploitation may have caused *C. gigantea* populations to decline (Chiappone et al. 2001).

More recently, Rhyne et al. (2009) conducted a comprehensive survey of ornamental coral reef invertebrates that are collected from the Florida Keys. There was a dramatic increase in the collection of ornamental invertebrates, including *C. gigantea*, from 1994 to 2007, with much of the collection concentrated on a small number of species (Rhyne et al. 2009). During this time the catch of *C. gigantea* declined precipitously: 227,328 anemones were harvested in 1994, compared to just 91,737 in 2007 (Rhyne et al. 2009). The declining catch could not be attributed to change in demand or restrictions on fishing; instead it was caused by increasing rarity due to over harvesting for the ornamental coral reef wildlife trade (Rhyne et al. 2009). Although collection is restricted to a limited number of license holders, there are no limits on how many anemones each collector can harvest, leading to overharvesting (Rhyne et al. 2009).

The study by Chow et al. (1994) is the best examination available for the causes and rates of anemonefish mortality during transportation. There are other reports of mortality at early stages in the supply chain. For instance, one export facility in Indonesia experienced 100% mortality among clownfish in the sub-family Amphiprioninae that had been in the stock system for more than four days (Schmit and Kunzmann 2005). In this case the cause of death was an outbreak of rapidly-spreading *Brooklynella hostilis* infections (Schmit and Kunzmann 2005). Diseases like *B. hostilis* are common causes of anemonefish mortality in holding tanks, import/export facilities, and aquariums, especially when fish are not well quarantined (e.g., Nelson and Ghiorse 1999).

In addition to over-collection from the coral reef wildlife trade, giant anemones and anemonefish face a number of threats to their long-term survival. Bleaching resulting from elevated water temperatures has been strongly indicated as a cause of anemone and anemonefish decline (Jones et al. 2008), including localized extinctions (Hattori 2002). The death of a giant anemone from bleaching forces the resident anemonefish to move great distances over the reef in search of a new host, which exposes the fish to predators (Hattori 2005). Loss of anemones from bleaching has also forced at least one Clark’s anemonefish, *A.*

clarkii, into an unusual symbiosis with a soft coral (Arvedlund and Takemura 2006). This unusual host association was likely a result of coral bleaching eliminating the availability of anemone hosts (Arvedlund and Takemura 2006). However, *A. clarkii* is one of the most symbiotically flexible anemonefish species (Table 4) and most species will not be able to respond in this way.

In addition to bleaching, higher ocean temperatures from climate change increases the growth rate of clownfish (i.e., *A. melanopus*), but decreased the swimming ability at settlement and pelagic larval duration (Green and Fisher 2004). Together these physiological changes could potentially compromise dispersal ability of anemonefish, thereby reducing the capacity of populations to recover from collection (Green and Fisher 2004). Beyond warming the planet, the burning of fossil fuels increases the dissolved carbon dioxide concentration and the acidity in the oceans, a process known as ocean acidification. Ocean acidification affects the ability of larval clownfish (*A. percula*) to detect predatory olfactory cues (Dixson et al. 2010). Newly hatched and settlement-stage larval fish exposed to acidified sea water were actually attracted to predators and unable to differentiate predators from non-predators (Dixson et al. 2010). Therefore ocean acidification may cause anemonefish larvae to be predated at higher rates, decreasing recruitment and lowering the recovery capacity from over exploitation and beaching (Dixson et al. 2010).

Conservation of giant anemones and anemonefish:

Several different measures have been attempted to improve anemonefish and giant anemone conservation, including captive-breeding programs, marine protected areas, and a quota-based collection system. This section presents an overview of these three measures, noting the efficacy of each for anemones and anemonefish.

As mentioned above, captive breeding of anemeonefish has been successfully achieved by hobbyists, commercial breeders, scientific researchers, and public aquaria (Dawes 2003, Hall and Warmolts 2003, Moe 2003). Anemonefish aquaculture was one of the first efforts to rear a group of coral reef fish and it remains one of the most successful efforts to this day (Dawes 2003). However, caring for, breeding, and raising coral reef fishes present many challenges, with feeding and caring for larval anemonefish and other coral reef fishes during their larval stage being one of the most difficult problems (Anto and Turingan 2010). Diets need to be precisely calibrated to meet the needs of the developing animals and, even with adequate care, mortality rates of larvae can be very high (Olivotto et al. 2008, 2010, Anto and Turingan 2010). Beyond issues with feeding, bacterial, dinoflagellate, and other types of infections frequently cause health problems in captive raised fish (Cobb et al. 1998, Nelson and Ghiorse 1999, Dhayanithi et al. 2010). The combined stressors experienced in captivity can lead to frequent mortalities and high rates of skeletal growth abnormalities (reviewed in Avella et al. 2010).

Despite these challenges, aquaculturists have recently developed many different methods to improve captive breeding, including exposure to lactic acid probacteria (Avella et al. 2010), treatments to avoid infection (Cobb et al. 1998, Dhayanithi et al. 2010), methods to improve growth and skin coloration (Avella et al. 2007, Yasir and Qin 2010), and other improvements. Captive rearing programs have been so successful that captive bred *A. bicinctus* have even been introduced into the wild (Maroz and Fishelson

1997). Despite this, anemonefish are still collected from the wild to satisfy market demand.

In contrast to the successes of captive bred anemonefish, captive-bred giant anemones are not widely available. This is surprising because many giant anemone species can be propagated asexually by longitudinal fission (Olivotto et al. 2011) and the feasibility of captive breeding has also been demonstrated (Scott and Harrison 2007a,b, 2009). There are reports of successful propagation of some giant anemones by hobbyists, commercial breeders, scientists, and public aquariums (Hall and Warmolts 2003, Moe 2003), although captive breeding attempts for *Stichodactyla* spp. anemones have not succeeded (Moe 2003). The slow growth rates and sporadic reproduction of giant anemones, combined with the availability of animals from the wild, likely makes captive breeding unprofitable under current laws and market conditions.

Beyond captive breeding programs, efforts are underway throughout the world to make the collection of wild anemones and anemonefish ecologically sustainable. For example, one measure suggested by Shuman et al. (2005) was to selectively harvest juvenile and male fish, leaving mature females on the reef, and occasionally allowing young male fish to replace females. The logic behind this suggestion was based on the protandrous hermaphrodism and the high recruitment rate of anemonefish. If the most reproductively productive individuals are left in place, it could insure a supply of future recruits to replace those that were collected (Shuman et al. 2005). (Note that this strategy is contingent upon healthy populations of anemones on the reef and source populations for new recruits.) Collectors in Australia have adopted a version of this strategy, but unfortunately, it is the youngest fish, not the oldest animals, that Australian collectors leave on the reef (Jones et al. 2008).

One of the most common measures aimed at coral reef wildlife conservation, including anemonefish and giant anemones, are no-take areas or marine protected areas. For anemonefish, a network of marine protected areas that enable replenishment (self-recruitment) and dispersal to new areas have been recommended (Planes et al. 2009). In some cases, existing marine protected area networks, such as the system in Papua New Guinea, appear to be sufficient for replenishment to occur (Planes et al. 2009). In other cases, such as protected areas in the central islands of the Philippines, the current sites are too dispersed and additional protected sites are necessary to insure connectivity and replenishment (Pinsky et al. 2010). Virtually nothing is known about the dispersal ability of anemone larvae and therefore the effectiveness of marine protected area systems cannot be evaluated from the anemone perspective.

The Great Barrier Reef marine protected area system provides a suitable model to examine in greater detail. The Great Barrier Reef Marine Park Authority (GBRMPA) is responsible for zoning different reefs within the system as either open or closed to recreational or commercial aquarium harvesting (Jones et al. 2008). The Department of Primary Industries and Fisheries issues licenses for commercial collection. Harvesting is conducted year round, with peaks in harvesting during February, March, July, October, and November (Jones et al. 2008). Harvesters typically remove a pair of (older) fish from an anemone, leaving a few (younger) fish behind (Jones et al. 2008). Recreational harvesting is subject to gear and bag limits, and anemones cannot be collected recreationally, but the extent of this harvest is unknown (Jones et al. 2008).

Scott et al. (2011) examined the change in anemone and anemonefish populations in response to changing management at North Solitary Island on the Great Barrier Reef. In 1991 a marine reserve was established that included “no-take” zones on the north and west sides of North Solitary Island (Scott et al. 2011). From 1994 to 2008, the protected area dramatically increased in both anemones and anemonefish populations (Scott et al. 2011). The percentage cover of the dominant anemone, *E. quadricolor*, increased by 86% to 450% during this time period (Scott et al. 2011). Anemone density also increased by up to 533% (Scott et al. 2011). The dominant anemonefish in this region, *A. akindynos*, increased in density by 42% to 133% during that same time interval (Scott et al. 2011). Other species of anemone (*H. crispa*) and anemonefish (*A. latezonatus* and *A. melanopus*) were less common at North Solitary Island and showed less of a clear increase in response to protection (Scott et al. 2011). However, *A. melanopus* did disperse and establish itself at locations where it had not previously existed (Scott et al. 2011). The overall result of the GBRMPA system was a dramatic increase in both anemones and anemonefish populations (Scott et al. 2011). Protection provided by the marine protected area system enabled recovery from population declines due to collection or bleaching (Scott et al. 2011).

The final example of anemone and anemonefish management presented here is the quota system used in the Maldives. Since the trade began in the Maldives, the government has closely monitored collection and export for ornamental aquarium fish (Edwards and Shepherd 1992). In 1988, concerns about sustainability led to the establishment of a combined total annual export quota of 100,000 fish and invertebrates for all allowable coral reef species (Edwards and Shepherd 1992). The government requires collectors and exporters to report the fish exported to customs officials; once the quota is reached no additional fishing is permitted until the following year (Edwards and Shepherd 1992). The analysis of Edwards and Shepherd (1992) identified a number of species that were either being over-exploited or were at risk of over-collection under current practices. The government therefore provisionally implemented a species-specific plan that included species-specific quota system for 22 species (Saleem and Islam 2008). Additional measures were implemented over the years. In 1995 and 1999, for instance, twenty-five sites were designated as areas protected from collection (Saleem and Islam 2008). Despite these improvements, enforcement for the system was lacking (Saleem and Islam 2008).

In recent times, the Maldives established a species-based quota system (Saleem and Islam 2008). This system bans the export of parrotfish, puffer fish, porcupine fishes, eels, giant clams, and hard corals besides *Tubipora musica* (Saleem and Islam 2008). Fish that are used for pole and line bait in tuna fishing are also banned from export (Saleem and Islam 2008). This has resulted in a ban on the export of *Chromis viridis*, which is the most commonly traded ornamental coral reef fish species in the world (Wabnitz et al. 2003). The new system includes three tiers of species, those that are banned entirely, those that have a species-specific quota, and those that are subjected to a general quota (see text above). This species-based quota system is still quite new and there has not been sufficient time for scientific studies to evaluate its efficacy. Concerns about the sustainability of anemonefish collection (Edwards and Shepherd 1992) resulted in the creation of species-specific quotas for *Amphiprion clarkii* and *A. nigripes* (Saleem and Adam 2004).

Saleem and Islam (2008) described new system as “quite effective”, but they note a number of challenges: (1) enforcement is still weak, (2) the program is governed by several different government agencies resulting in some jurisdictional conflicts and unnecessary bureaucracy, (3) the use of common names for exported species and a lack of familiarity with scientific names causes confusion, (4) licenses are issued based on the value of fish not the number of fish; to avoid paying for additional licenses, collectors underreport the value of their collection, and (5) collectors are moving from area to area as stocks decline. Although this system represents one of the most comprehensive national management schemes in existence, additional improvements are necessary to achieve sustainability and additional data are necessary to effectively monitor the trade in the Maldives (Saleem and Islam 2008).

Chapter 6
Seahorses

An introduction to the biology of seahorses:

Seahorses are a diverse group of unusual fishes found throughout the temperate to tropical oceans of the world. All seahorses are members of the genus *Hippocampus* and family Syngathidae, a family of teleost fishes which also includes pipefishes, pipehorses, and seadragons (Lourie et al. 1999). Defining the total number of seahorse species has challenged scientists. There have been more than 120 species of seahorses described in the scientific literature, but many of these were erroneously identified and are not true species (Scales 2010). The difficulty results from the great plasticity in seahorse appearance that caused different morphological variants to be mistakenly classified as different species (Scales 2010). Recently Lourie et al. (2004) revised many species' descriptions and condensed the number of valid *Hippocampus* species down to 33. Since then, several more species were described, but not all of these new species are accepted (Scales 2010). Molecular genetic analyses have revealed additional complexities, with cases of multiple genetically-distinct taxa lumped into a single species (e.g., *H. trimaculatus* and *H. erectus*). Vincent et al. (2011) places the number of valid seahorse species at 48. A list including the 37 most recognized *Hippocampus* species and their involvement in trade is provided in Table 5, but this list cannot be considered a definitive species list without additional validation.

Table 5: Species of seahorses, their conservations status according to the International Union for Conservation of Nature (IUCN), and their involvement in the dry (curio and traditional medicine) and live (aquarium) trades. Table adapted from Vincent et al. (2011).

Species	**IUCN status**	**Involved in Trade?**	
		Dry	**Live**
Hippocampus abdominalis	DD	Yes	Yes
H. algiricus	DD	Yes	Yes
H. barbouri	VU	Yes	Yes
H. bargibanti	DD	Yes	Yes
H. borboniensis	DD	Yes	Yes
H. campelopardalis	DD	Yes	Yes
H. capensis	EN	No	No
H. comes	VU	Yes	Yes
H. coronatus	DD	No	Yes
H. denise	DD	No	Yes
H. erectus	VU	Yes	Yes
H. fisheri	DD	No	No
H. fuscus	DD	Yes	Yes
H. guttulatus	DD	Yes	Yes

H. hendriki	DD	No	No
H. hippocampus	DD	Yes	Yes
H. histrix	DD	Yes	Yes
H. ingens	DD	Yes	Yes
H. jayakuri	DD	No	No
H. kelloggi	DD	Yes	Yes
H. kuda	VU	Yes	Yes
H. lichtensteinii	DD	No	No
H. minotaur	DD	No	No
H. mohnikei (*H. japonicus*)	DD	Yes	Yes
H. montebelloenis	-	No	Yes
H. pontohi	DD	No	No
H. reidi	DD	Yes	Yes
H. satomiae	DD	No	No
H. severnsi	DD	No	No
H. sindonis	DD	No	No
H. spiosissimus	VU	Yes	Yes
H. subelongatus	DD	No	Yes
H. trimaculatus	VU	Yes	Yes
H. whitei	DD	Yes	Yes
H. zebra	DD	Yes	Yes
H. zosterae	DD	Yes	Yes

DD = data deficient
VU = vulnerable
EN = endangered
- = status has not been evalutated by the IUCN

Depending on the species, seahorses range in length from 10–20 mm (*H. minotaur*) to 300 mm (*H. ingens*; Vincent 1996). The name seahorse comes from these fishes' horse-like appearance, including heads that form right angles relative to the body and tube-like snouts used for capturing food (Vincent 1996). Seahorses are relatively weak swimmers; they lack pelvic or caudal fins and their anal and pectoral fins are relatively small (Vincent 1996). Instead of swimming in pursuit of prey, they wait for food to approach while grasping seagrass or another holdfast with their fully prehensile tails (Vincent 1996). Their unique body form likely evolved to improve reach and strike capacity of this feeding strategy (Van Wassenbergh et al. 2011).

General characteristics of seahorses are often extrapolated from a few studies on a handful of species. Detailed biological information is sparse or lacking for most seahorse species. Seahorses are found throughout the world's oceans and seas from 45 degrees N to 45 degrees S latitude (Vincent 1996). Most species are found in relatively shallow water marine communities, from 0.5 to 100 meters in depth depending on the species (Vincent 1996, Scales 2010). *Hippocampus bargibanti*, for example, ranges from 45 to 60 meters deep (Vincent 1996, Scales 2010). Some individuals migrate into deeper waters during the winter, however, this behavior is poorly understood (Scales 2010).

Seahorses and other synganathids are a dominant fish family in seagrass habitats across the world (Pollard 1984). They are also found in a wide variety of habitat types including coral reefs as well as sponges, seaweed habitat, mangroves, soft-bottom and rocky-bottom areas, lagoons, estuaries, harbors, and soft coral or gorgonian fields, among other habitats (see Table 1 of McPherson and Vincent 2004 for a breakdown of habitat by species). All seahorse species are benthic as adults, with the possible exception of *H. fischeri* (Scales 2010). *Hippocampus comes*, for example, is found on coral reefs, soft corals, sponges, sea grass, soft sediments, and *Sargassum* from the surface to 20 meters deep (Martin-Smith et al. 2004). Some species even shift habitat preferences as they transition from juvenile to adult life stages (Scales 2010).

Most seahorse species have patchy, sparse distributions (Vincent 1996). Seahorses tend to be poor swimmers and this leads to low mobility and small home ranges, with females tending to range further than males (Foster and Vincent 2004, Vincent et al. 2005). The area ranged by an individual varies by the degree of monogamy vs. polygamy the species exhibits across breeding events. Species that are monogamous across breeding events tend more toward smaller ranges (e.g., 1–20 m^2 in *H. breviceps*; Foster and Vincent 2004), whereas species that are more socially polygamous tend toward larger home ranges (e.g., *H. abdominalis* migrates 100s of meters per day; Vincent et al. 2005). Moreover, seahorses typically have low population densities of just one individual per several square meters of suitable habitat (Vincent 1996). As a result, most seahorses are vulnerable to over-exploitation.

Seahorses are active predators that feed on live, slow-moving benthic organisms (Vincent 1996, Kendrick and Hyndes 2005). Their tube-like snouts are used to suction up just about anything of appropriate size, including small crustaceans, nematodes, and small fish (Vincent 1996, Castro et al. 2008, Storero and Gonzalez 2008). The typical hunting strategy is to sit and wait for prey while remaining attached to a holdfast, relying on their cryptic coloration, ability to remain immobile, skin filaments that mimic the surrounding habitat, and ability to change color to match their surroundings to camouflage them from approaching prey (note that the details of the camouflage varies among species; Vincent 1996). As prey drifts by, seahorses can strike rapidly without leaving their holdfasts (Vincent 1996, Kendrick and Hyndes 2005). Seashorse are likely important as predators of benthic organisms in sea grass environments (Vincent 1996) that can influence population structure of their prey communities (Tipton and Bell 1998). Therefore the removal of seahorses could potentially alter seagrass ecosystem community structure (Vincent 1996).

Seahorses have a unique reproductive biology that makes them both fascinating to study and vulnerable to over-exploitation. Most remarkably, seahorses are the only known animals where males become pregnant (reviewed in Scales 2010). Additionally, seahorses are notable for their monogamous reproductive pairings. Individual male and female seashorse form tight pair bonds that typically last through multiple mating events and sometimes even through multiple breeding seasons (Vincent and Sadler 1995). During this time, the pair mates exclusively with one another (Vincent 1996). The partnership is reinforced through daily courtship dances (Vincent 1996). When mating, the dance can last for hours; it culminates with the pair aligning as they rise through the water (Vincent 1996). The female then deposits eggs in the male's brood pouch with her ovipositor (Vincent 1996). Surprisingly, the male releases sperm into the water and the sperm must be captured within the pouch within 6 seconds of its release (reviewed in Scales 2010). Nevertheless, fertilization rates of deposited eggs are high (reviewed in Scales 2010). Neither sex will mate again while the male partner is pregnant (Vincent 1996). Genetic studies confirm that males only mate with one female at a time and that all the eggs in a given brood belong to a single female (Scales 2010).

Monogamy likely evolved as a successful reproductive strategy in the context of seahorses' low-density populations. Because finding mates is challenging, pairing exclusively helps to reduce the inter-brood interval and increases reproductive output (Scales 2010). Furthermore, seahorses have characteristics that are conducive to monogamy: (1) reproduction is asynchronous among different pairs allowing little opportunity for infidelity, (2) there are only small differences in the potential reproductive rates of males and females causing there to be no advantage to seeking another mate, and (3) mate familiarity increases reproductive success, and therefore fitness, in successive matings (Vincent 1996, Vincent et al. 2004). As a result of all these benefits, a paired individual will typically seek a new mate only if its partner is lost (Vincent 1996).

Fertilization occurs once eggs and sperm are deposited in the male broodpouch. Fertilized eggs then imbed in the pouch wall, where they are enveloped in tissue, provided with oxygen through capillaries, and nourished in a placental fluid (Linton and Soloff 1964, Haresign and Shumway 1981). The pregnancy lasts between 9 and 45 days, culminating in a lengthy labor where males pump and thrust for several hours of giving birth (Vincent and Sadler 1995, Scales 2010). The offspring emerge as fully-developed, but miniature, seahorses that are independent from birth (Foster and Vincent 2004). Seahorse males can harbor between 5 (*H. zosterae*) and 2,000 (*H. reidi*) offspring per pregnancy, with 100–300 young being a common range of brood sizes (Vincent 1990, Scales 2010). Young seahorses develop rapidly and are often capable of reproduction within six months to a year following birth (Vincent 1996). Upon reaching maturity, many seahorses breed year round (Martin-Smith et al. 2004). Overall, seahorse reproduction is characterized by relatively low fecundity, lengthy parental care (i.e., male pregnancy), and high mate fidelity (Vincent 1996). Taken together, these characteristics render seahorses susceptible to over collection.

Unlike most coral reef fishes, seahorses do not have a larval dispersal phase (although some juvenile seahorses are briefly planktonic following birth) (Foster and Vincent 2004). The lack of a larval phase results in a limited dispersal capacity among seahorses (Vincent 1996). As noted above, the seahorse body plan is adapted for maneuverability in complex habitats and not for speed or sustained swimming (Blake 1976). As a result, dispersal distances can be as low as several hundred meters or less (Vincent and Sadler 1995, Vincent 1996). Longer-distance dispersal may occur through rafting while attached to debris that is cast adrift by storms (Vincent 1996, Teske et al. 2007).

Several population genetic and phylogeographic studies have examined the dispersal ability and genetic breaks among seahorse populations. Seahorse populations oftentimes exhibit isolation by distance and phylogeograpic breaks across a species range. Examples include *H. kuda* in the Andaman Sea vs. Gulf of Thailand (Panithanarak et al. 2010), *H. ingens* in the Gulf of California vs. other populations (Saarman et al. 2010), *H. kuda* and *H. trimaculatus* between the two coasts of India (Goswami et al.

2009), and *H. trimaculatus* along Wallace's Line in Southeast Asia (Lourie and Vincent 2004). Lourie et al. (2004) examined the genetic structure of four Southeast Asian seahorse species using mitochondrial DNA. They found shared haplotypes over an average of 1,169 kilometers in *H. trimaculatus*, indicating some dispersal ability to maintain genetic connectivity in this species (Lourie et al. 2005). By comparison, *H. barbouri* haplotypes averaged just 67 kilometers in range, suggesting a low dispersal ability leading to differentiation between populations (Lourie et al. 2005). While much remains to be learned, the overall low dispersal ability of seahorses renders recovery from over harvesting difficult for many species (Scale 2010).

Finally, the lifespan of seahorse individuals is not well known. It appears that most species have low natural adult mortality due to predation (Vincent 1996). Seahorses have few known predators, although they are occasionally found in the guts of tuna, sharks, or rays and may be commonly eaten by crabs and sea birds (reviewed in Vincent 1996, Lourie et al. 1999). Estimates of seahorse lifespans range from as low as one year in *H. zosterae* (Strawn 1953) to as many as four years in most Indo-Pacific species (Vincent 1996).

Collection of seahorses and declines in seahorse populations:

Since the mid-1980s, seahorses have been collected and traded internationally to supply the aquarium, curio/home décor, and traditional medicine industries (Vincent 1996, Baum and Vincent 2005). Trade has grown rapidly, including a ten-fold increase in volume from the mid-1980s to the mid-1990s (Vincent 1996). As of 1995 there were over 20 million seahorses and 32 countries involved in the seahorse trade (Vincent 1996). Since the mid-1990s, seahorse collection and export continued to grow both in volume (Baum and Vincent 2005, Giles et al. 2006, Vincent et al. 2011) and in the number of countries involved (at least 72, including 46 exporting and 45 importing nations; Vincent et al. 2011). However, in recent years, trade has ebbed slightly, possibly as a result of a collection ban in the Philippines (Vincent et al. 2011).

Since at least 1996, Thailand has been a major exporter of seahorses (Vincent et al. 2011). The role of other countries has varied with time over the past 15 years. India, the Philippines, Vietnam, Mexico, Tanzania, and China have all been major sources of dried seahorses at various times (Vincent et al. 2011). Indonesia, Vietnam, Sri Lanka, and until 2005 the Philippines were the major source countries for live seahorses (Vincent et al. 2011). In contrast to the previous case studies where the U.S. has primarily driven demand, China, Taiwan, and the Hong Kong Special Administrative Region drive the demand for dried seahorses (Vincent 1996, Vincent et al. 2011). For live seahorses, the dominant markets are similar to the rest of the coral reef wildlife trade: the U.S. and the European Union are the primary markets (Vincent et al. 2011).

The majority (approx. 95%) of internationally traded seahorses are used in traditional medicine (Vincent 1996, Vincent et al. 2011). However, several hundred thousand live seahorses are collected each year for the aquarium trade, and this collection can place localized pressure populations of seahorses (Vincent 1996, Vincent et al. 2011). The number of animals collected as curios is unknown (Vincent 1996), but seahorses are the second most imported group or species of marine fish in the U.S. curio trade (Grey et al. 2005). Common medicinal uses for seahorses include treatments for respiratory problems (e.g., asthma), sexual dysfunction, incontinence, general lethargy, and pain (Vincent 1996, Vincent et al. 2011). Seahorses are also used as a medicinal resource in Brazil and in parts of Latin and South America (Alves and Rosa 2006, Baum and Vincent 2005). Their widespread application in traditional medicine suggests that seahorses may have true medicinal value (Vincent 1996). Based on this, Vincent (1996) argued that instead of allowing continued exploitation of seahorses from the wild, natural populations should be protected, with efforts directed at further scientific study and potentially pharmaceutical development. Traditional medicine is not a focus of this review and will not be discussed in detail in the following sections. However, the mixed use of seahorses for three different industries makes it challenging to ascribe impacts to any one trade. Unless otherwise noted, the impacts described below should be considered a cumulative consequence of the traditional medicine (dead seahorses), curio/home décor (dead seahorses), and aquarium (live seahorses) industries.

Tracking the seahorse trade is a significant challenge. Seahorses are collected by either small-scale, artisanal operations (approx. 5% of trade) or as the result of bycatch from shrimp and demersal-fish trawling (approx. 95% of trade; Scales 2010, Vincent et al. 2011). Seahorses are very common bycatch species in demersal shrimp trawling because they are similar in their size and habitat requirements to shrimp, and are poor swimmers (Vincent et al. 2011). In both types of fisheries, catch data are rarely recorded, making it difficult to monitor patterns over time (Scales 2010). When seahorses are declared on import and export forms, they are rarely differentiated by species and mixed species assemblages are often shipped together in the same container (Scales 2010, Vincent et al. 2011). Listing of seahorses on Appendix II of CITES has improved that situation in recent years (Vincent et al. 2011), but around 23% of seahorse shipments are still listed under the generic name of *Hippocampus* sp. (Evanson et al. 2011). The two most common species in trade are *H. kuda* and *H. erectus* (Wabnitz et al. 2003), but many other species are also collected. According to CITES trade data, 28 out of 48 known species are involved in trade, including 18 species harvested for traditional medicine and/or curios whereas 27 species are used in home or public aquariums (Vincent et al. 2011). As described above, seahorses have considerable variability within and between species, as well as poorly known species boundaries, which hinders proper identification, monitoring, and management.

Martin-Smith et al. (2004) and Vincent et al. (2007) described artisanal seahorse fisheries from coral reefs of the central Philippines. At least 200 fishers actively collect seahorses along 150 km of the Danajon Bank reef system (Martin-Smith et al. 2004). Collectors harvest seahorses by free diving (>75% of fishers) or hooka, a type of surface-supply breathing apparatus (<25% of fishers; Martin-Smith et al. 2004, Vincent et al. 2007). Collecting often takes place at night using kerosene lanterns (Martin-Smith et al. 2004). About 40% of collectors' income comes from seahorse collection; fishers also make a living by collecting food fish using a spear and other valuable species for the ornamental trade by hand (Martin-Smith et al. 2004). Collectors in the Danajon Bank region target mostly *H. comes*, but *H. spinosissimus* is also harvested (Martin-Smith et al. 2004). Collectors sell their catch for traditional medicine, curios, or aquariums (Martin-Smith et al. 2004, Vincent et al. 2007). The destination of each fish largely depends on the size of the animal – smaller fish are sold to aquarium trade, where fish are priced by the individual (Martin-Smith et al. 2004). Larger fish are sold for traditional medicine, which assesses value according to seahorse length (Martin-Smith et al. 2004).

Because of the artisanal or bycatch nature of seahorse collection, collectors will continue capturing seahorses, even when populations become vulnerable and numbers decrease to critically low levels (Vincent 1996). In the case of bycatch fisheries, fishers are primarily targeting shrimp or demersal fish, and fishers' behavior will be driven by trends in the populations of their target species, not seahoarses (Vincent 1996). In the case of artisanal fisheries, collectors venture out to collect many different species and will opportunistically capture those seahorses that they happen to encounter (Vincent 1996). Therefore, collection effort is maintained even when seahorse catch declines. In many artisanal fisheries, collectors have very few other economic opportunities and demand for seahorses remains high (Vincent 1996). Collectors have a strong incentive to continue collecting regardless of seahorse population sizes (Vincent 1996). As a result, collection could reduce seahorse populations to the point of local extinction (Vincent et al. 2007).

The biology of seahorses makes many species and populations vulnerable to over exploitation. As described in the previous section and reviewed by Vincent (1996), Foster and Vincent (2004), and Scales (2010), seahorses are naturally sparse in their distribution, which makes it difficult to find new mates. When a mate is located, partners maintain high fidelity to one another. This social and reproductive system is easily disrupted by removing one partner. The loss of a mate forces the remaining seahorse to seek a new partner, re-pair, and mate again. This can take a considerable amount of time and energy due to their innately low dispersal capacities, sparse distributions, and low encounter rates between individuals. Seahorses' limited mobility, high site fidelity, and small home ranges further restrict re-colonization of areas subject to over-exploitation. The unique reproductive traits of seahorses, including male pregnancy and monogamy, further exacerbate the vulnerability of seahorses to overharvesting. Collection of pregnant males negates the time and energy a pair invested in reproduction. Male pregnancy also results in small brood sizes and high levels of parental care, both of which hinder recovery from population depletion. Finally, low natural mortality rates means that fishing exerts a substantial pressure on seahorse populations. Although these general trends outlined by Vincent (1996), Foster and Vincent (2004), and Scales (2010) are true for many seahorse species, they do not apply universally. Curtis and Vincent (2006) provide a contrasting example in *H. guttulatus* from Portugal. Based on its life-history characteristics, *H. guttulatus* has the capacity to recover quickly from direct (exploitation) and indirect (habitat loss, bycatch) population impacts (Curtis and Vincent 2006).

There have been few controlled scientific studies of the ecological impacts of collection on seahorse populations that compared equivalent locations with and without collection (*sensu* Tissot and Hallacher 2003, Tissot et al. 2004). As an alternative to this approach, a number of scientists, most notably Amanda Vincent and the non-profit organization Project Seahorse, have inferred the status of seahorse populations through examination of trade data and catch reports, as well as interviews with fishers from various countries. Interviews with fishers and exporters commonly indicate declining abundances and catches in many countries across the world. Based on interviews with fishers, Vincent (1996) inferred that seahorse catch declined by 15-20% between 1990 and 1995 in Southeast Asia (Vincent 1996). In five countries, seahorse population declines were estimated at 50% (Vincent 1996). At that time, large seahorses had become increasingly rare, which drove collection to less desirable specimens, including juveniles (Vincent 1996). Demand far exceeded supply, raising concerns about the long-term sustainability of seahorse fishing (Vincent 1996).

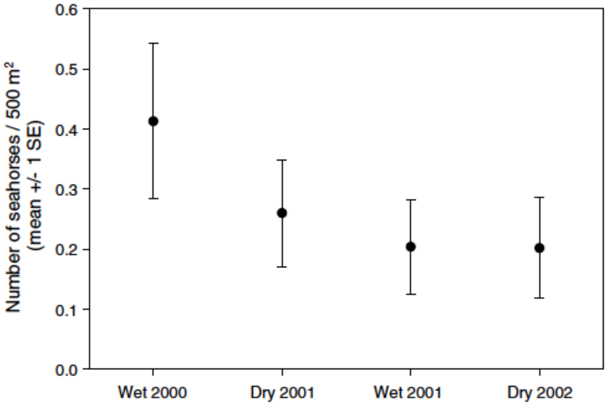
In the years since Vincent's (1996) report, a number of studies have further documented trends in seahorse populations and collection patterns by country and by region. Perhaps the best-studied examples come from the central Philippines. Martin-Smith et al. (2004) used six different metrics to qualitatively assess the sustainability of the central Philippines seahorse fishery and found evidence for overfishing according to all six metrics. Furthermore, assessments of seahorse collection based on criteria devised by Carl Walters and catch data from the fishery, biological data from other sources, and estimates of fishery parameters found evidence of overfishing under every criterion considered (Martin-Smith et al. 2004). Based on these results, Martin-Smith et al. concluded that seahorses were overfished in the central Philippines and additional management action was required.

Vincent et al. (2007) also studied the impacts of seahorse fishing in the central Philippines by examining catch per unit effort (CPUE) from 1996 to 1999. That analysis found that between 2.94 and 3.43 seahorses were collected per fisher per night (Vincent et al. 2007). These values are considered to be very low and are likely indicative of depleted populations (Vincent et al. 2007). Comparisons between CPUE data and information gleaned from interviews with fishers and buyers indicated that there were major declines in seahorse populations over time (Vincent et al. 2007). Fishers reported 50–100 seahorses were collected per fisher per night during the 1960s and 1970s (Vincent et al. 2007). This number declined to 10–50 seahorses per fisher per night from 1980–1985 to less than 4 seahorses per fisher per night in 2000 (Vincent et al. 2007). Based on interviews with 21 collectors, O'Donnell et al. (2010) estimated 75–93% declines in CPUE from the central Philippines over three decades ending in 1994 (O'Donnell et al. 2010). Furthermore, the portion of brooding males caught has declined over time (Vincent et al. 2007). The lack of pregnant males could be interpreted as a failure in seahorse reproduction – low populations sizes can lead to Allee effects where reproduction begins to fail (Vincent et al. 2007). Alternatively, conservation training may be causing collectors to avoid pregnant males (Vincent et al. 2007). Regardless of the cause, the diminished CPUE suggested that seahorses are overfished in central Philippines.

In addition to heavy collection, habitat decline contributes to seahorse decline. Surveys of seahorse fishing grounds in the central Philippines from 2000 to 2002 found extremely degraded coral reefs, with 69% of seahorse habitat being comprised of non-living or dead structures (i.e., rubble, dead corals, etc.; Marcus et al. 2006). Fishing using dynamite, cyanide, plant-based poisons, and other illegal, destructive fishing methods were likely causes of this situation (Marcus et al. 2006). Areas with regular patrols by law enforcement had low coral rubble and high quality habitat (Marcus et al. 2006). As distance increased from the enforcement areas, habitat quality declined (Marcus et al. 2006). Habitat quality also declined over time; comparisons to historical accounts show dramatic declines in habitat quality since the 1950s when destructive fishing began (Marcus et al. 2006). In just 2–4 years, coral cover dropped from 31% to 15% (Marcus et al. 2006). Collectors acknowledged that the decline resulted from cyanide and dynamite fishing (Marcus et al. 2006).

Within the study area examined by Marcus et al., seahorse populations were extremely low. Seahorses were not found in more than 90% of the 487 transects examined by Marcus et al. When seashores were found, fish density was just one to three fish in the survey area (Marcus et al. 2006). Total seahorse density throughout the area Marcus et al. surveyed was 0.262 fish per 500 m^2, making it one of the lowest seahorse population density ever reported (Marcus et al. 2006, Figure 12). Comparisons of seahorse population densities in collection areas versus marine protected areas revealed an order of magnitude difference in seahorse density, suggesting that heavy collection and habitat degradation were responsible for the decline (Marcus et al. 2006).

Figure 12: Decline in seahorse density from 2000 to 2002 during the rainy (wet) and dry seasons. Note the very low overall density throughout the study interval. Figure taken from Marcus et al. (2006).



Similar indicators of seahorse decline have been reported in other areas of Southeast Asia. In Vietnam, for example, seahorses are collected as bycatch from shrimp and demersal fish trawling (Giles et al. 2006). Seven species are collected by the Vietnamese fishery, with *H. spinosissmus*, *H. trimaculatus*, and *H. kuda* being the most common species in trade (Giles et al. 2006). From 1980 to 2001, trawling effort increased 250% and the associated bycatch concomitantly increased as a result (Giles et al. 2006). Around 2.275 million seahorses are collected each year from Vietnam, with most animals exported through unofficial channels to China for TM (Giles et al. 2006). CPUE was estimated to range from 0.33 to 2.50 depending on region and year (Giles et al. 2006). These consistently low CPUE values indicate a seahorse population that is dispersed, patchy, and in decline (Giles et al. 2006). This finding was corroborated through interviews with fishers and buyers (Giles et al. 2006). Specifically, 122 out of the 143 fishers and 21 out of the 27 buyers that Giles et al. interviewed reported a recent decrease in seahorse abundance. Overall, seahorses were estimated to have declined by 30–60% during the preceding 2 to 5 years (Giles et al. 2006, but see Meeuwig et al. 2006). As with multispecies artisanal fisheries (e.g., Vincent et al. 2007), collection effort continues in bycatch fisheries even as seahorse populations decline (Giles et al. 2006).

Fishers and traders also reported decreasing seahorse abundance in nearby Malaysia and Thailand (Perry et al. 2010). In Malaysia, fishers indicated population declines of 68 ± 24% over 12.5 years (Perry et al. 2010). Most interviewees simply stated that there were now considerably fewer seahorses than in previous years (n=28 of 37), however, some (n=9 of 37) indicated that the reductions resulted from over fishing (Perry et al. 2010). In Thailand, 81% (n=30 of 37) of the seahorse collectors and traders interviewed suggested that seahorse catches were declining and none (n=0 of 37) of the interviewees thought that seahorse populations had grown (Perry et al. 2010). Fishers and traders estimated that the seahorse catch had declined by 22–96% over 2.5–15 years in Thai waters (Perry et al. 2010). As was the case for central Philippines and Vietnam, the causes of seahorse declines in Malaysia and Thailand were attributed to over-exploitation and habitat decline (Perry et al. 2010).

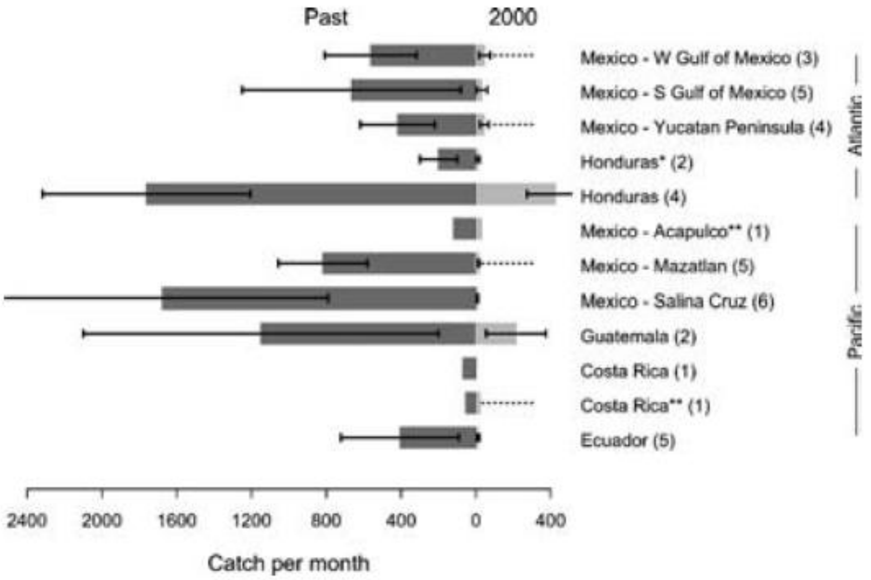
In many locations, the ecological impacts of the seahorse trade are unknown or poorly documented. One ambiguous example comes from the east African nations of Kenya and Tanzania (McPherson and Vincent 2004). The seahorse trade in east Africa has been ongoing since 1975 (McPherson and Vincent 2004). As of 2000, there were four seahorse species from the Kenyan and Tanzanian coastline that were actively traded: *H. borboniensis, H. camelopardalis, H. histrix*, and *H. kelloggi* (McPherson and Vincent 2004). All four species were typically found in association with seagrasses in shallow water habitats (McPherson and Vincent 2004). In Kenya, the seahorse trade was negligible (i.e., approximately 10 seahorses per year; McPherson and Vincent 2004). However, trade was more substantial in Tanzania, with 634–937 kg of seahorses (over 250,000 animals) exported in 2000 (McPherson and Vincent 2004). (It was presumed that a similar volume was exported from Tanzania every year [McPherson and Vincent 2004]). Most exports were of dead seahorses. Exporters discouraged capture of live seahorses because of high mortality in captivity (i.e., approximately 20%) and the large amount of shipping space required to transport live animals (McPherson and Vincent 2004). Protection of seahorses is weak in east Africa, with no conservation measures in place that directly protect seahorses in this region and enforcement of existing fishing regulations being relatively weak (McPherson and Vincent 2004). There was also cause for concern about seahorse populations due to habitat decline; coastal degradation, mangrove cutting, destructive fishing, and high fishing pressure have all degraded seahorse habitats (McPherson and Vincent 2004). Despite these concerns, there was no clear consensus among fishers about whether seahorse population numbers have declined (McPherson and Vincent 2004). Some dealers reported patterns of decline, but overall there was not a consistent pattern to report from this fishery (McPherson and Vincent 2004). Additional study is necessary to fully assess the situation.

In at least one location, low-level collection does not appear to have major impacts on seahorse populations. Australia is home to the highest sygnathid diversity of any country in the world, including 14 species of seahorses (Lourie et al. 1999, Martin-Smith and Vincent 2006). Seahorses are not collected in high volume in Australia (Stobutzki et al. 2001). To better document this situation, Martin-Smith and Vincent (2006) collected trade data based on government reports and interviewed persons involved in the seahorse trade. Although the seahorse trade in Australia is very minor compared to Asian and Latin American countries, starting in 2000, there were major increases in exports of live seahorses for the aquarium trade (Martin-Smith and Vincent 2006). Discrepancies in the trade data hindered a precise determination of trade impacts. For example, Australian export records indicate that the post-2000 increased exports were captive bred fish, whereas U.S. import records indicated an increase in both wild-caught and captive-bred fish (Martin-Smith and Vincent 2006). Despite some problems in the trade data, Martin-Smith and Vincent (2006) concluded that international trade was not a major threat to Australian seahorse populations. Instead, habitat loss was considered to be the greatest problem for Australian sygnathids (Martin-Smith and Vincent 2006).

Seahorses are also collected in large numbers from the Gulf of Mexico, Caribbean Sea, eastern Pacific Ocean, and western Atlantic Ocean. In the Gulf of Mexico, seahorses are captured incidentally as bycatch in bait-shrimp trawl fishing (Baum et al. 2003). One documented example comes from the fleet from Hernando Beach, Florida, U.S., which catches approximately 72,000 seahorses annually (Baum et al. 2003). The overall size of this population is unknown, and the lack of specific information about seahorse populations creates uncertainty about the impacts of fishing (Baum et al. 2003). Because of temporal and spatial variation in CPUE and population size as well as possible segregation of males and females in the environment, trawling in the Gulf of Mexico removes a greater proportion of female seahorses than males (Baum et al. 2003). Observers aboard fishing vessels report that most of the captured seahorses were returned to sea, with 1% dying during towing, 4.7% being injured during capture, and an unknown number dying after release (Baum et al. 2003). The mortality rates are likely to be higher in fishing vessels that drag the tow for longer periods of time (Baum et al. 2003). Overall there is a lack of definitive data on the impacts of seahorse bycatch, highlighting the need for additional research on this topic (Baum et al. 2003).

Seahorses are collected throughout Latin and South American waters including Mexico, Belize, Costa Rica, Guatemala, Honduras, Nicaragua, Panama, Ecuador, and Peru among other nations (Baum and Vincent 2005). Between 1998 and 2001, Baum and Vincent (2005) interviewed 486 people involved in trade at 63 sites across these nine countries as well as an additional 49 retailers in Mexico. Additionally, the authors examined customs and trade data from throughout Latin and South America (Baum and Vincent 2005). At least four species of seahorses occur in this region, including *H. zosterae, H. erectus,* and *H. reidi* in the Atlantic Ocean/Caribbean Sea as well as *H. ingens* on the Pacific coast (Baum and Vincent 2005). Although *H. zosterae* did not appear to be involved in trade, the remaining three species were collected, with *H. erectus* and *H. ingans* dominating trade in Latin and South America (Baum and Vincent 2005). In this region, seahorses are used as dried curios (e.g., on key chains, as unadorned trinkets, etc.), as jewelry (e.g., as “dragons” with artificial eyes and wings attached), or as shell craft with other ornamental species (i.e., mollusk shells, sea stars, etc.; Baum and Vincent 2005). Seahorses are only rarely used in TM in Latin and South America, but they are sometimes powdered and consumed in a drink to treat asthma (Baum and Vincent 2005). Live seahorses from Latin and South America are also collected to supply the aquarium trade (Baum and Vincent 2005). During the interviews by Baum and Vincent, approximately 76.5% of interviewees that commented on population trends reported declines in Latin and South American seahorse populations (Figure 13). The reasons most commonly given for the decline were bycatch from shrimp fishing and overcollection for trade (Baum and Vincent 2005). In the countries with the heaviest levels of trade, catch had declined by more than 75% (Baum and Vincent 2005). Existing trade controls, including protection by CITES, were insufficient protections against seahorse depletion (Baum and Vincent 2005).

