Threats and Stressors to U.S. Coral Reef Ecosystems

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Human activity is commonly identified as a major contributor to the observed global deterioration of coral reef ecosystem health, with loss of live coral cover, declining species diversity, and reduced abundance reported in many areas (NOAA, 2002a; Wilkinson, 2002; Turgeon et al., 2002). Degradation in the structure and functioning of coral reef ecosystems results in a concomitant loss in the intrinsic value of the ecological system, as well as a significant loss in the provision of goods and services for society. Approximately 8% of the global population live within 100 km of a coral reef (Bryant et al., 1998) and many local communities and national economies are directly dependent on coral reef ecosystems for tourism revenue, food, and coastal protection (Spurgeon, 1992). As such, human pressures can be intense, and developing strategies to mitigate stressors is a complex task.

Shallow-water coral reef ecosystems experience a wide range of physical, biological, and chemical threats and stressors, which stem from both anthropogenic and natural causes. Threats are defined as environmental trends with potentially negative impacts. Stressors are defined as factors or processes that harm ecosystem components, causing lethal or sublethal negative effects. Categories of stressors include chemical (e.g., pollution), physical (e.g., extreme events), and biological (e.g., invasive species) stressors, and the relationship between key stressors and the threats discussed in this document are listed in Table 3.1. The relative importance of each threat varies substantially among jurisdictions and individual reefs.

Table 3.1. This table is a crosswalk between the threats identified in "A National Coral Reef Action Strategy" (NOAA, 2002a) and the stressors identified by the National Science and Technology Council's Committee on Environmental and Natural Resources. Source: CENR, 2001.

STRESSORS	POLLUTION	INVASIVE SPECIES	EXTREME EVENTS	RESOURCE AND LAND USE	CLIMATE CHANGE
Climate Change and Bleaching					X
Diseases	X				
Tropical Storms			X		
Coastal Development and Runoff	X			X	
Coastal Pollution	X				
Tourism and Recreation				X	
Fishing				X	
Trade in Coral and Live Reef Species				X	
Ships, Boats and Groundings				X	
Marine Debris	X				
Aquatic Invasive Species		X			
Security Training Activities				X	
Offshore Oil and Gas Exploration				X	

Multiple Stressors

The occurrence of multiple sequential stressors and the synergistic interaction between stressors can be especially detrimental to coral reef ecosystems. For example, in many parts of the Caribbean, the compounding effects of eutrophication, decline of key herbivores from disease and overfishing, and impacts of hurricanes and coral bleaching have likely led to the observed shifts in community structure from coral-dominated to macroalgal-dominated reefs (Hughes, 1994; McManus et al., 2000). Generally, the effects of multiple stressors are poorly understood, making it difficult or even inappropriate to assign a single cause to local or regional

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5 NOAA Oceans and Coasts, Special Projects Office

widespread decline. The challenge now is to understand the complex interactions among stressors by refining existing techniques and developing new multidisciplinary approaches aimed at detailing mechanisms and predicting effects at multiple spatial and temporal scales.

Determining how humans utilize coral reef ecosystems and estimating the social and economic costs and benefits of those uses are key steps for resource managers. Techniques such as causal chain analysis (e.g., in Belausteguigoitia, 2004) may provide a useful approach for modeling and communicating the many significant cause-effect linkages between human systems and coral reef ecosystems.

Climate Change and Coral Bleaching

Climate change refers to any change in climate over time, whether due to natural variability or human activity (IPCC, 2001). Over the 20th century, mean near-surface air temperature over land and mean sea surface temperature (SST) increased 0.6 ± 0.2°C, with the 1990s being the warmest decade and 1998 being the warmest year since 1861 when instrumental records began (IPCC, 2001; Figure 3.1).

Most of the observed warming over the last 50 years may have resulted from an increase in concentrations of greenhouse gases such as carbon dioxide (CO_2) and methane (CH_4) in the atmosphere (IPCC, 2001; NRC, 2001). The atmospheric concentration of CO_2 has increased by 31% since the beginning of the industrial revolution, and represents a level that has not been exceeded in at least the last 420,000 years (Petit et al., 1999), and probably not exceeded in over 24 million years (Pearson and Palmer, 2000). The rate of increase of CO_2 concentration has been about 0.4% per year over the last two decades (IPCC, 2001). Such increases have been shown to decrease the calcium carbonate ($CaCO_3$) saturation state of seawater and the calcification rates of corals (Kleypas et al., 1999; Feely et al., 2004). In combination with potentially more frequent bleaching episodes, reduced calcification could reduce the energy that a coral would otherwise apply to reproduction and thereby impede a reef's ability to keep pace with sea level rise (IPCC, 2001) or recover from other potential impacts of climate change.