Figure 13: Historical catch per month compared to catch per month in 2000 in Mexico and various Latin and South American countries. Figure taken from Baum and Vincent (2005).



Among Latin and South American nations, Brazil is one of the main suppliers of live seahorses for the ornamental aquarium trade (Wood 2001b, Baum and Vincent 2005, Rosa et al. 2006). The Brazilian seahorse fishery provides another example of an artisanal fishery, with collection practices similar to those of the central Philippines. Collectors harvest seahorses in shallow water (less than 7 m depth) by free diving or occasionally by hooka (Rosa et al. 2006). Capture occurs by hand or using nets (Rosa et al. 2006). As with most artisanal seahorse fisheries, collectors take other marine ornamentals and spear fish or invertebrates for human consumption while collecting seahorses (Rosa et al. 2006). The fishery targets two seahorse species, *H. reidi* and *H.* cf. *erectus* (Rosa et al. 2006). Pregnant male seahorses are commonly collected (Rosa et al. 2006). From 1997–2005, Rosa et al. collected data on this fishery through a collaborative monitoring program with 152 fishers as well as various retailers. During this time, 9,793 seahorses were collected (Rosa et al. 2006). The seahorse catch varied from year to year and exhibited a general trend of declining catch over time (Rosa et al. 2006). The official trade statistics from Brazil generally underestimated the magnitude of this trade and overall lack of data and accurate record keeping was clearly a problem (Rosa 2005). Declining catch may suggest a decline in seahorse populations or it may reflect other trends, such as reduced collection effort (Rosa et al. 2006). As with seahorse populations from many parts of the world, additional study would better illuminate the status of Brazilian seahorse populations.

Once seahorses are collected in Brazil, they are often stored in small tanks without holdfasts and other necessary and appropriate habitat (Rosa et al. 2006). This results in seahorses grasping each other or the tank’s aeration hoses with their prehensile tails (Rosa et al. 2006). This can lead to high rates of mortality when the gill area of a seahorse is constrained thus preventing the grasped seahorse from breathing (Rosa et al. 2006). Like other ornamental aquarium fishes, Brazilian seahorses are starved for several days leading up to shipping (to prevent them from contaminating their containers with waste during shipping) and then transported in transparent bags filled approx. 1/3 full of seawater (Rosa et al. 2006).

Wright et al. (2007) examined stress experience by *H. abdominalis* during and after transportation using several different bio-indicators. Stress levels were significantly elevated during transportation of *H. abdominalis*, however, transportation did not have any apparent long-term effects on the animals (Wright et al. 2007). Indicators of seahorse stress returned to normal levels within 6 hours of transportation (Wright et al. 2007). Once live seahorses arrive at their final destination, a home or public aquarium, they often fare poorly; however, there have been considerable advances in the past decade that have improved success in captivity (Vincent and Koldewey 2006, Koldewey and Martin-Smith 2010).

Other than over collection for the ornamental coral reef wildlife and TM trades, seahorses are primarily threatened by habitat loss. Seahorses live in habitats that are in global decline, including coral reefs, mangroves, seagrass beds, and estuaries (Hodgson 1999). Habitat decline has various causes. Climate change is one major threat to seahorse habitats, most especially coral reefs (Hodgson 1999). Bottom trawling for shrimp and demersal fish also damages seahorse habitats (Environmental Justice Foundation 2003). Recreational diving can even damage seahorse habitat, although the spatial extent of diving impacts is limited (Uyarra and Côté 2007). Chemical pollutants, eutrophication, diminished water quality, and invasive species also contribute to habitat and seahorse population declines (Curtis and Vincent 2005, Vincent et al. 2011). In some cases, seahorse population declines occur in the absence of fishing pressure or habitat loss. Martin-Smith and Vincent (2005) observed seahorse population declines without a clear cause. The authors concluded the loss of seahorses was due to disease, invasive species interactions, and Allele effects (Martin-Smith and Vincent 2005).

Conservation of seahorses:

Concern about the status of seahorses resulted in the listing of many *Hippocampus* species in the IUCN Red List of Threatened species (www.iucnredlist.org). This includes seven *Hippocampus* spp. that are involved in trade and are considered to be vulnerable because of population declines exceeding 30% over 10 years. Most of the *Hippocampus* spp. recognized by IUCN (n=29) are considered data deficient and more research is required to evaluate their status. One species, *H. capensis*, is listed as endangered, but this species is not involved in the TM, curio, home décor, or aquarium trades.

Amid worries about seahorse population declines and their vulnerability to localized extinction in many regions, in November of 2002 the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) designated all species in the genus *Hippocampus* on CITES Export permits for Appendix II specimens can be issued by the exporting country only when the following conditions are met (CITES Treaty, Article IV): (a) “[a] Scientific Authority of the State of export has advised that such export will not be detrimental to the survival of that species"; (b) "[a] Scientific Authority in each Party shall monitor both the export permits granted by that State for specimens of species included in Appendix II and the actual exports of such specimens. Whenever a Scientific Authority determines that the export of specimens of any such species should be limited in order to maintain that species throughout its range at a level consistent with its role in the ecosystems in which it occurs and well above the level at which that species might become eligible for inclusion in Appendix I, the Scientific Authority shall advise the appropriate Management Authority of suitable measures to be taken to limit the grant of export permits for specimens of that species"; and (c) "a Scientific Authority of the State of introduction advises that the introduction will not be detrimental to the survival of the species involved". Seahorses were the first commercially-important marine fish to be CITES listed (Martin-Smith and Vincent 2006) and seahorses may be the most voluminously traded CITES species group (Bruckner et al. 2005). Unfortunately the population biology data required to render a robust non-detriment finding is often lacking for seahorse species. Furthermore, issues with proper identification of seahorses species remain a challenge for CITES.

In the lead up to implementation of CITES Appendix II listing, a workshop was held in Mexico in February 2004 in order to increase the effectiveness of seahorse management and conservation. Participants in that meeting made the following recommendations (Bruckner et al. 2005):

(1) Implement voluntary interim measures, such as minimum export sizes, export quotas, and a cap on the number of new licenses, that allow for sustainable management while a comprehensive management program is developed and implemented.

(2) Source countries should collect and maintain catch and effort data as well as data on seahorse population trends over time.

(3) Source countries should review established marine protected areas and establish new no take areas to sufficiently protect vulnerable life stages and maintain minimum viable populations.

(4) Source countries should develop and implement a set of adaptive management tools, improve communication and cooperation between stakeholders, and monitor sentinel species as indicators of the seahorse fishery's overall sustainability.

(5) There should be a universal set of export standards, including universal reporting volumes (likely by weight or mass for dried seahorses), separation of seahorses from other tropical fishes, transparent packaging materials for live animals, and resolution of taxonomic and identification issues.

(6) Countries should manage aquaculture to address production capabilities, degree of reliance on wild populations, and environmental concerns associated with these businesses. Ideally, a tagging system should be developed that differentiates between captive-raised vs. wild-caught seahorses. Until such a system is developed, a robust system of documentation is needed.

(7) Support a new Project Seahorse trade report to identify fisheries of concern.

(8) Communicate workshop findings to appropriate governmental and international bodies, including the CITES parties.

Although these recommendations would likely improve seahorse conservation, to date, most of these recommended measures have not been implemented.

As an interim measure to protect seahorses, a minimum-size limit of 10 cm for *Hippocampus* spp. was proposed by the CITES Animals Committee. The reasoning behind this size limit was that it would allow for sufficient breeding to occur in the time before seahorses were collected, thus maintaining population viability over time (Foster and Vincent 2005). The efficacy of the size limit was evaluated by various studies, with mixed results. For example, Foster and Vincent (2005) evaluated the 10-cm size limit on a species-specific basis and concluded that at a length of 10 cm, 15 species of the 32 species considered would be reproductively active (Foster and Vincent 2005). Of the remaining 17 species, 16 were not commonly involved in the international trade, were safeguarded under domestic legislation, or were partially protected by this size limit (Foster and Vincent 2005). Only one species, *H. kelloggi*, was not well served by the 10-cm minimum size limit in Foster and Vincent's determination (2005). Foster and Vincent (2005) noted the need for complementary conservation measures, especially for seahorses captured as bycatch, but regarded the minimum-size limit as a reasonably effective protection measure. By contrast, Wood (2005) evaluated the size limit in New Zealand's *H. abdominalis* and concluded that the size limit was insufficient because few male *H. abdominalis* were reproductively mature at this size. Similarly, Curtis and Vincent (2008) examined the 10-cm size limited and judged it to be an insufficient preventative measure against over collection and population declines for many species (Curtis and Vincent 2008). The authors recommended increasing the minimum-size limit to the size reached after one complete reproductive cycle (Curtis and Vincent 2008). Such a measure would reduce the chance of making populations vulnerable to extinction by half, while only decreasing the long-term catches by 5.6% (Curtis and Vincent 2008). It is also important to note that seahorses shrink by 14-44% when dehydrated and preserved, and this presents a problem for the implementation and enforcement of the minimum size limit recommendation (Nadeau et al. 2009). Increasing the size limit to 11 cm or more could help account for the size lost during preservation (Nadeau et al. 2009).

Many of the studies on population declines cited above are based on trends predating CITES Appendix II listing of *Hippocampus* spp. As of the writing of this report, it is still too early to conclude whether or not this measure has improved seahorse conservation or put the seahorse trade on a sustainable trajectory. This will be an important topic for future studies. Across the 46 exporting nations involved in trade, many different conservation measures have been put in place to protect seahorses and it is not possible to comprehensively review all of these measures here (see Vincent et al. 2011 for a recent overview of seahorse conservation). Since the proposal and listing of seahorses under CITES Appendix II, the Philippines and Brazil have drastically changed their management practices and we will briefly examine those improved practices here.

As a result of concerns about the population declines and CITES Appendix II listing of *Hippocampus* spp., the government of the Philippines banned seahorse collection in 2004 (Vincent et al. 2011). By 2006, seahorse exports from the Philippines had ceased (Vincent et al. 2011, but see Scales 2010 which reports continued fishing). Recent CITES data have shown a reduction in seahorse trading globally, which may be related to the ban the Philippines placed on capturing and trading seahorses (Evanson et al. 2011). This dramatic reduction in fishing pressure should allow seahorse populations to recover, although this recovery will take time for populations suffering from Allee effects. Using *H. comes* as a model organism, Martin-Smith et al. (2004) recently developed a management plan for the Philippines. That plan was based on input gathered from a diverse group of stakeholders, including fishers, traders, consumers, conservation groups, management agencies, and fisheries experts (Martin-Smith et al. 2004). Among these groups, there was considerable support for no take marine protected areas and minimum size limits, as well as moderate to high support for tenure systems and temporal closures (Martin-Smith et al. 2004). Management options with mixed or low support included reducing fisher numbers, gear restrictions, sex-selective fishing, caging pregnant males, total allowable catch, and maximum size limits (Martin-Smith et al. 2004). Based on these results, Martin-Smith et al. recommended implementation of no take marine protected areas, minimum size limits, and a tenure system. Vincent et al. (2007) made similar recommendations for the Philippines, including development of a community co-management system, no-take marine protected areas, temporal closures during period of high seahorse recruitment (i.e., February through April), minimum size limits, maximum size limits to keep productive breeders in population, and a ban on fishing pregnant males (Vincent et al. 2007). Some of these recommendations have now gone into effect. With the assistance of Project Seahorse, the Philippines established 34 community-managed no-take marine protected areas that used seahorses as sentinel species for ecosystem health (Vincent et al. 2011). The results of this measure have been mixed thus far. Although many fish species are doing well inside the marine protected areas, there is no evidence for increased abundance of seahorses as a result of this protective measure (Vincent et al. 2011). One possible cause for this result is an increase in predator abundance within the reserves, which may reduce seahorse numbers (Vincent et al. 2011). Seahorses within the marine protected area have increased in average size, however, and over time this may contribute to increased reproductive success and population growth (Vincent et al. 2011).

Prior to the listing of seahorses under CITES, seahorse collection in Latin and South America was largely unmonitored (Baum and Vincent 2005). Few countries maintained customs records or any other official records. Where records were kept, there was little agreement in trade data among different exporting and importing countries suggesting that the trade was highly underreported (Baum and Vincent 2005). Permits were generally not required for collection and where they were required, few traders actually had them (e.g., dried seahorses in Honduras and Nicaragua, live seahorses in Costa Rica) (Baum and Vincent 2005). Furthermore, seasonal and area closures for trawl fisheries were often not respected, especially in remote areas where enforcement is difficult (e.g. Ecuador, Caribbean coasts of Honduras and Nicaragua) (Baum and Vincent 2005). Mexico and Guatamala did establish marine reserves that protected seahorses, but enforcement and effectiveness of these areas were unknown (Baum and Vincent 2005). However, following the listing of all seahorses under CITES Appendix II, management has improved in some countries. Brazil, for instance, has established export quotas for seahorses that have been progressively reduced every year since 2002 (Rosa et al. 2006). In 2004 Brazil produced its first list of endangered, over-exploited, or threatened of exploitation aquatic invertebrates and fish species (MMA 2004, Rosa et al. 2006). This measure necessitates development of a recovery plan for seahorses as well as several other species (Rosa et al. 2006). With time, hopefully such measures will improve the status of Brazilian seahorse populations.

Demand for seahorses continues to be high throughout the world and wild populations are often unable to support consumers' needs for these animals. One way of satisfying this demand and reducing pressure on wild populations is through robust aquaculture and captive rearing programs. Research into seahorse aquaculture and captive rearing has been an active area of study for the last 50 years (Zhang et al. 2010). Seahorses generally fare poorly in captivity (Vincent et al. 2011), but major breakthroughs over the past 10 years are making aquaculture and captive rearing a viable alternative to wild-caught seahorses **(**Koldewey and Martin-Smith 2010, Zhang et al. 2010).

Hobbyists began raising and breeding seahorses in the 1960s and these efforts expanded into attempts at commercial operations in the 1970s (Koldewey and Martin-Smith 2010). Technical issues with diseases and feeding hindered advancement of this business model for many years (Koldewey and Martin-Smith 2010). However, by the late 1990s, aquaculture operations became a successful business (Koldewey and Martin-Smith 2010). Today there are at least 13 species of seahorses that are either commercially cultured or under research for their potential, including *H. barouri*, *H. kuda*, *H. reidi*, *H. erectus*, and *H. abdominalis* among others (Koldewey and Martin-Smith 2010). These are reared at 28 (or more) seahorse aquaculture facilities, most of which are located in developed nations (Koldewey and Martin-Smith 2010). Captive reared seahorses now make up a sizeable portion of the live seahorse trade (i.e., the aquarium trade), including 36% of live traded seahorses as of 2004 and 80% of live traded seahorses as of 2008 (Koldewey and Martin-Smith 2010). Unfortunately, captive raised seahorses do not comprise a significant portion of the trade in dead seahorses (Koldewey and Martin-Smith 2010), which comprise the vast majority of the overall seahorse trade.

The economic viability of seahorse aquaculture remains a challenge to this business. The greatest challenge is wild-caught animals are often cheaper and more readily available than captive-raised seahorses (Koldewey and Martin-Smith 2010). This is especially true for bycatch fisheries, where no extra effort or costs go into harvesting seahorses (Koldewey and Martin-Smith 2010). Some traditional medicine consumers are also concerned that aquacultured seahorses are less medicinally effective compared to their wild-caught counterparts (Koldewey and Martin-Smith 2010). However, for aquarium hobbyists, captive-raised seahorses offer several advantages (Koldewey and Martin-Smith 2010). Captive-raised seahorses are better acclimated to *ex situ* environments, less prone to stress-related diseases, more adaptable to aquarium foods, and can be bred for colors, shapes, and sizes that appeal to hobbyists (Koldewey and Martin-Smith 2010). These advantages have led to better survival of seahorses in captivity and a more sustainable and humane aquarium trade.

Chapter 7
Giant Clams

An introduction to giant clam biology:

Giant clams are bivalve mollusks in the family Tridacnidae (Schneider and Foighil 1999). These animals are easily recognized by their ornate, fluted shells and enormous size. Tridacnids grow to be the largest bivalves in the world, with *Tridacna gigas* reaching up to 1.5 m in length and 263.4 kg in mass (Ruscoe 1962). Their large, attractive shells and bright mantle colorations have made them popular animals in the coral reef wildlife trade (Wabnitz et al. 2003). There are two genera of giant clams, *Tridacna* and *Hippopus,* containing ten species distributed from the east coast of Africa to the Polynesian Islands, 30°E to 120°W and 36°N to 30°S (bin Othman et al. 2010, see Table 6 for species names and ranges). As with many marine animals, there is genetic evidence for cryptic species and true diversity may exceed the ten nominal species. Genetic analyses of *Tridacna maxima*, for example, revealed four genetically-distinct groups, each of which may be a separate species, throughout the “species” range (Nuryanto and Kochzius 2009). These four clades were geographically partitioned, with one clade per region in the Red Sea, eastern Indian Ocean and Java Sea, Indonesia throughflow and seas in the east of Sulawesi, and Western Pacific (Nuryanto and Kochzius 2009). Potentially cryptic species also occur in the boring giant clam, *T. crocea* (DeBoer et al. 2008), and possibly other groups.

Table 6: Species of giant clams, their distribution, and occurence in the coral reef wildlife trade.

Scientific Name	**Common Name**	**Range**	**Occurrence in Trade?**
Hippopus hippopus	Bear paw, horse hoof, or strawberry giant clam	West coast of Malaysian Pennisula, South China Sea, Coral Sea, southern Japan, southern Australia, and east to Micronesia and Palau	Yes
Hippopus porcellanus	China clam	Only found in parts of Indonesia, the Philippines, and Palau	Uncertain
Tridacna costata	N/A	Northern Red Sea	No
Tridacna crocea	Crocus or boring giant clam or saffron-colored clam	West coast of Malaysian Pennisula, South China Sea, Coral Sea, southern Japan, southern Australia, and east to Micronesia and Palau	Yes, especially in the aquarium trade
Tridacna derasa	Smooth giant clam or southern giant clam	As far west as the eastern Malasian Pennisula, ranges throughout Coral Triangle, souther Japan, Great Barrier Reef, Papua New Guinea and as far east as the Cook Islands.	Yes
Tridacna gigas	True giant clam	As far west as the western Malasian Pennisula, ranges throughout Coral	Yes

		Triangle, souther Japan, Great Barrier Reef, Papua New Guinea and as far east as the Cook Islands.	
Tridacna maxima	Rugose or small giant clam	From east Africa, throughout the Indian Ocean and as far east as Polynesia and Pitcairn	Yes, especially in the aquarium trade
Tridacna mbalavuana = *T. teveroa*	Deep water devil clam or tevero clam	Northern Tonga Islands and eastern Fiji Islands	Uncertain
Tridacna rosewateri	Bénitier de rosewater	Found only in Saya de Malha Bank, Mauritius in the Indian Ocean	Uncertain
Tridacna squamosa:	Fluted or scaly giant clam	East coast of Africa to Polynesian Islands; north to Japan and south to the Great Barrier Reef	Yes, especially in the aquarium trade

Giant clams occur in shallow-water coral reef and shoreline communities. Most species are found from just below the surface to 20 m deep, but *T. teveroa* occurs in deeper habitats. Most species occur in the tidal and sub-tidal zone and some can even tolerate exposure to air for extended periods of time (i.e., up to 27 hours; Mingoalicuanan 1993). They typically occur at low densities, between 10^{-3} and 10^{-5} individuals per m^2, but occasionally populations exceed 100 individuals per m^2 (bin Othman et al. 2010). Tridacnids have a tendency to aggregate together and exhibit positive chemotaxis for conspecifics – in other words they follow chemical cues to move towards other giant clams (Huang et al. 2007). This is possibly a means of increasing spawning density and avoiding Allee effects (Huang et al. 2007). It may also be a means of protecting against predators or stabilizing substrate (Huang et al. 2007). Giant clams provide structural complexity on coral reefs and act as habitat for a number of reef associated fish and invertebrates (e.g., Craig et al. in press). Most species live exposed on the reef, but *T. crocea* actually burrows into the reef substrate (Todd et al. 2009). Giant clams are long-lived and largely sedentary. These characteristics enable scientists to reconstruct historical environmental and climatic conditions by examining the chemical composition of their calcium carbonate shells (Watanabe et al. 2004, Aubert et al. 2009).

Tridacnid clams have two modes of nutrition: filter feeding and symbiosis with photosynthetic dinoflagellates in the genus *Symbidonium* (zooxanthellae). Giant clams form associations with several different lineages of *Symbiodinium*, including various types of clades A and C (Baillie et al. 2000). Within a clam there is typically one dominant *Symbiodinium* lineage and one or more additional symbiont types present at lower densities (Carlos et al. 2000). Photosynthetic *Symbiodinium* cells release glucose to their hosts (Ishikura et al. 1999) and these photosynthetic symbionts provide the majority of the clam’s energetic needs (Trench et al. 1981, Klumpp et al. 1992, Hawkins and Klumpp 1995). Around 80–100% of their energy for growth and metabolism is derived from their symbionts, making these clams primarily phototrophic (Klumpp and Griffith 1994, Klumpp and Lucas 1994). Even the deep-dwelling species *T. teveroa* derives most of its energy from its symbionts (Klumpp and Lucas 1994). As with corals, anemones, and other hosts of *Symbiodinium*, efficient nutrient recycling between host and symbiont allows high growth and productivity in the otherwise low-nutrient reef waters (Muscatine and Porter 1977, Hawkins and Klump 1995). The importance of phototrophy generally increases with clam size (Klumpp and Griffith 1994).

Despite the energetic importance of symbiosis, giant clams are still efficient filter feeders. Heterotrophy can supply a great deal of the required energy for metabolism and growth, especially in young and rapidly growing clams (Klumpp et al. 1992). Jantzen et al. (2008) examined co-occurring *T. maxima* and *T. squamosa* in the Red Sea. On the reefs examined, *T. maxima* occurred in the shallow reef crest whereas *T. squamosa* lived on the lower fore-reef slope (Jantzen et al. 2008). Based on measurements of the clams' photosynthetic rates, shallow *T. maxima* was able to meet all of its metabolic needs through photosynthesis (Jantzen et al. 2008). By contrast, *T. squamosa* was mixotrophic, obtaining energy from both photosynthesis and filter feeding (Jantzen et al. 2008). The different dietary modes of the two species may explain their different distributions on the reef and ability to co-exist in the same environment (Jantzen et al. 2008).

Compared to other bivalves, giant clams have evolved unique body plan and structures in order to accommodate *Symbiodinium* (Venn et al. 2008). The body of all tridacnids is rotated 180 degrees relative to the shell hinge (Venn et al. 2008). This results in the gape of the shell revealing the mantle and siphon instead of the foot as is the case for other bivalves (Venn et al. 2008). Furthermore, the digestive tract of tridacnids harbors *Symbiodinium*. Symbionts reside in a series of dichotomously branching diverticula of the stomach that project outwards toward the light like the branches of a tree (Norton et al. 1992). Unlike symbioses between *Symbiodinium* and cnidarians, symbionts are extracellular in giant clams (Venn et al. 2008). The clam expands its mantle by day to expose the *Symbiodinium* cells to sunlight. Giant clams have several hundred pinhole eyes on the exposed region of their mantle (Land 2003). These eyes enable the clam to withdraw or extend their mantle in response to changes in daylight or nearby movement (Land 2003).

Giant clams accumulate large amounts of dimethylsulfoniopropionate (DMSP) from their resident *Symbiodinium* communities (Hill et al. 2004). The level of DMSP within giant clams is the highest recorded concentration within an animal (Hill et al. 2004). High levels of DMSP are cause for concern because giant clams are popular as food. After a giant clam is collected and killed, its meat rapidly accumulates a foul odor and taste. This taste and odor likely occurs from breakdown of DMSP in the clam's tissues into dimethylsulfide (Hill et al. 2004). Because of this, giant clams are typically consumed locally. In fact, large-scale commercial aquaculture of giant clams as food has been largely unsuccessful because it is so difficult to preserve the clam tissue long enough for overseas shipping.

All tridacnids are either simultaneous or sequential hermaphrodites (Keys and Healy 1999). They reproduce by broadcast spawning sperm and eggs into the water column, followed by external fertilization (Lucas 1988). Some species are even capable of self fertilization (Murakoshi and Hirata 1993). Fertilized eggs develop into a veliger larva that disperses for approximately 9–10 days and then settles onto a suitable location (Copland and Lucas 1988, Lucas 1988). Most settlement cues are unknown, but *T. squamosa* larvae prefer substrates of crustose coralline algae-covered coral rubble (Neo et al. 2009).

Genetic studies have examined the dispersal capacity and genetic connectivity of some tridacnids. In *T. maxima*, data from allozymes indicated low genetic structure and evidence for a high dispersal capacity along the Great Barrier Reef and western Coral Sea (Benzie and Williams 1992a,b). By contrast, *T. derasa* from the Great Barrier Reef had high genetic structure reflecting low dispersal ability indicating that these dispersal patterns vary by species (Macaranas et al. 1992). At broad geographic scales, there is often isolation between populations. An example of this comes from *T. maxima* in the Indo-Pacific. There are at least four genetic breaks between *T. maxima* populations, with some populations diverging to the point of incipient speciation (Nuryanto and Kochzius 2009).

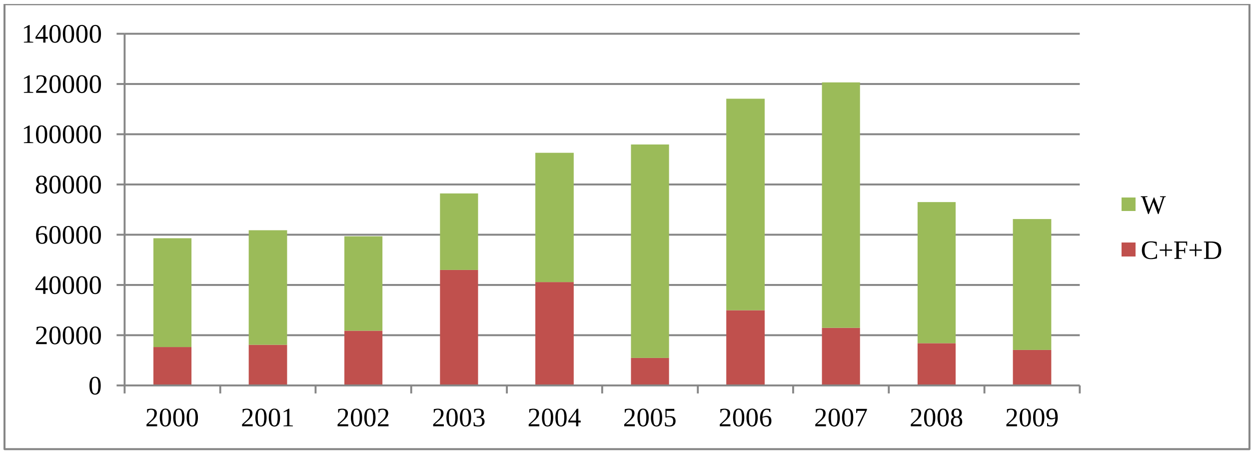
Several studies have examined dispersal capacity in the boring giant clam, *T. crocea*. In the Philippines, *T. crocea* has high dispersal capacity, but low realized dispersal (Juinio-Menez et al. 2003, Ravago-Gotanco et al. 2007). DeBoer et al. (2008) used molecular genetic tools to examine *T. crocea* across much of its range. This analysis revealed three deeply-diverged groups, with phylogenetic separation concordant with geography (DeBoer et al. 2008). The three clades included central Indonesia, Sumatra, and northern Papua (DeBoer et al. 2008). The genetic distance between clades represented strong and significant dispersal barriers. DeBoer et al. (2008) proposed that each clade could be a cryptic species deserving of special conservation status due to its evolutionary distinctiveness. Kochzius and Nuryanto (2008) also examined *T. crocea* population structure across the Indo-Malay Archipelgo and found strong population genetic structure with evidence for limited dispersal. At least four major phylogeographic groups of *T. crocea* were detected, including the eastern Indian Ocean, Java Sea, South China Sea/Indonesian throughflow/seas in the east of Sulawesi, and western Pacific Ocean (Kochzius and Nuryanto 2008). In both studies, populations were highly structured. The average dispersal distance of *T. crocea* larvae is approximately 25–50 km, which is considerably lower than their theorhetical-maximum capability based on larval duration and current velocity (DeBoer et al. 2008). Notably, the marine protected area system in the Indo-Malay Archipelgo did not provide sufficient coverage to maintain connectivity between reefs and regions (Kochzius and Nuryanto 2008).

Upon settlement, larvae metamorphose into juvenile clams. *Symbiodinium* cells are acquired after metamorphosis (Hirose et al. 2006). The young clams then grow rapidly for several years before reaching reproductive maturity, at which point the clams grow more slowly (Pearson and Munro 1991). For example, *T. gigas* reaches adulthood after 6 to 9 years (Okuzawa et al. 2008). Giant clams are incredibly long-lived; lifespan in the wild is estimated to exceed 100 years.

Collection, trade, and over-exploitation of giant clams:

Humans have harvested giant clams as food since prehistoric times (Fitzpatrick and Donaldson 2007). Giant clams have huge adductor muscles which hold their shells closed and this muscle is considered a delicacy in East Asia, where this meat commands a high price (i.e., $50 U.S. per kg of meat) and is often eaten raw. Today, giant clams are taken from the wild (1) for their meat as food, delicacies, and aphrodisiacs, (2) for their shells for curios, ornaments, water vessels, pots for plants, and home decorations, and (3) as pets for the marine aquarium industry (Wells 1997, Hodgson and Liebeler 2002, Wabnitz et al. 2003, Gomez and Mingoa-Licuanan 2006, Craig et al. in press).[1] Over the past 30 to 40 years, wild stocks of giant clams have been over collected and heavily depleted to supply these three trades (Wabnitz et al. 2003, CITES 2004a,b,c,d). Collection has driven giant clams to localized extinction in the Philippines, Indonesia, Micronesia (Gomez et al. 1994, Lucas 1994), Malaysia (Tan and Zulfigar 2003), Singapore (Guest et al. 2008) and elsewhere. The largest giant clams, including *T. derasa*, *T. tevoroa*, and most especially *T. gigas*, are the species most severely affected by this intense exploitation (Wabnitz et al. 2003, CITES 2004b,c, Okuzawa et al. 2008).

Figure 14: Quantity (number of clams, y axis) of *Tridacna* spp. imports by source and year (x axis). (W=Wild caught, C+F+D=captive bred, captive born, and bred in captivity for commercial purposes). Figure taken from Craig et al. (in press).



Giant clams one of the most heavily traded invertebrates in the world (Craig et al. in press). Wild-caught tridacnids are collected primarily in Vietnam, Solomon Islands, Tonga, Fiji, Vanuatu, the Marshall Islands, and Micronesia (Wabnitz et al. 2003, Craig et al. in press). The Philippines was once a major source country, but export and trade in giant clams has been banned there since 1996 (Wabnitz et al. 2003). There are also giant clam aquaculture programs that export clams from Solomon Islands, the Marshall Islands, Fiji, and Tonga (Wabnitz et al. 2003). Giant clam meat, shells, and whole animals are imported primarily by the U.S., European Union, Hong Kong, and Japan (Wabnitz et al. 2003). Most of the giant clams collected for the aquarium trade come to the U.S. (e.g., 70% of imports in 2002; Craig et al. in press). Despite the success of giant clam aquaculture operations, the majority of tridacnids imported into the U.S. are still wild caught (Craig et al. in press, Figure 14). The most popular aquarium species are those with brightly colored mantles: *T. maxima*, *T. crocea*, and *T. derasa* (Wabnitz et al. 2003). From 2002 to 2007, giant clam imports into the U.S. grew from 60,000 to

---

[1] Note that the majority of giant clams in trade are collected for human consumption as food. Food fisheries are not the subject of this report and will not be discussed in detail. Because giant clams are used by multiple trades, it is not possible to disentangle the impacts of collection for food, décor, and aquarium purposes. The text in this section examines over-exploitation of giant clams by all three industries. As a result, the impacts reviewed in this report cannot be attributed solely, or even primarily, to the trade in coral reef wildlife for ornamental purposes.

120,000, but have since decreased (Craig et al. in press), possibly due to the global economic downturn.

Because giant clams are sessile, brightly colored, and found in shallow water, they are easy to find and collect (Craig et al. in press). Giant clams take years to reach reproductive maturity, reproduce sporadically, and have low recruitment rates, all of which hinder the ability of tridacnid populations to recover from heavy collection (Wabnitz et al. 2003). As of 2010, the IUCN Red List of Threatened Species listed four tridacnid species as vulnerable (i.e., *T. derasa*, *T. gigas*, *T. rosewateri*, and *T. tevoroa*) and four as low risk/conservation dependent (i.e., *H. hippopus*, *H. porcellanus*, *T. maxima*, and *T. squamosa*). Only *T. crocea* was listed as a species of lower risk/least concern. At that time, *Tridacna costata* was not yet a recognized species, but Richter et al. (2008) proposed “Critically Endangered” as the correct category. No tridacnid species has been reassessed since 1996, making it challenging to estimate if the conservation status of giant clams has become better, deteriorated, or remained stable (Craig et al. in press).

Tridacna gigas, the world’s largest bivalve, provides an illustrative example of giant clam decline. *T. gigas* has been heavily targeted as food, and as a result, the species suffered significant reductions and local extinctions throughout its range (Wabnitz et al. 2003). For example, populations have been greatly reduced in Japan, Taiwan, Tuvalu, Micronesia, and Vanuatu (Wabnitz et al. 2003). Full-grown *T. gigas*, those individuals approaching 1.5 m in length, no longer exist in the wild (Hodgson and Liebeler 2002). The largest shells can be found in museums and churches where they are used as storage vessels for holy water (Hodgson and Liebeler 2002). This species was listed under CITES Appendix II over 25 years ago, yet it still remains vulnerable today (Craig et al. in press). Very few countries still export *T. gigas*, with the majority of exports coming from Tonga (Craig et al. in press). Despite this, the majority of *T. gigas* imports into the U.S. are still from wild populations (Craig et al. in press). Imports of *T. gigas* in the U.S. have declined from approximately 4,000 individual clams in 2005 to about 1,000 in 2009 (Craig et al. in press) This decline has been attributed to a lack of available animals to collect (Craig et al. in press)**.**

Hodgson (1999) conducted a global survey of coral reefs, using giant clams as indicators of overfishing and curio collection. Giant clams were missing from more than 90% of reefs surveyed in both the Indo-Pacific and Red Sea, locations where they were expected to occur (Hodgson 1999). Tridacnids are large, sessile, and highly-visible species, and as a result, their absence from the surveys could not be attributed to an oversight by the field researchers. The mean density of giant clams was just eleven clams per reef in the Indo-Pacific (Hodgson 1999). Sites in the Red Sea and Australia with low collection impacts had 150–250 giant clams per reef; the absence of clams at other locations suggested that over collected was common (Hodgson 1999).

Hodgson and Liebeler (2002) followed in the footsteps of the 1999 study, using giant clams as indicators for overfishing, curio collection, and harvesting for home aquaria. They monitored 869 reefs from 1997 to 2001 and found an average of 3.9 ± 19.1 giant clams per $100m^2$ of coral reef surveyed (Hodgson and Liebeler 2002). The abundance of clams was skewed by several sites that harbored high numbers of *T. crocea*, a relatively small species that burrows into sediment and is therefore harder to collect. Tridacnids were missing from 29% of surveyed reefs (Hodgson and Liebeler 2002).

There were significantly ($p \leq 0.01$) more clams inside of marine protected areas than in unprotected areas, indicating that collection was responsible for their low abundance. Comparison of areas where aquarium collection occurred to non-fished areas revealed higher numbers of clams in the locations without collection (i.e., 4.2 ± 21.4 per 100 m^2 in non-fished areas versus 1.6 ± 3.9 per 100 m^2 in fished areas). Clam abundance was also lower in areas with cyanide fishing than in areas without this destructive fishing practice (Hodgson and Liebeler 2002).

Many giant clam populations in Southeast Asia are either in dramatic decline or extinct (bin Othman et al. 2010). In Malaysia, five out of six species had declining or unstable populations (Tan and Zulfigar 2003). In Singapore, *Hippopus hippopus* and *T. maxima* were locally and functionally extinct (Guest et al. 2008). The remaining species, *Tridacna crocea* and *T. maxima*, had very small populations left in Singapore (Guest et al. 2008). In Thailand, two out of three giant clam species were very scare (Thamrongnavasawat et al. 2001). Cinner et al. (2006) compared periodic closure areas to open collection areas in Indonesia and Papua New Guinea. At Muluk, Papua New Guinea, there were no giant clams whatsoever in the open area whereas there were 410 ± 288 clams per hectare in the area with rotating closures. Juinio et al. (1989) surveyed reefs in the Philippines and found few large tridacnids with many species over-exploited or locally extinct due to ornamental shell collection and subsistence fishing. *Tridacna gigas* was driven to virtual extinction in the Philippines due to collection, with juvenile shells recieving a price of $25 to 45 U.S. per shell in recent years (Gomez and Mingoa-Licuanan 2006). At one time the Philippines dominated international exports of giant clams, but the country prohibited all exports starting in1996 (Wabnitz et al. 2003). Despite the ban, poaching and illegal trade continued in the Philippines, until at least 2002 (Craig et al. in press). For other Southeast Asian countries, including Cambodia, Laos, Myanmar, and Vietnam, little is known about the status of giant clams (bin Othman et al. 2010).

The conservation status of giant clams in Australia, Melanesia, Micronesia, and Polynesia is less dire than the situation in Southeast Asia (bin Othman et al. 2010). Giant clam diversity naturally decreases with distance from the coral triangle, from seven species in Australia to one species in the central Pacific (bin Othman et al. 2010). Although there remain areas with an abundance of giant clams, populations of giant clams in the Pacific region are generally dwindling (bin Othman et al. 2010). For example, Helen Reef in Palau harbors just 1 *T. gigas* individual per hectare (Weng and Guilbeaux 2000). Similarly, Zann (1994) noted that giant clams are overfished and locally extinct in some southwestern Pacific islands. Ironically, aquaculture operations in Micronesia and the Hawaiian Islands have even introduced giant clams to areas where they do not naturally occur (bin Othman et al. 2010).

Bin Othman et al. (2010) reviewed the distribution and status of giant clam species. The general trend revealed from bin Othman et al.'s analysis is that the density of giant clams is low throughout the world. Giant clam populations have been depleted by fishing pressure and habitat destruction (bin Othman et al. 2010). Fishing and habitat destruction have been accelerated by three factors: (1) human population growth, (2) improved access to clams due to improvements in technology, and (3) expanded trade for food, shells, and live animals for aquariums (Lucas 1994). Furthermore, illegal trade and poaching continue to be major problems (Lucas 1994, Craig et al. in press). Efforts to monitor giant clam population status are limited (bin Othman et al. 2010).

Although the ornamental trade contributes to giant clam decline throughout the world, collection for food fisheries is one of the largest problems facing these species. In parts of the Egyptian coast of the Red Sea, giant clams are collected by reef top gathering, a form of subsistence fishing (Ashworth et al. 2004). Populations of giant clams were very low in areas where reef top gathering takes place compared to protected areas (Ashworth et al. 2004). Control species that were not targeted by reef top gathering did not show a difference between collection and protected areas, indicating that subsistence fishing was responsible for the status of giant clams (Ashworth et al. 2004). In this case and others, food fisheries, not the ornamental trade, are the major threats to giant clams.

Collection of giant clams can also cause severe damage to the structural integrity of coral reefs. *Tridacna crocea* is a small species of giant clam that burrows into reef sediments and over time becomes part of the reef structure. Collection of *T. crocea* causes the destruction of corals and local damage to the reef (Wells 1997). Such destructive practices contribute to the overall decline of many coral reefs (Burke et al. 2011).

Giant clams are imperiled by many threats beyond over-exploitation for food fisheries, the curio/décor trade, and the aquarium trade. As with many coral reef organisms, climate change is one of the greatest threats. Giant clams form symbioses with *Symbiodinium*, the same group of dinoflagellate symbionts that associate with corals. As a result of these associations, giant clams are prone to bleaching under high temperature stress (Buck et al. 2002, Leggat et al. 2003, 2004, Venn et al. 2008). Leggat et al. (2003) found that the 1998 mass bleaching event decreased *Symbiodinium* populations by 30 fold in their *T. gigas* hosts. This resulted in profound physiological changes in *T. gigas*, including altered haemolymph pH and glucose concentration as well as an impaired ability to assimilate ammonium (Leggat et al. 2003). Bleaching also causes a loss of chlorophyll per cell in giant clam symbionts, retention of only small zooxanthellae, a release of ammonium, and a decreasing ability for giant clams to absorb ammonium (Buck et al. 2002). Ultimately, bleaching deprives giant clams of their primary energy source leading to starvation and death.

Unfortunately the threats facing giant clams do not end there. In Indonesia, coral mining has been shown to dramatically decrease giant clam populations (Caras et al. 2009). Land-based pollution, heavy metal pollution, sedimentation, cyanide fishing, and blast fishing with dynamite also cause population declines (Elfwing et al. 2001, bin Othman et al. 2010, Craig et al. in press). Parasites, diseases, and predators threaten both wild and aquacultured giant clams (Govan et al. 1993, Newman et al. 1993, Sutton and Garrick 1993, Cumming and Alford 1994, Okuzawa et al. 2008). Ingestion of toxic cyanobacteria can pose a threat to giant clams and poison people who consume their meat (Laurent et al. 2008, Méjean et al. 2010). Reducing the threat posed by over-exploitation would better enable giant clam populations to withstand the many threats they face.

Giant clam conservation:

Many different conservation measures have been attempted in order to reverse the severe declines in giant clam populations throughout the world. Over the past 30 years, giant clam aquaculture has been one of the most successful of these efforts (Heslinga and Watson 1985, Hodgson and Liebeler 2002), including projects in Tonga, Palau, Fiji, Solomon Islands, the Cook Islands, and the Philippines (bin Othman et al. 2010). Aquaculture is used to supplement wild sources and thereby reduce collection pressure on wild populations. It is also used to stock wild populations in an effort to restore coral reefs to a less anthropogenically-impacted state. Giant clams are highly amenable to commercial culturing due to their high spawning rates in captivity, short larval duration, and feeding through symbiosis and filter feeding leading to little maintenance and rapid growth rate to maturity (Bonham 1965, Beckvar 1981, Foyle et al. 1997). The aquarium market is generally more lucrative for aquaculture operations than the food market because of the way animals are priced (Gomez and Mingoa-Licuanan 2006). Aquarium buyers purchase clams by the individual, not by the kilogram (Bell et al. 1997, Gomez and Mingoa-Licuanan 2006). Therefore, aquaculture has not taken over the giant clam food fishery. Another limiting factor to aquaculture for the food industry is that demand for giant clam meat vastly exceeds cultured supply (Hodgson and Liebeler 2002). As a result, most internationally traded giant clams come from wild sources (Wabnitz et al. 2003, Craig et al. in press).

The Philippines once dominated giant clam exports for the shell and aquarium trades (Wabnitz et al. 2003). In 1996, in response to the CITES Appendix II listing of all giant clam species, the Philippines banned giant clam exports (Wabnitz et al. 2003). Additionally, the Philippines has ongoing restocking programs through the Marine Science Institute of the University of the Philippines (Lebata-Ramos et al. 2010). Clams are grown in aquaculture facilities until they reach a certain size and are then transplanted onto the reef. To date, over 45,000 *T. gigas* and 30,000 individuals of other species have been reintroduced to more than 40 sites across the Philippines (Gomez and Mingoa-Licuanan 2006, bin Othman et al. 2010, Lebata-Ramos et al. 2010). Following introduction into the wild, many giant clams survived. Poaching, typhoons, and fouling resulted in losses of up to 27% of the introduced tridacnids, with the remaining ≥73% of individuals surviving (Gomez and Mingoa-Licuanan 2006). The highest survivorship was achieved with *T. gigas* of at least 8 to 10 cm in size (Okuzawa et al. 2008). Restocking in the Philippines has successfully increased coral and giant clam cover resulting in an increase in fish diversity, abundance, and recruitment (Cabaitan et al. 2008). However, restocking efforts failed in the Vasayas region due to problems with poaching combined with local sociological and environmental factors (Gomez and Mingoa-Licuanan 2006, Lebata-Ramos et al. 2010). Efforts are currently underway to improving restocking success on protected reefs in the Vasayas (Lebata-Ramos et al. 2010). The primary challenges to the restocking efforts in the Philippines are transportation of live animals, selection of an appropriate stocking location, and protecting clams from predation and poaching until they reach reproductive size (i.e., 6 years for *T. gigas*; reviewed in Okuzawa et al. 2008, Craig et al. in press). Protection against poaching can be enhanced by involving local community members in monitoring and actively protecting the aquacultured clams (Gomez and Mingoa-Licuanan 2006).

Southeast Asia is the epicenter of giant clam diversity and the nations of this region have tried several different approaches such as the Philippines' collection prohibition and restocking efforts. Preliminary work in Singapore suggests that restocking efforts could be successful despite severely overharvested populations, high sedimentation from poor land use practices, and no evidence for recent giant clam recruitment (Guest et al. 2008). In Japan, declines in giant clam populations from 1975 to 1995 led to stricter regulations and the introduction of an aquaculture and restocking program (Okada 1997, bin Othman et al. 2010). In Indonesia, seven marine protected areas were recently established covering 9,000 square kilometers in the Raja Ampat Archipeligo in western Papua (MPA News 2007). The structure of this reserve area is conducive to the limited dispersal in species such as *T. crocea*, giving the marine protected area network a high probability of success in maintaining connectivity and potentially restocking depleted populations (DeBoer et al. 2008). In Papua New Guinea, periodic closures of the fishery appear to be an effective measure, even for long-lived species like giant clams (Cinner et al. 2006). Because the harvest of giant clams generally occurs through small-scale operations, local-community-based management efforts and designated protected areas are the conservation measures with the highest probability of success (Johannes 1998, bin Othman et al. 2010).

In the Pacific, Solomon Islands once only allowed cultured specimens to be exported (Wabnitz et al. 2003), but wild-caught giant clams have appeared on export forms. Furthermore, marine conservation areas do not appear to be effective in preventing overharvesting for local subsistence fisheries. A comparison of conservation areas and harvesting areas found no difference in *T. maxima* abundance between the two types of sites (Lincoln-Smith et al. 2006). Elsewhere, Vanuatu banned the collection and export of *T. crocea* and set quotas for all other giant clam species (Wabnitz et al. 2003). Vanuatu also has many different local reserves with different giant clam harvest regulations and this situation enabled a comparison of the success of different management practices (Barlet et al. 2009). In Vanuatu, certain areas banned harvest altogether (i.e., permanent reserves) whereas others had periodic closures (i.e., rotating closures) or no restrictions whatsoever (i.e., open areas) (Bartlet et al. 2009). Comparisons between sites revealed that tridacnid abundance was higher in the rotating closure areas than the adjacent open areas that were actively fished (Bartlet et al. 2009). However, there were no differences found between permanent reserve and open fishing areas or rotating closure and permanent reserve areas (Bartlet et al. 2009). This result was somewhat counterintuitive and may be caused by some open areas being actively managed by locals under a marine-tenure system (Bartlet et al. 2009). Open areas adjacent to permanent reserves had higher densities of clams than areas adjacent to periodic closures (Bartlet et al. 2009). Rotating closure areas were also examined before and after a collection season. There were large decreases in giant clam numbers after harvest (Bartlet et al. 2009). After harvest, there was also no difference in giant clam abundance between the periodic harvest area and the control sites (Bartlet et al. 2009). These data highlight the vulnerability of tridacnids to overharvesting and suggest that rotating closures are not sufficient measures to prevent over-exploitation (Bartlet et al. 2009). Dumas et al. (2010) also looked at the effect of closures in Vanuatu and found that after heavy collection, 4 years of closures was an insufficient amount of time to see statistically significant population increases in giant clams. If combined with other protection measures, such as limiting effort or only opening the fishery to harvest when populations exceed a set threshold, then periodic closures might be more effective (Bartlet et al. 2009).

As with other marine fisheries, protected areas or marine protected areas are popular measures to prevent over collection of giant clams and maintain viable populations to resupply fished areas. In Egypt, concerns about overfishing of fin fish led to the establishment of no-take zones (NTZs), known as the Nabq Managed Resource Protected Area, on the southern Gulf of Aqaba coast. These NTZs also protect giant clams and other invertebrates from harvest. Ashworth et al. (2004) looked at the effects of reef-top gathering of giant clams in NTZs compared to collection in Egyptian areas of the Red Sea. They found much lower abundances of clams in collection areas when compared to protected areas (Ashworth et al. 2004). Control species didn't follow a similar pattern indicating that population differences were due to harvesting (Ashworth et al. 2004). It remains to be seen whether or not the NTZs supply collection areas with new giant clam recruits. At a minimum, however, they prevent the local extirpation of giant clams in this region.

Concern about the global decline in giant clam populations led to the listing of all tridacnid species on Appendix II of CITES. Along with sea turtles (Appendix I), seahorses (Appendix II), humphead wrasse (Appendix II), and scleractinian corals (Appendix II), giant clams are one of the few coral reef species to have received CITES protection. Appendix II requires (1) a permit to export listed species, (2) countries to submit an annual report on the quantities of giant clam species involved in trade, and (3) for there to be a non-detriment finding insuring that trade does not have long-term detrimental consequences for the species. Unfortunately, implementation and enforcement of CITES protections is relatively weak and some source countries are not CITES signatories. In addition to the regulations covered under CITES, the European Union independently reviews the status of species in trade and bans the import of wild specimens from locations where the sustainability of collection is questionable. As of 2003, this included *H. hippopus* from New Caledonia, *T. crocea* from Vietnam, *T. derasa* from Tonga, *T. gigas* from Guam, Micronesia, Fiji, Indonesia, the Marshall Islands, Palau, Papua New Guinea, and Vanuatu, *T. rosewateri* from Mozambique, *T. squamosa* from Fiji, Mozambique, Vanuatu, and *T. trevora* from Tonga (Wabnitz et al. 2003). If the other large-volume importing countries, like the U.S., were to implement higher standards, it could transform the market for coral reef wildlife for ornamental purposes (Tissot et al. 2010).

Chapter 8
Scleractinian Corals

An introduction to stony coral biology:

The cnidarian order Scleractinia consists of 17 families, 109 genera, and more than 800 species (Veron 2000), at least 140 of which are involved in the coral reef wildlife trade (Wabnitz et al. 2003). Because of widespread hybridization between closely-related "species," there is considerable evolutionary and taxonomic complexity to corals (Veron 1995, Willis et al. 2006). The tremendous diversity combined with prevalent hybridization makes it challenging to accurately identify stony corals (Jones 2008).

Reef-building corals occur in shallow hard-bottom marine environments throughout the tropical and sub-tropical oceans (Kleypas et al. 1999). Coral reef ecosystems are typically found in locations that are warmer than 18 °C throughout the year, that experience high light intensity, and where the water is highly saturated in aragonite, a mineral form of calcium carbonate (Kleypas et al. 1999).

The common characteristic of scleractinians is an aragonite skeleton. This "stony" or limestone skeleton earned scleractinians their common name, "stony corals." Corals secrete this structure on the underside of their thin tissues (Barnes and Chalker 1990). Coral skeletons grow in a wide variety of forms, including branching, tabulate, digitate, boulder, mushroom, foliase, encrusting, and plate-like forms, among others. Cumulatively, the skeletal growth of stony corals and other marine calcifying organisms forms the structural framework of coral reefs. This structure provides the home, nursery, and feeding grounds for myriad other species. Scleractinian corals grow by adding skeleton and polyps at a rate approximately 0.5 to 20 cm per year. Most species grow indefinitely and can reach enormous sizes.

Corals are ancient animals with a relatively simple anatomy. Each coral individual is a structure known as a polyp. A typical polyp is small in size, ranging from 1–3 mm in diameter. A polyp consists of a ring of tentacles, a mouth that serves as the site of ingestion and excretion, and a body trunk that contains a simple digestive cavity, the gastrovascular cavity. Most coral reef scleractinians are colonial animals consisting of many interconnected polyps.

Polyps actively feed on zooplankton and detritus by extending their tentacles, particularly at night. The tentacles contain harpoon-like stinging cells known as nematocysts that both mechanically and chemically immobilize prey. Nematocysts also function in defense against coral predators. In addition to heterotrophic feeding, corals obtain energy by absorption of dissolved organic matter and through symbiosis with photosynthetic dinoflagellates in the genus *Symbiodinium* (Sorokin 1973, Trench 1974). These highly-diverse endosymbionts reside within the digestive cells of their hosts where they contribute up to 90% of the coral's metabolic needs (Muscatine et al. 1981, Coffroth and Santos 2005).

Corals can be either dioecious or hermaphroditic. Approximately 80% of scleractinian corals reproduce by mass spawning of sperm and eggs into the water column. Spawning typically occurs just a few nights per year, often in the late summer after a full moon. A fertilized egg matures into a planula larva that disperses (actively and passively) for several hours to several weeks before settlement in a suitable habitat. The location of settlement is highly important. As juveniles and adults, corals compete intensely with other sessile invertebrates for space on the reef.

The minority of corals, approximately 20%, have internal fertilization and brood larvae internally, with larval release happening shortly before settlement. Brooding corals tend to reproduce continually throughout the year. As a result of these varied strategies, some coral individuals and species can disperse great distances, whereas others have highly structured populations with limited dispersal. Corals are also capable of asexual reproduction through fragmentation.

Because of the diversity and varied biology of scleractinians, it is not possible to exhaustively review their biology here without resorting to overgeneralizations. Veron (2000) provides a comprehensive review of the identification, taxonomy, distribution, and basic biology of most reef corals. Here I will highlight the biology of a mushroom coral, *Heliofungia actiniformis*, as an example scleractinian species. This coral is one of the most heavily exploited scleractinians in the live coral trade (Green and Shirley 1999).

Heliofungia actiniformis is a fungid coral that occurs on flat reef substrates, including soft bottom and rubble habitats (Knittweis and Wolff 2010). It is typically found at depths of less than 16 m, often in turbid habitats (Knittweis and Wolff 2010). *Heliofungia actiniformis* ranges throughout the Indo-Pacific from eastern Sri Lanka to the western Pacific Islands, as far north as Okinawa and as far south as the southern Great Barrier Reef (Veron 2000). Unlike many coral reef scleractinians, adult *H. actiniformis* consist of a single large polyp with long fleshy tentacles that does not affix itself to the reef.

Reproductively, *H. actiniformis* is a brooding coral that releases well-developed larvae from the maternal parent (Abe 1937, Knittweis et al. 2009a). Larvae are negatively phototactic and settle within about 2 days (Abe 1937). Knittweis et al. (2009a) examined the genetic structure of *H. actiniformis* based on nuclear ribosomal genes and spacer regions. Gene flow of *H. actiniformis* was restricted among the investigated populations and significant isolation by distance (Knittweis et al. 2009a). There were five genetic regional groupings of *H. actiniformis* in Southeast Asia, including (1) Adi, (2) Barrang Lompo, Cebu, Gilli Trawangan, Manado, Kapoposang, Komodo, Lanyukang, Samalona, Sarappokeke, Pulau Sembilan, (3) Pulau Seribu, (4) Saboeda, and (5) Tilmuta (Knittweis et al. 2009a). The average distance at which dispersal was noted was 52 km (Knittweis et al. 2009a). Upstream populations, those acting as a source for other locations but without a supply of external recruits, were especially vulnerable to over collection or other threats (Knittweis et al. 2009a).

Following larval settlement, young *H. actiniformis* attach themselves to a substrate via a stalk, known as an anthocaulus (Bourne 1893). The anthocaulus forms multiple connected polyps which grow for a period of time (Knittweis et al. 2009b). Once reaching a size of 3–4 cm, individual polyps detach and become free-living (Knittweis et al. 2009b). Prior to detachment, the anthocaulus may generate new polyps (Hoeksema 1989). *Heliofungia actiniformis* polyps reach reproductive maturity at approximately 8 cm in size or at about 9 years of age (Knittweis et al. 2009b). As a result of their reproductive mode and anthocaulus, the polyps often occur in natural clusters (Knittweis et al. 2009b). The natural population demographics exhibit a decline in abundance of individuals with age (Knittweis et al. 2009b). Knittweis et al. (2009b) found that approximately 42% of polyps were attached to an anthocaulus, with the abundance of older age classes growing increasingly smaller with colony age. Sedimentation, predation by fish and gastropod coralivores, competition for space, and collection for the aquarium trade are the most common sources of *H. actiniformis* mortality (Knittweis et al. 2009b).

Ecological impacts of the trade in scleractinian corals:

Although many species of scleractinian coral can be captive-bred or proliferated by fragmentation, the majority of internationally traded corals are still taken from the wild (Craig et al. in press, Figure 2). The primary source countries for wild caught corals have changed over time. During the 1980s, the Philippines was the center of coral collection for international trade (Ross 1984). Scleractinian corals were listed under CITES Appendix II in 1990, which led the Philippines to ban their collection and export. Indonesia has since become the primary source country for scleractinian corals; as of 2005, Indonesia supplied 91% of live corals to the world market (Green and Shirley 1999, Suharsono and Bruckner 2003, Wabnitz et al. 2003, Bruckner and Borneman 2006, Jones 2008, Figures 15-16). Other major exporters include Fiji (8% in 2005), Solomon Islands (less than 1% in 2005), and Tonga (less than 1% in 2005) (Green and Shirley 1999, Wabnitz et al. 2003, Jones 2008). The closure of the Philippine coral fishery also shifted some demand to Australia (Jones 2011). In recent years, China has become an exporter of coral products, particularly for the export of curios and jewelry that are processed in China but collected elsewhere (Craig et al. in press). Most live corals and coral products go to the U.S., Europe, and Japan. The U.S. is the largest consumer of corals for aquariums and curios (Moore and Best 2001, Jones 2008, Jones 2011). As of 2005, 65% of corals in trade were imported into the U.S., followed by the European Union with 24% of imports (Jones 2008).

Figure 15: The top exporting countries and percentage of global exports in the coral trade circa 2005. Figure taken from Jones (2008).

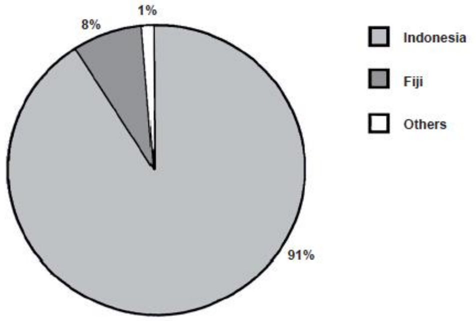
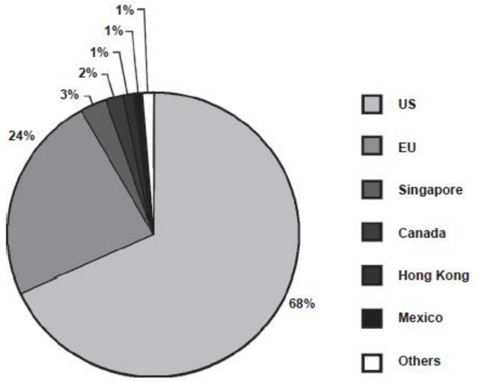


Figure 16: The top importing countries and percentage of global imports in the coral trade, circa 2005. Figure taken form Jones (2008).



The trade in ornamental corals has grown considerably over time (Green and Shirley 1999, Wabnitz et al. 2003, Jones 2008, Craig et al. in press). From 1988 to 2007, imports of live corals into the U.S. increased by 600% (Tissot et al. 2010). During that same time period, global imports increased by 1500% (Tissot et al. 2010). Wabnitz et al. (2003) estimated that 11 to 12 million pieces of coral were traded every year. Estimating the precise volume is difficult for several reasons: most stony corals are colonial animals that vary in size and can be fragmented into multiple pieces, reporting of trade volume varies between pieces and weight, live rock and corals are sometimes confused with one another, and the complexity of coral taxonomy hinders the accurate reporting of trade data. There is also a substantial illegal trade in corals that goes unreported (Green and Shirley 1999). Based on a 2006 study by the European Union Wildlife Trade Regulation, Jones (2008) estimated that the illegal trade in corals constituted 20% of the overall trade. However, because of its clandestine nature, it is impossible to precisely quantify the size of this illegal trade.

Throughout the 1980s and early 1990s, more than 90% of corals in trade were dead skeletons for use as home decorations and curios (Green and Shirley 1999). Since that time, however, the aquarium trade in live corals has developed into a robust global industry (Green and Shirley 1999). This change was largely driven by increased interest in marine home aquaria in North America and Europe as well as fast and frequent air travel from previously inaccessible parts of the world (Green and Shirley 1999). Collection for public aquariums constitutes a small percentage of the overall coral trade and public aquariums likely produce as many corals as they consume (Jones 2008). By contrast, more than 2 million households keep marine aquaria and these home hobbyists drive the demand for live corals (Green 2003). As a result, live aquarium corals now account for the majority of the coral trade (Green and Shirley 1999, Jones 2008).

Over 140 coral species are involved in the ornamental coral reef wildlife trade (Wabnitz et al. 2003). The ten most-traded coral genera, in order of volume, are *Acropora*, *Euphyllia, Goniopora*, *Trachyphyllia, Plerogyra, Montipora, Heliofungia, Lobophyllia, Porites,* and *Turbinaria* (Jones 2008, see alsoWabnitz et al. 2003). Although there is some overlap between the two trades, there are notable differences in the types of corals targeted by the aquarium trade versus the curio trades. Aquarium collectors primarily remove colorful species with large polyps (known as large polyp species or LPS), especially those that extend their tentacles during the day (Green and Shirley 1999, Wabnitz et al. 2003). By contrast, the curio and home décor trade targets species with attractive skeletal features, particularly corals with branching or mushroom forms (Green and Shirley 1999). Examples of coral genera commonly collected for the aquarium trade include *Euphyllia, Goniopora, Catalaphyllia, Trachyphyllia,* and *Heliofungia*, whereas *Fungia, Pocillopora, Porites,* and *Acropora* are more commonly taken for home décor (Green and Shirley 1999). A typical aquarium coral is 3 years old, with a mass of 206.1±13.1 g, a height of 5.86±3.1 cm, and an area of 180.1±10.1 cm^2 (Green and Shirley 1999, data presented as means ± 95% confidence intervals). The curio trade generally involves larger coral pieces than the live aquarium trade, with a typical home décor coral ranging from 13–18 cm in height (Ross 1984), compared to the approximately 6-cm size of aquarium corals (Green and Shirley 1999).

One illustrative example of the organization of the coral trade is that of the mushroom coral *Heliofungia actiniformis*. This coral has been harvested in the

Spermonde Archipelago of South Sulawesi, Indonesia since the mid-1990s (Knittweis and Wolff 2010). This coral is managed through a quota system for South Sulawesi, with the quota increased from 6,000 pieces in 2002 to 7,000 pieces in 2003, to 9,500 pieces from 2004–2006 (Knittweis and Wolff 2010). A total of 10 companies collect *H. actiniformis* in the Spermonde Archipelago, with each company employing an average of 5.6 ± 2.9 collectors (mean ± standard deviation; Knittweis and Wolff 2010). Collectors are typically men between the ages of 19 and 26 (Knittweis and Wolff 2010). They travel on 2–3 day trips to collect corals aboard company boats using scuba gear (Knittweis and Wolff 2010). Fishermen prefer to work at the mid-outer shelf zones where there are more abundant colorful polyps and visibility and working conditions are good (Knittweis and Wolff 2010). Many fishermen are unlicensed and actively avoid law enforcement patrols, making a portion of the trade illegal (Knittweis and Wolff 2010, see also Bentley 1998). There are also independent family groups that collect a wide range of organisms for various purposes, including corals for the ornamental trade (Knittweis and Wolff 2010). These independent family groups harvest animals while free-diving or diving using a hookah (Knittweis and Wolff 2010). Once corals are collected they are sold to middlemen for transportation to exporters (Knittweis and Wolff 2010). Middlemen fix prices and discard much of what is collected; Knittweis and Wolff (2010) observed large piles of discarded ornamental coral skeletons at holding facilities. High supply-chain mortality also occurs during export (Jones 2008). Paperwork delays, overcrowded transportation bags, and other problems during shipping can lead to low temperatures, poor water quality, and physical damage to corals that result in high rates of death (Jones 2008).

Removal of scleractinians for the ornamental trade is removal of the reef structure itself (McManus 2001). Collectors sometimes use crowbars to pry corals from the reef, thereby further damaging the reef structure (Ross 1984). Corals are relatively fragile organisms and they can be damaged or killed during the collection of other coral reef wildlife using destructive moxy nets, poisons like cyanide, or other harmful practices (Öhman et al. 1993). Because stony corals form the structural and trophic framework for the ecosystem, coral (and live rock) collection has potentially serious consequences for this ecosystem; Table 7 provides a summary of potential ecological and socio-economic problems in the stony coral trade.

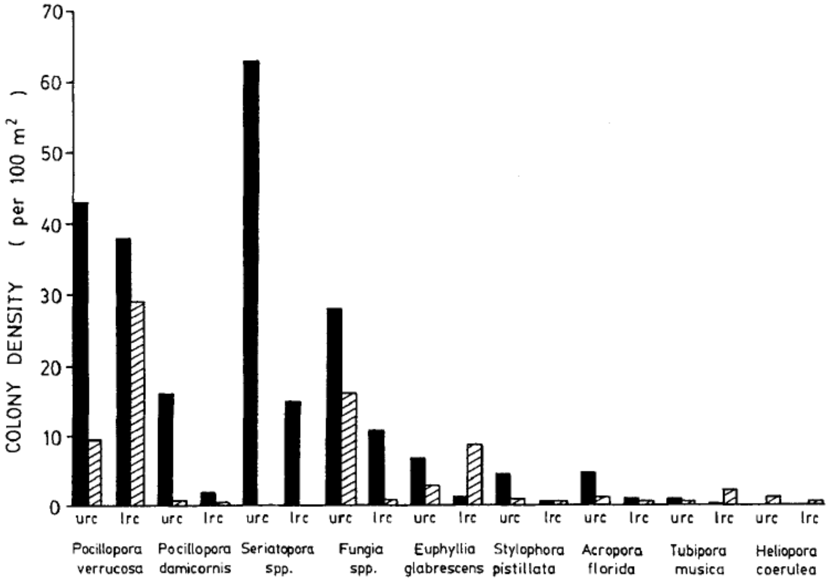
Table 7: Potential ecological and socio-economic effects of the ornamental coral trade. Table adapted from Knittweis (2008) and Bruckner (2002).

Potential Impact	**Description**
Effect on target population	Over-exploitation and localized extinctions
	Reduced reproduction and recruitment; removal of large reproductive colonies for home décor removes the main source of recruits; removal of small, pre-reproductive colonies for the aquarium trade prevents recruits from reproducing
Habitat impacts	Reduced coral cover
	Reduced coral diversity
	Reduced coral rugosity
	Loss of slow-growing reef structure

	Generation of coral rubble from collection leads to abrasion damage in other habitat forming species
Effect on associated species	Decreased abundance, biomass and diversity of reef fish, invertebrates, and other species due to loss or destruction of habitat
Ecosystem impacts	Increased degradation and thus erosion of the reef structure, associated islands, and coastal environments
	Loss of resilience; exacerbation of the impacts of the larger threats to reefs (e.g. climate change)
Socio-economic impacts	Conflicts with other uses/user groups, including traditional uses, fishing, and tourism

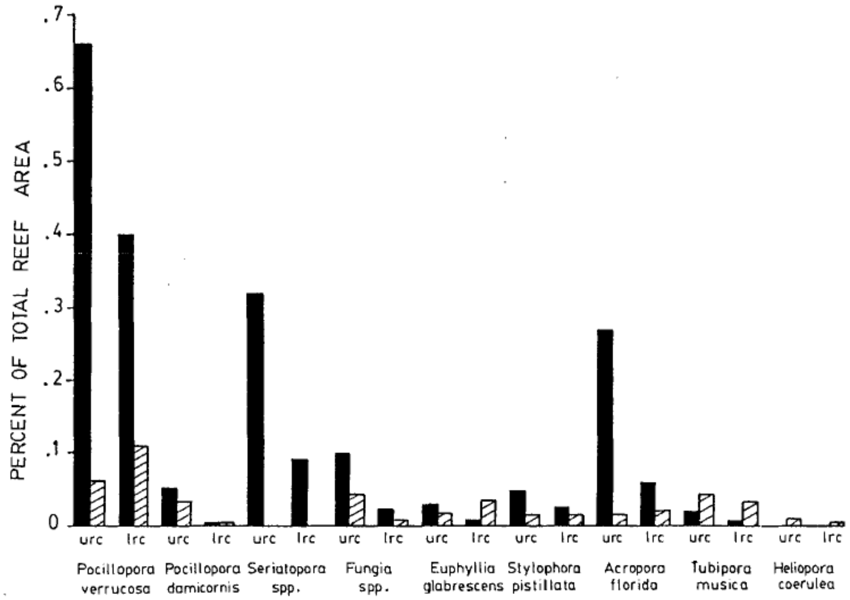
Several studies have examined the population-level effects of coral harvesting. Ross (1984) was the first to systematically address this topic and to demonstrate the impacts of the ornamental trade on corals. During the 1980s the Philippines was the center of collection for the international trade in corals for the curio market (Ross 1984). In 1981, Ross compared stony coral populations in two different types of locations, very lightly harvested areas (i.e., baseline controls) and heavily collected areas (i.e., impacted sites). Baseline indicators of these two sites, including overall percentage of coral cover, species and substrate composition, and other metrics indicated that the collection and control study locations were comparable (Ross 1984). Ross measured coral populations along 100 m transects and found significant impact to the species targeted by the curio trade. Collectors removed at least 28 species in 17 genera (Ross 1984). A total of 81% of the targeted corals had mushroom, brown-stem cluster, brain, or branching morphologies

Figure 17: Colony density of nine coral groups that are harvested for the curio trade at collected (hatched bars) and control (solid bars) sites. URC = upper reef crest; LRC = lower reef crest. Figure taken from Ross (1984).



(Ross 1984), all growth forms that are popular in home décor. Sites where collection occurred experienced large reductions in the density and percentage cover of popular curio species except for *Euphyllia glabrescens*, *Tubipora musica* (a non-scleractinian coral), and *Heliopora coerulea* (Ross 1984, Figures 17–18). In the upper reef crest at the collected location, there was a 77.8% reduction in coral density and 86.3% reduction in percent cover in species collected for the curio trade (Ross 1984). In the lower reef crest at the collection site, the reductions were 30.9% for density and a 64.3% in percent cover of curio species (Ross 1984). *Seriatopora* spp. were abundant at the control site but were totally absent, save for one colony, in the collection zone (Ross 1984). Ross (1984) also interviewed collectors who confirmed that the curio harvest had driven down coral populations and converted some sites to commercial inviablity (Ross 1984).

Figure 18: Colony area of nine coral groups that are harvested for the curio trade at collected (hatched bars) and control (solid bars) sites. URC = upper reef crest; LRC = lower reef crest. Figure taken from Ross (1984).

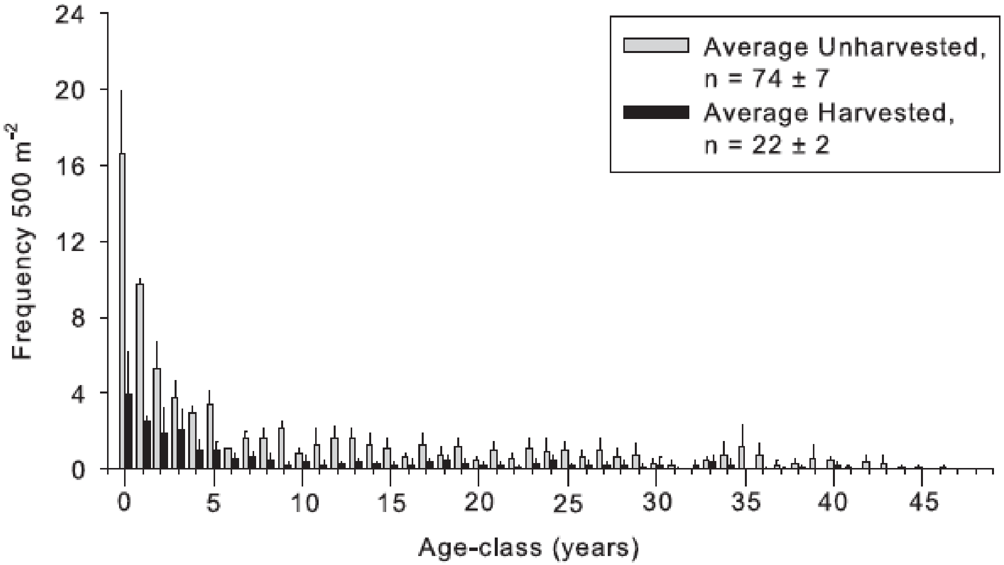


Beyond the reef-wide surveys reviewed above, Ross (1984) examined the population demographics of *Pocillopora verrucosa*, a popular branching coral in the curio trade. At the collection site, 71.8% of *P. verrucosa* colonies were small and reproductively immature (Ross 1984). At the unharvested control sites, the population demographics were different and only 50.8% of colonies were immature (Ross 1984). Notably, large colonies were missing from the location open to collection (Ross 1984). Similar patterns were observed for *Fungia* spp., *Pocillopora damicornis, Acropora florida,* and *Seriatopora* spp. (Ross 1984). Many corals exhibit size-dependent reproduction; by removing large colonies, collection is reducing the population's reproductive output and driving these corals toward reproductive failure (Ross 1984). Ross (1984) characterized this situation as a "major concern" for fishery management in the Cebu region. Despite these significant impacts from collection, Ross (1984) did not observe larger effects in non-collected corals or overall coral diversity, suggesting that removal of curio corals was not having larger effects on other non-target scleractinians.

The study did not examine the effects of harvesting on the associated fish, invertebrate communities, or the reef itself (Ross 1984).

It is worth noting the early publication date of the Ross study. During the 1980s, the ornamental coral reef wildlife trade was small compared to today's trade (Tissot et al. 2010). It is notable that even at this lower volume, trade had significant negative effects on coral populations. Because the volume of imports has continued to grow, it is likely that the impacts of trade on stony coral populations have also grown with time.

Figure 19: The size frequency distribution of *H. actiniformis* from harvested and unharvested locations. Note the lower abundance of large colonies at harvested sites. Figure taken from Knittweis and Wolff (2010).



Despite the volume of trade and the number of species involved, there are relatively few studies on the direct effects of collection on stony coral populations. One recent exception is work by Knittweis and Wolff (2010) which examined the aquarium trade's impacts on *H. actiniformis* in Indonesia. *Heliofungia actiniformis* is notable as a top five coral species by volume in the live coral trade (Green and Shirley 1999). To determine trade's impacts on this species, Knittweis and Wolff (2010) conducted interviews with people involved in trade, examined government records, and performed reef surveys that compared harvested and unharvested sites. The researchers observed a size-selective *H. actiniformis* fishery (Knittweis and Wolff 2010). Specifically, collectors favored small colorful polyps, which require only a small amount of space in an aquarium, over larger brown polyps (Knittweis and Wolff 2010). This size-selective and color-selective fishery caused a marked decline in coral abundance and a shift in the size-frequency distribution at collection sites (Knittweis and Wolff 2010). There was a significant absence of corals between 4 and 11 cm in size (Figure 19). Comparison of these data to previous research (i.e., Hoeksema 1989) suggests that the change in *H. actiniformis* populations was driven by collection for the aquarium trade (Knittweis and Wolff 2010).

In addition to changes in the demographics of *H. actiniformis* populations, Knittweis and Wolff found much lower polyp abundances at collection sites compared to locations where collection did not occur. At collection sites, coral mortality was high for corals 11 cm in size or smaller and mortality was significantly lower for corals above this size (Knittweis and Wolff 2010). This difference in mortality reflects the collectors' and hobbyists' preferences for smaller *H. actiniformis* (Knittweis and Wolff 2010). Removal of young corals is concerning because *H. actiniformis* takes approximately 9 years to reach reproductive maturity; many corals are removed before having the opportunity to spawn (Knittweis and Wolff 2010). Furthermore, *H. actiniformis* naturally grows as a stalked colonial form before splitting into independent polyps (Knittweis and Wolff 2010). Collectors oftentimes do not wait for the coral polyps to break apart and become free-living colonies (Knittweis and Wolff 2010). Instead, collectors commonly break off anthocauli (stalks connecting the polyps) and take only the largest polyp in the colony (Knittweis and Wolff 2010). Smaller polyps are discarded and are often left to die (Knittweis and Wolff 2010). This practice dramatically reduces asexual reproduction, thereby causing negative effects on *H. actiniformis* populations (Knittweis and Wolff 2010). Finally, collectors tend to remove every coral they find, but middlemen select only the best corals to export (Knittweis and Wolff 2010). Unfortunately, the remaining corals

Precious corals

Precious corals are diverse group of animals that are highly valued for their attractive hard skeletons, which are composed of dense complexes of protein and/or calcium carbonate (Tsounis et al. 2010). Skilled craftsmen refine and carve these skeletons into high-value beads, pendants, gemstones, sculptures, and other art and jewelry objects (Tsounis et al. 2010). Unfortunately, precious corals are the slowest growing commercially harvested species in the world, making them virtually irreplaceable once they are over collected (Tsounis et al. 2010). These animals naturally occur in many deep-water, rocky-bottom habitats on the outer continental shelf, continental slope, and seamounts (Tsounis et al. 2010). Although they are typically not found on coral reefs, precious corals do form reef like structures that provide habitat to many other marine species (Tsounis et al. 2010).

Throughout the world, precious coral fisheries have been heavily depleted by collection for the coral jewelry trade (Grigg 1989, Santangelo and Abbiati 2001, Tsounis et al. 2010). For example, Mediterranean red corals (*Corallum rubrum*) along the Spanish and Moroccan coasts have dramatically dwindled in number (Tsounis et al. 2006, Zoubi 2009, Tsounis et al. 2010). Few large and old colonies – the colonies that produce the most offspring – can be found today (Tsounis et al. 2010). Only young and slowly-regenerating remnant populations of the Mediterranean red coral remain (Tsounis et al. 2010). These problems are almost certainly a result of over collection; Mediterranean red coral has been heavily exploited for centuries by dredging and, in recent years, divers, leading to the near-complete depletion of accessible populations (Tsounis et al. 2010).

Significant declines in precious coral populations have also been seen in the Pacific and Ocean, with drops in harvest levels by 95% over the past 30 years (Tsounis et al. 2010). For instance, black corals (*Antipathes* spp.) in Hawaii have been reduced in abundance and have lost the older coral colonies because of harvesting (Grigg 1976). As a result, few Hawaiian black corals remain accessible to divers today (Tsounis et al. 2010). Similarly, red and pink corals (*Corallum* spp.) at the Emperor Seamounts have been eradicated from over-exploitation (Tsounis et al. 2010). These examples highlight how precious coral fishing can only continue through the discovery of new unexploited populations and harvesting increasingly remote and inaccessible coral beds (Tsounis et al. 2010). Unfortunately, most of the known precious coral stocks have been over-exploited (Tsounis et al. 2010). For further details, Tsounis et al. (2010) provide a comprehensive review of precious coral extraction and conservation.

are discarded as waste (Knittweis and Wolff 2010). This leads to much higher levels of collection than is permitted under the annual quota system (Knittweis and Wolff 2010). Based on demographic data, Knittweis and Wolf (2010) calculated that current fishing practices had over-exploited *H. actiniformis*.

The studies by Ross (1984) and Knittweis and Wolff (2010) demonstrate the trade can and does have direct negative effects on coral populations in nature. Several studies and reports have attempted to further infer the severity of putative impacts (Table 7) through an examination of catch reports, trade data, and various analyses. For instance, Green and Shirley (1999) conducted a lifespan analysis based on CITES trade data from 1985-1997. These authors assumed that corals must, at a minimum, survive as long in captivity as the corals' age at the time of collection in order for the coral trade to be sustainable (Green and Shirley 1999). Based on this assumption, Green and Shirley (1999) concluded that certain coral species were harvested sustainably whereas others were not. Although the local scale effects can be profound, globally, the collection of live corals was concluded by Green and Shirley (1999) to be small relative to other threats facing coral reefs (see also Surharsono 1999). However, this type of analysis is difficult because of the severe lack of data for the age of corals in trade and mortality rates often vary (Green and Shirley 1999). Furthermore, the Green and Shirley analysis failed to consider many essential parameters such as age of reproductive maturity, existing population size, coral growth rate, dispersal ability, recruitment variability, and waste/loss in the supply chain. As a result the Green and Shirley (1999) conclusions must be treated with caution as they likely underestimated trade's impacts on coral populations.

The impacts of the stony coral trade vary by country. In Vietnam, corals that are popular in the curio and aquarium trades are increasingly rare in some places, for instance at the epicenter of the Vietnamese coral trade in Nha Trang Bay (Tuan 2002). Vietnam has relatively little regulation and few formal statistics on trade activities making it difficult to determine the impact of trade in other areas (Tuan 2002). In near-by Indonesia, the large-scale impacts of trade are somewhat unclear. Samedi and Liman (2002) concluded that although trade does impact the reef, the damage is relatively small and controlled (see also Suharsono 1999). By contrast, Wicaksono et al. (2002) indicated that trade degrades more value from coral reefs than it provides in economic benefits. There is also a severe lack of data and monitoring in Fiji (Lovell 2001). Although the impacts of the Fijian coral trade appear limited (Lovell 2001), much more information is required. This severe lack of data is a widespread problem that stymies a full understanding of trade's impacts on coral reefs.

Australia is home to one of the largest reef ecosystems in the world and has an active coral trade. Harriott (2001) indicated that coral collection is small compared to the entire reef system and myriad other threats that imperil coral reefs. However, this conclusion was recently called into question by Jones (2011). Jones (2011) examined CITES export data and annual Queensland Coral Fisheries reports from 2006 to 2009 to estimate the impacts of trade. There was a large increase in the Australian coral harvest from 2006 to 2009, but a decline in CPUE over the same time period, suggestive of over harvesting (Jones 2011). The increased harvest was driven by increased demand from U.S. markets (Jones 2011). Several corals appeared to be depleted, including *Blastomussa* and *Scolymia*, as well as the anemone *H. crispa* (Jones 2011). There was an increase in harvest of small specimens concomitant with decreasing populations, suggesting that harvesting exceeded the capacity for coral populations to regenerate (Jones 2011). Fishing quotas were based on outdated growth rates for rapidly-growing branching corals and not the slower growth rates of the species primarily targeted for the coral trade (Jones 2011). Based on these findings, Jones (2011) concluded that the Australian coral harvest may not be ecologically or economically sustainable, which is surprising given the high level of regulation and protections in place for Australian corals.

Conservation and the trade in ornamental corals:

Sustainable harvest of corals is theorhetically possible and many different strategies have been attempted to achieve that goal (Bruckner 2002, Knittweis 2008). Bruckner (2002) put forth several recommendations for sustainable coral collection, including:

(1) Licensing collectors and limiting entry into the fishery
(2) Limiting the number, size, and species of corals that are permitted for collection based on biological and population information
(3) Targeting only species with appropriate life history traits including fast growth rate, high reproductive output, high captive survival rates
(4) Spatial management with designated no-take protected areas, collection areas, and possible rotational closures to facilitate recovery
(5) Holistic and flexible ecosystem management that considered the ecosystem role of target species, species interactions on the reef, and other stressors facing these ecosystems
(6) Prohibition of destructive collection methods like moxy nets, breaking coral colonies, and cyanide fishing, as well as use of non-destructive methods (e.g., diving with fins instead of walking on the reef)
(7) Implementing a robust monitoring program to evaluate the resource status and the efficacy of management over time
(8) Eco-labeling of sustainably harvested products
(9) Minimizing mortality and managing trade at all levels of the supply chain from collection to sale
(10) Consultation with various stakeholders in a transparent and well-organized manner

Several of these strategies have been implemented in various locations, with varying success.

The IUCN has classified 837 species of scleractinian corals according to their conservation status. As of the time of this writing, scleractinian corals are rated as Critically Endangered (6 species), Endangered (23 species), Vulnerable (199 species), Nearly Threatened (174 species), Data Deficient (146 species), or Least Concern (289 species) (IUCN Red List version 2010.4). As the IUCN Red List highlights, there are concerns about the state of many coral populations and species. As a result of these types of concerns, all scleractinians are now listed and international trade is regulated under CITES.

CITES is the primary instrument for monitoring and regulating the international trade in wildlife (Lieberman and Field 2001). This agreement went into force in 1975 and now includes 175 signatory countries, which are known as Parties (Jones 2008). CITES Parties are obligated to regulate international trade in accordance with the provisions of the Convention, which differ depending on which of three Appendices species are listed. Those listed on Appendix I cannot be traded internationally for commercial purposes. Appendix II can be traded for commercial purposes as long as certain findings are made (CITES Treaty, Article IV): (a) "[a] Scientific Authority of the State of export has advised that such export will not be detrimental to the survival of that species"; (b) "[a] Scientific Authority in each Party shall monitor both the export permits granted by that State for specimens of species included in Appendix II and the actual exports of such specimens. Whenever a Scientific Authority determines that the export of specimens of any such species should be limited in order to maintain that species throughout its range at a level consistent with its role in the ecosystems in which it occurs and well above the level at which that species might become eligible for inclusion in Appendix I, the Scientific Authority shall advise the appropriate Management Authority of suitable measures to be taken to limit the grant of export permits for specimens of that species"; and (c) "a Scientific Authority of the State of introduction advises that the introduction will not be detrimental to the survival of the species involved". Those on Appendix III can be traded as long as the specimens have been legally acquired and living specimens are prepared and shipped so as to minimize injury, mortality and cruel treatment. CITES Parties are obligated to provide annual reports to the CITES Secretariat on trade in CITES-listed species. A CITES Trade Database, available on the CITES Secretariat website (www.cites.org), allows one to analyze these trade records to determine trade levels, source countries, importing countries, the purpose of trade and the source (wild for example) of specimens in trade. CITES includes measures to address illegal trade in specimens of Appendix I-listed species and detrimental trade in specimens of Appendix II-listed species.

There are more than 30,000 species listed under CITES (http://www.cites.org/eng/disc/what.php), including more than 2,000 species of hard corals (Jones 2008). As noted above, hard corals can be challenging to identify and this problem can lead to lax enforcement of protections (Green and Shirley 1999, McManus 2001, Jones 2008). There is also considerable scientific uncertainty about the sustainability of exploitation due to a dearth of data and monitoring (Lieberman and Field 2001); however, many Parties nonetheless provide export permits. As a result, inspection officers in importing countries are often forced to release many shipments that are accompanied by a CITES export permit where the sustainability of collection is uncertain (Jones 2008). When confiscation of corals does occur, it is more often a result of paperwork irregularities, such as a lack of permits or inaccurate information on a permit, instead of environmental concerns (Jones 2008). If one port of entry enforces the rules strictly, importation often shifts to other locations (Jones 2008). When wildlife is confiscated, the sanctions include loss of the goods, fines, and persecution leading to prison sentences; however, the maximum penalties are rarely enforced (Jones 2008). Additionally, CITES leaves several gaps in coverage that weaken protection. Once corals are imported to a country, there is no mechanism for tracking where the coral colony originated, thereby potentially creating misleading trade data when corals are transshipped through an intermediary country (Jones 2008). There is also an exemption for maricultured corals that may provide a loophole; wild caught corals can be broken up into smaller pieces, attached to a base, and passed off as a maricultured specimen (Jones 2008). Although CITES does provide a minimum level of protection, there are significant problems with implementation and enforcement of the Convention's provisions that limit the effectiveness of this measure for protection of corals.

Exporting countries have tried a range of different legal and management strategies to protect corals. The coral trade once centered on the coral reefs of the Philippines. However, collection and export of ordinary hermatypic corals was banned by Presidential Decree 1219 (Ross 1984). For several years after Decree 1219, commercial collection continued, especially in the Cebu region (Ross 1984). However, the official Philippine coral trade has largely desisted (e.g., see recent trade volumes in Craig et al. in press), but corals may still be laundered through neighboring countries. An all out ban is one way to protect coral reefs, but it does not support local livelihoods, allow for consumers to purchase coral products, or prevent the creation of a market for illegal trade. Measures in other countries have attempted to maintain an active and sustainable coral trade.

Following the Philippine ban of the coral trade, collection shifted to other countries. Indonesia is now the primary source of corals for the trade. Indonesia allows only live corals to be collected; collection and trade in dead corals for curios and décor has been banned since 1997 and coral mining is prohibited in sensitive areas (Bruckner and Borneman 2006). Collection of corals is allowed in ten Indonesian provinces that fall outside of designated tourism and protected areas (Bruckner and Borneman 2006). Collectors and collection companies must be licensed to take Indonesian corals (Knittweis 2008). There are size limits as well; the maximum allowable size is 25 cm for branching species and 15 cm for boulder species (Bruckner 2002, Bruckner and Borneman 2006).

In collaboration with the CITES Secretariat, Indonesia established quotas for collection of certain coral genera (Green and Shirley 1999). Green and Shirley (1999) evaluated the efficacy of these quotas and found that for the eight commonly traded coral genera the number of exports fell far below the established quota. However, for *Acropora* and *Pocillopora*, the number of coral pieces traded was much greater than the quota and yet trade in these genera had not been restricted (Green and Shirley 1999). (It is worth recognizing, however, that *Acropora* spp. and *Pocillopora* spp. have relatively rapid growth rates.) There was little to no scientific justification for the quota value set for each genus (Green and Shirley 1999). Taken together, Green and Shirley (1999) concluded that the quotas were arbitrary and potentially ineffective. Knittweis (2008) reach a similar conclusion for the management of Indonesian corals, stating that the "present management plans are as a result based on little more than arbitrary decisions."

Bruckner and Borneman (2006) followed up on the work of Green and Shirley (1999) by comparing field data on the abundance of Indonesian corals to the allowable export quota set by the Indonesian government. There was a notable disconnect between a species' abundance and its export quota (Bruckner and Borneman 2006). Depending on the coral species examined, the quota system varied from allowing the take of less than 1% of certain corals to taking 96% of the population of other coral species each year (Bruckner and Borneman 2006). The quota system did not reflect the biological and ecological reality; some coral genera were heavily collected despite small populations and limited distributions (Bruckner and Borneman 2006). Species that occurred in isolated patches were especially prone to over collection under this system (Bruckner and Borneman 2006). For example, the coral *Nemenzophyllia turbida* was completely extirpated from one site due to collection (Bruckner and Borneman 2006). Fast-growing branching corals were more resilient against collection compared to other morphologies with slower growth rates (Bruckner and Borneman 2006).