Elevated water temperatures cause corals to bleach, a process that is characterized by the loss of zooxanthellae (a symbiotic alga) from coral tissues. Increased ultraviolet irradiance, typically from unusually calm, clear waters, may aggravate the impact of increased temperatures (Lesser and Lewis, 1996). Although corals may recover from brief episodes of bleaching, if ocean temperatures warm too much or remain high for an extended period, bleached corals often will die. Several correlative field studies show a close association between warmer than normal conditions (at least 1°C higher than the annual maximum) and the incidence of bleaching (Hoegh-Guldberg, 1999). In 1997-1998, an estimated 16% of the world's coral reefs were seriously damaged in a global coral bleaching event associated with high SST

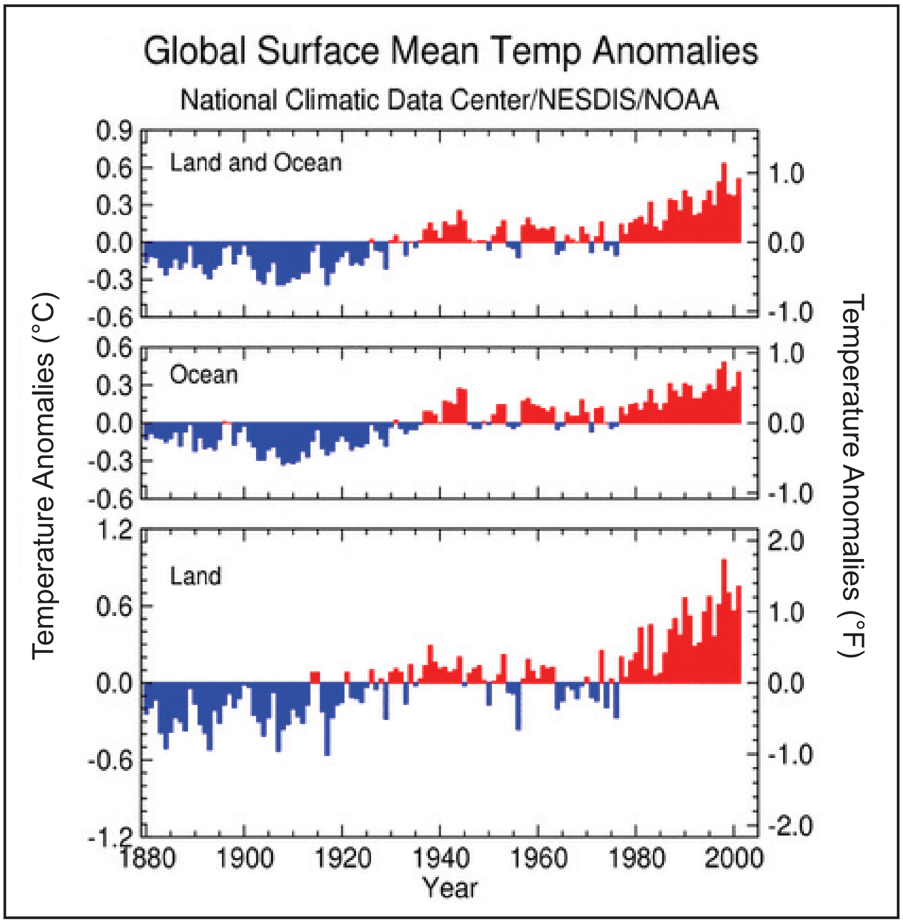

Figure 3.1. Mean global temperature anomalies over the period 1880-2001. Zero line represents the long term mean temperature throughout the period, while red and blue bars indicate annual departures from that mean. Source: NOAA's National Climactic Data Center.

which was apparently enhanced by an extreme El Niño event (Wilkinson, 1998). A U.S. Department of State report to the U.S. Coral Reef Task Force (USCRTF; Pomerance, 1999) concluded that the severity and extent of the 1998 event cannot be explained by El Niño alone, and that the "...geographic extent, increasing frequency, and regional severity of mass bleaching events are likely a consequence of a steadily rising baseline of marine temperatures..."

Several bleaching events in Florida, the U.S. Caribbean, and the U.S. Pacific have been associated with elevated SST events during the 1980s and 1990s, and especially in 1997-1998. The occurrence of bleaching is highly variable in both time and space, but generally affects shallow-water reefs with reduced water circulation. In U.S. waters, substantial bleaching has been observed on shallow reefs off the coasts of Florida, the Commonwealth of the Northern Mariana Islands (CNMI), Palmyra Atoll (PRIAs), and portions of the Northwestern Hawaiian Islands (NWHI), and recent data suggest that elevated SST is still a significant threat to coral reefs in the U.S. Caribbean (Nemeth and Slakek-Nowlis, 2001). Palau suffered the worst coral bleaching mortality of any U.S. associated region during the 1997-1998 global bleaching event (Wilkinson, 2000). During a 2002 summertime warm water event in the higher latitudes of the mid-Pacific, Midway Atoll (NWHI) experienced unprecedented bleaching, including considerable mortality (Liu et al., 2004). Mass bleaching episodes are predicted to reoccur in the future with increasing frequency (IPCC, 2001).

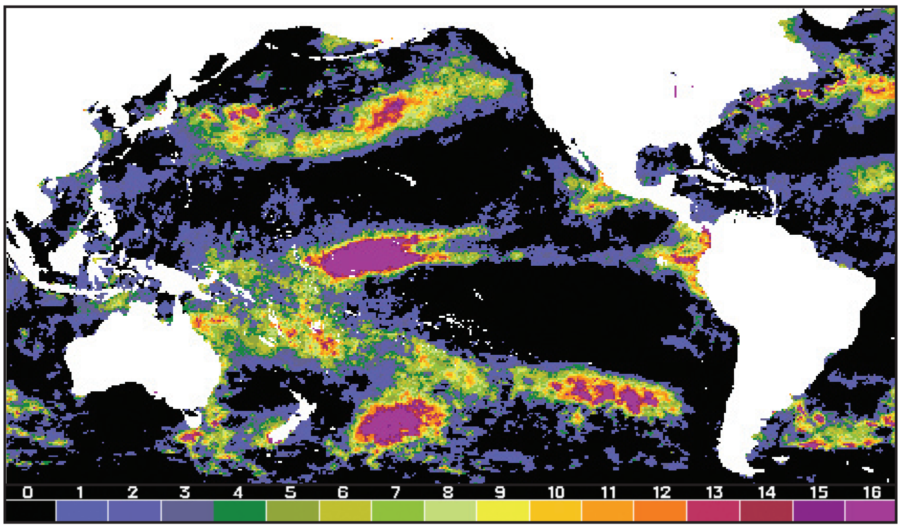

Figure 3.2. 2002 Maximum annual DHW values indicate locations that experienced significant thermal stress, which has been shown to be highly correllated with coral bleaching. Values above 4 represent areas that are likely to experience bleaching, while values above 8 represent areas that are likely to experience significant bleaching with widespread mortality. Source: NOAA's Coral Reef Watch Program.

Coral reef ecosystem managers and stakeholders consistently use one particular satellite-derived index–the Degree Heating Week (DHW)–to gauge accumulated thermal stress on reef ecosystems. The DHW, which was developed by scientists in the National Oceanic and Atmospheric Administration's (NOAA) Coral Reef Watch (CRW) Program, represents the accumulated temperature stress for each 50 x 50 km^2 pixel during the preceding 12-week period as compared to the baseline value calculated for that pixel. The unique baseline value, roughly equal to the expected annual maximum temperature, was empirically determined for each of the 250 km^2 pixels shown in Figure 3.2. To calculate the DHW, temperature deviations (in degrees Celsius) above this baseline are multiplied by the duration of the elevated temperature event (in weeks). For example, if there is a sustained SST of 1°C above the threshold for one week , during a 12-week period, the DHW value will be one; if SST is 2°C above the threshold for three weeks, the DHW value will be six. Figure 3.2 illustrates the distribution of the maximum DHW values for each pixel for 2002.

In-situ observations show that widespread bleaching is most likely to occur at locations where DHW≥4; significant bleaching with widespread mortality is expected where the DHW >8. Table 3.2 shows the maximum annual DHW value in the 14 U.S. jurisdictions with coral reefs for 2001-2003. The DHW values are color-coded to reflect the intensity of accumulated thermal stress [Blue, DHW=0; Green, DHW <4; Orange, 4≤ DHW ≤8; Red, DHW >8]. If a thermal stress event spans two calendar years (e.g., November-January), then the maximum DHW for each of those years may occur during that single event. This is most likely to occur at reefs located near the equator. Such occurrences are shown in Table 3.2 as DHW values enclosed in a grey box. The CRW Program utilizes satellite and *in situ* tools for near real-time, hindcast, and long-term monitoring, modeling, and reporting of environmental conditions that affect domestic and foreign coral reef ecosystems. A full list of the CRW Program's operational products can be found on-line at http://coralreefwatch.noaa.gov (Accessed 2/16/05).