In 1998 and 2000 the European Union suspended import of 25 Indonesian coral species because of concerns about the sustainability of collection. Bruckner and Borneman (2006) examined the status of these 25 species and found that as little as 1% to as high as 96% of the existing population could be collected each year under the current quota system, depending on the coral species examined. For example, collectors were permitted to harvest just 2.2% of the *Cynarina lacrymalis* population compared to a full 30% of the *Blastomussa merleti* population that could be collected each year under this system (Bruckner and Borneman 2006). For many species, collectors were permitted to take corals at all size classes available, leaving corals with no demographic refuge from collection (Bruckner and Borneman 2006). To address the deficits of the quota system, Bruckner and Borneman (2006) recommended decreasing quotas for species prone to over collection and implementing an adaptive management strategy to protect vulnerable species.

For *H. actiniformis*, Knittweis and Wolf (2010) calculated that collecters were in the process of over-exploiting the population. However, these researchers estimated that a 20% reduction in fishing mortality would have minimum impact on profit (i.e., $16.8 vs. $16.7 U.S. per 1000 coral colonies). A full 60% reduction in fishing pressure was needed in order for natural mortality to equal fishing mortality, which is a common standard for sustainable fisheries (Knittweis and Wolff 2010 following Gulland 1971). Knittweis and Wolff (2010) also calculated that a 5-cm minimum size limit for *H. actiniformis* could both increase profit by 58% and lower the number of polyps harvested. This proposed size limit would also protect anthocauli which would insure an asexual source of recruits into the population (Knittweis and Wolff 2010). Although size limits would be an important step towards making this coral fishery more sustainable, a size limit alone would not ensure sustainability (Knittweis and Wolff 2010). Knittweis and Wolff (2010) suggested combining this size limit with reduced fishing pressure, reduced harvest quotas, enforcement of protected areas, and expansion of the marine protected area system. As mentioned above, Indonesia also has a system of reserves designated to protect coral reef wildlife. Knittweis et al. (2009a) examined the efficacy of this marine protected area system in the context of *H. actiniformis* dispersal. Based on population genetic data, the authors concluded that the existing system needed to be larger and more reserves needed to be added to the system (Knittweis et al. 2009a). Although Indonesia has a wide range of conservation and management tools at its disposal, the consensus of work by various scientists is that additional measures are needed to achieve sustainability (Green and Shirley 1999, Bruckner and Borneman 2006, Knittweis 2008, Knittweis et al. 2009a, Knittweis and Wolff 2010).

Although it is not one of the largest suppliers of stony corals on the international market, there is an active coral trade on the Great Barrier Reef in Australia. Most collection happens in the Great Barrier Reef World Heritage area, which is managed by the Great Barrier Reef Marine Park Authority (GBRMPA) independently of the Queensland Fisheries Service (QFS) (Breen and Owens 2002). Collection began in 1930 and has continued to support a growing curio and souvenir industry (Breen and Owens 2002). Live aquarium collection started in 1985 and represented 90% of the Australian domestic coral trade in 2001 (Breen and Owens 2002). Collection is permitted in only a small area of the total Great Barrier Reef World Heritage Area (Breen and Owens 2002). Export of corals from Australia requires ministerial approval, which has only been granted recently (Breen and Owens 2002, Jones 2011). As a result, for most of the history of the Australian trade, collection supplied only domestic markets (Breen and Owens 2002).

Today, stony corals from Australia are largely exported to the U.S. (Jones 2011). Collection licenses in Australia were restricted by location until approximately 2006 (Jones 2011). The depletion of certain collection sites led to a shift from site-specific licenses to general collection licenses that allowed collectors to rove from site to site (Jones 2011). Fishing quotas for Australian corals were also based on outdated growth rate estimates for rapidly-growing branching corals and not the slower growth rates of the species primarily targeted for the coral trade (Jones 2011). Jones (2011) examined CITES export data and Queensland Coral Fisheries data and concluded that the coral trade may be unsustainable.

Regulation and management of the coral trade is not limited to the source countries. The U.S. is the largest consumer of corals for aquariums and curios (Moore and Best 2001), and is well positioned to drive collection practices throughout the world (Tissot et al. 2010). The U.S. prohibits collection of stony corals within its own states and territories, which is a tacit recognition of the threat trade posed by collection (McManus 2001, Moore and Best 2001, Lieberman and Field 2001). The U.S. also protects two coral species, *Acropora palmata*, and *Acropora cervicornis*, under the Endangered Species Act (ESA), with 83 additional species currently under consideration for ESA listing. The U.S. Caribbean Fisheries Management Council has formally declared that “The principal value of coral reefs (including live-rock) is considered to be non-consumptive and are viewed as essentially non-renewable resource[s]...The importance of corals and associated plants and invertebrates lies in their relationship to the marine ecosystem.” (CFMC 1999, as quoted in Lieberman and Field 2001). Despite the United States’ recognition of the risks of coral collection, the U.S. does not restrict the importation of CITES Appendix II coral species (Lieberman and Field 2001).

In contrast to the U.S., the European Union (E.U.) can place restrictions on the importation of certain species or animals from certain countries if there are concerns about the sustainability of collection (Jones 2008). Wildlife Trade Rgulation (EC 338/97) supplements CITES for 27 E.U. nations, which allows an importing country to raise concern about a species and enables an E.U. Scientific Review Group to investigate (Jones 2008). If the Review Group reaches a negative ruling, imports of that species from a particular country can be banned (Jones 2008). In 1998 and 2000 the E.U. temporarily suspended imports for 25 species until Indonesia could provide data to show that the harvest was sustainable (Bruckner and Borneman 2006), suggesting that CITES alone is not sufficient to regulate the coral trade (Jones 2008). If the U.S. were to implement a similar system, it could greatly reform the international trade in scleractinian corals and other coral reef wildlife (Tissot et al. 2010).

Finally, the captive breeding and propagation of scleractinian corals has taken off in recent years (e.g., Franklin et al. 1998). Although this practice only supplies a tiny portion of the overall trade, aquaculture has the potential to significantly alleviate the impacts of collection on coral reefs. For this to succeed, it will require a consumer base that is willing to pay a price premium for captive-raised corals to fuel considerable growth in the aquaculture industry, retailers that are willing to promote captive-bred animals, as well as changes in the marketplace such that wild source corals cannot undersell captive-raised ones.

Part II:
Broader Impacts of Trade

Chapter 9
Injury and Death in the Supply Chain: Accelerating Collection on Reefs

Causes and rates of injury and death in the coral reef wildlife trade:

High supply-chain mortality is a major ecological concern in the aquarium trade in coral reef wildlife. Death of coral reef animals between collection and retail sale causes waste (i.e., fishing effort, income, animal lives) and drives additional collection to satisfy market demand, thereby exacerbating pressure on resources (Green and Shirley 1999, Wood 2001b, Sadovy 2002, Wabnitz et al. 2003, Schmidt and Kunzmann 2005). Furthermore, trade in live animals, without sufficient attention to their health and welfare, has been characterized as unethical behavior (e.g., Wood 2001b). Mortality in the coral reef wildlife trade is caused by poor capture, handling, husbandry, and transportation methods, as well as inadequate facilities, poor water quality (i.e., ammonia build up), and the collection of unsuitable species (Wood 2001b). The use of sodium cyanide or other poisons to capture fish also increases mortality (see Chapter 10). Much of this mortality could be reduced or even avoided with additional training, adoption of alternative methods, changing perceptions, and even new regulations to establish better industry standards (Sadovy 2002).

Estimates of losses due to severe injury or death in the supply chain range from a few percent to over 80% of animals collected (Sadovy 2002). Losses of animals at each stage of the supply chain can be 10−20% of animals, and are sometimes even higher (Vallejo 1997a,b). Businesses and locations with a more integrated supply chain with fewer steps in that supply chain have much lower mortality rates (Sadovy 2002). In the Pacific during the early 1990s, pre-shipment mortality averaged 5−10%, and was as low as 1−2%, including release of sub-standard or injured fish after collection (Pyle 1993). In Puerto Rico, mortality from capture to export was estimated to be 10−20% of collected animals, depending on collection methods, collector skill level, and quality of holding facilities (Sadovy 1992). In Sri Lanka during the 1980s, about 15% of collected animals died during or immediately after collection, 10% died during shipping, and another 5% died at importer and retailer holding facilities (Wood 1985). For aquarium fish collection in the Philippines, Gonsales and Savaris (2005) estimate mortality to be 90% along the supply chain. Rubec and Soundararajan (1991) estimated cumulatively mortality from the reef to the retailer at over 90%.

Because the coral reef wildlife trade is a data-poor fishery that involves thousands of different species (Rhyne et al. 2012), it is hard to precisely estimate the rates of death from capture. The range of mortality and loss rates cited above highlight this situation. Mortality rates vary widely among collectors, collection methods, location of the collection site, time spent in holding and transport, and target species. Typically mortality rates are undocumented, so estimates must be based on interviews or inferences based on incomplete information (Wood 2001b, Sadovy 2002, Wabnitz et al. 2003, Rubec and Cruz 2005). People involved in this trade understandably fear regulation or persecution making them hesitant to disclose accurate information (Rubec and Cruz 2005). Those collectors, exporters, and importers engaged in practices that minimize mortality are often more willing to share information, potentially biasing the estimate. The method of calculation also affects the estimate of mortality rate. For example, calculation of percentage change depends heavily on the baseline used for comparison; that baseline could be the number of animals taken at collection, number of animals brought in from the reef, number of animals entering at a given stage of the supply chain, or some other metric. Care must be taken in summarizing total mortality rates because the benchmark may vary between (or even within) studies. As a result, the rates of mortality cited here should be considered cautiously due to the incomplete data available.

Mortality and injury to coral reef wildlife begin on the reef. Fish and invertebrates regularly experience injuries from rough handling during collection. Sensitive coral reef fish species easily experience abrasion that removes mucus protection and subjects the fish to osmotic stress (Rubec and Cruz 2005). This physical damage to the organism can raise its susceptibility to other stressors (Hanawa et al. 1998), and can result in rejection of that animal by middlemen and exporters.

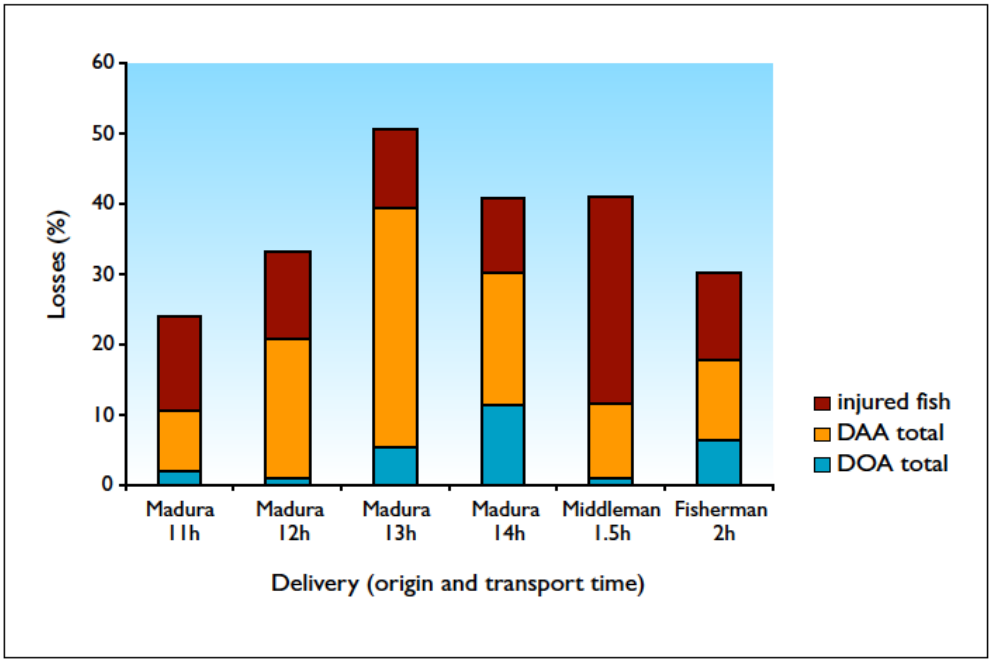
Fish captured at depth and surfaced suffer a variety of injuries, known as barotrama, from changing pressure (Wilde 2009). The most common injury is an expanded swim bladder that may rupture and inflate the body cavity (Wilde 2009). Barotrama can also include emboli (i.e., gas bubbles) in the blood and organs like the eyes, brain, heart, arteries, gills, spleen, fins, muscles, and dermis (Feathers and Knable 1983, Parrish and Moffitt 1993, Brown et al. 2007). The severity of the injury increases with capture depth (Casillas et al. 1975, Rogers et al. 1986, St. John and Syers 2005, Hannah et al. 2008).

To reduce pressure in the expanded swim bladder, many fishers deflate, vent, or “fizz” the swim bladder. This is typically achieved by inserting a wide-bore hypodermic needle into the swim bladder or body cavity in the event of a bladder rupture (Wilde 2009). Venting is somewhat controversial; it may reduce injury to the fish by relieving excess pressure in the bladder or body cavity, but it can also cause additional injury or even kill the fish (reviewed in Wilde 2009). Studies examining the efficacy of venting have yielded conflicting results. To better understand the effects of venting, Wilde (2009) reviewed studies of venting using survival rates in captivity and capture-release-recapture rates as metrics of survival. In total, 39 species-study combinations were examined, including 18 examinations of fish survival and 21 samples of capture-release-recapture rates (Wilde 2009). The studies reviewed included 4 freshwater and 17 marine fish species or composite species-groups (i.e., *Plectrodomus* spp.), but unfortunately, no coral reef ornamental fish were exained. Venting had no survival effect in 32 of 39 samples (Wilde 2009). Venting reduced survival in 2 samples (i.e., red grouper and red snapper) whereas it increased survival in 5 species samples (i.e., black sea bass, crimson snapper, gag, walleye, and yellow perch) (Wilde 2009). Red snapper, crimson snapper, and walleye were examined by multiple studies, enabling replicate examinations of venting. For these three species, only one showed a significant effect of venting, whereas the remainder exhibited no consistent trend (Wilde 2009). When the results of all studies were pooled together, there was no evidence that venting positively or negatively affected fish survival (Wilde 2009). There was also no difference depending on whether the venting was done by anglers or biologists (Wilde 2009). Wilde (2009) attributed the inefficacy of venting to the severity of barotrauma in damaging all organs, not just the swim bladder. When the results were partitioned by depth, there was slight evidence for improved survival with venting in fish captured at depths of shallower than 20 m and increased death following venting with fish captured at depths greater than 20 m (Wilde

2009). This is important for coral reef ornamentals because most fish are captured at relatively shallow depth (e.g., Stevenson et al. 2011).

Once fish are collected, poor holding conditions and inadequate husbandry cause mortality through the following stages of the supply chain. For example, Gonsales and Savaris (2005) observed fish being packed in plastic bags aboard collection vessels in the Philippines. They estimated mortality and rejection of collected fish aboard the collection vessel to be 10% of collected animals (Gonsales and Savaris 2005). The mortality aboard the collection vessel often resulted from holding bags bursting aboard the ship (Gonsales and Savaris 2005). Baquero (1995) reports that poor conditions cause mortality rates of 20% between collection and trade with middlemen at a major aquarium fish source in the Philippines. Similar observations were made of fish collected in Indonesia (Schmidt and Kunzmann 2005). Fish experienced rough treatment following collection; in one case fish from Madura spent several days stored in plastic bags on fishers' boats, often in the direct sunlight (Schmidt and Kunzmann 2005). Once arriving, bags of fish were tossed onto the ground and left exposed to full sunlight (Schmidt and Kunzmann 2005). This contributed to high rates of mortality (i.e., 24–51%) between collection and export for Indonesian fish (Schmidt and Kunzmann 2005).

Figure 20: Percent losses on several shipments of aquarium fish as reported by Schmidt and Kunzmann (2005). DAA = dead after arrival. DOA = dead on arrival.



Water quality for collected animals is also a major issue. Fish are often held in sealed polyethylene bags or in closed tank systems. Under these conditions, environmental parameters can rapidly change. Water has a low capacity to hold oxygen, confined conditions prevent metabolic wastes from dispersing, and water temperature in holding bags and tanks can fluctuate greatly (Rubec and Cruz 2005). As fish consume oxygen and excrete waste, dissolved oxygen, dissolved carbon dioxide, pH, ammonia, and temperature all change (McFarland and Norris 1958, Fry and Norris 1962), often with deleterious effects on the animals (Schmidt and Kunzmann 2005). For example, aquarium fish collected in the Philippines are held for three to five days in a financier's community holding facility before transport to exporters (Gonsales and Savaris 2005). Here, mortality is approximately 30%, due to stress of capture, ammonium accumulation in holding tanks, and salinity or temperature fluctuations (Gonsales and Savaris 2005).

Mortality rates can also be high at holding and export facilities. For instance, Phillipines facilities experienced mortality in 30−40% of fish (as estimated by Vallejo 1997a,b). Based on interviews, Rubec and Cruz (2005) reported that mortality at a major exporter in the Philippines at 30−40% of fish, resulting in $250,000 U.S. in annual losses. In one of the most quantitative mortality studies available, Schmidt and Kunzmann (2005) examined post-harvest mortality at an Indonesian export facility. The authors followed six deliveries from collection to export, including 2,576 fish from 120 species (Schmidt and Kunzmann 2005). Injury and death related losses from these six deliveries ranged from 24−51% of animals entering the facility (Schmidt and Kunzmann 2005, Figure 20). The variation between shipments was large. Eleven to 40% of the delivered fish were dead on arrival (Schmidt and Kunzmann 2005). An additional 8.5−34% of fish died in holding facilities prior to export (Schmidt and Kunzmann 2005). High rates of death after arrival resulted from parasites and diseases at the stocking facilities (Schmidt and Kunzmann 2005). Clownfish, for example, experienced 100% mortality of fish remaining in the stock system for four or more days due to a rapidly-spreading *Brooklynella hostilis* infection (Schmidt and Kunzmann 2005). High disease rates may have resulted from the prevalent cyanide use (i.e., in an estimated 80% of fish captured), causing stress and raising the susceptibility to other threats (Schmidt and Kunzmann 2005). Water quality (e.g., ammonia and ammonium) during transport and in the tanks, high stocking density of fish in tanks, poor handling at the facility, conditions before arrival, and collection of unsuitable species also caused death in the fish (Schmidt and Kunzmann 2005). Death was not the only source of losses; severely injured fish that could not be transported to the next stage of the supply chain comprised 25−76% of the total loss (Schmidt and Kunzmann 2005). Rates of death at each stage depended on the species collected, with tendencies for certain species to die during transport whereas others died in holding systems (Schmidt and Kunzmann 2005). Death rates also depended heavily on the quality of each holding facility's equipment and husbandry practices, suggesting that improved protocols could reduce losses (Schmidt and Kunzmann 2005). From 0.8 to 11% of the fish tracked by Schmidt and Kunzmann (2005) died during transportation. Water quality was poor within the transportation bags, with pH and levels of oxygen, ammonium, and ammonia being outside of the acceptable range of conditions for fish health (Schmidt and Kunzmann 2005). The amount of time fish spent in transit was loosely correlated to the loss rate (Schmidt and Kunzmann 2005).

Several other studies examined the effects of transportation conditions on fish health and mortality. Chow et al. (1994), for example, investigated the physiological response of clownfish (*A. ocellaris*) to routine transportation conditions in the supply chain of the coral reef wildlife trade. Under these conditions, 10% of fish died within 48 hours of exposure (Chow et al. 1994). Clownfish were sensitive to large and sudden changes in temperature (Chow et al. 1994). Temperature fluctuations were a common source of mortality during transportation (Chow et al. 1994). Packages of fish and other wildlife are commonly exposed to unsuitable temperatures within the cold cargo holds of aircraft or on the tarmac at the airport, often leading to mortality of transported animals. Rubec et al. (2001), for example, noted an instance of 100% fish mortality in packages left on a cold tarmac. Clownfish needed to be maintained at 24−32 °C or they experienced stress and increased risk of death (Chow et al. 1994). The median tolerance limits, wherein 50% of fish would die after 48 hours, of *A. ocellaris* were 34.46 and 19.49 °C (Chow et al. 1994). The authors also examined water quality conditions within the transportation bags. The water remained well oxygenated over 48 hours, provided fish were packaged in 3:1 pure oxygen to water ratio (Chow et al. 1994). Carbon dioxide and acidity levels increased rapidly during the first 24 hours of transportation due to handling stress (Chow et al. 1994). Ammonia concentration increased over the full 48 hours of transportation (Chow et al. 1994). The median tolerance limits for carbon dioxide, acidity, and ammonia were not reached during the 48 hours of testing (Chow et al. 1994). Despite this, water quality still deteriorated to stressful levels (Chow et al. 1994). As a result, 10% of fish died from the combined effects of temperature change and diminished water quality (Chow et al. 1994). Reports from importers corroborate the findings of Chow et al. (1994). Three Florida importers reported acidic conditions in shipping bags from the Philippines (i.e., ph 6.1 or 6.5); these conditions caused stress and mortality in the imported fishes (Rubec and Cruz 2005). Similarly, a Canadian importer reported mortality rates of 10−15% in shipments of net caught fish and 30−60% in fish captured by cyanide fishing (Rubec and Cruz 2005).

Prior to shipping, exporters commonly clip the spines, barbs, and sharp fins of exported animals. This is done to minimize the risk of puncturing the transport bag during shipping, but may cause nociception and pain, as well as induce stress to the animal. Scientific studies on the effects or effectiveness of this practice in common carp and Nile tilapia (which are not coral reef wildlife species) indicate that this procedure induces a strong stress response, mucus secretion, and immune system mobilization (Roques et al. 2010). Based on these findings, Roques et al. (2010) inferred that fin clipping was a painful procedure for animals subjected to this treatment.

It is also common practice to starve fish for several days prior to shipping. This is done to reduce the accumulation of metabolic wastes and fouling of the water during shipping. Hall and Bellwood (1995) examined this practice in the damselfish, *Pomacentrus coelestis*. Over 16 days, starvation caused atrophy of the animals' digestive systems (Hall and Bellwood 1995). The observed changes included a reduction in the length of mucosal folds that decreased intestinal surface area and absorptive capacity, a reduction in height of columnar epithelial cells that decreased the thickness of intestinal mucosa, and a decrease in the area and thickness of mucosa that reduced the ability of the digestive system to absorb food (Hall and Bellwood 1995). Similar results were observed in the three-spot Dascyllus, *Dascyllus trimaculatus*, where cyanide exposure, starvation and stress were correlated to deterioration of the stomach and intestinal tract's mucosal lining, including decreased size of mucosal folds, increased autolysis, and gastric mucosa sloughing (Bellwood 1981). Hall and Bellwood (1995) also examined the effects of starvation on fish mortality. Starvation alone did not cause *P. coelestis* to die (Hall and Bellwood 1995). However, starvation increased the fishes' susceptibility to other stressors (e.g., handling stress, cyanide) leading to higher rates of death (Hall and Bellwood 1995). For instance, starvation and handling stress combined resulted in 66.7%

mortality, the highest mortality rate observed among the experimental treatments of Hall and Bellwood (1995) (Table 9).

Injuries and fatalities continue to occur in coral reef wildlife after the animals are imported. During the mid-1980s, approximately 30% of fish imported from the Philippines died at import facilities (Rubec 1986, Rubec and Sundararajan 1991). Heidel and Miller-Morgan (2004) examined fish mortality at U.S. importers, including 79 different species of fish from Indonesia and the Philippines (reviewed in Rubec and Cruz 2005). Mortality at the facilities was 0–16% depending on the species examined (Heidel and Miller-Morgan 2004, reviewed in Rubec and Cruz 2005). The researchers also observed physical injuries, infections, and behavioral signs of stress in many of the surviving fish (Heidel and Miller-Morgan 2004, reviewed in Rubec and Cruz 2005).

Fish are distributed from import facilities to retail businesses for sale. The International Marinelife Alliance surveyed 300 aquarium fish retailers in the U.S.; the surveyed retailers reported that 30–60% of fish from the Philippines died within the first three days of arrival to the store (Rubec et al. 2000). According to this survey, mortality rates varied by region and time spent in shipping from Indo-Pacific locations (Rubec et al. 2000, 2001). Similarly, Rubec and Cruz (2005) interviewed several retailers in Canada and the U.S. who report mortality rates of 30% at the retail level.

In the end, all animals collected from coral reefs for the wildlife trade are destined to die in captivity. Animals that survive the process of collection, handling, transportation, export, import, and retail sale, will ultimately die in a home or public aquarium. In terms of their ecosystem function, these animals are already dead; they are finished contributing offspring to the next generation of wild animals. Many of these aquarium animals have short captive lifespans. According to surveys of over 200 households with reef tanks in the United Kingdom, about 50% of coral reef animals die within 6 months of purchase and 70% die within one year (Wood 1985). In the U.S., surveys of 900 households with reef tanks indicated that the average captive lifespan of a marine ornamental fish was two to three years (reefkeeping.com/issues/2005-12/eb/index.php). Note that technology has improved since these surveys were conducted, making these data outdated. Furthermore, hobbyist skill level will have a large influence on survival of the fish. Well-maintained and modern reef tank systems can achieve mortality rates below 1% (Wood 2001b).

Throughout the supply chain, the process of capturing, handling, transporting, and holding coral reef wildlife causes stress in animals. For teleost fish, stress is characterized by a generalized endocrine response followed by the mobilization of energy reserves through metabolic and osmotic changes (reviewed in Hall and Bellwood 1995). These complex physiological changes are an adaptive response that evolved to meet energy demands of the exposure to stressors (Hall and Bellwood 1995). Extended periods of stress have deleterious effects on organisms. One example is the stress-induced gastric ulcers that result from decreased mucosal blood flow and increased gastric acid secretion during the stress response (Gray and Ramsey 1957, Schellerer 1974, Peters 1982). As seen in various fish species, stress can eventually culminate in mortality through osmoregulatory dysfunction and increased disease susceptibility (Berka 1986).

Collecting, transporting, and holding conditions experienced by coral reef wildlife cause stress. In the pot-bellied seahorse, *Hippocampus abdominalis*, transportation and confinement induced an adrenergic and stress response, including elevated cortisol, glucose, and lactate levels (Wright et al. 2007). Crowded holding tanks and poor water quality cause chronic stress that ultimately reduces an animal's food intake and growth (Santos et al. 2010). Moreover, stress increases an animal's susceptibility to mortality. In *Dascyllus auranus*, for example, non-lethal doses of cyanide became lethal when fish were stressed due to bagging (Hanawa et al. 1989). Similarly, Hall and Bellwood (1995) examined the effects of cyanide, starvation, stress, and handling on mortality in a popular damselfish (*P. coelestis*) in the aquarium trade. Stress caused the greatest overall mortality of all the treatments they examined, including cyanide exposure and starvation (Hall and Bellwood 1995, Table 9). Stress or handling alone caused the death of 25% of examined animals (Hall and Bellwood 1995). Mortality rates were as high as 66.7% when stress was combined with other factors (Hall and Bellwood 1995).

Table 9: Mortality rates under different treatment conditions, adapted from Hall and Bellwood (1995).

Treatment	**Percent Mortality**
Starvation	0%
No handling control	16.7%
Handling control	25%
Stress	25%
Cyanide + stress	25%
Cyanide + starvation	33.3%
Cyanide	37.5%
Cyanide + stress + starvation	41.7%
Stress + starvation	66.7%

The time it takes for animals to recover from stress varies from a few hours to a few days. Recovery time depends on the duration and degree of the stressor, the species of animal, and the way that recovery time is measured. Seahorse (*H. abdominalis*) cortisol and glucose levels returned to pre-stress conditions within as little as six hours of experiencing chronic stress from confinement and transportation (Wright et al. 2007). In contrast, confinement stress elevated stress hormone levels for 5−7 days in largemouth bass (Carmichael et al. 1984a,b). None of these fish died during transportation, but 88% were dead within four days of the confinement and transportation stress (Carmichael et al. 1984a,b). If analogous processes occur in marine ornamentals, the impacts of stress may be long lasting and severe.

Finally, certain coral reef species are simply unsuitable for life in captivity. Many coral reef ornamentals have specific dietary, habitat, or other requirements that are difficult to replicate in captivity (Schmidt and Kunzmann 2005). Furthermore, some species are so extremely vulnerable to over collection that they simply should not be harvested. Different authors have assembled lists of these unsuitable species. For example, Sadovy (2002) listed select angelfish and seahorses as examples of inappropriate species. Sadovy and Vincent (2002) characterized ≤40% of species in trade as having dietary or husbandry requirements that are beyond the skill level of most hobbyists (Sadovy 2002). Despite this, methods of reef tank husbandry are constantly improving (Rhyne and Tlusty 2012). What was once an unsuitable species could become the next success story in the reef aquarium hobby. The mandarinfish provides one notable example; it has specialized feeding requirements and does poorly in captivity (see Chapter 4 and references within). However, advances in captive breeding have produced captive-bred mandarinfish that are trained to eat a prepared diet. In general, species with difficult feeding requirements, short captive lifespans, or highly vulnerable wild populations should be avoided.

Conservation methods to avoid injury and mortality to coral reef wildlife:

In theory, injury and mortality can be nearly eliminated from the coral reef wildlife trade through improved practices and a shortened supply chain (Sadovy 2002, Wabnitz et al. 2003). Perhaps the most obvious reforms are the use of non-destructive collection methods rather than damaging practices like cyanide fishing, smashing corals to uncover fish, abrasive nets, and spearfishing. The use of hand nets, barrier nets, or hook and line fishing, combined with proper swim bladder deflation can substantially lower mortality rates (Rubec et al. 2001, Sadovy 2002). In the Philippines where cyanide use is prevalent, the use of improved handling practices along with the adoption of hand nets reduced mortality from 30% to less than 5% for fish from capture to export (Rubec and Cruz 2002).

Improvements could also be made throughout the supply chain. Many fish are stored in local villages in sealed plastic bags awaiting onward transportation (Baquero 1995). Replacing this system with floating cages, submerged nets, or regional holding facilities would reduce post-harvest stress (Rubec and Cruz 2002). When fish arrive at an importing or exporting facility, it is helpful to acclimatize animals to holding tanks by slowly improving water quality and not suddenly shocking the animal with different environmental conditions (Rubec and Cruz 2005). Mortality is also minimized in facilities with excellent filtration systems and practices like quarantining of animals upon arrival to prevent the spread of disease, long acclimatization time to aquarium conditions, and feeding a diet tailored to the needs of each species (Wood 2001b, Sadovy 2002, Rubec and Cruz 2005). Crowding of fish into confined conditions during shipping and holding at import or export facilities is a potentially significant contributor to wildlife stress and mortality. Reducing crowding by stocking fewer fish per container would alleviate this problem, but would increase costs and space requirements (Rubec and Cruz 2005).

During transportation, special techniques have been shown to alleviate signs of stress in salmon (Farrell et al. 2010) and it is possible that similar techniques could be applied to marine ornamentals (Rubec and Cruz 2005). Mortality of freshwater fish during transport has been reduced by adding chemicals to water that sedate the fish, reduce bacterial growth, neutralize ammonia, and buffer pH; Rubec and Cruz (2005) recommend that these methods be adapted for marine ornamentals. Addition of tris buffer, for example, controls the acidity of shipping water, decreases carbon dioxide accumulation, and considerably prolongs the survival of killifish (*Fundulus parvinnus*) compared to non-buffered controls (McFarland and Norris 1958). Sedatives and lowered water temperatures (within the tolerance range of fish) can also lower the rates of fish metabolism and decrease the fouling of shipping water (Rubec and Cruz 2005). There are chemical treatments that remove ammonia after it is released as waste (Bower and Turner 1982), control bacterial growth in shipping bags (Amend et al. 1982, Teo et al. 1989, 1994), enhance fish health (Lim et al. 2002, 2003), and anesthetize the fish during shipment (McFarland 1960, Teo et al. 1989, 1994, Kwan et al. 1994, Cole et al. 2001, Lim et al. 2003). However, some of these chemicals cause health problems in fish and additional study is required to fully access their efficacy on ornamental marine fish.

The approaches outlined above could be combined in the context of improving coral reef resource management and conservation (Sadovy 2002). Theorhetically, one way of holistically improving practices and reducing mortality in the supply chain is through a certification program that creates market incentives for environmentally responsible products. Ideally, a successful certification program would provide consumers with the opportunity to make informed choices about the products they purchase and sellers with a market advantage over non-eco-labeled products. Programs for sustainable seafood provide an interesting and somewhat successful example of this approach. The success of a sustainable product certification program depends heavily on the program’s credibility. Credibility requires an independent program with standards based on scientifically-meaningful information and the presence of a third-party verified chain of custody. Success also requires the right economic incentives to drive demand towards the certified product.

One initiative, the Marine Aquarium Council (MAC), attempted to create a certification program targeting the marine aquarium trade that strived for ecologically sustainable marine aquarium fish, corals, and other reef wildlife (reviewed in Mathews Amos and Claussen 2009). The MAC program developed standards covering practices throughout the supply chain. It outlined requirements for third-party certification for sustainability, including fishery and ecosystem management, collection, handling, husbandry and transport, and aquaculture management. MAC began in 1998, with certification starting in 2001 (Mathews Amos and Claussen 2009). Unfortunately, MAC’s major initiative in Indonesia and the Philippines, the Marine Aquarium Market Transformation Initiative (MAMTI), failed to accomplish its goals and funding was terminated after just 3.5 years (Mathews Amos and Claussen 2009). By 2009, very little certified product had reached consumers and a lack of funding caused the program to largely shut down (reviewed in Mathews Amos and Claussen 2009), although certain importers still stock MAC certified fish.

Many different challenges limited the success of MAC. First and foremost, MAC’s claims of improved sustainability could not be verified and therefore lacked credibility (Mathews Amos and Claussen 2009). This resulted from a severe lack of scientific, administrative, management and legal frameworks for certification in source countries, especially Indonesia and the Philippines (Mathews Amos and Claussen 2009). Claims of sustainability were further challenged by (1) a lack of enforcement capacity for fisheries management in many source countries, (2) the large number of species in trade making traditional management approaches challenging, (3) the complexities of coral reefs making it difficult to study and manage these ecosystems, (4) the severe lack of baseline un-impacted coral reefs for comparisons and monitoring, (5) the large number of collectors that did not understand the need for sustainability and failed to implement the MAC standards once training was completed, and (6) unaccountable roving collectors who move from place to place depleting one area after another (Mathews Amos and Claussen 2009). In the absence of support from governments, businesses, and other NGOs, MAC faced a tremendous burden compared to other certification schemes

(Mathews Amos and Claussen 2009). MAC acted as the standard setting body, as the implementer of training in the field, and as a monitor of the success of the program, which created a potential conflict of interest and undermined credibility of the program (Mathews Amos and Claussen 2009).

Secondly, MAC certification failed to clearly and consistently document sustainability throughout the supply chain (Mathews Amos and Claussen 2009). This occurred because of the numerous steps in the supply chain, the lack of formal integration between the thousands of collectors, traders, and exporters/importers involved in the industry, the lack of understanding of the value of physical documentation in the source countries, and the lack of oversight from any outside entity (Mathews Amos and Claussen 2009). Furthermore, the ever-changing demand for many different species often requires mixing certified and uncertified specimens (Mathews Amos and Claussen 2009). Most collectors, traders and exporters lacked the facilities or incentives to separate certified and uncertified fish at every step in the chain (Mathews Amos and Claussen 2009). The result was a lack of credibility of sustainable practices from collection to sale (Mathews Amos and Claussen 2009).

Finally, MAC certification alone did not create the economic incentives necessary for reform (Mathews Amos and Claussen 2009). Collectors often lacked the resources necessary for moving into compliance. Although exporters and importers had resources to comply with standards, most lacked the competitiveness and profit incentives to move into compliance. To overcome these challenges and to create a demand for sustainable products, certification programs must create value for the consumer and cost reduction for the industry. Unfortunately, MAC did not result in the price premiums, quality improvements or mortality reductions necessary to drive this change in demand (Mathews Amos and Claussen 2009). The cost of implementing voluntary standards exceeded the cost of mortality for collectors and exporters (Mathews Amos and Claussen 2009). Operations that collected sustainably were undercut by collectors that use the same reefs but did not pay the costs of certification (Mathews Amos and Claussen 2009). Most consumers did not know about or support MAC standards and the few consumers that were aware of MAC were skeptical of the MAC standard’s efficacy (Mathews Amos and Claussen 2009). Fundamentally, the large supply of cheap and uncertified fish made price premiums for sustainable products challenging to implement (Mathews Amos and Claussen 2009).

Despite the inadequacies of MAC, it is possible that coupling renewed effort at improving standards with legal reforms could be more successful. New industry standards could be successful if implemented in conjunction with holding industry participants accountable through new regulatory requirements and adequate enforcement that create proper market incentives to comply with higher standards.

Chapter 10
Cyanide Fishing

Cyanide fishing is a widespread problem with severe environmental consequences. Fishing with cyanide and other poisons is likely the single largest source of mortality and environmental damage in the coral reef wildlife trade. As a result, this topic is examined separately from other sources of injury and mortality in the supply chain (Chapter 9). Cyanide fishing is commonly used in both the aquarium and live reef fish food (LRFF) trades. Here the text will maintain its focus on the coral reef wildlife trade (i.e., the aquarium trade), but studies of cyanide's extent of use and impacts are drawn from both trades.

Figure 21: Photograph of collector demonstrating the use of cyanide. Image taken from Cervino et al. (2003).

Cyanide is used for fishing because of its rapid anesthetic effects, ease of application, and low cost (Rubec 1988a, McManus et al. 1997, Conservation and Community Investment Forum 2001). Typically, aquarium collectors place sodium cyanide (NaCN) tablets in a squirt bottle (i.e., 1–3 20 gram tablets in a 1 liter container; Rubec et al. 2000, 2001, Gonzales and Savaris 2005). Additional chemicals (e.g., detergent or kerosene) can be added to the bottle to improve the visibility of the poison underwater (Debenham 2008). The result is a white plume of hydrogen cyanide (HCN), un-dissolved cyanide particulates, and sometimes other chemicals dispensed into the environment (Rubec et al. 2000, 2001, Gonzales and Savaris 2005). In order to stun and capture live fish, collectors disperse the solution on coral colonies and around the reef while snorkeling, free-diving, or using a hooka (Rubec et al. 2001, Cervino et al. 2003, Fig. 21). Stunned fish can then be easily captured by hand or with nets. Fish will also swim out of coral crevices to escape the poison, enabling their capture. The resulting concentration of dissolved cyanide on the reef is approximately 1,500−2,000 mg per liter (Pet and Djohani 1998), with some doses as high as 120,000 mg per liter (Johannes and Riepen 1995, Barber and Pratt 1998, Jones et al. 1998). Collectors are unable to control the dose of cyanide administered to fish. The resulting physiological effects on wildlife range from non-lethal anesthesia to immediate death. For marine fishes, the range of acute toxicity for CN^- is 0.05 to 5 mg per liter (reviewed by Hanawa et al. 1998). Hanawa et al. (1998) found that the range of concentrations and exposure times separating non-lethal anesthesia and immediate death to be very narrow, leaving a very small window for safety.

Besides the squirt-bottle method, there are several other means of administering cyanide to capture coral reef fish (reviewed in Jones and Steven 1997). A cyanide tablet can be attached to a stick and held in the vicinity of a fish (McManus et al. 1997). Cyanide can also be placed in bait that is cast overboard (Jones and Steven 1997). Fish that eat the poisoned bait will rise, stunned and vomiting, to the surface where they are easily captured (Jones and Steven 1997). Cyanide is sometimes pumped from 5-gallon containers onto the reef (Jones and Steven 1997). In some cases, 55-gallon drums of cyanide are cast overboard directly onto reefs in order to capture fish for food (del Norte et al. 1989, Johannes and Riepen 1995, Bruckner and Borneman 2008). Although cyanide is the most common poison used to capture reef fish, alternative poisons are also used, including clove oil, quinaldine, bleach, and gasoline (Randall 1987, Sadovy 1992, Bruckner 2000).

Cyanide fishing began in 1962 in the Philippines; over time the practice spread to much of Southeast Asia, Oceania, and the tropical Indian Ocean (Barber and Pratt 1997a,b). As of the 1980s, an estimated 1,000 aquarium fish collectors in the Philippines were each using 75 kg of sodium cyanide per year, for an annual total of 75,000 kg of cyanide dumped onto the coral reefs of the Philippines (McAllister 1988). Those numbers approximately doubled to an estimated total of 150,000 kg when live reef fish food collection was also considered (McAllister 1988). Between the 1960s and late 1990s, at least 1 million kg of cyanide was dispensed onto the coral reefs of the Philippines alone (Bryant et al. 1998). Estimates vary and the number may (or may not) be even higher than the values cited above. Based on a very rough calculation built upon the estimates of previous studies, Rubec et al. (2001) estimated that cyanide fishing in the Philippines “might exceed 500,000 kg per year”.

Many studies and reports suggest that approximately 70–90% of coral reef fish in the aquarium trade are captured using cyanide. For instance, Hingco and Rivera (1991) estimated that 70% of aquarium fishers in the Philippines used cyanide. However, such estimates are often presented without a corresponding reference or methodology, so it is difficult to determine their accuracy. One exception is the International Marinelife Alliance (IMA) which tested 48,000 fish in the Philippines and found evidence for cyanide fishing in 25% of aquarium fish going to U.S. and Europe and 44% of live groupers and humphead wrasse going to Hong Kong for the LRFF trade (Debenham 2008, http://www.aquariumcouncil.org/). However, detection of cyanide is confounded by the rapid conversion of cyanide into other chemicals and therefore the reports of detection rates should be interpreted very cautiously (Mak et al. 2005, Bruckner and Roberts 2008, but see Rubec et al. 2008, Vaz et al. 2012).

Cyanide fishing is illegal in most countries, but the practice persists and has even spread to new locations (Johannes and Riepen 1995, Barbor and Pratt 1997a). The Philippines, for example, banned cyanide fishing in 1979, but the practice has continued ever since (McManus et al. 1997, Conservation and Community Investment Forum 2001). Enforcement of cyanide bans is often weak and corruption commonly confounds enforcement efforts; cyanide fishers sometimes pay off those responsible for enforcing anti-cyanide regulations (Barber and Pratt 1998).

Today, fishing with cyanide occurs in around 15 countries, many of which are major exporting nations in the coral reef wildlife trade (Bruckner and Roberts 2008). Bruckner and Roberts (2008) characterized cyanide fishing as "pervasive" in the aquarium and LRFF trades. Countries with confirmed cyanide fishing include the Philippines, Indonesia, Vietnam, Malaysia, and Papua New Guinea (Rubec 1986, Johannes and Riepen 1995, Barber and Pratt 1998). Use of this poison is strongly suspected in Eritrea, the Marshall Islands, Solomon Islands, the Maldives, and Sri Lanka, among others (Barber and Pratt 1998, Debenham 2008). Cyanide is rampantly used in Indonesia and certain regions of the Philippines (Conservation and Community Investment Forum 2001, see also Pat 1997 for an alternative perspective on Indonesia and Gonzales and Savaris 2005 for a breakdown on prevalence of cyanide use by region in the Philippines). Cyanide fishing does not regularly occur in many Pacific Island nations (e.g., Fiji), the U.S. (i.e., Hawaii or Florida), or Australia (Conservation and Community Investment Forum 2001, Debenham 2008).

Cyanide is predominantly used when fishing for high-value species such as surgeonfish in the marine aquarium trade or groupers or wrasses in the live reef fish food trades (Bruckner and Roberts 2008). Aquarium fish captured with cyanide include blue ring angelfish (*Pomacanthus annularis*), emperor angelfish (*Pomacanthus imperator*), blue surgeonfish (*Paracanthurus* sp.), and among many others (Fahrudin 2003, Bruckner and Borneman 2008).

Cyanide is effective as a fish anesthetic because it is a rapidly-acting, highly-toxic poison. Hydrogen cyanide travels quickly across cell membranes (Duodoroff 1980) and often inhibits membrane-bound proteins. For example, cyanide will terminate cellular respiration by inhibition of important membrane-bound enzymes (Buchel and Garab 1995), such as the mitochondrial electron transport enzyme cytochrome C oxidase (Egekeze and Oehme 1980, Leduc 1984, Ballantyne 1987, Buchel and Garab 1995). Blocking cytochrome C oxidase prevents normal oxygen utilization, preventing adenosine tri-phosphate production, and thereby leading to the failure of mitochondrial respiration and ultimately cell death (Egekeze and Oehme 1980, Leduc 1984, Ballantyne 1987, Buchel and Garab 1995). Cyanide also reacts with various heme and cobalamins compounds, thus blocking other biochemical components of metabolic redox processes (Ballantyne 1987). In coral reef ecosystems, cyanide inhibits carbonic anhydrase, an enzyme necessary for coral calcification and reef accretion (Hayes and Goreau 1977). In photosynthetic organisms like symbiotic corals, cyanide inhibits photosynthesis at various biochemical targets, including ribulose-1,5-bisphosphate carboxylase/oxygenase, plastoquinone-oxidoreductase, and/or photosystem II (Wishnik and Lane 1969, Buchel and Garab 1995, Jones et al. 1998, Jones and Hoegh-Guldberg 1999). In all of these different problems, acute cyanide toxicity occurs when absorption rate exceeds an organism's detoxification rate (Magos 1992).

Because of its toxicity and rapid action, cyanide commonly causes severe injury in coral reef wildlife. Eisler (1991) conducted a literature review on the effects of cyanide in a broad range of terrestrial, aquatic, and marine taxa. Of the organisms examined, fish were one of the most highly-sensitive groups to cyanide poisoning (Eisler 1991). As little as 0.5 μg of cyanide per liter of water adversely affected fish swimming ability and reproduction (Eisler 1991). Cyanide can cause strange feeding behaviors ranging from total loss of appetite to impaired digestion, which ultimately leads to starvation (Herwig 1976, 1977). In the three-spot dascyllus, *Dascyllus trimaculatus*, cyanide exposure was correlated to deterioration of the stomach and intestinal tract's mucosal lining (Bellwood 1981); however, stress and starvation likely contributed to (or even caused) this finding (Hall and Bellwood 1995). Dempster and Donaldson (1974) experimentally exposed marine fish from California to sodium cyanide and found that the poison damaged the fishes' liver, heart, kidneys, spleen, and brains. Similar effects have been seen in marine aquarium fish from coral reefs. Hanawa et al. (1998), for example, examined response of *Dascyllus aruanus* to cyanide and found that the surviving fish had impaired liver oxygen consumption that persisted for 2.5 weeks after cyanide exposure.