Table 3.2. Maximum annual DHWs for each of the 14 jurisdictions for 2001-2003. The DHW values are color-coded to reflect the intensity of accumulated thermal stress [Blue, DHW=0; Green, DHW<4; Orange, 4≤DHW≤8; Red, DHW>8]. If a thermal stress event spans two calendar years (e.g., November-January), then the maximum DHW for each of those years may occur during that single event. Such occurrences are shown by enclosing the DHW values in a grey box.

JURISDICTION	LOCATION	2001	2002	2003
USVI		Green	Green	Orange
Puerto Rico		Green	Green	Orange
Navassa		Green	Blue	Orange
Florida		Green	Green	Orange
Flower Garden Banks		Blue	Green	Blue
Hawaii	Hawaii	Blue	Blue	Blue
	Oahu	Blue	Blue	Green
	Kauai	Blue	Blue	Blue
Northwestern Hawaiian Islands	Nihoa	Blue	Blue	Blue
	French Frigate Shoals	Blue	Blue	Blue
	Maro Reef	Blue	Blue	Blue
	Lisianski	Blue	Green	Blue
	Midway	Blue	Orange	Blue
	Kure	Green	Red	Blue
American Samoa	Tutuila	Green	Green	Orange
	Rose Atoll	Blue	Green	Orange
USPRIAs	Johnson	Blue	Blue	Blue
	Palmyra	Green	Red (grey box)	Orange (grey box)
USPRIAs (cont.)	Kingman	Green	Red (grey box)	Orange (grey box)
	Baker	Green	Red (grey box)	Red (grey box)
	Wake	Blue	Blue	Blue
	Jarvis	Green	Red (grey box)	Red (grey box)
	Howland	Green	Red (grey box)	Red (grey box)
Marshall Islands	Bikini	Green	Green	Green
	Kwajalein	Green	Orange	Green
	Majuro	Blue	Blue	Green
Federated States of Micronesia	Yap	Green (grey box)	Green (grey box)	Green
	Chuuk	Green (grey box)	Green (grey box)	Green
	Pohnpei	Orange (grey box)	Green (grey box)	Green
	Kosrae	Green	Green	Orange
CNMI	Asuncion	Red	Green	Orange
	Agrihan	Red	Green	Orange
	Pagan	Red	Blue	Green
	Saipan	Green	Green	Green
Guam		Green	Blue	Green
Palau		Green	Green	Green

Diseases in Coral Reef Ecosystems

In the past two decades, there has been a worldwide increase in the reporting of diseases affecting marine organisms, with the Caribbean Basin emerging as a hot spot (Harvell et al., 1999). The first documented coral reef epizootic was the mass mortality of the keystone herbivore, *Diadema antillarum*, which was caused by an unknown waterborne pathogen (Figure 3.3). This disease spread throughout the Caribbean between 1982 and 1983, moving with Caribbean oceanic currents and causing the loss of up to 90-95% of the *Diadema* population (Lessios et al., 1984). Mass mortalities of *Diadema* have contributed to phase-shifts from coral- to algal-dominated reefs in many locations, and the recovery of urchin populations has been slow. Another Caribbean-wide epizootic observed during the 1980s was attributed to a fungal infection in *Thalassia testudinum* seagrasses. In Florida Bay, an estimated 4,000 ha of seagrasses were lost and severe declines were observed across an additional 23,000 ha (Roblee et al., 1991). During one of the best documented of coral disease outbreaks which occurred in the 1980s, two of the dominant reef-building coral species on shallow western Atlantic reefs (*Acropora palmata* and *A. cervicornis*) were virtually eradicated by white-band disease (Aronson and Precht, 2001). The frequency and severity of outbreaks of common as well as newly emerging diseases may increase with changing environmental conditions such as a rise in SST and anthropogenic impacts that: 1) increase the prevalence and virulence of pathogens; 2) facilitate invasions of new pathogens from terrestrial or aerial sources; and 3) reduce host resistance and resilience, thereby facilitating pathogen transmission and infection (Sutherland et al., 2004).