The concentrations of cyanide used to capture fish are sufficient to cause death by acute toxicity (Leduc 1984). As a result, many (i.e., approximately 50%) of the fish exposed to cyanide die on the reef (Rubec 1986, Rubec et al. 2001). Delayed mortality, known as "sudden death syndrome," is also a common consequence of cyanide exposure. According to Perino (1990), 5–25% of fish collected with cyanide or other chemicals die shortly after capture, with 20–40% more perishing within several hours to days. Rubec (1986) reported that more than 80% of fish that initially survive cyanide exposure die within about 6 weeks of capture.

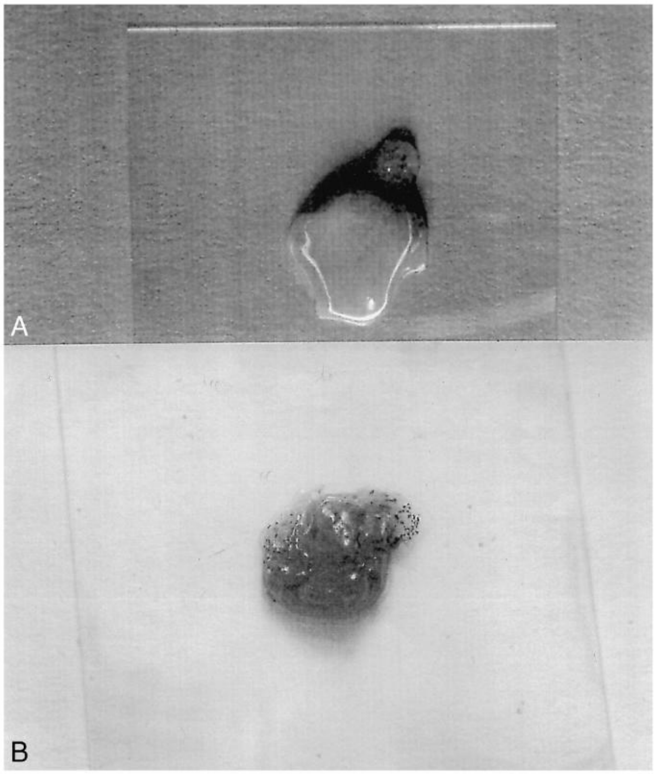
Hall and Bellwood (1995) examined mortality in the neon damselfish, *Pomacentris coelestis,* after cyanide exposure, handling stress, and starvation, as well as combinations these stressors over 13 days. Nineteen percent of fish that were exposed to cyanide at concentrations of 10 mg per liter for 30 seconds immediately died (Hall and Bellwood 1995). All of the fish exposed to 10 mg of cyanide per liter of water for 2 minutes died (Hall and Bellwood 1995). No delayed effects of cyanide exposure, as measured by digestive system condition, were observed (Hall and Bellwood 1995). Note that when cyanide is used to capture coral reef wildlife, fish are potentially exposed to this poison at concentrations that are multiple orders of magnitude higher than the concentrations used by Hall and Bellwood (Johannes and Riepen 1995, Barber and Pratt 1998, Jones et al. 1998, Pet and Djohani 1998).

In a related study Hanawa et al. (1998) examined response of the whitetail dascyllus, *Dascyllus aruanus,* to cyanide. Fish were exposed to cyanide at 25 and 50 mg per liter concentrations for time intervals of 10, 60, or 120 seconds. Exposure to cyanide at any concentration or time interval caused "severe gasping, followed by loss of balance and a complete loss of all respiratory activity" (Hanawa et al. 1998). Within 40 seconds of exposure, *D. aruanus* were immobilized (Hanawa et al. 1998). Swimming ability and fin movement were all impaired by cyanide, with fish swimming on their sides during recovery (Hanawa et al. 1998). Greater cyanide exposure led to a longer recovery time; fish that were not killed immediately by the cyanide took up to 50 minutes to return to normal behavior (Hanawa et al. 1998).

Hanawa et al. (1998) tracked the mortality rate of cyanide-exposed fish over 2.5 weeks. In the absence of stress, exposure to 25 or 50 mg of cyanide per liter for 10 or 60 seconds did not cause mortality (Hanawa et al. 1998). However, fish exposed to low doses of cyanide for 120 seconds died (i.e., 60% mortality at 25 mg and 100% mortality at 50 mg). To replicate conditions during collection a subset of fish were netted and bagged. The stress of netting and bagging caused previously non-lethal exposure to cyanide to become 100% lethal (Hanawa et al. 1998). The combined stress of cyanide and handling increased both the mortality rate and the metabolic load on the fish (Hanawa et al. 1998). Handling stress and cyanide exposure could explain the delayed mortality or “sudden death syndrome” observed in many coral reef wildlife in the aquarium trade.

After 2.5 weeks of recovery, Hanawa et al. (1998) examined the internal state of the surviving fish. A brief pulse of cyanide exposure appeared to cause irreversible and long-lasting damage to liver tissues (Hanawa et al. 1998). The livers of exposed fish were covered in dark red blood clots whereas control fish didn’t have this response (Hanawa et al. 1998, Figure 22). Oxygen consumption in the liver of the surviving fish remained impaired, even after 2.5 weeks of normal conditions (Hanawa et al. 1998). Cyanide exposed and stressed (i.e., bagged) fish experienced abnormal liver function, including a 6 to 17 fold increase in oxygen consumption rates after 2.5 weeks (Hanawa et al. 1998). Abnormal conditions were observed − including pycnosis, a contraction of the cell nucleus and condensation of chromatin into stained clumps, and cytolysis − with severity of these conditions increasing with the dose of cyanide (Hanawa et al. 1998).

Figure 22: Complete livers from (a) control and (b) cyanide-exposed fish showing extensive clotting. Figure taken from Hanawa et al. (1998).



The harmful effects of cyanide fishing extend beyond the individual fish targeted for capture. Individuals that are not removed by collectors experience longer-duration exposures to cyanide that often kill the organism. According to Rubec (1988b), only 10% of fish emerging from cyanide-treated refuges were captured, the rest were left on the reef to die. Many more fish and invertebrates remained entombed within the coral reef structure (Rubec 1988b). Species that are unsuitable for the live reef fish food or aquarium trades, as well as individuals that are the wrong size, are discarded to die (Wood 2001b).

Corals, anemones, and other habitat-forming species are also severely and negatively affected by cyanide fishing. Cyanide commonly induces localized bleaching in the exposed corals (Mak et al. 2005) that often culminates in the death of the colony.

Figure 23: Photograph of a tabulate coral that was smashed by collectors during cyanide fishing in Indonesia. Photo by Crissy Hufford, Conservation International.

Collectors also commonly break apart corals to access the stunned and poisoned fish (Figure 23). Over the years, research effort has been directed towards assessing the effects of cyanide on corals and anemones. For example, Chalker and Taylor (1975) showed that 5 mg of cyanide per liter of water inhibited photosynthesis and calcification in the corals *Acropora formosa* and *A. cervicornis*. Barnes (1985) showed that the coral *A. formosa* experienced declines in respiration, but not total respiratory inhibition, when exposed to 1 x 10^{-4} M NaCN. Similar results were obtained for the anemone *Aiptassia pulchella* (Pickles 1992).

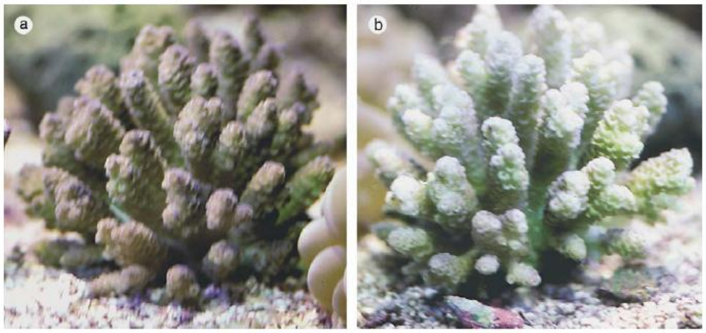
In recent years, physiological, biochemical, and lethal effects of cyanide on corals have been rigorously examined. Jones and Steven (1997), for instance, subjected *Pocillopora* and *Porites* spp. corals to levels of cyanide experienced during cyanide fishing and examined the physiological consequences. Exposure to cyanide at concentrations of 5,200 mg per liter for 10−30 minutes resulted in the death of the coral (Jones and Steven 1997). The lethality of cyanide in *Pocillopora* spp. was also confirmed in unpublished work by Dr. R. Richmond at the University of Guam (as cited in Johannes and Riepen 1995). Application of lower cyanide concentrations induced bleaching, impaired photosynthesis, and inhibited respiration by 10−90%, each of which could eventually kill the coral colony (Jones and Steven 1997). The level of respiratory inhibition increased with cyanide concentration and duration of exposure (Jones and Steven 1997). Exposure to cyanide-free water for one to two hours restored corals to their normal respiratory rates (Jones and Steven 1997).

Jones et al. (1998) exposed small branches of the corals *Stylophora pistillata* and *Acropora aspera* to low doses of cyanide. At cyanide exposure levels that replicated cyanide fishing, photosynthetic electron transport totally ceased (Jones et al. 1998). Jones and Hoegh-Guldberg (1999) conducted similar work on the coral *Plesiastrea versipora*. Exposure to very low concentrations of cyanide (i.e., less than 10^{-5} M NaCN for 3 hours) under saturating light intensity caused a long-term decrease in photosystem II function and bleached the experimental corals (Jones and Hoegh-Guldberg 1999). These results demonstrate that cyanide causes symbiotic breakdown through impairment of the endosymbiont photosystem (Jones et al. 1998, Jones and Hoegh-Guldberg 1999). Because *Symbiodinium* supply a large portion of the corals' energy budget (Muscatine et al. 1981), the impairment of photosynthesis and concomitant bleaching could have long-term negative effects on coral reefs.

Figure 24: Coral experiencing tissue detachment as a result of cyanide exposure. Figure taken from Cervino et al. (2003).

Cervino et al. (2003) further examined the effect of cyanide on symbiotic cnidarians. Ten species of corals and anemones were exposed to various concentrations (i.e., 50,100, 300 and 600 mg per liter) of cyanide for up to 2 minutes (Cervino et al. 2003). The cyanide-exposed cnidarians were tracked for up to 7 months following exposure (Cervino et al. 2003). In all the corals examined, cyanide caused tentacle retraction, copious mucus production, and changes in protein production (Cervino et al. 2003). The cyanide-exposed corals also experienced gastrodermal disruption, mesoglea degradation, tissue detachment, changes in their associated bacterial communities, and/or an impaired relationship with their dinoflagellate symbionts (*Symbiodinium* spp.) (Cervino et al. 2003, Figure 24). *Symbiodinium* cells within these corals experienced pigment loss, swelling, and deformation (Cervino et al. 2003). *Symbiodinium* cell density was reduced (i.e., bleached) in six out of seven species examined at this level (Cervino et al. 2003, Figure 25). Coral and anemone mortality varied by species, cyanide concentration, and time after cyanide exposure. Specifically, 30−50% of corals died at 50 mg per liter, 50−66% died at 100 mg per liter, 22.2−100% died at 300 mg per liter, and 50−100% died at 600 mg per liter (Cervino et al. 2003). Thicker-tissue species responded more slowly than thin-tissue species (Cervino et al. 2003). In Cervino et al.'s study, *Acropora* spp. were the most susceptible to cyanide exposure, with all *Acropora* colonies

Figure 25: A *Pocillopora* sp. coral before (a) and after (b) cyanide exposure. Note the bleached state of the cyanide-exposed colony. Figure taken from Cervino et al. (2003).



dying within a day of exposure. Branching *Acropora* spp. corals are likely the most commonly poisoned corals in nature because fish take refuge in their branches to avoid collectors. Unfortunately, *Acropora* spp. may be both the most cyanide-exposed and most cyanide-susceptible groups of corals (Cervino et al. 2003).

Downs et al. (2010) recently isolated and cultured single-cell suspensions of the coral *P. damicornis.* These cell suspensions are useful for precise ecotoxicology studies of cyanide and other toxins. To demonstrate this, Downs et al. examined the response of coral cell suspensions to 10 μM, 100 μM, 1 mM, and 10 mM concentrations of potassium cyanide for 4 hours (Downs et al. 2010). At all doses, cyanide exposure caused a statistically-significant drop in mitochondrial membrane potential, thereby affecting cellular respiration (Downs et al. 2010). The degree of decline increased with increasing cyanide concentration (Downs et al. 2010). The decrease was small at 10 and 100 μM cyanide, but at 1 mM cyanide membrane potential decreased by approximately 85% (Downs et al. 2010). Cyanide also caused significant *P. damicornis* cell mortality at concentrations at and above 100 μM, with greater than 90% cell death at 10 mM cyanide (Downs et al. 2010). The concentration of cyanide required to kill 50% of cells after 3 hours, known as LC_{50}, was just 5.1 mg per liter (Downs et al. 2010). Observational studies suggest that much higher concentrations of this poison are dispersed onto the reef during cyanide fishing (Johannes and Riepen 1995, Barber and Pratt 1998, Jones et al. 1998, Pet and Djohani 1998), likely resulting in severe coral mortality.

Cyanide fishing is one of the most destructive fishing practices in the world. Its impacts surpass the impacts of dynamite fishing by destroying thousands of acres of coral reefs every year (Mak et al. 2005). Cyanide fishing indiscriminately injures or kills large numbers of corals, fish, and other reef wildlife, thereby reducing coral reef biodiversity and causing lasting ecosystem degradation. Although threats like climate change, acidification, overfishing, and pollution are widely regarded as the largest threats facing coral reefs (e.g., Hoegh-Guldberg 1999, Hughes et al. 2003), in a number of countries destructive fishing represents the largest and most immediate threat to the continued existence of this ecosystem (Pet-Soede 2001). Cyanide fishing is now well recognized as one of the major factors responsible for the ongoing degradation of coral reef ecosystems worldwide (Rubec and Pratt 1984, Rubec 1986, 1988a, Barber and Pratt 1997a,b, 1998, Burke et al. 2011).

There are many case studies in the scientific and gray literature documenting cyanide’s broad negative effects on coral reef ecosystems. In the waters around Mactan Island in the central Philippines, many corals died after just two incidents of cyanide fishing over a period of 4 months (Bureau of Fisheries and Aquatic Resources unpublished data as cited in Rubec 1986, 1987a). McLarney (1986) described the destroyed reefs of the Danajan Bank region between the islands of Cebu and Bohol in the Philippines. Much of the devastation that McLarney documented, especially the loss of reef fish communities, could be attributed to cyanide. Allen and Werner (2002) surveyed reef fish communities in four areas of the Coral Triangle and found low biodiversity in Indonesia and the Philippines due to habitat homogeneity and wide-spread destructive fishing practices, including cyanide use. The areas affected by cyanide included the Calamianes Islands in northern Palawan Province, the Philippines, Togean and Banggai Islands in central-eastern Sulawesi, Indonesia, and Weh Island in Aceh Province, Sumatra, Indonesia (Allen and Werner 2002). The Calamines Islands had low fish diversity, few large fish and widespread evidence of cyanide fishing (Allen and Werner 2002). The Togean and Banggai Islands had widespread cyanide and dynamite fishing, plus large human populations that had negative impacts on fish communities (Allen and Werner 2002). Overall, cyanide fishing contributes to the loss of habitat and biodiversity on coral reefs; this causes collectors to move on to less degraded locations and thereby increases the reach of cyanide's negative impacts (Rubec 1986, Johannes and Riepen 1995, Barber and Pratt 1998).

Several studies attempted to quantify the long-term impacts of cyanide fishing. Saila et al. (1992) modeled destructive fishing, including dynamite and cyanide fishing as well as anchor damage, in the Philippines. At that time, the corals at Santiago Island area of Luzon, the Philippines were already 50% lost due to destructive fishing practices (Saila et al. 1992). The model estimated that the loss due to cyanide fishing under three different scenarios: a best, worst, and moderate case for future cyanide fishing (Saila et al. 1992). Under the worst case scenario, the model predicted that cyanide fishing would destroy 9.5% of the remaining reef every year (Saila et al. 1992). Reef degradation would continue at 2% per year in the moderate case scenario and 0.002% per year in the best case scenario where cyanide fishing was eliminated (Saila et al. 1992). The replacement rate of corals is approximately 1% per year in the Philippines; therefore only the best case scenario prevented continued reef degradation from cyanide fishing (Saila et al. 1992). However, the best case scenario remains unlikely. Cyanide fishing has already been banned in the Philippines but the practice continues (McManus et al. 1997, Conservation and Community Investment Forum 2001). Of all the scenarios Saila et al. modeled, elimination of cyanide fishing had the greatest positive effect on Philippine coral reefs.

McManus et al. (1997) developed a simple balance sheet model to estimate the impacts of destructive fishing, including cyanide, on coral reefs of Bolinao, the Philippines. The authors based their model inputs on data from field surveys, interviews with fishers, and estimates based on the best information available (McManus et al. 1997). Cyanide was estimated to be used by 70% of fishers (based on Hingco and Riviera 1991). At Bolinao, blast fishing (which is not associated with the ornamental trade) had caused the largest amount of damage (McManus et al. 1997). However, cyanide fishing was estimated to be responsible for the destruction of up to 8% of the reef area each year, the equivalent of 55,845 coral colonies killed per km^2 of reef area (McManus et al. 1997). Under the most likely scenario modeled, cyanide caused a loss of 0.4% of the reef corals per year, the equivalent of 7,008 colonies per km^2 of reef area (McManus et al. 1997).

Conservation science and cyanide fishing:

Cyanide has remained popular as a fishing method because collectors can take more fish with less effort than could be taken using less damaging methods (Rubec et al. 2001). Non-destructive hand or barrier nets are an obvious alternative. The Haribon Foundation and Ocean Voice, for example, trained fishers in the use of non-damaging collection methods as alternatives to cyanide (Bruckner and Roberts 2008). Unfortunately, many fishers trained in alternative methods quickly revert to using poison because of the economics of aquarium collecting (Rubec et al. 2001). In order for the practice to be eradicated, reforms are needed that change the economic incentives. Rubec et al. (2001) encouraged higher payments to collectors for sustainably-collected fish. In the Philippines, the exporters' purchasing price is divided unevenly between the middlemen (85% of the price) and fishers (15% of the price) (Rubec et al. 2000, Gonzales and Savaris 2005). Changing this profit structure could have the combined benefit of improving the livelihood of collectors and maintaining healthier fish populations on the reef (Rubec et al. 2001).

In addition to hand or barrier nets, alternative chemicals have been suggested as more ecologically-friendly alternatives to cyanide fishing (Erdmann 1999). Clove oil is one potential candidate. It can be used in the same manner as cyanide fishing and it effectively calms and sedates fish without killing them (Munday and Wilson 1997, Keene et al. 1998, Ackerman and Bellwood 1999, Erdmann 1999). It is also readily available in Indonesia, one of the primary countries where cyanide fishing takes place (Soto and Burhanuddin 1995). In the coral *P. damicornis*, the sustainability of clove oil fishing is dependent on the dose used (Frisch et al. 2007). Clove oil kills corals at high concentrations (i.e., 50 ppt) for any duration (Frisch et al. 2007). Lower concentrations (i.e., 5 ppt) had effects that depended on the exposure duration, ranging from no effect after 1 minute exposure, to bleaching after 10 minutes of exposure, to death after 60 minutes of exposure (Frisch et al. 2007). Among corals that survived the exposure, there were no lingering effects after 63 days (Frisch et al. 2007). Therefore, Frisch et al. suggested that using clove oil sparingly is a more ecologically sound alternative to cyanide fishing. However, controlling the dosage of any chemical is extremely difficult in the marine environment. Quinaldine is another chemical that is used to collect fish; however, this chemical causes bleaching in corals and may only be moderately better than cyanide (Jaap and Wheaton 1975). Unlike cyanide, quinaldine is used to collect fish in U.S. waters (Jaap and Wheaton 1975).

Many countries have already banned cyanide fishing and yet use of this poison continues. Enforcement of anti-cyanide laws is often non-existent or weak. Some conservation efforts to combat cyanide have been undermined by corruption. Barber and Pratt (1998) describe the failure of efforts in Milne Bay, Papua New Guinea to curb cyanide fishing by Hong Kong live fish traders. The government of Papua New Guinea introduced economic ventures to generate employment opportunities in fisheries (Barber and Pratt 1998). In collaboration with D'Entrecasteaux Youth Marine Supplies (DYMS), Charter Wide Investments (CWI) was brought in to develop fishing operations that trained and employed young collectors in hook and line fishing (Barber and Pratt 1998). Instead of training collectors in sustainable methods, CWI trained 40 fishermen to fish with cyanide (Barber and Pratt 1998). The CWI cyanide fishers were then deployed to collect fish for the live reef fish food trade using poison (Barber and Pratt 1998). Contrary to its responsibilities, CWI did not make certain that the economic benefits of fish collection were locally realized (Barber and Pratt 1998). CWI even forced fishermen to work extremely long hours (12 hr / day) for a month at a time, under arduous conditions, with minimal food, and very low pay (Barber and Pratt 1998). The working conditions were so poor and the concentrations of cyanide used were so high that many of the fishermen experienced symptoms consistent with cyanide poisoning (Barber and Pratt 1998). In the end, the whole operation was shut down and CWI and DYMS's permits were revoked (Barber and Pratt 1998).

One way of improving enforcement of anti-cyanide fishing laws is through testing of marine aquarium and LRFF trade fish for cyanide. During the 1990s and early 2000s, the International Marinelife Alliance (IMA) and the Philippine Bureau of Fisheries and Aquatic Resources (BFAR) established a set of Cyanide Detection Laboratories (CDL) to detect this poison in live marine fish in trade (Rubec et al. 2000, 2001). The CDLs used a detection method that was based on ion selective electrodes sensitive to cyanide (Mak et al. 2005). Between 1993 and 2000 the CDLs tested over 32,000 fish (Rubec et al. 2000). The CDLs detected high rates of cyanide in the tested fish, including 25% of aquarium fish going to the U.S. and Europe and 44% of live groupers and humphead wrasse going to Hong Kong for the LRFF trade (Debenham 2008, http://www.aquariumcouncil.org/). From 1996 to 1999 the CDLs also found a reported a decline in the use of cyanide over time (from 43% to 8%), suggesting that the fishery was headed in a positive direction (Rubec et al. 2001). However, when organisms receive sub-lethal doses of cyanide, they quickly convert it into thiocyanate or other metabolites which are then excreted (Eisler 1991, Bruckner and Roberts 2008). The relatively rapid breakdown of cyanide makes detecting this toxin technically difficult (reviewed in Mak et al. 2005 but see Rubec et al. 2008). As a result of this and other technical concerns, the accuracy of the CDL tests has been questioned (Mak et al. 2005, Bruckner and Roberts 2008). Many of the CDLs are now closed and cyanide fishing in the Philippines has apparently increased since the late 1990s (Bruckner and Robers 2008)

In 2008 an international workshop was held to identify a rapid, cost-effective, and efficient way of detecting cyanide in marine fish (reviewed in Bruckner and Roberts 2008). The workshop participants reviewed various cyanide detection methods including "colorimetric methods, enzyme-based biosensors, cyanide-ion selective electrodes (ISE), and biomarker approaches" (Bruckner and Borneman 2008, see also Mak et al. 2005 for a review of these techniques). Of the methods examined, cyanide-ion selective electrodes were the preferred method for detection (Bruckner and Roberts 2008). The ISE method has been around for a long period of time and many people are familiar and comfortable with this approach (Bruckner and Roberts 2008). However, the limitations of cyanide-ISE are also well known (Bruckner and Roberts 2008). Electrodes have limited sensitivity for cyanide detection, are difficult to employ in a field setting, potentially yield false readings, and metabolic conversion of cyanide into other compounds causes cyanide collected fish to escape detection (e.g., Mak et al. 2005, Bruckner and Roberts 2008, but see Rubec et al. 2008). Further verification of the efficacy of ISE was still required, especially to meet the legal requirements for effective enforcement (Bruckner and Roberts 2008). Issues that remained to be addressed are (1) the background levels of cyanide, thiocyanate and related compounds in coral reef fish, (2) the metabolic half-life of these compounds, (3) validation of methods in homogenized fish tissue under experimental conditions, and (4) verification that detection methods are sensitive, specific, accurate, and precise (Bruckner and Roberts 2008). The workshop participants saw a need for establishing new labs in three major exporting countries: Indonesia, the Philippines, and Vietnam (Bruckner and Roberts 2008). Export facilities were identified as a logistically feasible place for testing to occur (Bruckner and Roberts 2008), but these facilities are often far-removed from where collection takes place, allowing time for the breakdown of cyanide (Bruckner and Roberts 2008). The workshop recommended further partnerships between governments and conservation organizations to raise awareness and improve detection capacity among stakeholders in exporting countries (Bruckner and Roberts 2008). Better integration and data sharing between importing and exporting countries were also recommended to improve enforcement of cyanide bans (Bruckner and Roberts 2008).

Fortunately, there has been a major recent advance in cyanide detection technology. A new detection tool using optical fiber technology enables the rapid detection of thiocyante in seawater (Silva et al. 2011). This is important because thiocyanate is excreted for several weeks by fish that are exposed to cyanide (Vaz et al. 2012). Vaz et al. (2012) tested this optical fiber technique on clownfish (*A. clarkii*) that had been exposed to cyanide concentrations consistent with cyanide fishing. Vaz et al. (2012) were able to confirm the exposure of fish to cyanide for at least 28 days following exposure. This tool has the potential to dramatically improve the detection of cyanide fishing and therefore the enforcement of anti-cyanide laws. This optical fiber technology can be deployed quickly and easily without harming the fish being examined (Vaz et al. 2012). However, one potential limitation is that this method requires artificial seawater because background levels of thiocyanate in natural seawater exceed the levels excreted by fish that were captured using cyanide (Silva et al. 2011, Vaz et al. 2012).

Beyond cyanide detection laboratories, the IMA jointly developed the Cyanide Fishing Reform Program with the Philippines BFAR in the late 1980s and early 1990s (Barber and Pratt 1997b, Barber 1999). Since then the organization partnered with local communities and governments with financial help from USAID and the Asian Development Bank (Barber 1999). The Cyanide Fishing Reform Program had success in combating cyanide fishing through increased enforcement, enhanced public awareness via campaigns, training over 2,000 fishers in alternative capture techniques, providing aid to help collectors obtain a better price for their catch, and development of community-based management (Barber and Pratt 1997b, Barber 1999). Despite these efforts, Barber (1999) noted an overall lack of progress in addressing the issue of cyanide fishing. In 1998 the IMA expanded this program to address the issue more broadly across the Indo-Pacific (Barber 1999). The IMA and World Resources Institute launched the Indo-Pacific Destructive Fishing Reform Initiative, a campaign to protect the best reefs of the Indo-Pacific (Barber 1999). Efforts such as this have the potential to reduce cyanide fishing and improve coral reef conservation. For example, Conservation International’s Marine Rapid Assessment Program has had success combating cyanide use in Milne Bay, the Philippines (Allen and Werner 2002). The Rapid Assessment Program was based on engaging locals to report cyanide use, which ceased cyanide fishing in Milne Bay (Allen and Werner 2002).

Barber and Pratt (1997b, 1998) recommended that a comprehensive program to address cyanide fishing should include effective anti-cyanide policies, rigorous law enforcement, efforts to address corruption, cyanide monitoring systems in the field and at export/import centers (potentially using the new cyanide detection method of Silva et al. 2011 and Vaz et al. 2012), media outreach to raise public awareness, training programs to teach collectors reef ecology, dive safety, and sustainable harvesting techniques, changing economic incentives to insure fishers take home a larger profit per fish, and community-based management (see also Bruckner and Borneman 2008). Most of these recommendations target changes in the countries where cyanide fishing occurs. However, there are also opportunities for reform in the importing countries. In the aquarium fish market, the U.S. remains the largest importer (Wabnitz et al. 2003). Through the Lacey Act, the U.S. already bans the importation of animals that were illegally collected in another country. Despite this, the Lacey Act has not been widely applied to combat cyanide fishing because cyanide is difficult to detect in imported fish so as to prove it was used during collection (but see Vas et al. 2012). Cyanide fishing is illegal in much of the world and yet cyanide-collected fish continue to be imported into the U.S. Pet-Soede (2001) noted, in a summary of the results of a destructive fishing minisymposium at the 9th International Coral Reef Symposium in Bali, Indonesia, that there is an untapped opportunity to reform cyanide fishing through the U.S. market. Improved standards on marine fish imported into the U.S. could shift the burden of proof and better-insure that animals were harvested sustainably and legally (Pet-Soede 2001).

Chapter 11:
Invasive Species Introductions

The ecological consequences of the coral reef wildlife trade include introductions of exotic and invasive species into new environments (Smith et al. 2009). Invasive species are a primary threat to biodiversity throughout the world (Wilcove et al. 1998, Bax et al. 2001, D'Antonio et al. 2001) and are responsible for enormous economic losses (Pimental et al. 1999). When introduced to environments where they do not belong, invasive species proliferate and outcompete native taxa, thereby reducing biodiversity and threatening ecosystem function. Invasive species are the second leading cause of extinctions across the world (Wilcove et al. 1998). Once invasive species are introduced into a new environment, they are difficult to control and nearly impossible to eradicate.

Although ballast water is a commonly-cited source for invasive species introductions in freshwater and marine habitats (Ruiz et al. 1997), the trade in coral reef wildlife for the aquarium industry is increasingly recognized as an important source of these introductions (Padilla and Williams 2004). In freshwater ecosystems, for example, the aquarium trade is already well known as a leading cause of fish invasions (Courtenay and Robins 1973, Courtenay and Stauffer 1990). In marine ecosystems, the aquarium trade is less recognized as a source of invasive organisms, but the ecological impacts of these invasive species are significant. For instance, the invasive marine algae *Caulerpa taxifolia* was introduced into the Mediterranean Sea via the aquarium trade (Verlaque and Fritayre 1994, Jousson et al. 1998).

Several risk assessments of the aquarium trade have highlighted the threat posed by the trade (Chang et al. 2009, Weigle et al. 2004). For coral reef ornamental species, locations with the highest risk of invasive species are countries that import a high volume of aquarium species and have suitable habitat and temperature conditions. In the U.S., the states of Florida and Hawaii, as well as tropical island territories (e.g., Puerto Rico), face the greatest risk. Semmens et al. (2004) surveyed 32 locations in southern Florida, Bermuda, and other areas of the western Atlantic Ocean for the presence of non-native fish. The investigators identified 16 species of tropical marine fish that were putatively introduced to these locations via the aquarium trade (Semmens et al. 2004). This estimate has been subsequently increased the estimate to 33 species of nonindigenous fishes introduced to Florida waters (Schofield et al. 2009). Notably, all 16 species of the species discovered by Semmens et al. were high-volume species in the marine aquarium trade, suggesting that there is a connection between the volume of importation and the probability of introduction into locations where these fish do not belong (Semmens et al. 2004). Recently, Rhyne et al. (2012) examined US import records and compared the volume of species in trade to the nonindigeous species in Florida documented by Schofield et al. (2009). Rhyne et al.'s analysis indicated that species that were rare in trade had been introduced to Florida at a greater frequency than could be explained by chance; thus, trade volume alone did not explain the patterns of introduction.

Semmens et al. noted the risks the coral reef aquarium trade posed to Caribbean coral reefs and encouraged robust measures to prevent ecological impacts from these invaders (Semmens et al. 2004). Unfortunately, the protection measures proposed by Semmens et al. have not occurred. As a result, one of the species identified by Semmens et al., the red lionfish, has caused severe and negative ecological impacts to Caribbean coral reef ecosystems. The remainder of this chapter will focus on the lionfish introduction and its consequences for Caribbean coral reefs.

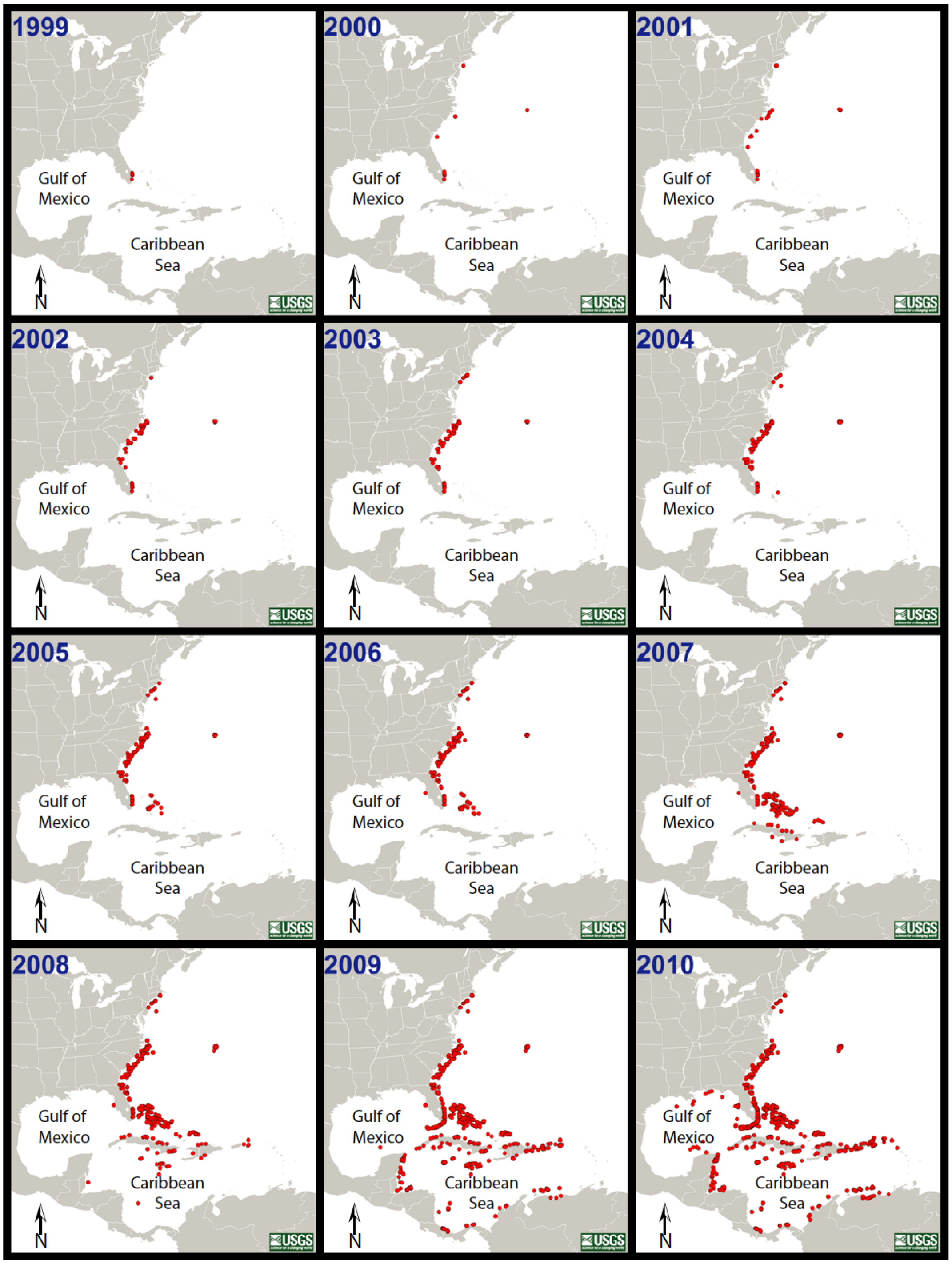
The case of the Indo-Pacific lionfish introduction in the Caribbean Sea and western Atlantic Ocean:

Lionfish, *Pterois* spp., are coral-reef predators in the family Scorpaenidae (Scorpionfishes). Because of their striking markings, ornate fins, venomous spines, and exotic biology, lionfish are popular species in the ornamental aquarium trade. The red lionfish, *Pterois volitans*, was rated as one of the 10 most valuable species of marine fish imported in the United States (Balboa 2003). Trade in lionfish may have depleted populations in certain coral reefs, such as the reefs of the Hawaiian Islands. However, the greatest ecological concern about lionfish is their introduction into locations outside of their native range.

Lionfish are native to the Indo-Pacific and do not naturally occur in the Atlantic Ocean, Gulf of Mexico, or Caribbean Sea (Schultz 1986). Sometime around the early 1990s, several lionfish were released into the marine waters of southern Florida. The best available evidence indicates that the source of the introduction was either the intentional or unintentional release of home-aquarium animals (Whitfield et al. 2002, Ruiz-Carus et al. 2006). Genetic studies indicate that the invasion resulted from the introduction of a small number of fish to the Atlantic coast of Florida (Hamner et al. 2007, Freshwater et al. 2009, Betancur et al. 2011). The release may have happened at Biscayne Bay, Florida when several lionfish were released from an outdoor aquarium during Hurricane Andrew in 1992 (Courtenay 1995). However, there are also reports of lionfish being captured in Dania, Florida from early as 1985 (Morell 2010). As a result, the precise time and place of the introduction remain unknown. Nevertheless, it is widely acknowledged that an aquarium pet, and therefore the aquarium trade, was the source of the introduction (Whitfield et al. 2002, Ruiz-Carus et al. 2006, Morell 2010).

Lionfish are the first ornamental marine fish introduced to the waters of Florida and the southeastern U.S. From the U.S. east coast, lionfish spread to other regions of the western Atlantic (Whitfield et al. 2002), such as Bermuda (Whitfield et al. 2002), the Bahamas (in 2004, Snyder and Burgess 2007, Schofield and Fuller 2009), the Gulf of Mexico (Betancur et al. 2011), and throughout the Caribbean Sea as far south as South America (Betancur et al. 2011) (Figure 26). In actuality, two closely-related species of lionfish have been released into U.S. waters, *P. volitans* and *P. miles* (Hamner et al. 2007). *Pterois miles* appears to be less common and restricted to the U.S. east coast at this time, whereas *P. volitans* has spread southward into the Caribbean Sea (Hamner et al. 2007). Lionfish are now abundant off the coast of the southeastern U.S. (i.e., Georgia, Florida, and the Carolinas) and have even been found as far north as Long Island, New York (Meister et al. 2005). Temperature limits how far north lionfish can disperse. Lionfish die at temperatures below 10°C, so fish spreading too far north will not live for more than a year (Kimball et al. 2004). Unfortunately, there is no temperature limitation on the spread of lionfish into the Caribbean (Kimball et al. 2004). As a result, *P. volitans* now occurs throughout much of the Caribbean and Gulf of Mexico (Betancur et al. 2011). Over the past 10 years, *P. volitans* has rapidly invaded the continental shelf

Figure 26: The expanding distribution of lionfish over time. Red dots highlight locations with the confirmed presense of lionfish. Figure taken from Betancur et al. (2011).

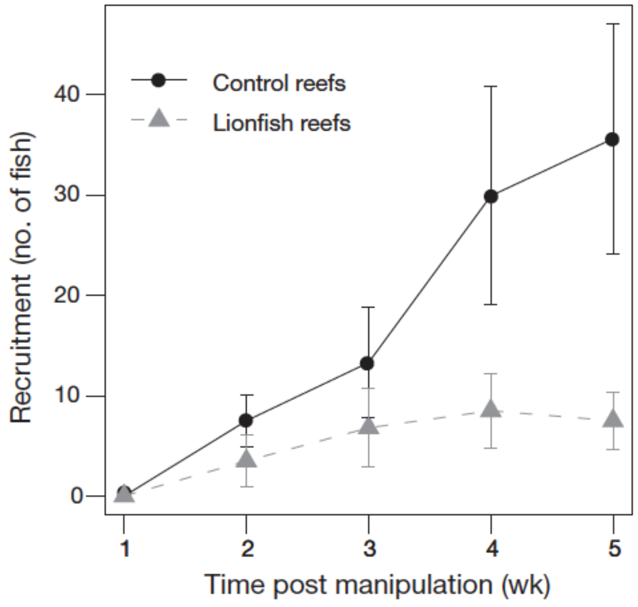


(Whitfield et al. 2002), mangrove and coral reef ecosystems (Barbour et al. 2010, Claydon et al. 2012) throughout the southeastern U.S., Bermuda, the Bahamas, and much of the Caribbean Sea. At depths of 35 to 100 m off of North Carolina, for example, lionfish are now the second most abundant fish species (Whitfeld et al. 2007). In the Bahamas, for example, these lionfish now occur at a larger body size, higher biomass, and greater density in their non-native habitat when compared to similar habitat in their natural range (Darling et al. 2011).

Pterois volitans has spread widely and rapidly – the fastest invasion ever documented by a marine fish (Claydon et al. 2012) – in part because of its high fecundity. A single female lionfish can produce up to 30,000 eggs in floating clusters that develop into planktonic larval fish (Fishelson 1975, Imamura and Yabe 1996). Additionally, lionfish reproduce throughout the year when conditions are suitable (Morris et al. 2011a). As with many other coral reef fishes, it is the larval life stage that allows lionfish to drift around on the currents and spread widely. They spend about a month (averaging 26.2 days) as a larval fish before metamorphosing into their adult form (Ahrenholz and Morris 2010).

The *P. volitans* invasion is cause for concern because lionfish are voracious predators that consume several fish per hour (Côté and Maljković 2010). They eat a very wide range of fish and crustacean species and appear to eat whatever species are locally abundant (Morris and Akins 2009, Muñoz et al. 2011, Valdez-Moreno et al. 2012). This has the potential to decimate native fish populations, to displace native predators like grouper, and to out-compete commercially-important species. Although lionfish appear to be docile and slow-moving animals, these fish are actually highly-efficient predators. Using their large and ornate pectoral fins, lionfish herd prey until it is cornered and then rapidly strike the pretty to capture it (Allen and Eschmeyer 1973, Fishelson 1997). Lionfish strike by quickly expanding their buccal and opercular cavities, which sucks prey into the lionfish’s mouth (Morris and Akins 2009). They have also been shown to confuse and distract prey using a directed jet of water (Albins and Lyons 2012).

Figure 27: The effect of a single lionfish on fish recruitment over time. Control reefs had no lionfish whereas lionfish reefs harbored a single *P. volitans.* Figure taken from Albins and Hixon (2008).



One highly-cited study examined the effects of the introduction of one lionfish to experimental patch reefs in the Bahamas (Albins and Hixon 2008, Figure 27). Albins and Hixon (2008) paired reefs for comparison; half of the study sites had an introduced lionfish and the remainder of study sites acted as controls. The researchers then tracked fish recruitment over many weeks (Albins and Hixon 2008). Over time, the presence of a single lionfish reduced the recruitment of Atlantic coral reef fishes by an average of 79% (Albins and Hixon 2008). This severe effect on fish recruitment raises grave concerns about the consequences of the invasive lionfish introduction. By reducing recruitment (Albins and Hixon 2008) lionfish could impact populations of parrotfish and other herbivores that are crucial for controlling macroalgae on reefs (Williams and Polunin 2001, Mumby et al. 2006). For example, a survey of lionfish feeding behavior in the Bahamas found consumption of surgeonfish and parrotfish, including *Scarus iserti*, *S. viride*, and *Acanthurus bahianus*, albeit at low abundance relative to their total diet (Morris and Akins 2009).

Live Rock as an Invasive-Species Vector:

Live rock is limestone, calcareous structures, and other rock-like material that contains microorganisms, algae, and invertebrates from coral reef ecosystems. It is popular and valuable in the aquarium trade because it enhances the function and appearance of a reef tank (Wabnitz et al. 2003, Bolton and Graham 2006). Most live rock is collected from tropical islands in the Indo-Paicfic and imported by the U.S., Europe, and Japan (Wabnitz et al. 2003).

In the U.S. and many other major importing nations, there are no quarantine rules for live rock importation (Bolton and Graham 2006). Additionally, live rock is commonly cultured upon arrival into the U.S. This creates many opportunities for the microbes and other fauna associated with live rock to be released into non-native environments (Bolton and Graham 2006).

Bolton and Graham (2006) demonstrated that live rock has the capacity to vector upside-down jellyfish, *Cassiopea* spp., and presumably other organisms, from the Indo-Pacific into U.S. waters. Jellyfish are common invasive species in marine environments that can cause significant economic and ecological harm (Graham et al. 2003). As a result, better conservation measures are needed to prevent a harmful live rock-vectored introduction.

Unfortunately, the concerns about ecosystem-level effects resulting from these invasive and voracious predators have already been realized on reefs in the Bahamas. Lesser and Slattery (2011) surveyed reefs from 30−150 m in depth before and after the lionfish introduction. The arrival of lionfish decimated many important fish groups, including herbivores (Lesser and Slattery 2011). As a result, there was a dramatic increase in algal cover from 30% or less before the introduction to over 90% cover after the introduction at depths of 46 and 61 m (Lesser and Slattery 2011). Concomitant declines in coral and sponge cover were also observed (Lesser and Slattery 2011). These changes could not be explained by coral bleaching, overfishing, or other factors that are typically associated with phase shifts on coral reefs (Lesser and Slattery 2011). Similarly, Green et al. (2012) surveyed reefs of the Bahamas from 2004 to 2010. They found an increase in lionfish abundance, to the point where lionfish comprised 40% of the predator biomass, was accompanied by a 65% decline in prey abundance (Green et al. 2012). Furthermore, mathematical models predict that lionfish will have a significant effect on the abundance of many Caribbean fishes unless heavy and continuous mitigation efforts are implemented (Arias-González et al. 2011).The results of these studies indicate that lionfish, and therefore the coral reef wildlife trade, have had significant and profound ecosystem-level consequence on Caribbean coral reefs.

In general, few native Atlantic predators will feed on lionfish. In their natural Indo-Pacific range, their only major cause of predation is large lionfish cannibalizing smaller lionfish (Bernadsky and Goulet 1991). This lack of natural predators likely results from lionfish's venomous fin rays and spines. In humans, a lionfish puncture and envenomation wound can cause serious injury (Vetrano et al. 2003), including edema, intense pain, and necrosis at the site of sting (Balasubashini et al. 2006). There is at least one case where a person was paralyzed by a lionfish sting (Badillo et al. 2009). People have been injured by lionfish both in the wild (e.g., Chan et al. 2010) and in a home aquarium setting (Aldred et al. 1996, Schaper et al. 2009). As a result, lionfish are considered to be a threat to the safety of divers, fishers, and home aquarium owners (Anonymous 2004).

Although lionfish's poisonous spines ward off most predators, there are recent reports of tiger and Nassau groupers feeding on small lionfish (Maljković and Leeuwen 2008). In parts of Honduras, sharks are even being trained to consume lionfish (National Geographic 2010). One study demonstrated that large grouper populations can biologically control lionfish (Mumby et al. 2011). However, native predators like grouper are chronically overfished throughout the Caribbean (Mumby et al. 2011). Populations of grouper would need to be an order of magnitude larger in order to effectively reduce lionfish populations (Mumby et al. 2011). Thus chronic overfishing and the introduction of an aggressive non-native predator cause substantial problems on Caribbean coral reefs (Mumby et al. 2011).

Resource managers have responded to invasive lionfish with various mitigation efforts. In some locations, fishing derbies and tournaments attempt to reduce lionfish populations (Williams 2010). Despite the danger of envenomation, lionfish meat is considered flavorful and many managers are encouraging spear fishers to harvest as many lionfish as possible. However, there may be some danger in this suggestion as the U.S. Food and Drug Administration recently found dangerous ciguatera toxin, the product of a toxic dinoflagellate, in 4 out of 7 lionfish meat samples that were assayed (Cearnal 2012). Furthermore, concerns have been raised that creating a demand for fishing lionfish and other invasive species may result in public support for maintaining, instead of eradicating, the invasive species (Nuñez et al. 2012). Smaller lionfish are desirable for the aquarium trade and some managers encourage shifting collection for the aquarium trade to the Caribbean. This strategy leaves native lionfish populations intact, while reducing non-native lionfish from places where they do not belong.

Because lionfish have very high fecundity, it will be extremely difficult to totally remove them from Caribbean reefs. A pair of lionfish can produce an enormous number of offspring that effectively replace those that are removed by fishing. As a result, Albins and Hixon (2008) argue that management needs to focus on control and mitigation, as opposed to eradication, of lionfish. Additionally, population modeling research indicates that targeting both larval survival and adult fish are necessary for mitigation and that continuously heavy harvest will be needed to control lionfish populations (Arias-González et al. 2011, Barbour et al. 2011, Morris et al. 2011b). For example, Barbour et al. (2011) estimated that an annual harvest level of 35–65% would be required to effectively overfish introduced lionfish populations and that harvest would need to be continued in order to avoid the rebound of lionfish populations. Morris et al. (2011b) estimated that a monthly reduction of 27% of the adult lionfish population was necessary to reduce their populations. Overall, current research suggests that a combined approach of fishing lionfish for food (larger lionfish) and for the aquarium trade (smaller lionfish), biological control through robust native predator populations, and measures targeting larval survival has the *potential* to reduce the lionfish problem, but such measures will require considerable effort.

Chapter 12:
Ecosystem Level Consequences of the Coral Reef Wildlife Trade

The preceding sections of this report examined the ecological consequences of the coral reef wildlife trade on specific taxonomic groups (Chapters 2−8), from the results of harmful and destructive practices (Chapters 9−10), and from the introduction of invasive species (Chapter 11). In each of these cases, trade has been demonstrated to have specific and significant negative effects on coral reef wildlife. However, several larger unanswered questions remain. What are the cumulative impacts of the ornamental coral reef wildlife trade? Are the impacts significant at the level of an entire ecosystem? How does the wildlife trade compare to other threats faced by reefs? Unfortunately, answering such questions has proven very difficult for ecosystems as complex and diverse as coral reefs. The combination of direct harvesting, secondary injury and mortality, destructive practices, and invasive species could cause broad community impacts that could alter reef ecosystem dynamics, but additional data are required to validate this conjecture. In this final chapter, I will briefly review the very limited evidence for such ecosystem level consequences on reefs and place the coral reef wildlife trade within the context of the many other threats that reefs face. No studies directly address the larger ecosystem effects of trade; however, a cursory synthesis of the previous chapters points to the potential for larger ecosystem impacts.

Collection of wildlife for the coral reef ornamental trade reduces biodiversity at nearly every trophic level (Andrews 1990, McManus 2001, Wabnitz et al. 2003). In many locations, collectors acknowledge that fish and invertebrates collected for the coral reef wildlife trade are declining in abundance, with concomitant increases in catch effort required to sustain their livelihoods (Wilhelmsson et al. 2002). The majority of this report was devoted to reviewing population-level declines in marine fish and invertebrates (Chapters 2−9). Additional documented examples are listed in Table 10. Although Chapters 2−9 and Table 10 are not an exhaustive list of all the taxa known to be negatively affected by trade, these examples provide a sense of the size and scope of trade's impacts on coral reef wildlife.

Table 10: Examples of reduced abundance and/or species richness in coral reef wildlife as a result of collection and trade. Information drawn primarily from Wood (2001b) and United Nations Environment Programme and World Conservation Monitoring Centre (2008).

Location	Wildlife group	Source
Australia	Black angelfish, *Chaetodontoplus personifer*, and scribbled angelfish, *C. duboulayi*	QFMA 1999
Brazil	French and grey angelfish	Gasparini et al. 2005
Indonesia (Seribu Islands)	Various ornamental species	Soegiarto and Polunin 1982
Indonesia	Pacific blue tang, powder blue tang, blue girdled fish and emperor angelfish	Boggiatto and Reiser 2004

Kenya	Damselfish: *Pomacanthus maculosus* and *P. chtysurus*	Samoilys 1988
Mexico	*Holacanthus clarionensis*	Almenara-Roland and Ketchum 1994
Philippines	Butterflyfish, triggerfish and angelfish	Albaladejo and Corpuz 1984
	Angelfish (*Centropyge bispinosus*), emperor angelfish (*Pomacanthus imperator*), and blue tang (*Paracanthurus hepatus*)	Rubec 1987
	Coral reef fish	Nañolla et al. 2011
Sri Lanka	Butterflyfish	Lubbock and Polunin 1975
U.S. (Florida)	Angelfish	Noyes 1976
	Various ornamental invertebrates	Rhyne et al. 2009
U.S. (Hawaii)	Various ornamental fishes, including *Acanthurus achilles*, *Chaetodon multicinctus*, *C. quadrimaculatus*, *Centropyge potteri*, *Forcipiger* spp., and *Zanclu cornutus*	Tissot and Hallacher 1999, 2003, Tissot et al. 2004

Removal of a wide diversity of species by fisheries can result in the reduction or elimination of many important functional groups from coral reef ecosystems across the world (Jackson et al. 2001, Bellwood et al. 2004), and this pattern may also apply to ornamental fisheries. Top predators, including sharks, are collected for their jaws and teeth (Grey et al. 2005). These predators exhibit top-down controls that help to maintain ecosystem and community structure. Predatory shrimp in the genus *Hymenocera*, which feed on crown-of-thorns sea stars (*Acanthaster* spp.), are targeted by collectors for the aquarium trade (Calado et al. 2003). Crown-of-thorns sea stars eat stony corals and outbreaks of these corallivorous asteroids have devastating effects on coral reefs throughout the world; removal of natural predators could exacerbate or even cause *Acanthaster* outbreaks. Removal of butterflyfish and other corallivores (Lubbock and Polunin 1975, Albaladejo and Corpuz 1984) may reduce asexual reproduction (fragmentation) in corals and dispersal of coral endosymbionts (e.g., Castro-Sanguino and Sánchez 2012). Hermit crabs, which play an important role in recycling detritus, are also caught in large numbers for trade (Calado et al. 2003). Collectors even take the reef-building species, including corals and live rock, which provide crucial habitat for all other reef-dwelling species (Knowlton 2001, Chapter 8). In some cases, corals are deliberately crushed to collect other species (Gasparini et al. 2005), which destroys the structural basis of the reef itself. Fishing with poisons has devastating effects to all reef-dwelling wildlife (Chapter 10). From the reef-building species to the top predators, collection affects all trophic levels and types of coral reef wildlife.

The collection of two important functional groups – parasite cleaners and herbivores – is worth highlighting in additional detail. Parasite cleaners are a relatively low abundance group of species that serve an important functional role on many coral reefs (reviewed in Waldie et al. 2011). These fish and invertebrates remove damaged skin/scales as well as parasites that reduce the health and fitness of other coral reef species (reviewed in Waldie et al. 2011). Typically a fish in need of cleaning, known as a client fish, will stop at a "cleaning station" where parasite-cleaning species will approach and clean the client (reviewed in Waldie et al. 2011). The client and the cleaners both benefit from this mutualistic interaction: the client has its parasite load reduced and its flesh cleaned and the cleaners benefit from a reliable source of food (reviewed in Waldie et al. 2011). This interesting biology and behavior makes parasite cleaners desirable in the ornamental trade. Cleaner wrasse (*Labroides* spp.), gobies, and cleaning shrimp (e.g., *Lysmata* and *Stenopus* spp.) are all collected in large numbers for use in home aquaria (Edward and Shepherd 1992, Wood and Rajasuriya 1999, Côté 2000, Calado et al. 2003, Monteiro-Neto et al. 2003, Wabnitz et al. 2003). As a result, there is concern that collection will have negative effects on the ecosystem health and parasite loads in reef fish (Edward and Shepherd 1992, Côté 2000, Calado et al. 2003, Monteiro-Neto et al. 2003). Removal of the cleaner fish *Labroides dimidiatus* has been linked to reduced client fish size, abundance, recruitment, and diversity on coral reefs (Waldie et al. 2011). Thus trade could be having similar significant impacts on coral reefs wherever parasite cleaners are taken.

Cortez Rainbow Wrasse, *Thalassoma lucasanum*

The Cortez rainbow wrasse is a sequential (protogynous) hermaphroditic wrasse (family Labridae) that occurs in shallow (approx. 2–25 m) coral habits of the tropical eastern Pacific (Michael 2005, Froese and Pauley 2012). These fish cluster in small groups on coral reefs where they eat zooplankton and benthic invertebrates (Michael 2005, Froese and Pauley 2012). Fish begin their adult lives as females, known as the initial phase or IP, and later change into males fish, known as the terminal phase or TP (Warner 1982). TP males are larger and more colorful than females and as a result, male fish are valued more highly by hobbyists (Warner 1982, McCauley et al. 2008).

Cortez rainbow wrasses are commonly collected in the Costa Rican aquarium trade (McCauley et al. 2008). Although the species does well in captivity (Michael 2005) and is considered a species of least concern by the IUCN, collection has had negative effects on certain Cortez rainbow wrasse populations (McCauley et al. 2008). McCauley et al. (2008) conducted a comparison of high collection pressure and low collection pressure sites in Guanacaste Province of Costa Rica. They found that Cortez rainbow wrasse were considerably more abundant and larger in areas with low collection pressure. On the low collection pressure reefs, IP fish were approx. 6 times more abundant, whereas TP fish were appox. 2.5 times more abundant and almost twice as large when compared to fish on the high collection pressure sites (McCauley et al. 2008). The differences were attributable to several causes, but fishing pressure accounted for the largest portion of the declines (McCauley et al. 2008). McCauley et al. (2008) concluded that "these stocks may be at risk of overexploitation."

Herbivorous fish and invertebrates are essential functional components of coral reef ecosystems. A number of different taxa serve this functional role, notably parrotfish, surgeonfish, and sea urchins. These herbivores feed on various types of algae (e.g., macroalgae, filamentous algae, turf algae, etc.) that compete for space on coral reefs

(Hughes et al. 2003, Bellwood et al. 2004). The removal of herbivores for food fisheries, in the presence of low-level nutrient enrichment (e.g., through nutrient pollution from sewage or agricultural runoff), has been demonstrated to shift coral reef ecosystems from coral-dominated systems to fields of algae (reviewed in Bellwood et al. 2004). Unfortunately, herbivorous animals like surgeonfish (e.g., yellow tang) and grazing invertebrates are also increasingly targeted for the aquarium trade (e.g., Rubec 1987, Rhyne et al. 2009, Stevenson et al. 2011). Herbivores are desirable aquarium species because they control algal growth in a reef tank, similar to the role these species play in nature. The removal of herbivorous animals from coral reefs reduces the redundancy in this important functional role, which *potentially* imperils the local reef ecosystem. The experience of Jamaica provides an illustrative example. In the 1980s, overfishing of parrotfish and surgeonfish for food and a disease outbreak in sea urchins, combined with nutrient pollution and hurricanes, devastated the entire Jamaican reef ecosystem (reviewed in Aronson and Precht 2000). In this case, the effects of severe overfishing of parrotfish and surgeonfish (i.e., the loss of functional redundancy) remained hidden and Jamaican reefs appeared healthy until the urchin die-off (Aronson and Precht 2000). Nearly 30 years later, recovery of Jamaican reefs has been minimal (Aronson and Precht 2000).

Globally, coral reefs are imperiled by many different factors, ranging from climate change and ocean acidification to overfishing, pollution, invasive species introductions, and destructive fishing, among other threats (Wilkinson 2008, Burke et al. 2011). In the context of these severe and growing problems, collection of coral reef wildlife for trade may seem insignificant (Rhyne et al. 2012). However, removal of animals for trade reduces species richness and abundance which removes or reduces the importance of key functional groups as well as intra- and inter-species interactions (see references above). Some authors have suggested that the high fecundity of many coral reef species replaces the losses associated with the trade (e.g., Rhyne et al. 2012); however, the majority of species have not been scientifically evaluated (e.g., the stock status of 64% of coral reef fish imported into the US has not been assessed; Rhyne et al. 2012). Even though problems like climate change and ocean acidification pose larger threats, the science reviewed in this report demonstrates that the negative effects of the coral reef wildlife trade are significant and widespread. Furthermore, the threats posed by nutrient pollution, climate change, and/or ocean acidification and collection for trade are not mutually exclusive. In fact, in many cases these multiple threats compound one another. The loss of coral reef biological and functional diversity from trade may reduce the ability of reefs to recover from disturbance while maintaining ecosystem functions and services (Hughes et al. 2003, Bellwood et al. 2004, Mumby and Steneck 2008, Hoegh-Guldberg and Bruno 2010). As the loss of Jamaican coral reefs illustrated, the loss of redundancy within a functional group can lead to the loss of a functional group and critical ecosystem functions all together.

Reefs are rapidly shifting to alternative states that are less able to support the services – including tourism, recreation, fishing, storm protection, educational and cultural value, and the ornamental trade – that millions of people depend upon for their livelihoods, recreation, and sustenance (Bellwood et al. 2004). Improved conservation and management of trade has the *potential* to both increase the resistence of coral reefs to other threats and to provide a sustainable livelihood for collectors, middlemen, exporters, importers, and retailers throughout the world. These potentially beneficial results will be hard fought and success is far from guaranteed. Coral reefs are perhaps the most complex ecosystem on the planet, and these ecosystems are threatened by myriad problems. As a result, management measures need to be tailored to each reef ecosystem, local people, and problems unique to that location. However, improving the sustainability of trade is certainly an achievable goal (e.g., Chapter 2) that could improve resilience to buy time to deal with more challenging problems like climate change and ocean acidification, thereby ensuring more robust fisheries and ecosystem services for local communities.

References:

Abe N (1937) Post-larval development of *Fungia actiniformis*. Palao Tropical Biological Station Studies 1: 73–79.

Ackerman JL, Bellwood DR (2002) Comparative efficiency of clove oil and rotenone for sampling tropical reef fish assemblages. Journal of Fish Biology 60: 893–901.

Ahrenholz DW, Morris JA (2010) Larval duration of the lionfish, *Pterois volitans* along the Bahamian Archipelago. Environmental Biology of Fishes 88: 305–309.

Albaladejo VD, Corpuz VT (I984) Marine aquarium fish research and resource management. Bureau of Fisheries and Aquatic resources, Republic of the Philippines.

Albins MA, Hixon MA (2008) Invasive Indo-Pacific lionfish *Pterois volitans* reduce recruitment of Atlantic coral-reef fishes. Marine Ecology Progress Series 367: 233–238.

Albins MA, Lyons PJ (2012) Invasive red lionfish *Pterois volitans* blow directed jets of water at prey fish. Marine Ecology Progress Series 448: 1–5.

Aldred B, Erickson T, Lipscomb J (1996) Lionfish envenomations in an urban wilderness. Wilderness & Environmental Medicine 7: 291–296.

Allen GR (1975) Anemonefishes- their classification and biology. TFH, Neptune City.

Allen GR (1980) The anemonefishes of the world. Species, care and breeding. Aquarium Systems, Mentor, OH.

Allen GR (2000) Threatened fishes of the world: *Pterapogon kauderni* Koumans, 1933 (Apogonidae). Environmental Biology of Fishes 57: 142–142.

Allen GR, Eschmeyer WN (1973) Turkeyfishes at Eniwetok. Pacific Discovery 26: 3–11.

Allen GR, Steene RC (1995) Notes on the ecology and behaviour of the Indonesian cardinalfish (Apogonidae) *Pterapogon kauderni* Koumans. Revue fr. Aquariol. 22: 7–9.

Allen GR, Werner TB (2002) Coral reef fish assessment in the 'coral triangle' of southeastern Asia. Environmental Biology of Fishes 65: 209–214.

Allen GR, Donaldson TJ (2007) *Pterapogon kauderni*. In: IUCN 2010. IUCN Red List of Threatened Species. Version 2010.4 http://www.iucnredlist.org/search/details.php/63572/summ

Almany GR, Beruman ML, Thorrold SR, Planes S, Jones GP (2007) Local replenishment of coral reef fish populations in a Marine Reserve. Science 316: 742–744.

Almenara-Roland S, Ketchum JT (1994) Forgotten islands of the Mexican Pacific. OFI Journal 9: 12–14.

Alves RRN, Rosa IL (2006) From cnidarians to mammals: The use of animals as remedies in fishing communities in NE Brazil. Journal of Ethnopharmacology 107: 259–276.

Amend DF, Croy TR, Goven BA, Johnson KA, McCarthy DH (1982) Transportation of fish in closed systems: Methods to control ammonia, carbon dioxide, pH, and bacterial growth. Transactions of the American Fisheries Society 111: 603–611.

Andrews C (1990) The ornamental fish trade and fish conservation. Journal of Fish Biology 37: 53–59.

Anonymous (2004) Venomous invasive lionfish: a danger to divers and deepwater fishermen. Marine Pollution Bulletin 49: 7.

Anto J, Turingan RG (2010) Relating the ontogeny of functional morphology and prey selection with larval mortality in Amphiprion frenatus. Journal of Morphology 271: 682–696.

Arias-González JE, González-Gándara C, Cabrer JL, Christensen V (2011) Predicted impact of the invasive lionfish *Pterois volitans* on the food web of a Caribbean coral reef. Environmental Research 111: 917–925.

Aronson RB, Precht WF (2000) Herbivory and algal dynamics on the coral reef at Discovery Bay, Jamaica. Limnology and Oceanography 45: 251–255.

Arvedlund M, Takemura A (2005) Long-term observation in situ of the anemonefish *Amphiprion clarkii* (Bennett) in association with a soft coral. Coral Reefs 24: 698–698.

Arvedlund M, McCormick MI, Fautin DG, Bildsoe M (1999) Host recognition and possible imprinting in the anemonefish *Amphiprion melanopus* (Pisces : Pomacentridae). Marine Ecology Progress Series 188: 207–218.

Arvedlund M, Larsen K, Winsor H (2000a) The embryonic development of the olfactory system in *Amphiprion melanopus* (Perciformes : Pomacentridae) related to the host imprinting hypothesis. Journal of the Marine Biological Association of the United Kingdom 80: 1103–1109.

Arvedlund M, Bundgaard I, Nielsen LE (2000b) Host imprinting in anemonefishes (Pisces : Pomacentridae): Does it dictate spawning site preferences? Environmental Biology of Fishes 58: 203–213.

Arvedlund M, Iwao K, Brolund TM, Takemura A (2006) Juvenile *Thalassoma amblycephalum* Bleeker (Labridae, Teleostei) dwelling among the tentacles of sea anemones: a cleanerfish with an unusual client? Journal of Experimental Marine Biology and Ecology 329: 161–173.

Ashworth JS, Ormond RFG, Sturrock HT (2004) Effects of reef-top gathering and fishing on invertebrate abundance across take and no-take zones. Journal of Experimental Marine Biology and Ecology 303: 221–242.

Aubert A, Lazareth CE, Cabioch G, Boucher H, Yamada T, Iryu Y, Farman R (2009) The tropical giant clam *Hippopus hippopus* shell, a new archive of environmental conditions as revealed by sclerochronological and delta O-18 profiles. Coral Reefs 28: 989–998.

Avella MA, Olivotto I, Gioacchini G, Maradonna F, Carnevali O (2007) The role of fatty acids enrichments in the larviculture of false percula clownfish *Amphiprion ocellaris*. Aquaculture 273: 87–95.

Avella MA, Olivotto I, Silvi S, Place AR, Carnevali O (2010) Effect of dietary probiotics on clownfish: a molecular approach to define how lactic acid bacteria modulate development in a marine fish. American Journal of Physiology – Regulatory Integrative and Comparative Physiology 298: R359–R371.

Badillo RB, Banner W, Morris J, Schaeffer SE (2009) A case study of lionfish sting induced paralysis. Clinical Toxicology 47: 732.

Baillie BK, Belda-Baillie CA, Maruyama T (2000) Conspecificity and Indo-Pacific distribution of *Symbiodinium* genotypes (Dinophyceae) from giant clams. Journal of Phycology 36: 1153–1161.

Bagnara JT, Fernandez PJ, Fujii R (2007) On the blue coloration of vertebrates. Pigment Cell Research 20: 14–26.

Balasubashini MS, Karthigayan S, Somasundaram ST, Balasubramanian T, Viswanathan P, Menon VP (2006) In vivo and in vitro characterization of the biochemical and pathological changes induced by lionfish (*Pterios volitans*) venom in mice. Toxicology Mechanisms and Methods 16: 525–531.

Balboa CM (2003) The consumption of marine ornamental fish in the United States: A description from U.S. import data. In: Cato JC, Brown CL (eds.) Marine Ornamental Species: Collection, Culture, and Conservation. Iowa State Press Ames, Iowa, p. 65–76.

Ballantyne B (1987) Toxicology of cyanides. In: Ballantyne B, Marrs TC (eds.) Clinical and experimental toxicology of cyanides. Wright, Bristol, p. 41–126.

Baquero J (1995) The stressful journey of ornamental marine fish. Sea Wind 9: 19–21.

Barber CV, Pratt VR (1997a) Sullied seas: Strategies for combating cyanide fishing in Southeast Asia and beyond. Report prepared by World Resources Institute and International Marinelife Alliance, 73 p.

Barber CV, Pratt VR (1997b) Policy reform and community-based programmes to combat cyanide fishing in Philippines. SPC Life Reef Fish Information Bulletin 3: 26–35.

Barber CV, Pratt VR (1998) Poison for profits: cyanide fishing in the Indo-Pacific. Environment 40: 5–9, 28–34.

Barbour AB, Montgomery ML, Adamson AA, Diaz-Ferguson E, Silliman BR (2010) Mangrove use by the invasive lionfish *Pterois volitans*. Marine Ecology Progress Series 401: 291–294.

Barbour AB, Allen MS, Frazer TK, Sherman KD (2011) Evaluating the potential efficacy of invasive lionfish (*Pterois volitans*) removals. PLoS ONE 6: e19666.

Barlow GW (1974) Contrasts in social behaviour between Central American cichlid fishes and coral-reef surgeon fishes. American Zoologist 14: 9–34.

Barnes DJ (1985) The effects of photosynthetic and respiratory inhibitors upon calcification in the staghorn coral *Acropora formosa*. Proceedings of the 5th International Coral Reef Congress 6: 161–166.

Barnes DJ, Chalker BE (1990) Calcification and photosynthesis in reef-building corals and algae. In: Dubinski Z (ed.) Ecosystems of the world: 25 coral reefs. Elsevier Science, Amsterdam, p. 109–131.

Baum JK, Vincent ACJ (2005) Magnitude and inferred impacts of the seahorse trade in Latin America. Environmental Conservation 32: 305–319.

Baum JK, Meeuwig JJ, Vincent ACJ (2003) Bycatch of lined seahorses (*Hippocampus erectus*) in a Gulf of Mexico shrimp trawl fishery. Fishery Bulletin 101: 721–731.

Bax N, Carlton JT, Mathews-Amos A, Haedrich RL, Hogwarth FG, Purcell JE, Rieser A, Gray A (2001) The control of biological invasions in the world's oceans. Conservation Biology 15: 1234–1246.

Bay LK, Buechler K, Gagliano M, Caley MJ (2006) Intraspecific variation in the pelagic larval duration of tropical reef fishes. Journal of Fish Biology 68: 1206–1214.

Beckvar N (1981) Cultivation, spawning, and growth of the giant clams *Tridacna gigas*, *T. derasa*, and *T. squamosa* in Palau, Caroline Islands. Aquaculture 24: 21–30.

Bell JD, Lane I, Gervis M, Soule S, Tafea H (1997) Village-based farming of the giant clam, *Tridacna gigas* (L), for the aquarium market: Initial trials in Solomon Islands. Aquaculture Research 28: 121–128.

Bellwood DR (1981) Cyanide . . . An investigation into the long term histological effects of sodium cyanide doses upon the gastro-intestinal tract of *Dascyllus trimaculatus*. Part One. Freshwater and Marine Aquarium 4, 31–35, 75–76.

Bellwood DR, HughesTP, Folke C, Nyström M (2004) Confronting the coral reef crisis. Nature 429: 827–833.

Bentley N (1998) An overview of the exploitation, trade and management of corals in Indonesia. Traffic Bulletin 17: 67–78.

Benzie JAH, Williams ST (1992a) Genetic-structure of giant clam (*Tridacna maxima*) populations from reefs in the western Coral Sea. Coral Reefs 11: 135–141.

Benzie JAH, Williams ST (1992b) No genetic differentiation of giant clam (*Tridacna gigas*) populations in the Great-Barrier-Reef, Australia. Marine Biology 113: 373–377.

Berka R (1986) The transport of live fish — A review. European Inland Fisheries Advisory Commission Technical Paper 48. Rome: FAO.

Bernadsky G, Goulet D (1991) A natural predator of the lionfish, *Pterois miles*. Copeia 1991: 230–231.

Bernardi G, Vagelli A (2004) Population structure in Banggai cardinalfish, *Pterapogon kauderni*, a coral reef species lacking a pelagic larval phase. Marine Biology 4: 803–810.

Betancur RR, Hines A, Acero AP, Orti G, Wilbur AE, Freshwater DW (2011) Reconstructing the lionfish invasion: insights into Greater Caribbean biogeography. Journal of Biogeography 38: 1281–1293.

bin Othman AS, Goh GHS, Todd PA (2010) The distribution and status of giant clams (Tridacnidae) – A short review. The Raffles Bulletin of Zoology 58: 103–111.

Birkeland C (2001) Can ecosystem management of coral reefs be achieved? In: Best B, Borunbusch A (eds.) Global trade and consumer choices: coral reefs in crisis. Papers presented at a symposium held at the 2001 annual meeting of the American Association for the Advancement of Science, p. 15–18.

Blake RW (1976) On seahorse locomotion. Journal of the Marine Biological Association of the United Kingdom 56: 939–949.

Boggiatto K, Reiser A (2004) The marine aquarium trade in the Western Hemisphere and the Indo-Pacific region. Hawaii Audubon Society. 49 pp.

Bolton TF, Graham WM (2006) Jellyfish on the rocks: bioinvasion threat of the international trade in aquarium live rock. Biological Invasions 8: 651–653.

Bonham K (1965) Growth rate of giant clam *Tridacna gigas* at Bikini Atoll as revealed by radioautography. Science 149: 300–302.

Bourne (1893) On the post-embryonic development of *Fungia fungites*. Scientific Transactions of the Royal Dublin Society 5: 205.

Bower CE, Turner DT (1982) Ammonia removal by clinoptilolite in the transport of ornamental freshwater fishes. Progressive Fish Culturist 44: 19–23.

Breen S, Owens R (2002) The status of the ornamental trade in Australia: Collectors, collection areas and management strategies for coral resources in Queensland. In: Bruckner AW (ed.) Proceedings of the International Workshop on the Trade in Stony Corals: development of sustainable management guidelines. NOAA Technical Memorandum NMFS-OPR-23, Silver Spring, MD, p. 73–74.

Brown RS, Carlson TJ, Welch AE, Stephenson JR, Abernethy CS, McKinstry CA, Theriault MH (2007) Assessment of barotrauma resulting from rapid decompression of depth-accliminated juvenile Chinook salmon bearing radio telemetry transmitters. Report PNNL-16790, Pacific Northwest National Laboratory, Richland, Washington.

Bruckner AW (2000) New threat to coral reefs: trade in coral organisms. Issues in Science and Technology, Fall 2000.

Bruckner AW (2002) Proceedings of the International Workshop on the Trade in Stony Corals: development of sustainable management guidelines. NOAA Technical Memorandum NMFS-OPR-23, Silver Spring, MD, 152 pp.

Bruckner AW (2005) The importance of the marine ornamental reef fish trade in the wider Caribbean. Revista de Biología Tropical / International Journal of Tropical Biology and Conservation 53: 127–138.

Bruckner AW, Borneman EH (2006) Developing a sustainable harvest regime for Indonesia's stony coral fishery with application to other coral exporting countries. Proceedings of 10th International Coral Reef Symposium: 1692–1697.

Bruckner AW, Roberts G (2008) Proceedings of the international cyanide detection testing workshop. NOAA Technical Memorandum NMFS-OPR-40, Silver Spring, MD 164 pp.

Bruckner AW, Field JD, Daves N (2005). The Proceedings of the International Workshop on CITES Implementation for Seahorse Conservation and Trade. NOAA Technical Memorandum NMFS-OPR-27.

Bryant D, Burke L, McManus J, Spalding M (1998) Reefs at risk: a map-based indicator of threats to the world's coral reefs. World Resources Institute, Washington, D.C.

Buchel C, Garab G (1995) Evidence for the operation of a cyanidesensitive oxidase in chlororespiration in the thylakoids of the chlorophyll c containing alga *Pleurichloris meiringensis* (Xanthophyceae). Planta 197: 69–75.

Buck BH, Rosenthal H, Saint-Paul U (2002) Effect of increased irradiance and thermal stress on the symbiosis of *Symbiodinium microadriaticum* and *Tridacna gigas*. Aquatic Living Resources 15: 107–117.

Burke L, Reytar K, Spalding M, Perry A (2011) Reefs at risk revisited. World Resources Institute, Washington, DC. 130 p.

Bushnell ME, Claisse JT, Laidley CW (2010) Lunar and seasonal patterns in fecundity of an indeterminate, multiple-spawning surgeonfish, the yellow tang *Zebrasoma flavescens.* Journal of Fish Biology 76: 1343–1361.

Buston PM (2002) Group structure of the clown anemonefish *Amphiprion percula*. Ph.D. Dissertation, Cornell University, Ithaca, NY.

Buston PM (2003a) Mortality is associated with social rank in the clown anemonefish (*Amphiprion percula*). Marine Biology 143: 811–815.

Buston P (2003b) Forcible eviction and prevention of recruitment in the clown anemonefish. Behavioral Ecology 14: 576–582.

Buston PM (2004a) Does the presence of non-breeders enhance the fitness of breeders? An experimental analysis in the clown anemonefish *Amphiprion percula*. Behavioural Ecology and Sociobiology 57: 23–31.

Buston PM (2004b) Territory inheritance in clownfish. Proceedings of the Royal Society of London Series B – Biological Sciences 271: S252–S254.

Buston PM, Cant MA (2006) A new perspective on size hierarchies in nature: patterns, causes, and consequences. Oecologia 149: 362–372.

Buston PM, Garcia MB (2007). An extraordinary life span estimate for the clown anemonefish *Amphiprion percula*. Journal of Fish Biology 70: 1710–1719.

Buston PM, Bogdanowicz SM, Wong A, Harrison RG (2007) Are clownfish groups composed of close relatives? An analysis of microsatellite DNA variation in *Amphiprion percula*. Molecular Ecology 16: 3671–3678.

Cabaitan PC, Gomez ED, Alino PM (2008) Effects of coral transplantation and giant clam restocking on the structure of fish communities on degraded patch reefs. Journal of Experimental Marine Biology and Ecology 357: 85–98.

Calado R, Lin J, Rhyne AL, Araújo R, Narciso L (2003) Marine ornamental decapods – popular, pricey, and poorly studied. Journal of Crustacean Biology 23: 963–973.

Carmichael GJ, Tomasso JR, Simco BA, Davis KB (1984a) Confinement and water quality induced stress in largemouth bass. Transactions of the American Fisheries Society 113: 767–777.

Carmichael GJ, Tomasso JR, Simco BA, Davis KB (1984b) Characterization and alleviation of stress in largemouth bass. Transactions of the American Fisheries Society 113: 778–785.

Casillas E, Miller SE, Smith LS, D'Aoust BG (1975) Changes in hemostatic parameters in fish following rapid decompression. Undersea Biomedical Research 2: 267–276.

Capitini CA, Tissot BN, Carroll MS, Walsh WJ, Peck S (2004) Competing perspectives in resource protection: the case of marine protected areas in West Hawaii. Society and Natural Resources 17: 763–778.

Carlos AA; Baillie BK; Maruyama T (2000) Diversity of dinoflagellate symbionts (zooxanthellae) in a host individual. Marine Ecology Progress Series 195: 93–100.

Castro ALD, Diniz AD, Martins IZ, Vendel AL, de Oliveira TPR, Rosa IMD (2008) Assessing diet composition of seahorses in the wild using a non destructive method: *Hippocampus reidi* (Teleostei: Syngnathidae) as a study-case. Neotropical Ichthyology 6: 637–644.

Castro-Sanguino C, Sánchez JA (2012) Dispersal of *Symbiodinium* by the stoplight parrotfish *Sparisoma viride*. Biology Letters 8: 282–286.

Cearnal L (2012) Red lionfish and ciguatoxin: menace spreading through western hemisphere. Annals of Emergency Medicine 60: 21A–22A.

Cervino JM, Hayes RL, Honovich M, Goreau TJ, Jones S, Rubec PJ (2003) Changes in zooxanthellae density, morphology, and mitotic index in hermatypic corals and anemones exposed to cyanide. Marine Pollution Bulletin 46: 573–586.

Cesar H, vanBeukering P, Pintz S, Dierking S (2002) Economic valuation of the coral reefs of Hawaii. Final Report. Hawaii Coral Reef Initiative Research Program. 117 p.

Chadwick NE, Arvedlund M (2005) Abundance of giant sea anemones and patterns of association with anemonefish in the northern Red Sea. Journal of the Marine Biological Association of the United Kingdom 85: 1287–1292.

Chalker BE, Taylor DL (1975) Light-enhanced calcification, and the role of oxidative phosphorylation in calcification of the coral *Acropora cervicorni*s. Proceedings of the Royal Society of London Series B 190: 323–331.

Chan HY, Chan YC, Tse ML, Lau FL (2010) Venomous fish sting cases reported to Hong Kong Poison Information Centre: a three-year retrospective study on epidemiology and management. Hong Kong Journal of Emergency Medicine 17: 40–44.

Chan TC, Sadovy Y (1998) Profile of the marine aquarium fish trade in Hong Kong. Aquarium Sciences and Conservation 2: 197–213.

Chang AL, Grossman JD, Spezio TS, Weiskel HW, Blum JC, Burt JW, Muir AA, Piovia-Scott J, Veblen KE, Grosholz ED (2009) Tackling aquatic invasions: risks and opportunities for the aquarium fish industry. Biological Invasions 11: 773–785.

Chiappone M, Swanson DW, Miller SL (2001) *Condylactis gigantea* – A giant comes under pressure from the aquarium trade in Florida. Reef Encounter 30: 29–31.

Choat JH, Axe LM, (1996) Growth and longevity in acanthurid fishes an analysis of otolith increments. Marine Ecology Progress Series 134: 15–26.

Chow PS, Chen TW, Teo LH (1994) Physiological responses of the common clownfish, *Amphiprion ocellaris* (Cuvier), to factors related to packaging and long-distance transport by air. Aquaculture 127: 347–361.

Christie MR, Tissot BN, Albins MA, Beets JP, Jia YL, Ortiz DM, Thompson SE, Hixon MA (2010) Larval connectivity in an effective network of marine protected areas. PLoS ONE 5: e15715

Cinner J, Marnane MJ, McClanahan TR, Almany GR (2006) Periodic closures as adaptive coral reef management in the Indo-Pacific. Ecology and Society 11: Article #31.

Claisse JT, Kienzle M, Bushnell ME, Shafer DJ, Parrish JD (2009a) Habitat- and sex-specific life history patterns of yellow tang *Zebrasoma flavescens* in Hawaii, USA. Marine Ecology Progress Series 389: 245–255.

Claisse JT, Mctee SA, Parrish JD (2009b) Effects of age, size, and density on natural survival for an important coral reef fishery species, yellow tang, *Zebrasoma flavescens* . Coral Reefs 28: 95–105.

Claydon JAB, Calosso MC, Traiger SB (2012) Progression of invasive lionfish in seagrass, mangrove and reef habitats. Marine Ecology Progress Series 448: 119–129.

Cleveland A, Verde EA, Lee RW (2011) Nutritional exchange in a tropical tripartite symbiosis: direct evidence for the transfer of nutrients from anemonefish to host anemone and zooxanthellae. Marine Biology 158: 589–602.

Cobb CS, Levy MG, Noga EJ (1998) Acquired immunity to amyloodiniosis is associated with an antibody response. Diseases of Aquatic Organisms 34: 125–133.

Coffroth MA, Santos SR (2005) Genetic diversity of symbiotic dinoflagellates in the genus *Symbiodinium*. Protist 156: 19–34.

Cole B, Tamura CS, Bailey R, Brown C, Ako H (2001) Shipping practices in the ornamental fish industry. In: Paust BC, Rice AA (eds.) Marketing and shipping live aquatic products: Proceedings of the Second International Conference and Exhibition, November 1999, Seattle, Washington. Fairbanks: University of Alaska Sea Grant, AK-SG-01-03, p. 73–86.

Conservation and Community Investment Forum (CCIF) (2001) Analysis of destructive reef fishing practices in the Indo-Pacific. Conservation and Community Investment Forum, San Francisco CA. 38 p.

Convention on International Trade in Endangered Species of Wilde Fauna and Flora CITES (2004a) *Tridacna crocea.* AC22 Doc.10.2 Annex 8c. URL: http://www.cites.org/eng/com/AC/22/E22-10-2-A8c.pdf

Convention on International Trade in Endangered Species of Wilde Fauna and Flora CITES (2004b) *Tridacna derasa.* AC22 Doc.10.2 Annex 8d. URL: http://www.cites.org/eng/com/AC/22/E22-10-2-A8d.pdf

Convention on International Trade in Endangered Species of Wilde Fauna and Flora CITES (2004c) *Tridacna gigas.* AC22 Doc.10.2 Annex 8e. URL: http://www.cites.org/eng/com/AC/22/E22-10-2-A8e.pdf

Convention on International Trade in Endangered Species of Wilde Fauna and Flora CITES (2004d) *Tridacna squamosa.* AC22 Doc.10.2 Annex 8g. URL: http://www.cites.org/eng/com/AC/22/E22-10-2-A8g.pdf

Convention on International Trade in Endangered Species of Wilde Fauna and Flora (CITES) Fourteenth meeting of the Conference of the Parties. (2007) Consideration of Proposals for Amendment of Appendices I and II. Conference of Parties 14, Proposal 21. Geneva.

Cooper WJ, Smith LL, Westneat MW (2009) Exploring the radiation of a diverse reef fish family: Phylogenetics of the damselfishes (Pomacentridae), with new classifications based on molecular analyses of all genera. Molecular Phylogenetics and Evolution 52: 1–16.

Copland JW, Lucas JS (1988) Giant clams in Asia and the Pacific. Australian Center for International Agricultural Research, Canberra.

Côté IM (2000) Evolution and ecology of cleaning symbioses in the sea. In: Gibson RN, Barnes M (eds.) Oceanography and Marine Biology: An Annual Review. Taylor and Francis, London, p. 311–355.

Côté IM, Maljković A (2010) Predation rates of Indo-Pacific lionfish on Bahamian coral reefs. Marine Ecology Progress Series 404: 219–225.

Coughlin DJ (1993) Prey location by clownfish (*Amphiprion perideraion*) larvae feeding on rotifers (*Branchionus plicatilis*). Journal of Plankton Research 15: 117–123.

Coughlin DJ (1994) Suction prey culture by clownfish larvae (*Amphiprion perideraion*). Copeia 1994: 242–246.

Courtenay WR (1995) Marine fish introductions in south-eastern Florida. American Fisheries Society Introduced Fish News 14: 2–3.

Courtenay WR, Robins CR (1973) Exotic aquatic organisms in Florida with emphasis on fishes: a review and recommendations. Transactions of the American Fisheries Society 102: 1–12.

Courtenay WR, Stauffer JR (1990) The introduced fish problem and the aquarium fish industry. Journal of the World Aquaculture Society 21: 145–159.

Craig V, Allan C, Brittain A, Richman E, Lyet A (In press) Review of trade in coral, coral Products and reef associated species to the United States. TRAFFIC North America and World Wildlife Fund, Washington DC. In press.

Cumming RL, Alford RA (1994) Population dynamics of *Turbonilla* sp. (Pyramidellidae, Opisthobranchia), an ectoparasite of giant clams in mariculture. Journal of Experimental Marine Biology and Ecology 183: 91–111.

Curtis JMR, Vincent ACJ (2005) Distribution of sympatric seahorse species along a gradient of habitat complexity in a seagrass-dominated community. Marine Ecology Progress Series 291: 81–91.

Curtis JMR, Vincent ACJ (2006) Life history of an unusual marine fish: survival, growth and movement patterns of *Hippocampus guttulatus* Cuvier 1829. Journal of Fish Biology 68: 707–733.

Curtis JMR, Vincent ACJ (2008) Use of population viability analysis to evaluate CITES trade-management options for threatened marine fishes. Conservation Biology 22: 1225–1232.

D'Antonio C, Meyerson LA, Denslkow J (2001) Exotic species and conservation. In: Soulé ME, Orians GH (eds.) Conservation biology: research priorities for the next decade. Island Press, Washington, DC, p. 59–80.

Darling EM, Green SJ, O'Leary JK, Côté IM (2011) Indo-Pacific lionfish are larger and more abundant on invaded reefs: a comparison of Kenyan and Bahamian lionfish populations. Biological Invasions 13: 2045–2051.

Dawes J (2003) Wild-caught marine species and the ornamental aquatic industry. In: Cato JC, Brown CL (eds.) Marine Ornamental Species: Collection, Culture, and Conservation. Iowa State Press Ames, Iowa, p. 363–370.

de Beaufort LF (1940) The fishes of the Indo-Australian Archipelago. E. J. Brill, Leiden: 326–472.

Debenham P (2008) Background paper on cyanide detection tests for live fish. In: Bruckner AW, Roberts G (eds.). Proceedings of the International Cyanide Detection Testing Workshop. NOAA Technical Memorandum NMFS-OPR-40, Silver Spring, MD, p. 95–116.

DeBoer TS, Subia MD, Ambariyanto, Erdmann MV, Kovitvongsa K, Barber PH (2008) Phylogeography and limited genetic connectivity in the endangered boring giant clam across the coral triangle. Conservation Biology 22: 1255–1266.

del Norte AGC, Nanola CL, McManus JW, Reyes RB, Campos WL, Cabansag JBP (1989) Overfishing on a Philippine coral reef: a glimpse into the future. In: Magoon OT, Converse H, Miner D, Tobin LT, Clark D (eds.) Coastal Zone '89. Proceedings of the Sixth Symposium on Coastal and Ocean Management 4: 3089–3097.

Dempster RP, Donaldson MS (1974) Cyanide – tranquilizer or poison? Aquarium Digest International Tetra 2: 21–22.

Dhayanithi NB, Kumar TTA, Kathiresan K (2010) Effect of neem extract against the bacteria isolated from marine fish. Journal of Environmental Biology 31: 409–412.

Dierking J, (2007) Effects of the introduced predatory fish *Cephalopholis argus* on native reef fish populations in Hawaii. PhD Dissertation, Zoology Department, University of Hawaii at Manoa, Honolulu. 115 p.

Division of Aquatic Resources (DAR) (2010) A report on the findings and recommendations of the West Hawai'i Fisheries Management Area. Report to the Twenty-Fifth Legislature 2010 regular session. 53 p.

Dixson DL, Jones GP, Munday PL, Planes S, Pratchett MS, Srinivasan M, Syms C, Thorrold SR (2008) Coral reef fish smell leaves to find island homes. Proceedings of the Royal Society B – Biological Sciences 275: 2831–2839.

Dixson DL, Munday PL, Jones GP (2010) Ocean acidification disrupts the innate ability of fish to detect predator olfactory cues. Ecology Letters 13: 68–75.

Downs CA, Fauth JE, Downs VD, Ostrander GK (2010) In vitro cell-toxicity screening as an alternative animal model for coral toxicology: effects of heat stress, sulfide, rotenone, cyanide, and cuprous oxide on cell viability and mitochondrial function. Ecotoxicology 19: 171–184.

Drew J, Allen GR, Kaufman L, Barber PH (2008) Endemism and regional color and genetic differences in five putatively cosmopolitan reef fishes. Conservation Biology 22: 965–975.

Dumas P, Jimenez H, Leopold M, Petro G, Jimmy R (2010) Effectiveness of village-based marine reserves on reef invertebrates in Emau, Vanuatu. Environmental Conservation 37: 364–372.

Dunn DF (1981) The clownfish sea anemones: Stichodactylidae (Coelenterata: Actiniaria) and other sea anemones symbiotic with pomacentrid fishes. Transactions of the American Philosophical Society 71: 1–115.

Duodoroff P (1980) A critical review of recent literature on the toxicity of cyanides on fish. American Petroleum Institute, Washington, DC, 71 p.

Eble JA, Toonen RJ, Bowen BW (2009) Endemism and dispersal: comparative phylogeography of three surgeonfishes across the Hawaiian Archipelago. Marine Biology 156: 689–698.

Eda H, Takita T, Uno Y (1994a) Larval and juvenile development of two dragonets, *Repomucenus richardsonii* and *R. valenciennei*, reared in a laboratory. Japanese Journal of Ichthyology 41: 149–158.

Eda H, Fujiwara Y, Takita T (1994b) Embryonic, larval and juvenile development in laboratory-reared dragonets, *Repomucenus beniteguri*. Japanese Journal of Ichthyology 40: 465–473.

Eda H, Fujiwara T, Kuno Y, Takita T (1997) Larval and juvenile development of the dragonet, *Paradiplogrammus enneactis*, reared in a laboratory. Ichthyological Research 44: 77–81.

Edwards AJ, Shepherd AD (1992) Environmental implications of aquarium-fish collection in the Maldives, with proposals for regulation. Environmental Conservation 19: 61–72.

Egekeze JO, Oehme FW (1980) Cyanides and their toxicity: a literature review. Vet Q 2:104–114.

Eisler R (1991) Cyanide hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wildlife Service.

Elfwing T, Plantman P, Tedengren M, Wijnbladh E (2001) Responses to temperature, heavy metal and sediment stress by the giant clam Tridacna squamosa. Marine and Freshwater Behaviour and Physiology 34: 239–248.

Elliott JK, Mariscal RN (1997a) Acclimation or innate protection of anemone fishes from sea anemones? Copeia 1997: 284–289.

Elliott JK, Mariscal RN (1997b) Ontogenetic and interspecific variation in the protection of anemonefishes from sea anemones. Journal of Experimental Marine Biology and Ecology 208: 57–72.

Elliott JK, Lougheed SC, Bateman B, McPhee LK, Boag PT (1999)Molecular phylogenetic evidence for the evolution of specialization in anemonefishes. Proceedings of the Royal Society of London Series B – Biological Sciences 266: 677–685.

Environmental Justice Foundation (2003) Squandering the seas: how shrimp trawling is threatening ecological integrity and food security around the world. London: EJF.

Erdmann MV (1999) Clove-oil: an ‘eco-friendly’ alternative to cyanide use in the live reef fish industry? SPC Live Reef Fish Information Bulletin 5: 4–7.

Erdmann MV, Vagelli A (2001) Banggai cardinalfish invade Lembeh Strait. Coral Reefs 20: 252–253.

Evanson M, Foster SJ, Vincent ACJ (2011). Tracking the international trade of seahorses (*Hippocampus* species) – the importance of CITES. Fisheries Centre Research Reports 19 Vancouver, BC: Fisheries Centre, University of British Columbia.

Fahrudin A (2003) Extended cost benefit analysis of present and future use of Indonesian coral reefs: an empirical approach to sustainable management of tropical marine resources. Doctoral Dissertation, Institute fur Agrarokonomie, Jakarta, Indonesia, 225 p.

Farrell AP, Tang S, Nomura M, Brauner CJ (2010) Toward improved public confidence in farmed fish: a Canadian perspective on fish welfare during marine transport. Journal of the World Aquaculture Society 41: 225–239.

Fautin DG (1986). Why do anemonefishes inhabit only some host actinians? Environmental Biology of Fishes 15: 171–180.

Fautin D (1991) The anemonefish symbiosis: what is known and what is not. Symbiosis 10: 23–46.

Fautin D, Allen GR (1992) Anemonefishes and their host sea anemones. Western Australian Museum, Perth, 160 p.

Fishelson L (1975) Ethology and reproduction of pteroid fishes found in the Gulf of Aqaba (Red Sea), especially *Dendrochirus brachypterus* (Cuvier), (Pteroidae, Teleostei). Pubblicazioni Della Stazione Zoologica di Napoli 39: 635–656.

Fishelson L (1997) Experiments and observations on food consumption, growth and starvation in *Dendrochirus brachypterus* and *Pterois volitans* (Pteroinae, Scorpaenidae). Environmental Biology of Fishes 50: 391–403.

Fishelson L (1998) Behaviour, socio-ecology and sexuality in damselfishes (Pomacentridae). Italian Journal of Zoology 65: 387–398.

Fisher R, Bellwood DR (2001) Effects of feeding on the sustained swimming abilities of late-stage larval *Amphiprion melanopus*. Coral Reefs 20: 151–154.

Fisher R, Bellwood DR (2002) The influence of swimming speed on sustained swimming performance of late-stage reef fish larvae. Marine Biology 140: 801–807.

Fisher R, Bellwood DR, Job SD (2000) Development of swimming abilities in reef fish larvae. Marine Ecology Progress Series 202: 163–173.

Fitzpatrick SM, Donaldson TJ (2007) Anthropogenic impacts to coral reefs in Palau, western Micronesia during the Late Holocene. Coral Reefs 26: 915–930.

Foster SJ, Vincent ACJ (2004) Life history and ecology of seahorses: Implications for conservation and management. Journal of Fish Biology 65: 1–61.

Foster SJ, Vincent ACJ (2005) Enhancing sustainability of the international trade in seahorses with a single minimum size limit. Conservation Biology 19: 1044–1050.

Foyle TP, Bell JD, Gervis M, Lane I (1997) Survival and growth of juvenile fluted giant clams, *Tridacna squamosa*, in large-scale grow-out trials in the Solomon Islands. Aquaculture 148: 85–104.

Feathers MG, Knable AE (1983) Effects of depressurization upon largemouth bass. North American Journal of Fisheries Management 3: 86–90.

Franklin H, Muhando CA, Lindahl U (1998) Coral culturing and temporal recruitment patterns in Zanzibar, Tanzania. Ambio 27: 651–655.

Frédérich B, Fabri G, Lepoint G, Vandewalle P, Parmentier E (2009) Trophic niches of thirteen damselfishes (Pomacentridae) at the Grand Récif of Toliara, Madagascar. Ichthyological Research 56: 10–17.

Freshwater DW, Hines A, Parham S, Wilbur A, Sabaoun M, Woodhead J, Akins L, Purdy B, Whitfield PE, Paris CB (2009) Mitochondrial control region sequence analyses indicate dispersal from the US East Coast as the source of the invasive Indo-Pacific lionfish *Pterois volitans* in the Bahamas. Marine Biology 156: 1213–1221.

Fricke H (1979) Mating system, resource defense and sex change in the anemonefish *Amphiprion akallopisos*. Zeitschrift für Tierpsychologie 50: 313–326.

Fricke R (1996) Callionymid and tripterygiid fishes of Nusatenggara, Indonesian origins, endemism, and Wallace's Line. (Abstract) International Conference on Eastern Indonesia-Australian Vertebrate Fauna, Mataram, Indonesia.

Fricke HW, Fricke S (1977) Monogamy and sex change by aggressive dominance in coral reef fish. Nature 266: 830–832.

Frisch AJ, Ulstrup KE, Hobbs JPA (2007) The effects of clove oil on coral: An experimental evaluation using *Pocillopora damicornis* (Linnaeus). Journal of Experimental Marine Biology and Ecology 345: 101–109.

Froese R, Pauley D (2012) FishBase version 10/2012. www.fishbase.org

Fry FE, Norris KS (1962) The transportation of live fish. In: Borgstrom G (ed.) Fish as food, Volume II. New York and London: Academic Press, p. 595–608.

Galetto MJ, Bellwood DR (1994) Digestion of algae by *Stegastes nigricans* and *Amphiprion akindynos* (Pisces, Pomacentridae), with an evaluation of methods used in digestibility studies. Journal of Fish Biology 44: 415–428.

Gardner TA, Côté IM, Gill JA, Grant A, Watkinson AR (2003) Long-term region-wide declines in Caribbean corals. Science 301: 958–960.

Gasparini JL, Floeter SR, Ferreira CEL, Sazima I (2005) Marine ornamental trade in Brazil. Biodiversity and Conservation 14: 2883–2899.

Giles BG, KY TS, Hoang DU, Vincent ACJ (2005) The catch and trade of seahorses in Vietnam. Biodiversity and Conservation 15: 2497–2513.

Glaser KB, Mayer AMS (2009) A renaissance in marine pharmacology: from preclinical curiosity to clinical reality. Biochemical Pharmacology 78: 440–448.

Goda M, Fujii R (1995) Blue chromatophores in two species of callionymid fish. Zoological Science 12: 811–813.

Godinot C, Chadwick NE (2009) Phosphate excretion by anemonefish and uptake by giant sea anemones: Demand outstrips supply. Bulletin of Marine Science 85: 1–9.

Godwin J (1994) Behavioural aspects of protandrous sex change in the anemonefish, *Amphiprion melanopus*, and endocrine correlates. Animal Behaviour 48: 551–67.

Godwin J, Fautin D (1992) Defense of host actinians by anemonefishes. Copeia 1992: 902–908.

Goldshmid R, Holzman R, Weihs D, Genin A (2004) Aeration of corals by sleep-swimming fish. Limnology and Oceanography 49: 1832–1839.

Gomez ED, Mingoa-Licuanan SS (2006) Achievements and lessons learned in restocking giant clams in the Philippines. Fisheries Research 80: 46–52.

Gomez ED, Alino PM, Yap HT, Licuanan WY (1994) A review of the status of Philippine reefs. Marine Pollution Bulletin 29: 62–68.

Gonzales BJ, Okamura O, Taniguchi N (1996) Spawning behavior of laboratory-reared dragonet, *Repomucenus huguenini*, and development of its eggs and prolarvae. Suisanzoshoku 44: 7–15.

Gonzales E, Savaris J (2005) International seafood trade: Supporting sustainable livelihoods among poor aquatic resource users in Asia (EP/R03/014). Output 2 Marine Ornamentals trade in the Philippines and options for its poor stakeholders Poseidon Aquatic Resource Management Ltd, Network of Aquaculture Centres in Asia-Pacific (NACA), and the STREAM Initiative. 118 p.

Gopakumar G (2005) Marine ornamental fish culture status, constraints and potential. Ocean Life Food and Medicine Expo 2004 Proceedings. p. 347–359.

Goswami M, Thangaraj K, Chaudhary B, Bhaskar L, Gopalakrishnan A, Joshi M, Singh L, Lakra W (2009) Genetic heterogeneity in the Indian stocks of seahorse (*Hippocampus kuda* and *Hippocampus trimaculatus*) inferred from mtDNA cytochrome b gene. Hydrobiologia 621: 213–221.

Govan H, Fabro LY, Ropeti E (1993) Controlling predators of cultured tridacnid clams. In: Fitt WK (ed.) Biology and Mariculture of Giant Clams. ACIAR Proceedings, 47: 111–118.

Graham WM, Martin DL, Felder DL, Asper VL (2003) Ecological and economic implications of a tropical jellyfish invader in the Gulf of Mexico. Biological Invasions 5: 53–69.

Gray SJ, Ramsey CG (1957). Adrenal influences upon the stomach and the gastric responses to stress. Recent Progress in Hormone Research 13: 583–617.

Green BS, Fisher R (2004) Temperature influences swimming speed, growth and larval duration in coral reef fish larvae. Journal of Experimental Marine Biology and Ecology 299: 115–132.

Green BS, McCormick MI (2005a) Maternal and paternal effects determine size, growth and performance in larvae of a tropical reef fish. Marine Ecology Progress Series 289: 263–272.

Green BS, McCormick MI (2005b) O_2 replenishment to fish nests: Males adjust brood care to ambient conditions and brood development. Behavioral Ecology 16: 389–397.

Green E (2003) International trade in marine aquarium species: Using the global marine aquarium database. In: Cato JC, Brown CL (eds.) Marine Ornamental Species: Collection, Culture, and Conservation. Iowa State Press Ames, Iowa, p. 31–47.

Green EP, Shirley F (1999) The global trade in corals. World Conservation Monitoring Centre, World Conservation Press, Cambridge, UK.

Green SJ, Akins JL, Maljković A, Côté IM (2012) Invasive lionfish drive Atlantic coral reef fish declines. PLoS ONE 7: e32596.

Grey M, Blais A-M, Vincent ACJ (2005). Magnitude and trends of marine fish curio imports to the USA. Oryx 39: 413–420.

Grigg RW (1976) Fisheries management of precious and stony corals in Hawaii. UNIHI-SEAGRANT, TR-77–03. Honolulu: University of Hawaii Sea Grant Program.

Grigg RW (1989) Precious coral fisheries of the Pacific and Mediterranean. In Caddy JF (ed.) Marine Invertebrate Fisheries: Their Assessment and Management. New York: Wiley, p. 636–645.

Guest JR, Todd PA, Goh E, Sivaloganathan B, Reddy KP (2008) Can giant clam (*Tridacna squamosa*) populations be restored on Singapore's heavily impacted coral reefs? Aquatic Conservation – Marine and Freshwater Ecosystems 18: 570–579.

Gulland JA (1971) The fish resources of the oceans. West fleet survey. Fishing News Ltd., FAO.

Hall KC, Bellwood DR (1995) Histological effects of cyanide, stress and starvation on the intestinal mucosa of *Pomacentrus coelestis*, a marine aquarium fish species. Journal of Fish Biology 47: 438–454.

Hall H, Warmolts D (2003) The role of public aquariums in the conservation and sustainability of the marine ornamentals trade. In: Cato JC, Brown CL (eds.) Marine Ornamental Species: Collection, Culture, and Conservation. Iowa State Press Ames, Iowa, p. 307–323.

Hamner RM, Freshwater DW, Whitfield PE (2007) Mitochondrial cytochrome b analysis reveals two invasive lionfish species with strong founder effects in the western Atlantic. Journal of Fish Biology 71: 214–222.

Hanawa M, Harris L, Graham M, Farrell AP, Bendall-Young LI (1989) Effects of cyanide exposure on *Dascyllus aruanus*, a tropical marine fish species: lethality, anaesthesia and physiological effects. Aquarium Sciences and Conservation 2: 21–34.

Hanawa ML, Harris L, Graham M, Farrell AP, Bendell-Young LI (1998) Effects of cyanide exposure on *Dascyllus aruanus*, a tropical marine fish species: lethality, anaesthesia and physiological effects. Aquarium Sciences and Conservation 2: 21–34.

Hannah RW, Parker SJ, Matteson KM (2008) Escaping the surface: the effect of capture depth on submergence success of surface-released Pacific rockfish. North American Journal of Fisheries Management 28: 694–700.

Haresign TH, Shumway SE (1981) Permeability of the marsupium of the pipefish *Syngnathus fuscus* to [^{14}C]-Alpha amino isobutyric acid. Comparative Biochemistry and Physiology 69A: 603–604.

Harriott VJ (2001) The sustainability of Queensland's coral harvest fishery. CRC Reef Research Centre Technical Report No. 40. CRC Reef Research Centre, Townsville. 33 p.

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.

Hattori A (2005) High mobility of the protandrous anemonefish *Amphiprion frenatus*: Nonrandom pair formation in limited shelter space. Ichthyological Research 52: 5–63.

Hattori A (2006) Vertical and horizontal distribution patterns of the giant sea anemone *Heteractis crispa* with symbiotic anemonefish on a fringing coral reef. Journal of Ethology 24: 51–57.

Hattori A, Kobayashi M (2009) Incorporating fine-scale seascape composition in an assessment of habitat quality for the giant sea anemone *Stichodactyla gigantea* in a coral reef shore zone. Ecological Research 24: 415–422.

Hawkins AJS, Klumpp DW (1995) Nutrition of the giant clam *Tridacna gigas* (L.) II. Relative contribution of filter feeding and the ammonium acquired and recycled by symbiotic algae towards total nitrogen requirements for tissue growth and metabolism. Journal of Experimental Marine Biology and Ecology 190: 263–290.

Hayes RL, Goreau NI (1977) Cytodynamics of coral calcification. Proceedings 3rd International Coral Reef Symposium 2: 434–439.

Heidel J, Miller-Morgan TJ (2004) "Shipping fever" in marine ornamentals: Environmental and infectious factors predisposing to postshipping losses. Abstract. In: Program and abstracts for Marine Ornamentals '04 Collection, Culture, and Conservation Conference, held March 1–4, 2004 in Honolulu, Hawaii, p. 47.

Herwig N (1976) Starvation, a cyanide syndrome. Marine Aquarist 7: 5–11.

Herwig N (1977) Symptoms and diagnosis of cyanide poisoning. Marine Aquarist 8: 34–40.

Heslinga GA, Watson TC (1985) Recent advances in giant clam mariculture. Proceedings of the 5th International Coral Reef Congress. 5: 531–537.

Hill RW, Dacey JWH, Hill SD, Edward A, Hicks WA (2004) Dimethylsulfoniopropionate in six species of giant clams and the evolution of dimethylsulfide after death. Canadian Journal of Fisheries and Aquatic Sciences 61: 758–764.

Hingco TG, Rivera R (1991) Aquarium fish industry in the Philippines: Toward development or destruction. ICLARM Conference Proceedings 22: 249–253.

Hirose E, Iwai K, Maruyama T (2006) Establishment of the photosymbiosis in the early ontogeny of three giant clams. Marine Biology 148: 551–558.

Hodgson G (1999) A global assessment of human effects on coral reefs. Marine Pollution Bulletin 38: 345–355.

Hodgson G, Liebeler J (2002) The global coral reef crisis: trends and solutions. Reef Check Foundation report. 80 p.

Hoegh-Guldberg O (1999) Climate change, coral bleaching, and the future of the world's coral reefs. Marine and Freshwater Research 50: 839–866.

Hoegh-Guldberg O, Bruno JF (2010) Impacts of climate change on the world's marine ecosystems. Science 328: 1523–1528.

Hoegh-Guldberg O, Mumby PJ, Hooten AJ, Steneck RS, Greenfield P, Gomez E, Harvell CD, Sale PF, Edwards AJ, Caldeira K, Knowlton N, Eakin CM, Iglesias-Prieto R, Muthiga N, Bradbury RH, Dubi A, Hatziolos ME (2007) Coral reefs under rapid climate change and ocean acidification. Science 318: 1737–1742.

Hoeksema BW (1989) Taxonomy, phylogeny and biogeography of mushroom corals (Scleractinia: Fungiidae). Zoologische Verhandelingen 254: 1–295.

Hoffman EA, Kolm N, Berglund A, Arguello JR, Jones AG (2005) Genetic structure in the coral-reef-associated Banggai cardinalfish, *Pterapogon kauderni*. Molecular Ecology 14: 1367–1375.

Holbrook SJ, Schmitt RJ (2005) Growth, reproduction and survival of a tropical sea anemone (Actinaria): benefits of hosting anemonefish. Coral Reefs 24: 67–73.

Holt EWL (1898) On the breeding of dragonet (*Callionymus lyra*) at the Marine Biological Association’s Aquarium at Plymouth; with a preliminary account of the elements, and some remarks on the significance of the sexual dimorphism. Proceedings of the Zoological Society of London Part 2: 281–315.

Hoover JP (1993) Hawaii’s Fishes: A guide for snorkelers, divers and aquarists. Mutual Publishing, Honolulu, HI, USA, 183 p.

Houde ED (1984) Callionymidae: Development and relationships. In: Moser HG, Richards WJ, Cohen DM, Fahay MP, Kendall AW, Richardson SL (eds.) Ontogeny and systematics of fishes. American Society of Ichthyologists and Herpetologists Special Publication No. 1, p. 637–640.

Huang DW, Todd PA, Guest JR (2007) Movement and aggregation in the fluted giant clam (*Tridacna squamosa* L.). Journal of Experimental Marine Biology and Ecology 342: 269–281.

Hughes TP, Baird AH, Bellwood DR, Card M, Connolly SR, Folke C, Grosberg R, Hoegh-Guldberg O, Jackson JBC, Kleypas J, Lough JM, Marshall P, Nyström M, Palumbi SR, Pandolfi JM, Rosen B, Roughgarden J (2003) Climate change, human impacts, and the resilience of coral reefs. Science 301: 929–933.

Imamura H, Yabe M (1996) Larval record of a red firefish, *Pterois volitans*, from northwestern Australia (Pisces: Scorpaeniformes). Bulletin of the Faculty of Fisheries Hokkaido University 47:41–46.

Indrawan M (1999) Live reef food fish trade in the Banggai islands (Sulawesi, Indonesia): A case study. SPC Live Reef Fish Information Bulletin 6: 7–14.

Indrawan M, Suseno (2008) The complications of CITES inclusion of endemic species in Indonesia: Lessons learned from an in-country deliberation on protecting the Banggai cardinalfishi, *Pterapogon kauderni*. SPC Live Reef Fish Bulletin 18: 13–16.

Ishikura M, Adachi K, Maruyama T (1999) Zooxanthellae release glucose in the tissue of a giant clam, *Tridacna crocea*. Marine Biology 133: 665–673.

Iwata E, Nagai Y, Hyoudou M, Sasaki H (2008) Social environment and sex differentiation in the false clown anemonefish, *Amphiprion ocellaris*. Zoological Science 25: 123–128.

Iwata E, Nagai Y, Sasaki H (2010) Social rank modulates brain arginine vasotocin immunoreactivity in false clown anemonefish (*Amphiprion ocellaris*). Fish Physiology and Biochemistry 36: 33–345.

Jaap WC, Wheaton J (1975). Observations on Florida reef corals treated with fish collecting chemicals. Florida Marine Research Publications 10: 1–17.

Jackson JBC, Kirby MX, Berger WH, Bjorndal KA, Botsford LW, Bourque BJ, Bradbury RH, Cooke R, Erlandson J, Estes JA, Hughes TP, Kidwell S, Lange CB, Lenihan HS, Pandolfi JM, Peterson CH, Steneck RS, Tegner MJ, Warner RR (2001) Historical overfishing and the recent collapse of coastal ecosystems. Science 293: 629–663.

Jantzen C, Wild C, El-Zibdah M, Roa-Quiaoit HA, Haacke C, Richter C (2008) Photosynthetic performance of giant clams, *Tridacna maxima* and *T. squamosa*, Red Sea. Marine Biology 155: 211–221.

Jennison BL (1981) Reproduction in three species of sea anemones from Key West, Florida. Canadian Journal of Zoology 59: 1708–1719.

Johannes RE (1998) Government-supported, village-based management of marine resources in Vanuatu. Ocean & Coastal Management. 40: 165–186.

Johannes RE, Riepen M (1995) Environmental, economic and social implications of the live reef fish trade in Asia and the Western Pacific. South Pacific Commission, Forum Fisheries Agency, Report to the Nature Conservancy, 83 p.

Jones AM (2011) Raiding coral nurseries? Diversity 3: 466–482.

Jones AM, Gardner S, Sinclair W (2008) Losing 'Nemo': Bleaching and collection appear to reduce inshore populations of anemonefishes. Journal of Fish Biology 73: 753–761.

Jones GP, Planes S, Thorrold SR (2005) Coral reef fish larvae settle close to home. Current Biology 15: 1314–1318.

Jones R (2008) CITES, corals and customs: a review of the international trade in wild coral. In: Leewis RJ, Janse M (eds.) Advances in coral husbandry in public aquariums. Public Aquarium Husbandry Series 2. Burgers’ Zoo, Arnhem, The Netherlands.

Jones RJ, Steven AL (1997) Effects of cyanide on corals in relation to cyanide fishing on reefs. Marine and Freshwater Research 48: 517–522.

Jones RJ, Hoegh-Guldberg O (1999) Effects of cyanide on coral photosynthesis: implications for identifying the cause of coral bleaching and for assessing the environmental effects of cyanide fishing. Marine Ecology Progress Series 177: 83–91.

Jones RJ, Kildea T, Hoegh-Guldberg O (1998) PAM chlorophyll flourometry: a new in situ technique for stress assessment in scleractinian corals, used to examine the effects of cyanide from cyanide fishing. Marine Pollution Bulletin 38: 864–874.

Jousson O, Pawlowski J, Zainetti L, Meinesz A, Boudouresques CF (1998) Molecular evidence for the aquarium origin of the green alga *Caulerpa taxifolia* introduced to the Mediterranean Sea. Marine Ecology Progress Series 172: 275–280.

Juinio MAR, Menez LAB, Villanoy CL, Gomez ED (1989) Status of giant clam resources of the Philippines. Journal of Molluscan Studies 55: 431–40.

Juinio-Menez MA, Magsino RM, Ravago-Gotanco R, Yu ET (2003) Genetic structure of *Linckia laevigata* and *Tridacna crocea* populations in the Palawan shelf and shoal reefs. Marine Biology 142: 717–726.

Katekaru A (1978) Regulations of tropical fish collecting. Working Papers. Volume 34. Hawaii Sea Grant College Program. University of Hawaii, Honolulu. p. 35–42.

Keene JL, Noakes DG,Moccia RD, Soto CG (1998) The efficacy of clove oil as an anaesthetic for rainbow trout, *Oncorhynchus mykiss* (Walbaum). Aquaculture Resources 29: 89–101.

Kendrick AJ, Hyndes GA (2005) Variations in the dietary compositions of morphologically diverse syngnathid fishes. Environmental Biology of Fishes 72: 415–427.

Keys JL, Healy JM (1999) Sperm ultrastructure of the giant clam *Tridacna maxima* (Tridacnidae : Bivalvia : Mollusca) from the Great Barrier Reef. Marine Biology 135: 41–46.

Kimball ME, Miller JM, Whitfield PE, Hare JA (2004) Thermal tolerance and potential distribution of invasive lionfish (*Pterois volitans/miles* complex) on the east coast of the United States. Marine Ecology Progress Series 283: 269–278.

Kleypas JA, McManus JW, Lambert AB (1999) Environmental limits to coral reef development: where do we draw the line? American Zoologist 39: 146–59.

Klumpp DW, Griffith CL (1994) Contributions of phototrophic and heterotrophic nutrition to the metabolic and growth requirements of 4 species of giant clam (Tridacnidae). Marine Ecology Progress Series 115: 103–115.

Klumpp DW, Lucas JS (1994) Nutritional ecology of the giant clams *Tridacna tevoroa* and *T. derasa* from Tonga: Influence of light on filter-feeding and photosynthesis. Marine Ecology Progress Series 107: 147–156.

Klumpp DW, Bayne BL, Hawkins AJS (1992) Nutrition of the giant clam *Tridacna gigas* (L.). I Contribution of filter feeding and photosynthates to respiration and growth. Journal of Experimental Marine Biology and Ecology 155: 105–122.

Knittweis L (2008) Population demographics and life history characteristics of *Heliofungia actiniformis*: a fungiid coral species exploited for the live coral aquarium trade in the Spermonde Archipelago, Indonesia. Ph.D. Thesis, Centre for Tropical Marine Ecology Bremen, April 2008, 133 p.

Knittweis L, Wolff M (2010) Live coral trade impacts on the mushroom coral *Heliofungia actiniformis* in Indonesia: Potential future management approaches. Biological Conservation 143: 2722–2729.

Knittweis L, Kraemer WE, Timm J, Kochzius M (2009a) Genetic structure of *Heliofungia actiniformis* (Scleractinia: Fungiidae) populations in the Indo-Malay Archipelago: implications for live coral trade management efforts. Conservation Genetics 10: 241–249.

Knittweis L, Jompa J, Richter C, Wolff M (2009b) Population dynamics of the mushroom coral *Heliofungia actiniformis* in the Spermonde Archipelago, South Sulawesi, Indonesia. Coral Reefs 28: 793–804.

Knowlton N (2001) The future of coral reefs. Proceedings of the National Academy of Sciences USA 98: 5419–5425.

Koenig CC (2001) *Oculina* banks: habitat, fish populations, restoration and enforcement. Report to the South Atlantic Fishery Management Council. North Charleston, SC: South Atlantic Fishery Management Council.

Koldewey HJ, Martin-Smith KM (2010) A global review of seahorse aquaculture. Aquaculture 302: 131–152.

Kolm N (2002) Male size determines reproductive output in a paternal mouthbrooding fish. Animal Behaviour 63: 727–733.

Kolm N (2004) Female courtship in the Banggai cardinalfish: honest signals of egg maturity and reproductive output? Behavioral Ecology and Sociobiology 56: 59–64.

Kolm N, Berglund A (2003) Wild populations of a reef fish suffer from the "nondestructive" aquarium trade fishery. Conservation Biology 17: 910–914.

Kolm N, Olsson J (2003) Differential investment in the Banggai cardinalfish: can females adjust egg size close to egg maturation to match the attractiveness of a new partner? Journal of Fish Biology 63: 144–151.

Kolm N, Hoffman EA, Olsson J, Berglund A, Jones AG (2005) Group stability and homing behavior but no kin group structures in a coral reef fish Behavioral Ecology 16: 521–527.

Kochzius M, Nuryanto A (2008) Strong genetic population structure in the boring giant clam, *Tridacna crocea,* across the Indo-Malay Archipelago: Implications related to evolutionary processes and connectivity. Molecular Ecology 17: 3775–3787.

Kwan SK, Chen TW, Teo LH (1994) The control of total ammonia and carbon dioxide levels in the packing physiology of mollies. In: Chou LM et al. (eds) Proceedings of the Third Asian Fisheries Forum, held in Singapore. Manila, Philippines: Asian Fisheries Society, p. 888–891.

Land MF (2003) The spatial resolution of the pinhole eyes of giant clams (*Tridacna maxima*). Proceedings of the Royal Society of London Series B 270: 185–188.

Laurent D, Kerbrat AS, Darius HT, Girard E, Golubic S, Benoit E, Sauviat MP, Chinain M, Molgo J, Pauillac S (2008) Are cyanobacteria involved in Ciguatera Fish Poisoning-like outbreaks in New Caledonia? Harmful Algae 7: 827–838.

Lebata-Ramos MJHL, Okuzawa K, Maliao RJ, Abroguena JBR, Dimzon MDN, Doyola-Solis EFC, Dacles TU (2010) Growth and survival of hatchery-bred giant clams (*Tridacna gigas*) in an ocean nursery in Sagay Marine Reserve, Philippines. Aquaculture International 18: 19–33.

Leduc G (1984) Cyanides in water: toxicological significance. In Weber LJ (ed.) Aquatic Toxicology Vol. 2 Raven Press, New York, p. 113–153.

Leggat W, Buck BH, Grice A, Yellowlees D (2003) The impact of bleaching on the metabolic contribution of dinoflagellate symbionts to their giant clam host. Plant Cell and Environment 26: 1951–1961.

Leggat W, Whitney S, Yellowlees D (2004) Is coral bleaching due to the instability of the zooxanthellae dark reactions? Symbiosis 37: 137–153.

Leis JM Rennis DS (1983) The larvae of Indo-Pacific coral reef fishes. University of Hawaii Press, Honolulu, Hawaii. 269 p.

Lesser MP (2004) Experimental biology of coral reef ecosystems. Journal of Experimental Marine Biology and Ecology 300: 217–252.

Lesser MP, Slattery M (2011) Phase shift to algal dominated communities at mesophotic depths associated with lionfish (*Pterois volitans*) invasion on a Bahamian coral reef. Biological Invasions 13: 1855–1868.

Liberman T, Genin A, Loya Y (1995) Effects on growth and reproduction of the coral Stylophora pistillata by the mutualistic damselfish *Dascyllus marginatus*. Marine Biology 121: 741–746.

Lieberman S, Field J (2001) Global solutions to global trade impacts? In: Best B, Bornbusch A (eds.) Global trade and consumer choices: coral reefs in crisis. Papers presented at a symposium held at the 2001 annual meeting of the American Association for the Advancement of Science, San Francisco, California, February 19, 2001, p. 19–24.

Lieske E, Myers R (1994) Collins pocket guide. Coral reef fishes. Indo-Pacific & Caribbean including the Red Sea. Haper Collins Publishers. 400 p.

Lilley R (2008) The Banggai cardinalfish: An overview of conservation challenges. SPC Live Reef Fish Bulletin 18: 3–12.

Lim LC, Dhert P, Chew WY, Deraux V, Nelis H, Sorgeloos P (2002) Enhancement of stress resistance of the guppy *Poecilia reticulate* through feeding and Vitamin C supplement. Journal of the World Mariculture Society 33: 32–40.

Lim LC, Dhert P, Sorgeloos P (2003) Recent developments and improvements in ornamental fish packing systems for air transport. Aquaculture Research 34: 923–935.

Lincoln-Smith MP, Pitt KA, Bell JD, Mapstone BD (2006) Using impact assessment methods to determine the effects of a marine reserve on abundances and sizes of valuable tropical invertebrates. Canadian Journal of Fisheries and Aquatic Sciences 63: 1251–1266.

Linton JR, Soloff BL (1964) The physiology of the brood pouch of the male seahorse *Hippocampus erectus*. Bulletin of Marine Science Gulf and Caribbean 14: 45–61.

Loram JE, Trapido-Rosenthal HG, Douglas AE (2007) Functional significance of genetically different symbiotic algae *Symbiodinium* in a coral reef symbiosis. Molecular Ecology 16: 4849–4857.

Losey GS, Balaz SGH Privitera LA (1994) Cleaning symbiosis between the wrasse *Thalassoma duperrey*, and the green turtle *Chelonia mydas*. Copeia 1994: 684–690.

Lourie SA, Vincent ACJ (2004) A marine fish follows Wallace's Line: The phylogeography of the three-spot seahorse (*Hippocampus trimaculatus*, Syngnathidae, Teleostei) in Southeast Asia. Journal of Biogeography 31: 1975–1985.

Lourie SA, Vincent ACJ, Hall HJ (1999) Seahorses: An Identification Guide to the World’s Species and their Conservation. Project Seahorse, London.

Lourie SA, Foster SJ, Cooper EWT, Vincent ACJ (2004) A guide to the identification of seahorses. Washington, DC: University of British Columbia and World Wildlife Fund.

Lourie SA, Green DM, Vincent ACJ (2005) Dispersal, habitat differences, and comparative phylogeography of Southeast Asian seahorses (Syngnathidae : *Hippocampus*). Molecular Ecology 14: 1073–1094.

Lovell ER (2001) Status report: collection of coral and other benthic reef organisms for the marine aquarium and curio trade in Fiji. World Wide Fund for Nature South Pacific Program, GPO Suva, Fiji. 82 p.

Lubbock HR, Polunin NVC (1975) Conservation and the tropical marine aquarium trade. Environmental Conservation 2: 229–232.

Lucas JS (1988) Giant clams: description, distribution and life history. In: Copland JW, Lucas JS (eds.) Giant clams in Asia and the Pacific. Australian Centre for International Agricultural Research, Canberra, p. 21–32.

Lucas JS (1994) The biology, exploitation and mariculture of giant clams (Tridacnidae). Reviews in Fisheries Science 2: 181–223.

Lunn KE, Moreau MA (2004) Unmonitored trade in marine ornamental fishes: the case of Indonesia's Banggai cardinalfish (*Pterapogon kauderni*). Coral Reefs 23: 344–351.

Macaranas JM, Ablan CA, Pante MJR, Benzie JAH, Williams ST (1992) Genetic-structure of giant clam (*Tridacna derasa*) populations from reefs in the Indo-Pacific. Marine Biology 113: 231–238.

Magos L (1992) Chemical hazards and toxicology. In Luxton SG (ed.) Hazards in the chemical laboratory, 5th edition. Royal Society of Chemistry, London, p. 66–95.

Mai W (2000) Nachzucht des Mandarinfisches. Jahrgang 53: 8–9.

Mak KKW, Yanase H, Renneberg R (2005) Cyanide fishing and cyanide detection in coral reef fish using chemical tests and biosensors. Biosensors and Bioelectronics 20: 2581–2593.

Maljković A, van Leeuwen TE, Cove SN (2008) Predation on the invasive red lionfish, *Pterois volitans* (Pisces : Scorpaenidae), by native groupers in the Bahamas. Coral Reefs 27: 501–501.

Marcus JE, Samoilys MA, Meeuwig JJ, Villongco ZAD, Vincent ACJ (2006) Benthic status of near-shore fishing grounds in the central Philippines and associated seahorse densities. Marine Pollution Bulletin 54: 1483–1494.

Mariscal RN (1970a) The nature of the symbiosis between Indo-Pacific anemone fishes and sea anemones. Marine Biology 6: 58–65.

Mariscal RN (1970b). A field and laboratory study of the symbiotic behavior of fishes and sea anemones from the tropical Indo–Pacific. University of California Publications in Zoology 91: 1–43.

Maroz A, Fishelson L (1997) Juvenile production of *Amphiprion bicinctus* (Pomacentridae, Teleostei) and rehabilitation of impoverished habitats. Marine Ecology Progress Series 151: 295–297.

Martin-Smith KM, Vincent ACJ (2005) Seahorse declines in the Derwent estuary, Tasmania in the absence of fishing pressure. Biological Conservation 123: 533–545.

Martin-Smith KM, Vincent ACJ (2006). Exploitation and trade in Australian seahorses, pipehorses, sea dragons and pipefishes (Family Syngnathidae). Oryx 40: 141–151.

Martin-Smith KM, Samoilys MA, Meeuwig JJ, Vincent ACJ (2004) Collaborative development of management options for an artisanal fishery for seahorses in the central Phillipines. Ocean and Coastal Management 47: 165–193.

Mathews Amos A, Claussen JD (2009) Certification as a Conservation Tool in the Marine Aquarium Trade: Challenges to Effectiveness. Turnstone Consulting and Starling Resources. 52 p.

McAllister DE (1988) Environmental, economic, and social costs of coral reef destruction in the Philippines. Galaxea 7: 161–178.

McCauley DJ, Joyce FJ, Lowenstein JH (2008) Effects of the aquarium fish industry in Costa Rica on populations of the Cortez rainbow wrasse *Thalassoma lucasanum*. Ciencias Marinas 34: 445–451.

McFarland WN (1960) The use of anesthetics for the handling and transport of fishes. California Fish and Game 46: 407–431.

McFarland WN, Norris KS (1958) The control of pH by buffers in fish transport. California Fish and Game 44: 291–310.

McLarney WO (1986) Collecting coral reef fishes in the Philippines: Information and analogue. Scientists debate: more reefs, more fishes die. Annals of Earth (Part 2 of 4) 4: 16–20.

McManus JW, Reyes RBJ, Nañola CL (1997) Effects of some destructive fishing methods on coral cover and potential rates of recovery. Environmental Management 21: 69–78.

McManus R (2001) U.S. efforts to protect domestic and international coral reefs: trade in the larger context. In: Best B, Bornbusch A (eds.) Global trade and consumer choices: coral reefs in crisis. Papers presented at a symposium held at the 2001 annual meeting of the American Association for the Advancement of Science, San Francisco, California, February 19, 2001, p. 11–14.

McPherson JM, Vincent ACJ (2004) Assessing East African trade in seahorse species as a basis for conservation under international controls. Aquatic Conservation: Marine and Freshwater Ecosystems 14: 521–538.

Mebs D (1994) Anemonefish symbiosis – vulnerability and resistance of fish to the toxin of the sea-anemone. Toxicon 32: 1059–1068.

Mebs D (2009) Chemical biology of the mutualistic relationships of sea anemones with fish and crustaceans. Toxicon 54: 1071–1074.

Meeuwig JJ, Hoang DH, Ky TS, Job SD, Vincent ACJ (2006) Quantifying non-target seahorse fisheries in central Vietnam. Fisheries Research 81: 149–157.

Meister HS, Wyanski DM, Loefer JK, Ross SW, Quattrini AM, Sulak KJ (2005) Further evidence for the invasion and establishment of *Pterois volitans* (Teleostei : Scorpaenidae) along the Atlantic coast of the United States. Southeastern Naturalist 4: 193–206.

Méjean A, Peyraud-Thomas C, Kerbrat AS, Golubic S, Pauillac S, Chinain M 8, Laurent D (2010) First identification of the neurotoxin homoanatoxin-a from mats of *Hydrocoleum lyngbyaceum* (marine cyanobacterium) possibly linked to giant clam poisoning in New Caledonia. Toxicon 56: 829–835.

Michael S (1996) The Banggai cardinalfish: a newly available species that may become too popular for its own good. Aquarium Fish Magazine 8: 86–87.

Michael SW (2000) Fishes for the marine aquarium: the dragonets. Aquarium Fish Magazine September 2000: 23–32.

Michael SW (2005) Reef Aquarium Fishes. TFH Publications, Neptune City, NJ, 448 p.

Mingoalicuanan SS (1993) Oxygen-consumption and ammonia excretion in juvenile *Tridacna gigas* (Linne, 1758) – effects of emersion. Journal of Experimental Marine Biology and Ecology 171: 119–137.

Mitchell JS (2003) Mobility of *Stichodactyla gigantea* sea anemones and implications for resident false clown anemonefish, *Amphiprion ocellaris*. Environmental Biology of Fishes 66: 85–90.

Miyasaka A (1997) Status report, aquarium fish collections, fiscal year 1994–1995. Division of Aquatic Resources, Department of Land and Natural Resources, Honolulu.

MMA (2004) Lista Nacional das Espécies de Invertebrados Aquáticose Peixes Sobreexplotadas ou Ameaçadas de Sobreexplotação. Instrução Normativa nº 05, de 21 de Maio de 2004. Diário Oficial da União. 28 de Maio de 2004: 136–142.

Moe MA (2003) Culture of Marine Ornamentals: For love, for money, and for science. In: Cato JC, Brown CL (eds.) Marine Ornamental Species: Collection, Culture, and Conservation. Iowa State Press Ames, Iowa, p. 11–28.

Monteiro-Neto C, de Andrade Cunha FE, Carvalho Nottingham M, Araújo ME, Lucena Rosa I, Leite Barros GM (2003) Analysis of the marine ornamental fish trade at Ceará State, northeast Brazil. Biodiversity and Conservation 12: 1287–1295.

Moore A, Ndobe S (2007) Discovery of an introduced Banggai cardinalfish population in Palu Bay, Central Sulawesi, Indonesia. Coral Reefs 26: 569–569.

Moore F, Best B (2001) Coral reef crisis: causes and consequences. In: Best B, Bornbusch A (eds.) Global trade and consumer choices: coral reefs in crisis. Papers presented at a symposium held at the 2001 annual meeting of the American Association for the Advancement of Science, San Francisco, California, February 19, 2001, p. 5–10.

Morell V (2010) Mystery of the lionfish: don't blame Hurricane Andrew. Science, Science-Insider Article, April 29, 2010. http://news.sciencemag.org/scienceinsider/2010/04/mystery-of-the-lionfish-dont-bla.html

Morris JA, Akins JL (2009) Feeding ecology of invasive lionfish (*Pterois volitans*) in the Bahamian archipelago. Source: Environmental Biology of Fishes 86: 389–398.

Morris JA, Sullivan CV, Govoni JJ (2011a) Oogenesis and spawn formation in the invasive lionfish, *Pterois miles* and *Pterois volitans*. Scientia Marina 75: 147–154.

Morris JA, Shertzer KW, Rice JA (2011b) A stage-based matrix population model of invasive lionfish with implications for control. Biological Invasions 13: 7–12.

MPA News (2007) Indonesia protects local fisheries with 7 MPAs. MPA News June: 5.

Mumby PJ, Steneck RS (2008) Coral reef management and conservation in light of rapidly evolving ecological paradigms. Trends in Ecology & Evolution 23: 555–563.

Mumby PJ, Dahlgren CP, Harborne AR, Kappel CV, Micheli F, Brumbaugh DR, Holmes KE, Mendes JM, Broad K, Sanchirico JN, Buch K, Box S, Stoffle RW, Gill AB (2006) Fishing, trophic cascades, and the process of grazing on coral reefs. Science 311: 98–101.

Mumby PJ, Harborne AR, Brumbaugh DR (2011) Grouper as a natural biocontrol of invasive lionfish. PLoS ONE 6: e21510.

Munday PL, Wilson SK (1997) Comparative efficacy of clove oil and other chemicals in anaesthetization of *Pomacentrus amboinensis*, a coral reef fish. Journal of Fish Biology 51: 931–938.

Muñoz RC, Currin CA, Whitfield PA (2011) Diet of invasive lionfish on hard bottom reefs of the Southeast USA: insights from stomach contents and stable isotopes. Marine Ecological Progress Series 432: 181–193.

Murakoshi M, Hirata H (1993) Self-fertilization in 4 species of giant clam. Nippon Suisan Gakkaishi 59: 581–587.

Murata M, Miyagawa-Kohshima K, Nakanishi K, Naya Y (1986) Characterization of compounds that induce symbiosis between sea anemone and anemonefish. Science 234: 585–587.

Muscatine L, Porter J (1977) Reef corals: Mutualistic symbioses adapted to nutrient poor environments. Bioscience 27:454–460.

Muscatine L, McCloskey LR, Marian RE (1981) Estimating the daily contribution of carbon from zooxanthellae to coral animal respiration. Limnology and Oceanography 26: 601–611.

Myers RF (1999) Micronesian reef fishes: A comprehensive guide to the coral reef fishes of Micronesia, 3rd revised and expanded edition. Coral Graphics, Barrigada, Guam. 330 p.

Nadeau JL, Curtis JMR, Lourie SA (2009) Preservation causes shrinkage in seahorses: implications for biological studies and for managing sustainable trade with minimum size limits. Aquatic Conservation: Marine and Freshwater Ecosystems 19: 428–438.

Nañolla CL, Aliño PM, Carpenter KE (2011) Exploitation-related reef fish species richness depletion in the epicenter of marine biodiversity. Environmental Biology of Fishes 90: 405–420.

National Geographic (2010) Pictures: sharks taught to hunt alien lionfish. http://news.nationalgeographic.com/news/2011/03/pictures/110404-sharks-lionfish-alien-fish-invasive-species-science/?now=2011-04-04-00:01

Ndobe S, Moore A (2008) Banggai cardinalfish: towards a sustainable ornamental fishery. Proceedings of the 11th International Coral Reef Symposium, Ft Laurderdale, FL, 1026–1029.

Nelson EJ, Ghiorse WC (1999) Isolation and identification of *Pseudoalteromonas piscicida* strain Cura-d associated with diseased damselfish (Pomacentridae) eggs. Journal of Fish Diseases 22: 253–260.

Neo ML, Todd PA, Teo SLM, Chou LM (2009) Can artificial substrates enriched with crustose coralline algae enhance larval settlement and recruitment in the fluted giant clam (*Tridacna squamosa*)? Hydrobiologia 625: 83–90.

Newman LJ, Cannon LRG, Govan H (1993) *Stylochus* (*Imogene*) *matatasi* n-sp. (Platyhelminthes, Polycladida) – pest of cultured giant clams and pearl oysters from Solomon Islands. Hydrobiologia 257: 185–189.

Nolan RS (1978) Hawaii tropical fish study. Working Papers. Volume 34. Hawaii Sea Grant College Program. University of Hawaii, Honolulu. p. 27–34.

Norton JH, Shepherd MA, Long HM, Fitt WK (1992) Zooxanthellal tubular system in the giant clam. Biological Bulletin 183: 503–506.

Noyes JC (1976) Collecting without drugs. Marine Aquarist 6: 5–12.

Nuñez MA, Kuebbing S, Dimarco RD, Simberloff D (2012) Invasive species: to eat or not to eat, that is the question. Conservation Letters 5: 334–341.

Nuryanto A, Kochzius M (2009) Highly restricted gene flow and deep evolutionary lineages in the giant clam *Tridacna maxima*. Coral Reefs 28: 607–619.

O'Donnell KP, Pajaro MG, Vincent ACJ (2010) How does the accuracy of fisher knowledge affect seahorse conservation status? Animal Conservation 13: 526–533.

Öhman MC, Rahasuriya A, Liden O (1993) Human disturbances on coral reefs in Sri Lanka. Ambio 22: 474–480.

Okada H (1997) Market survey of aquarium giant clams in Japan. Access from Food and Agricultural Organizations Corporate Document Repository. URL: http://www.fao.org/docrep/005/AC892E/AC892E01.htm

Okuzawa K, Maliao RJ, Quinitio ET, Buen-Ursua SMA, Lebata MJHL, Gallardo WG, Garcia LMB, Primavera JH (2008) Stock enhancement of threatened species in Southeast Asia. Reviews in Fisheries Science 16: 394–402.

Olivotto I, Buttino I, Borroni M, Piccinetti CC, Malzone MG, Carnevali O (2008) The use of the Mediterranean calanoid copepod *Centropages typicus* in Yellowtail clownfish (*Amphiprion clarkii*) larviculture. Aquaculture 284: 211–216.

Olivotto I, Tokle NE, Nozzi V, Cossignani L, Carnevali O (2010) Preserved copepods as a new technology for the marine ornamental fish aquaculture: A feeding study. Aquaculture 308: 124–131.

Olivotto I, Planas M, Simoes N, Holt GJ, Avella MA, Calado R (2011) Advances in breeding and rearing marine ornamentals. Journal of the World Aquaculture Society. 42: 135–166.

Ollerton J, McCollin D, Fautin DG, Allen GR (2007) Finding NEMO: nestedness engendered by mutualistic organization in anemonefish and their hosts. Proceedings of the Royal Society Series B – Biological Sciences 274: 591–598.

Ortiz DM, Tissot BN (2008) Ontogenetic patterns of habitat use by reef-fish in a Marine Protected Area network: a multi-scaled remote sensing and in situ approach. Marine Ecology Progress Series 365: 217–232.

Padilla DK, Williams SL (2004) Beyond ballast water: aquarium and ornamental trades as sources of invasive species in aquatic ecosystems. Frontiers in Ecology and the Environment 2: 131–138.

Panithanarak T, Karuwancharoen R, Na-Nakorn U, Nguyen TTT (2010) Population genetics of the spotted seahorse (*Hippocampus kuda*) in Thai Waters: Implications for conservation. Zoological Studies 49: 564–576.

Parrish FA, Moffitt RM (1993) Subsurface fish handling to limit decompression effects on deepwater species. Marine Fisheries Review 43: 29–32.

Parmentier E, Colleye O, Mann D (2009) Hearing ability in three clownfish species. Journal of Experimental Biology 212: 2022–2025.

Pat J (1997) Destructive fishing methods in and around Komodo National Park. SPC Live Reef Fish Information Bulletin 2: 20–24.

Paxton JR Eschmeyer WN (1998) Encyclopedia of fishes, 2nd edition. Academic Press, San Diego, California. 240 p.

Pearson RG, Munro JL (1991) Growth, mortality and recruitment rates of giant clams, *Tridacna gigas* and *T. derasa*, at Michaelmas Reef, central Great-Barrier-Reef, Australia. Australian Journal of Marine and Freshwater Research 42: 241–262.

Perino L (1990) Assessment of the feasibility of establishing an aquarium fish industry in Papua New Guinea. FFA Report, no. 90/30. South Pacific Forum Fisheries Agency, Honiara, Solomon Islands. 35p.

Perry AL, Lunn KE, Vincent ACJ (2010) Fisheries, large-scale trade, and conservation of seahorses in Malaysia and Thailand. Aquatic Conservation – Marine and Freshwater Ecosystems 20: 464–475.

Pet JS, Djohani RH (1998) Combating destructive fishing practices in Komodo National Park: ban the hookah compressor! South Pacific Commission, Live Reef Fish Information Bulletin 4: 17–28.

Pet-Soede L (2001) Destructive fishing practices mini symposium. SPC Live Reef Fish Information Bulletin. 8: 16–19.

Peters G (1982). The effect of stress on the stomach of the European eel, *Anguilla anguilla* L. Journal of Fish Biology 21: 497–512.

Pickles AL (1992) Partitioning of respiration between the sea anemone *Aiptasia pulchella* and its symbiotic alga *Symbiodinium* sp. Honors Thesis, Sydney University.

Pimental D, Lach L, Zuniga R, Morrison D (1999) Environmental and economic costs associated with nonindigenous species in the United States. College of Agriculture and Life Sciences, Cornell University, Ithaca, NY.

Pinsky ML, Montes HR, Palumbi SR (2010) Using isolation by distance and effective density to estimate dispersal scales in anemonefish. Evolution 64: 2688–2700.

Planes S, Jones GP, Thorrold SR (2009) Larval dispersal connects fish populations in a network of marine protected areas. Proceedings of the National Academy of Sciences USA 106: 5693–5697.

Pollard DA (1984) A review of ecological studies on seagrass-fish communities, with a particular reference to recent studies in Australia. Aquatic Botany 18: 3–42.

Porat D, Chadwick-Furman NE (2004) Effects of anemonefish on giant sea anemones: Expansion behavior, growth and survival. Hydrobiologia 530/531: 513–520.

Porat D, Chadwick-Furman NE (2005) Effects of anemonefish on giant sea anemones: ammonium uptake, zooxanthellae content and tissue regeneration. Marine and Freshwater Behaviour and Physiology 38: 43–51.

Pyle RL (1993). Marine aquarium fish. in Wright A, Hill L (eds) Nearshore Marine Resources of the South Pacific. Information for fisheries development and management. Suva, Fiji: Institute of Pacific Studies, and Honiara, Solomon Islands: Forum Fisheries Agency. p. 135–176.

Queensland Fisheries Management Authority (QFMA) (1999) Discussion paper 10. Queensland marine fish and coral collecting fisheries. Prepared for the Queensland Fisheries Management Authority and Harvest MAC by the Aquarium Fish and Coral Fisheries Working Group. 84 p.

Quenouille B, Bermingham E, Planes S (2004) Molecular systematics of the damselfishes (Teleostei : Pomacentridae): Bayesian phylogenetic analyses of mitochondrial and nuclear DNA sequences. Molecular Phylogenetics and Evolution 31: 66–88.

Randall JE (1987) Collecting reef fishes for aquaria. In: Salvat B (ed.) Human Impacts on Coral Reefs: Facts and Recommendations, French Polynesia, Antenne Museum-EPHE, p. 30–39.

Randall JE, Allen GR, Steene RC (1990) Fishes of the Great Barrier Reef and Coral Sea. University of Hawaii Press, Honolulu, Hawaii. 506 p.

Rasotto MB, Sadovy de Mitcheson Y, Mitcheson G (2010) Male body size predicts sperm number in the mandarinfish. Journal of Zoology 281: 161–167.

Ravago-Gotanco RG, Magsino RM, Juinio-Menez MA (2007) Influence of the North Equatorial Current on the population genetic structure of *Tridacna crocea* (Mollusca: Tridacnidae) along the eastern Philippine seaboard. Marine Ecology-Progress Series 336: 161–168.

Ravindran VS, Kannan L, Venkateshvaran K (2010) Biological activity of sea anemone proteins: I. Toxicity and histopathology. Indian Journal of Experimental Biology 48: 1225–1232.

Reksodihardjo-Lilley G, Lilley R (2007) Towards a sustainable marine aquarium trade: An Indonesian perspective. SPC Live Reef Fish Information Bulletin 17: 11–19.

Rhyne AL, Tlusty MF (2012) Trends in the marine aquatic trade: the influence of global economics and technology. Aquaculture, Aquarium, Conservation & Legislation. 5: 99–102.

Rhyne A, Rotjan R, Bruckner A, Tlusty M (2009) Crawling to collapse: Ecologically unsound ornamental invertebrate fisheries. PLoS ONE 4: e8413.

Rhyne AL, Tlusty MF, Schofield PJ, Kaufman L, Morris JA Jr, et al. (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 DL, Harrison PL, Harriott VJ (1997a) Timing of spawning and fecundity of a tropical and subtropical anemonefish (Pomacentridae: *Amphiprion*) on a high latitude reef on the east coast of Australia. Marine Ecology Progress Series 156: 175–181.

Richardson DL, Harriott VJ, Harrison PL (1997b). Distribution and abundance of giant sea anemones (Actiniaria) in subtropical eastern Australian waters. Marine and Freshwater Research 48: 59–66.

Richter C, Roa-Quiaoit H, Jantzen C, Al-Zibdah M, Kochzius M (2008) Collapse of a new living species of giant clam in the Red Sea. Current Biology 18: 1349–1354.

Rogers SG, Langston HT, Targett TE (1986) Anatomical trauma to sponge-coral reef fishes captured by trawling and angling. Fishery Bulletin 84: 697–703.

Roopin M, Chadwick NE (2009) Benefits to host sea anemones from ammonia contributions of resident anemonefish. Journal of Experimental Marine Biology and Ecology 370: 27–34.

Roopin M, Henry RP, Chadwick NE (2008) Nutrient transfer in a marine mutualism: patterns of ammonia excretion by anemonefish and uptake by giant sea anemones. Marine Biology 154: 547–556.

Roopin M, Thornhill DJ, Santos SR, Chadwick NE (2011) Ammonia flux, physiological parameters, and *Symbiodinium* diversity in the anemonefish symbiosis on Red Sea coral reefs. Symbiosis 53: 63–74.

Roques JAC, Abbink W, Geurds F, van de Vis H, Flik G (2010) Tailfin clipping, a painful procedure studies on Nile tilapia and common carp. Physiology & Behavior 101: 533–540.

Rosa IL (2005) National Report - Brazil. In: Bruckner AW, Field JD, Daves N (Eds.). The proceedings of the International workshop on CITES implementation for seahorse conservation and trade. NOAA Technical Memorandum, p. 46–53.

Rosa IL, Sampaio CLS, Barros AT (2006) Collaborative monitoring of the ornamental trade of seahorses and pipefishes (Teleostei: Syngnathidae) in Brazil: Bahia State as a case study. Neotropical Ichthyology 4: 247–252.

Ross MA (1984) A quantitative study of the stony coral fishery in Cebu, Philippines. Marine Ecology 5: 75–91.

Ross RM (1978) Reproductive behavior of the anemonefish *Amphiprion melanopus* on Guam. Copeia 1978: 103–107.

Rubec PJ (1986) The effects of sodium cyanide on coral reefs and marine fish in the Philippines. In: Maclean JL, Dizon LB, Hosillos LV (eds.) Proceedings of the First Asian Fisheries Forum, held in Manila. Manila, Philippines: Asian Fisheries Society, p. 297–302.

Rubec PJ (1987) Fish capture methods and Philippine coral reefs - IMA Philippines visit. Part II. Marine Fish Monitor 2: 30–31.

Rubec PJ (1988a) The need for conservation and management of Philippine coral reefs. Environmental Biology of Fishes 23: 141–154.

Rubec PJ (1988b) Cyanide fishing and the International Marinelife Alliance Net-Training Program. Tropical Coastal Area Management. 3.

Rubec PJ, Pratt VR (1984) Scientific data concerning the effects of cyanide on marine fish. Freshwater And Marine Aquarium 7: 4–6, 78–80, 82–86, 90–91.

Rubec PJ, Soundararajan R (1991) Chronic toxic effects of cyanide on tropical marine fish. In: Chapman P et al. (eds) Proceedings of the Seventeenth Annual Toxicity Workshop: November 5–7, 1990, Vancouver, B.C. Canadian Technical Report of Fisheries and Aquatic Sciences, 1774: 243–251.

Rubec PJ, Cruz F (2002) Net-training to CAMP: Community-based programmes that benefit coral reef conservation and the aquarium trade. Ornamental Fish International, OFI Journal 40: 12–18.

Rubec PJ, Cruz FP (2005) Monitoring the chain of custody to reduce delayed mortality of net-caught fish in the aquarium trade. SPC Live Reef Fish Information Bulletin 13: 13–23.

Rubec PJ, Cruz F, Pratt V, Oellers R, Lallo F (2000) Cyanide-free, net-caught fish for the marine aquarium trade. SPC Live Reef Fish Information Bulletin 7: 28–34.

Rubec PJ, Cruz F, Pratt V, Oellers R, McCullough B, Lallo F (2001) Cyanide-free net-caught fish for the marine aquarium trade. Aquarium Sciences and Conservation 3: 37–51.

Rubec PJ, Frant M, Manipula B (2008) Methods for detection of cyanide and its metabolites in marine fish. In: Bruckner AW, Roberts G (eds.). Proceedings of the International Cyanide Detection Testing Workshop. NOAA Technical Memorandum NMFS-OPR-40, Silver Spring, MD, p. 43–63.

Ruiz G, Carlton J, Grosholz E, Hines AH (1997) Global invasions of marine and estuarine habitats by non-indigenous species: mechanisms, extent and consequences. American Zoologist 37: 621–632.

Ruiz-Carus R, Matheson RE, Roberts DE, Whitfield PE (2006) The western Pacific red lionfish, *Pterois volitans* (Scorpaenidae), in Florida: Evidence for reproduction and parasitism in the first exotic marine fish established in state waters. Biological Invasions 128: 384–390.

Ruscoe EJ (1962) Some records of large *Tridacna* specimens. Hawaiian Shell News 11: 8.

Saarman NP, Louie KD, Hamilton H (2010) Genetic differentiation across eastern Pacific oceanographic barriers in the threatened seahorse *Hippocampus ingens*. Conservation Genetics 11: 1989–2000.

Sadovy Y (1992). A preliminary assessment of the marine aquarium export trade in Puerto Rico. Proceedings of the Seventh International Coral Reef Symposium 2: 1014–1022.

Sadovy Y (2001) When being female is better. Porcupine! University of Hong Kong 23: 7–8.

Sadovy Y (2002) Death in the live reef fish trades. SPC Live Reef Fish Information Bulletin 10: 3–5.

Sadovy Y, Vincent ACJ (2002) Ecological issues and the trades in live reef fishes. In: Sale PF (ed.) Coral reef fishes: Dynamics and diversity in a complex ecosystem. Academic Press, San Diego, p. 391–420.

Sadovy Y, Mitcheson G, Rasotto MB (2001) Early development of the mandarinfish, *Synchiropus splendidus* (Callionymidae), with notes on its fishery and potential for culture. Aquarium Sciences and Conservation 3: 253–263.

Sadovy Y, Randall JE, Rasotto MB (2005) Skin structure in six dragonet species (Gobiesociformes; Callionymidae): interspecific differences in glandular cell types and mucus secretion. Journal of Fish Biology 66: 1411–1418.

Saenz-Agudelo P, Jones GP, Thorrold SR, Planes S (2009) Estimating connectivity in marine populations: an empirical evaluation of assignment tests and parentage analysis under different gene flow scenarios. Molecular Ecology 18: 1765–1776.

Saila SB, Kocic VLj, McManus JW (1992) Modelling the effects of destructive fishing practices on tropical coral reefs. Marine Ecology Progress Series 94: 51–60.

St. John J, Syers CJ (2005) Mortality of the demersal West Australian dhufish, (Richardson 1845) following catch and release: the influence of capture depth, venting and hook type. Fisheries Research 76: 106–116.

Sale PF, Eckert GJ, Ferrell DJ, Fowler AJ, Jones TA, Mapstone BD, Steel WJ (1986). Aspects of the demography of seven species of coral reef fishes. Townsville: Great Barrier Reef Marine Park Authority.

Saleem MR, Adam MS (2004) Review of the aquarium fish industry in the Maldives – 2003. Marine Research Center, Malé, Maldives, 33 p.

Saleem M, Islam F (2008) Management of the aquarium fishery in the Republic of the Maldives. Proceedings of the 11th International Coral Reef Symposium, Ft. Lauderdale, FL, 7–11 July 2008. Session 22: 1038–1042.

Samedi, Liman PD (2002) Management measures and CITES trade controls for the stony coral trade in Indonesia. In: Bruckner AW (ed.) Proceedings of the International

Workshop on the Trade in Stony Corals: development of sustainable management guidelines. NOAA Technical Memorandum NMFS-OPR-23, Silver Spring, MD, p. 64–67.

Samoilys MA (1988) Abundance and species richness of coral reef fish on the Kenyan coast: the effects of protective management and fishing. Proceedings of the Sixth International Coral Reef Symposium, Australia: 261–166.

Sano M, Shimuzu M, Nose Y (1984) Food habits of teleostean reef fishes in Okinawa Island, southern Japan. University Museum University of Tokyo Bulletin 25: 1–128.

Santini S, Polacco G (2006) Finding Nemo: Molecular phylogeny and evolution of the unusual life style of anemonefish. Gene 385: 19–27.

Santangelo G, Abbiati M (2001) Red coral: conservation and management of an over-exploited Mediterranean species. Aquatic Conservation Marine Freshwater Ecosystems 11: 253–259.

Santos GA, Schrama JW, Mamauag REP, Rombout JHWM, Verreth JAJ (2010) Chronic stress impairs performance, energy metabolism and welfare indicators in European seabass (*Dicentrarchus labrax*): The combined effects of fish crowding and water quality deterioration. Aquaculture 299: 73–80.

Scales H (2010) Advances in the ecology, biogeography and conservation of seahorses (genus *Hippocampus*). Progress in Physical Geography 34: 443–458.

Schaper A, Desel H, Ebbecke M, De Haro L, Deters M, Hentschel H, Hermanns-Clausen M, Langer C (2009) Bites and stings by exotic pets in Europe: An 11 year analysis of 404 cases from Northeastern Germany and Southeastern France. Clinical Toxicology 47: 39–43.

Schellerer W (1974). The role of mucosal blood flow in the pathogenesis of stress ulcers. Acta Hepato-Gastroenterology 21: 138–141.

Schmitt RJ, Holbrook SJ (2003) Mutualism can mediate competition and promote co-existence. Ecology Letters 6: 898–902.

Schmidt C, Kunzmann A (2005) Post-harvest mortality in the marine aquarium trade: A case study of an Indonesian export facility. SPC Live Reef Fish Information Bulletin 13: 3–12.

Schneider JA, Foighil DO (1999) Phylogeny of giant clams (Cardiidae : Tridacninae) based on partial mitochondrial 16S rDNA gene sequences. Molecular Phylogenetics and Evolution 13: 59–66.

Schofield PJ, Fuller PL (2009) *Pterois volitans/miles*. USGS nonindigenous aquatic species database, Gainesville, FL, USA. http://nas.er.usgs.gov/queries/FactSheet.asp?speciesID=963

Schofield PJ, Morris J, Akins L (2009) Field guide to nonindigenous marine fishes of Florida. Silver Spring, MD: NOAA Technical Memorandum NOS NCCOS 92 p.

Schultz ET (1986) *Pterois volitans* and *Pterolis miles*: two valid species. Copeia 1986: 686–690.

Scott A, Harrison P (2007a) Broadcast spawning of two species of sea anemone, *Entacmaea quadricolor* and *Heteractis crispa*, that host anemonefish. Invertebrate Reproduction and Development 50: 163–171.

Scott A, Harrison PL (2007b). Embryonic and larval development of the host sea anemones *Entacmaea quadricolour* and *Heteractis crispa*. Biological Bulletin 213: 110–121.

Scott A, Harrison PL (2008) Larval settlement and juvenile development of sea anemones that provide habitat for anemonefish. Marine Biology 154: 833–839.

Scott A, Harrison PL (2009) Gametogenic and reproductive cycles of the sea anemone, *Entacmaea quadricolor*. Marine Biology 156: 1659–1671.

Scott A, Malcolm HA, Damiano C, Richardson DL (2011) Long-term increases in abundance of anemonefish and their host sea anemones in an Australian marine protected area. Marine and Freshwater Research 62: 187–196.

Semmens BX, Buhle ER, Salomon AK, Pattengill-Semmens CV (2004) A hotspot of non-native marine fishes: evidence for the aquarium trade as an invasion pathway. Marine Ecology Progress Series 266: 239–244.

Shuman CS, Hodgson G, Ambrose R (2005) Population impacts of collecting sea anemones and anemonefish for the marine aquarium trade in the Philippines. Coral Reefs 24: 564–573.

Silbiger NJ, Childress MJ (2008) Interspecific variation in anemone shrimp distribution and host selection in the Florida Keys (USA): Implications for marine conservation. Bulletin of Marine Science 83: 329–345.

Silva LIB, Justino CIL, Lopes I, Pereira R, Freitas AC, Calado R, Rocha-Santos TAP, Panteleitchouk TSL, Pereira ME, Duarte AC (2011) Optical fiber based methodology for assessment of thiocyanate in seawater. Journal of Environmental Monitoring 13: 1811–1815.

Simpson SD, Yan HY, Wittenrich ML, Meekan MG (2005) Response of embryonic coral reef fishes (Pomacentridae : Amphiprion spp.) to noise. Marine Ecology Progress Series 287: 201–208.

Sin TM, Teo MM, Pkl NG, Chou LM, Khoo HW (1994) The damselfishes (Pisces – Osteichthyes, Pomacentridae) of Peninsular Malaysia and Singapore – Systematics, ecology and conservation. Hydrobiologia 285: 49–58.

Smith KF, Behrens MD, Max LM, Daszak P (2008) U.S. drowning in unidentified fishes: Scope, implications, and regulation of live fish import. Conservation Letters 1: 103–109.

Smith KF, Behrens MD, Shloegel LM, Marano N, Burgiel S, Daszak P (2009) Reducing the risks of the wildlife trade. Science 324: 594–595.

Snyder DB, Burgess GH (2007) The Indo-Pacific red lionfish, *Pterois volitans* (Pisces : Scorpaenidae), new to Bahamian ichthyofauna. Coral Reefs 26: 175–175.

Soegiarto A, Polunin NVC (1982). The marine environment of Indonesia. Report for the government of Indonesia under sponsorship of IUCN and WWF.

Sorokin YI (1973) On the feeding of some scleractinian corals with bacteria and dissolved organic matter. Limnology and Oceanography 18: 380–385.

Soto CG, Burhanuddin (1995) Clove oil as a fish anaesthetic for measuring length and weight of rabbitfish (*Siganus lineatus*). Aquaculture 136: 149–152.

Steinke D, Zemlak TS, Hebert PDN (2009) Barcoding Nemo: DNA-based identifications for the ornamental fish trade**.** PLoS ONE: 4: e6300.

Stevenson TC, Tissot BN, Diekering J (2011) Fisher behaviour influences catch productivity and selectivity in West Hawaii's aquarium fishery. ICES Journal of Marine Science 68: 813–822.

Stewart HL, Holbrook SJ, Schmitt RJ, Brooks AJ (2006) Symbiotic crabs maintain coral health by clearing sediments. Coral Reefs 25: 609–615.

Storero LP, Gonzalez RA (2008) Feeding habits of the seahorse *Hippocampus patagonicus* in San Antonio Bay (Patagonia, Argentina). Journal of the Marine Biological Association of the United Kingdom 88: 1503–1508.

Stobutzki IC, Miller MJ, Jones P, Salini JP (2001) Bycatch diversity and variability in a tropical penaeid fishery: The implications for monitoring. Fisheries Research 53: 283–301.

Strawn K (1953) Life history of the pygmy seahorse, *Hippocampus zosterae* Jordan and Gilbert, at Cedar Key, Florida. Copeia 1958: 16–22.

Suharsono (1999) Condition of coral reef resources in Indonesia. Indonesian Journal of Coastal and Marine Resources 1: 44–52.

Suharsono, Bruckner AW (2008) Evaluation of non-detriment finding for trade in stony corals from Indonesia NDF workshop case studies WG-9 Aquartic Invertebrates, Case Study 5 Stony Corals, Country- Indonesia.

Sutton DC, Garrick R (1993) Bacterial disease of cultured giant clam *Tridacna gigas* larvae. Diseases of Aquatic Organisms 16: 47–53.

Takita T (1980) Embryonic development and larvae of three dragonets. Bulletin of the Japanese Society of Science and Fisheries 46: 1–7.

Takita T (1983) Embryonic and larval development of the callionymid fish, *Callionymus calliste*. Japanese Journal of Ichthyology 29: 441–445.

Takita T, Okamoto E (1979) Spawning behavior of the two dragonets, *Callionymus flagris* and *C. richardsoni*, in the aquarium. Japanese Journal of Ichthyology 26:, 282–288.

Tan S, Zulfigar Y (2003) Status of giant clams in Malaysia. SPC *Trochus* Information Bulletin 10: 9–10.

Teo LH, Chen TW, Lee BH (1989) Packaging of the guppy, *Poecilia reticulata*, for air transport in a closed system. Aquaculture 78: 321–332.

Teo LH, Chen TW, Oei PP (1994) The uses of tris buffer, 2-phenoxyethanol and clinoptilolite in the long-distance transport of *Barbus tetrazona* Bleeker. In: Chou LM et al. (eds) Proceedings of The Third Asian Fisheries Forum, held in Singapore. Manila, Philippines: Asian Fisheries Society, p. 896–899.

Teske PR, Hamilton H, Matthee CA, Barker NP (2007) Signatures of seaway closures and founder dispersal in the phylogeny of a circumglobally distributed seahorse lineage. BMC Evolutionary Biology 7: Article 138.

Thamrongnavasawat T, Saisaeng A, Sittthaweepat N, Limviriyakul P, Woorachanant S, Patimanukasem O (2001) Survey report in Mu Ko Surin Marine National Park Area presented to UNESCO project.

Thresher RE (1984) Reproduction in reef fishes. TFH Publishing, NJ.

Thresher RE, Colin PL, Bell LJ (1989) Planktonic duration, distribution and population structure of western and central Pacific damselfishes (Pomacentridae). Copeia 1989: 420–434.

Timm J, Kochzius M (2008) Geological history and oceanography of the Indo-Malay Archipelago shape the genetic population structure in the false clown anemonefish (Amphiprion ocellaris). Molecular Ecology 17: 3999–4014.

Timm J, Figiel M, Kochzius M (2008) Contrasting patterns in species boundaries and evolution of anemonefishes (Amphiprioninae, Pomacentridae) in the centre of marine biodiversity. Molecular Phylogenetics and Evolution 49: 268–276.

Tipton K, Bell SS (1988) Foraging patterns of two syngnathid fishes: Importance of harpaticoid copepods. Marine Ecology Progress Series 47: 31–43.

Tissot BN (1999) Adaptive management of aquarium fish collecting in Hawaii. SPC Live Reef Fish Information Bulletin 6: 16–19.

Tissot BN, Hallacher LE (2003) Effects of aquarium collectors on reef fishes in Kona, Hawaii. Conservation Biology 17: 1759–1768.

Tissot BN, Walsh WJ, Hallacher LE (2004) Evaluating effectiveness of a marine protected area network in West Hawaii to increase productivity of an aquarium fishery. Pacific Science 58: 175–188.

Tissot BN, Walsh WJ, Hixon MA (2009) Hawaiian Islands marine ecosystem case study: Ecosystem- and community-based management in Hawaii. Coastal Management 37: 255–273.

Tissot BN, Best BA, Borneman EH, Bruckner AW, Cooper CH, D'Agnes H, Fitzgerald TP, Leland A, Lieberman S, Amos AM, Sumaila R, Telecky TM, McGilvray F, Plankis BJ, Rhyne AL, Roberts GG, Starkhouse B, Stevenson TC (2010) How US ocean policy and market power can reform the coral reef wildlife trade. Marine Policy 34: 1385–1388.

Todd PA, Lee JH, Chou LM (2009) Polymorphism and crypsis in the boring giant clam (*Tridacna crocea*): potential strategies against visual predators. Hydrobiologia 635: 37–43.

Trench RK (1974) Nutritional potentials in *Zoanthus sociathus* (Coelenterata, Anthozoa). Helgoländ Wiss Meer 26: 174–216.

Trench RK, Wethey DS, Porter JW (1981) Observations on the Symbiosis with zooxanthellae among the Tridacnidae (Mollusca, Bivalvia). Biological Bulletin 161: 180–198.

Tsounis G, Rossi S, Gili JM, Arntz WE (2006) Population structure of an exploited benthic cnidarian: the case study of red coral (*Corallium rubrum* L.). Marine Biology 149: 1059–1070.

Tsounis G, Rossi S, Grigg R, Santangelo G, Bramanti L, Gili J-M (2010) The exploitation and conservation of precious corals. Oceanography and Marine Biology: An Annual Review 48: 161–212.

Tuan VS (2002) Documentation of coral reefs and coral trade in Vietnam. In: Bruckner AW (ed.) Proceedings of the International Workshop on the Trade in Stony Corals: development of sustainable management guidelines. NOAA Technical Memorandum NMFS-OPR-23, Silver Spring, MD, p. 101–105.

Tyler WA (1995) The adaptive significance of colonial nesting in a coral-reef fish. Animal Behavior 49: 949–966.

United Nations Environment Programme World Conservation Monitoring Centre (2008) Consultation process on monitoring of international trade in ornamental fish, background information prepared for European Commission Directorate General E – Environment ENV.E.2. – Development and Environment. 43 p.

United Nations World Tourism Organization (2010) Compendium of tourism statistics, Data 2004 – 2008. Madrid, Spain: World Tourism Organization.

USAID Democratic Reform Support Program (USAID DRSP) (2006) Decentralization 2006: Stock taking on Indonesia's recent decentralization reforms. Jakarta: The Donor Working Group on Decentralization. 54 p. http://pdf.usaid.gov/pdf_docs/PNADH311.pdf

U.S. Commission on Ocean Policy (2004) An ocean blueprint for the 21st Century final report. Washington, DC: U.S. Commission on Ocean Policy.

Uyarra MC, Côté IM (2007) The quest for cryptic creatures: Impacts of species-focused recreational diving on corals. Biological Conservation 136: 77–84.

Vallejo B (1997a) Survey and review of the Philippine marine aquarium fish industry. Sea Wind 11: 2–16.

Vallejo B (1997b) An overview of the Philippine marine aquarium industry. Proc. 8th Int. Coral Reef Sym 2: 1981–1986.

Van Wassenbergh S, Roos G, Ferry L (2011) An adaptive explanation for the horse-like shape of seahorses. Nature Communications 2: Article #164.

Vagelli A (1999) The reproductive biology and early ontogeny of the mouthbrooding Banggai cardinalfish, *Pterapogon kauderni* (Perciformes, Apogonidae). Environmental Biology of Fishes 56: 79–92.

Vagelli AA (2004a) Ontogenetic shift in habitat preference by *Pterapogon kauderni*, a shallow water coral reef apogonid, with direct development. Copeia 2004: 364–369.

Vagelli AA (2004b) Significant increase in survival of captive-bred juvenile Banggai cardinalfish *Pterapogon kauderni* with an essential fatty acid-enriched diet. Journal of the World Aquaculture Society 35: 61–69.

Vagelli AA (2005) Reproductive Biology, Geographic Distribution and Ecology of the Banggai Cardinalfish *Pterapogon kauderni* Koumans, 1933 (Perciformes, Apogonidae), with Considerations on the Conservation Status of this Species on its Natural Habitat. Ph.D. Dissertation, University of Buenos Aires. 276 p.

Vagelli AA (2008). The unfortunate journey of *Pterapogon kauderni*: A remarkable apogonid endangered by the international ornamental fish trade, and its case in CITES. SPC Live Reef Fish Information Bulletin 18: 17–28.

Vagelli AA, Erdmann MV (2002) First comprehensive ecological survey of the Banggai cardinalfish, *Pterapogon kauderni*. Environmental Biology of Fishes 63: 1–8.

Vagelli AA, Volpedo AV (2004) Reproductive ecology of *Pterapogon kauderni*, an endemic apogonid from Indonesia with direct development. Environmental Biology of Fishes 70: 235–245.

Vagelli, Burford M, Bernardi G (2009) Fine scale dispersal in Banggai Cardinalfish, *Pterapogon kaudemi*, a coral reef species lacking a pelagic larval phase. Marine Genomics 1: 129–134.

Valdez-Moreno M, Quintal-Lizama C, Gómez-Lozano R, García-Rivas MdC (2012) Monitoring an alien invasion: DNA barcoding and the identification of lionfish and their prey on coral reefs of the Mexican Caribbean. PLoS ONE 7: e36636.

Vaz MCM, Rocha-Santos TAP, Rocha RJM, Lopes I, Pereira R, Duarte AC, Rubec PJ, Calado R (2012) Excreted thiocyanate detects live reef fishes illegally collected using cyanide – a non-invasive and non-destructive testing approach. PLoS ONE 7: e35355.

Venn AA, Loram JE, Douglas AE (2008) Photosynthetic symbioses in animals. Journal of Experimental Botany 59: 1069–1080.

Verlaque M, Fritayre P (1994) Mediterranean algal communities are changing in face of the invasive alga *Caulerpa taxifolia*. Oceanologica Acta 17: 659–672.

Veron JEN (1995) Corals in space and time. The biogeography and evolution of the Scleractinia. UNSW Press, Sydney.

Veron JEN (2000) Corals of the world, volumes 1–3. Australian Institute of Marine Science, Townsville, Australia. 1381 p.

Vetrano SJ, Lebowitz JB, Marcus S (2002) Lionfish envenomation. Journal of Emergency Medicine 23: 379–382.

Vincent ACJ (1990) Reproductive ecology of seahorses. Ph.D.Thesis. University of Cambridge.

Vincent ACJ (1996) The international trade in seahorses. TRAFFIC International, Cambridge, UK.

Vincent ACJ, Sadovy Y (1998) Reproductive ecology in the conservation and management of fishes. In: Caro T (ed.) Behavioural ecology and conservation biology. Oxford University Press, New York, New York, p. 209–245.

Vincent ACJ, Sadler LM (1995) Faithful pair bonds in wild sea horses, *Hippocampus whitei*. Animal Behavior 50: 1557–1569.

Vincent ACJ, Koldewey HJ (2006). An uncertain future for seahorse aquaculture in conservation and economic contexts. In Proceedings of the Regional Technical Consultation on Stock Enhancement of Species Under International Concern. Southeast Asian Fisheries Development Center (Primavera JH Ed.) p. 71–84.

Vincent ACJ, Marsden AD, Evans KL, Sadler LM (2004) Temporal and spatial opportunities for polygamy in a monogamous seahorse, *Hippocampus whitei*. Behaviour 141: 141–156.

Vincent ACJ, Evans KL, Marsden AD (2005) Home range behaviour of the monogamous Australian seahorse, *Hippocampus whitei*. Environmental Biology of Fishes 72: 1–12.

Vincent ACJ, Meeuwiga JJ, Pajaro MG, Perante NC (2007) Characterizing a small-scale, data-poor, artisanal fishery: Seahorses in the central Philippines. Fisheries Research 86: 207–215.

Vincent ACJ, Foster SJ, Koldewey HJ (2011) Conservation and management of seahorses and other Syngnathidae. Journal of Fish Biology 78: 1681–1724.

Vunisea A (2003) Coral harvesting and its impact on local fisheries in Fiji. SPC Women in Fisheries Information Bulletin 12: 17–20.

Wabnitz C, Taylor M, Green E, Razak T (2003) From Ocean to Aquarium: The global trade in marine ornamental species. UNEP-WCMC, Cambridge, United Kingdom. 65 p.

Waldie PA, Blomberg SP, Cheney KL, Goldizen AW, Grutter AS (2011) Long-term effects of the cleaner fish *Labroides dimidiatus* on coral reef fish communities. PLoS ONE 6: e21201.

Walsh WJ (1978) Aquarium fish collecting: promise or problem? Working Papers. Volume 34. Hawaii Sea Grant College Program. University of Hawaii, Honolulu. p. 8–12.
Walsh WJ (1984) Aspects of nocturnal shelter, habitat space, and juvenile recruitment in Hawaiian coral reef fishes. PhD Dissertation, Zoology Department, University of Hawaii at Manoa, Honolulu. p. 475.
Walsh WJ (1985) Reef fish community dynamics on small artificial reefs: the influence of isolation habitat structure, and biogeography. Bulletin of Marine Science 36: 357–376.
Walsh WJ (1987) Patterns of recruitment and spawning in Hawaiian reef fishes. Environmental Biology of Fishes 18: 257–276.
Walsh WJ, Cotton SP, Dierking J (2003) The commercial marine aquarium fishery in Hawaii 1976–2003. In: Friedlander AM (ed.) Status of Hawaii’s coastal fisheries in the new millennium: Proceedings of a symposium. Honolulu, HI, American Fisheries Society, Hawaii Chapter, p. 132–159.
Walsh WJ, Cotton SP, Dierking J, Williams ID (2004) The commercial marine aquarium fishery in Hawaii 1976–2003. In: Friedlander AM (ed.) Status of Hawaii’s coastal fisheries in the new millennium, revised. Proceedings of the 2001 fisheries Symposium sponsored by the American Fisheries Society, Hawaii Chapter. Honolulu, HI, p. 129–156.
Walster C (2008) The welfare of ornamental fishes. In: Branson EJ (ed.) Fish Welfare. Blackwell Publishing, Ames, Iowa, p. 271–290.
Warner RR (1982) Mating systems, sex change and sexual demography in the rainbow wrasse, *Thalassoma lucasanum*. Copeia 1982: 653–661.
Watanabe T, Suzuki A, Kawahata H, Kan H, Ogawa S (2004) A 60-year isotopic record from a mid-Holocene fossil giant clam (*Tridacna gigas*) in the Ryukyu Islands: physiological and paleoclimatic implications. Palaeogeography Palaeoclimatology Palaeoecology 212: 343–354.
Weber ES, Waltzek TB, Young DA, Twitchell EL, Gates AE, Vagelli A, Risatti GR, Hedrick RP, Frasca S (2009) Systemic iridovirus infection in the Banggai cardinalfish (*Pterapogon kauderni* Koumans 1933). Journal of Veterinary Diagnostic Investigation 21: 306–320.
Weigle SM, Smith LD, Carlton JT, Pederson J (2005) Assessing the risk of introducing exotic species via the live marine species trade. Conservation Biology 19: 213–223.
Wellington GM, Victor BC (1989) Planktonic larval duration of 100 species of Pacific and Atlantic Damselfishes (Pomacentridae). Marine Biology 101: 557–567.
Wells S (1997) Giant clams: Status, trade and mariculture, and the role of CITES in management. IUCN/SC Wildlife Trade Programme, Gland, Switzerland and Cambridge, UK. 77 p.
Weng K, Guilbeaux M (2000) Marine resources of Helen Reef in the Year 2000. Community Conservation Network, Honolulu, Hawaii, p. 6.
Whiteman EA, Côté IM (2004) Monogamy in marine fishes. Biological Reviews 79: 351–375.

Whitfield PE, Gardner T, Vives SP, Gilligan MR, Courtenay WR, Ray GC, Hare JA (2002) Biological invasion of the Indo-Pacific lionfish *Pterois volitans* along the Atlantic coast of North America. Marine Ecology Progress Series 235: 289–297.

Wicaksono A, Putrawidjaja M, Imran Amin M (2002) Overview of Indonesian coral trade: importance to coastal communities, health and safety issues, users' conflicts and illegal trade concerns. In: Bruckner AW (ed.) Proceedings of the International Workshop on the Trade in Stony Corals: development of sustainable management guidelines. NOAA Technical Memorandum NMFS-OPR-23, Silver Spring, MD, p. 68–72.

Wilcove DS, Rothstein D, Dubow A, Phillips E, Losos E (1998) Quantifying threats to imperiled species in the United States. BioScience 48: 607–615.

Wilde GR (2009) Does venting promote survival of released fish? Fisheries 34: 20–28.

Wilhelmsson D, Haputhantri SSK, Rajasuriya A, Vidanage SP (2002) Monitoring the trends of marine ornamental fish collection in Sri Lanka. In Linden O, Souter D, Wilhelmsson D, Obura D (eds.). Coral degradation in the Indian Ocean: Status Report 2002. CORDIO, Department of Biology and Environmental Science, University of Kalmar, Kalmar, Sweden. p. 158–166.

Wilkerson J (1996) C-Quest Hatchery – Innovations in captive ornamental marine fish culturing. Freshwater and Marine Aquarium 19: 123.

Wilkinson C (2008) Status of the coral reefs of the world: 2008. Townsville, Australia. Global Coral Reef Monitoring Network and Reef and Rainforest Research Centre.

Williams ID, Walsh WJ, Miyasaka A, Freelander AM (2006) Effects of rotational closure on coral reef fishes in Waikiki-Diamond Head Fishery Management Area, Oahu, Hawaii. Marine Ecology Progress Series 310: 139–149.

Williams ID, Walsh WJ, Claisse JT, Tissot BN, Stamoulis KA (2009) Impacts of a Hawaiian marine protected area network on the abundance and fishery sustainability of the yellow tang, *Zebrasoma flavescens*. Biological Conservation 142: 1066–1073.

Williams WI, Polunin PN (2001) Large-scale associations between macroalgal cover and grazer biomass on mid-depth reefs in the Caribbean. Coral Reefs 19: 358–366.

Williams N (2010) Major lionfish hunt launched. Current Biology 20: R1005–R1006.

Willis BL, van Oppen MJH, Miller DJ, Vollmer SV, Ayre DJ (2006) The role of hybridization in the evolution of reef corals. Annual Review of Ecology, Evolution, and Systematics 37: 489–517.

Wishnik M, Lane MD (1969) Inhibition of ribulose diphosphate carboxylase by cyanide inactive ternary complex of enzyme ribulose diphosphate and cyanide. Journal of Biological Chemistry 244: 55–59.

Wood EM (1985). Exploitation of coral reef fishes for the aquarium trade. Report to the Marine Conservation Society. 121 p.

Wood EM (2001a) Collection of reef fish for aquaria: conservation issues and management needs. In: Roberts CM, Hawkins JP, McAllister DE (eds.) Coral reef fish status report. Species Survival Commission, IUCN, Gland, Switzerland and Cambridge, UK.

Wood E (2001b) Collection of coral reef fish for aquaria: Global Trade, conservation issues and management strategies. Marine Conservation Society, Ross-on-Wye, UK. 56 p.

Wood E, Rajasuriya A (1999) Sri Lanka marine aquarium fishery conservation and management issues. Marine Conservation Society and National Aquatic Resources Agency. 11 p.

Wood EM, Malsch K, Miller J (2012) International trade in hard corals: review of management, sustainability and trends. Proceedings of the 12th International Coral Reef Symposium, Cairns, Australia, 9–13 July 2012, 19C Trade in coral reef wildlife.

Wright KA, Woods CMC, Gray BE, Lokman PM (2007) Recovery from acute, chronic and transport stress in the pot-bellied seahorse *Hippocampus abdominalis*. Journal of Fish Biology 70: 1447–1457.

Yasir I, Qin JG (2007) Embryology and early ontogeny of an anemonefish *Amphiprion ocellaris*. Journal of the Marine Biological Association of the United Kingdom 87: 1025–1033.

Zamzow JP (1998) Cleaning symbiosis between Hawaiian reef fishes and green sea turtles, *Chelonia mydas.* Proceedings of the Eighteenth International Sea Turtle Symposium. p. 235–237.

Zann LP (1994) The status of coral reefs in south western Pacific Islands. Marine Pollution Bulletin 29: 52–61.

Zeller D, Booth S, Pauly D (2005) Reconstruction of coral reef- and bottom fisheries catches for U.S. flag areas in the western Pacific. Western Pacific Regional Fishery Management Council Report, Honolulu, HI. 199 p.

Zhang D, Zhang YH, Lin JD, Lin QA (2010) Growth and survival of juvenile lined seahorse, *Hippocampus erectus* (Perry), at different stocking densities. Aquaculture Research 42: 9–13.