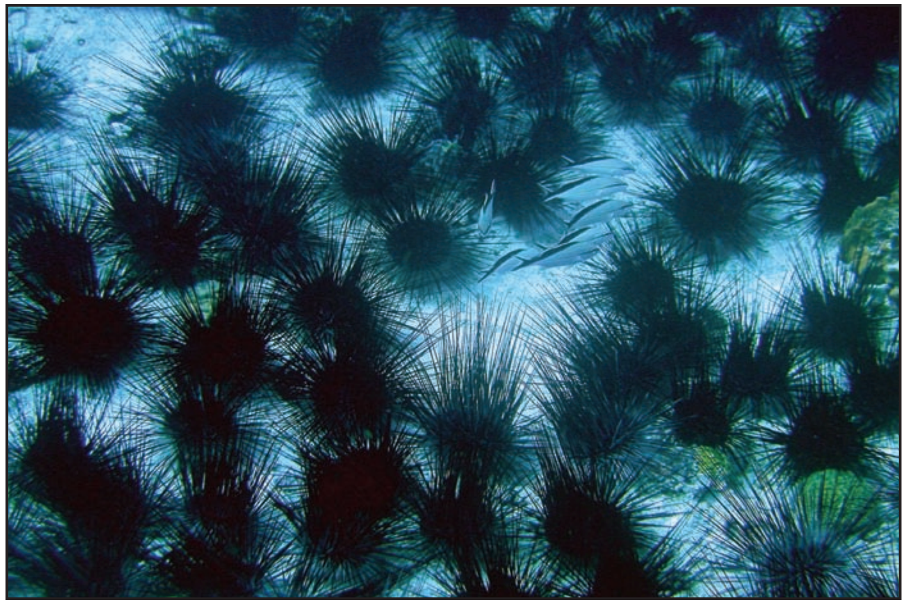

Figure 3.3. Coral disease and mortality from numerous pathogens have been reported with increased frequency since the 1970s. Disease in other ecosystem organisms can also result in cascading effects that can disrupt the entire system. Scientists believe that ~90% of the Caribbean population of *Diadema antillarum*, an important herbivore, was killed by disease in the late 1980s, and the subsequent reduction in grazing pressure allowed for algal overgrowth on many reefs. Populations are beginning to rebound as shown in this photo taken in St. Croix in October 2004. Photo: R. Clark.

Since the early 1990s, scientists have documented a rapid emergence of diseases among corals, with increases in the number of diseases reported, coral species affected, geographic extent, prevalence and incidence, and rates of associated coral mortality (Richardson, 1998; Harvell et al., 1999; Knowlton, 2001; Sutherland et al., 2004). A survey of the coral disease literature conducted by Green and Bruckner (2000) described 29 differently named diseases on 102 scleractinian coral species. At least 12 new syndromes have been reported in recent years, with a dramatic increase in observations from the Indo-Pacific. More than 150 scleractinian, gorgonian, and hydrozoan zooxanthellate species are now known to be susceptible to diseases (Sutherland et al., 2004). Despite an increase in coral disease research, the understanding of the causative agents, host-pathogen interactions, and impacts on host populations and associated communities is still very limited (Richardson, 1998; Harvell et al., 1999; Sutherland et al., 2004). For instance, microbial pathogens have been isolated, identified, and defined as the causative agent in only five diseases; while several other putative pathogens have been reported, it is unclear whether these are the cause or merely opportunistic infections (Bythell et al., 2002; Sutherland et al., 2004).

Diseases directly and indirectly alter reef community structure and function, and are considered to be playing an increasingly important role in regulating coral population size, diversity, and demographic characteristics (Porter and Tougas, 2001; Aronson and Precht, 2001; Bruckner, 2004). For example, the Caribbean-wide loss of Acroporid corals, the two dominant space occupants and most important framework builders in reef crest and forereef habitats, is the leading cause of the decline in coral cover in the Caribbean reported during the 1980s and 1990s (Richardson and Aronson, 2002). Coring studies from Belize and other locations revealed that mass mortalities at this scale had not occurred in at least the previous 3,000-4,000 years (Aronson et al., 2004). More recently, *Montastraea annularis* complex populations are experiencing significant declines as a result of multiple diseases including black-band disease, yellow-band disease, and white plague (Santavy et al., 1999; Kuta and Richardson, 2002; Gill-Agudelo and Garzon-Ferriera, 2001; Richardson and Aronson, 2002; Bruckner and Bruckner, 2003, 2004).

Understanding the relationships between coral health and environmental parameters is of key importance in the study of coral disease (Harvell et al., 1999; Green and Bruckner, 2000; Kuta and Richardson, 2002). Environmental stressors, including those associated with degraded water quality and climate change, are often cited as potential factors causing coral mortality, yet rarely have studies adequately identified causal linkages to specific environmental stressors (Woodley et al., 2003). In addition, human activity may enhance the global transport of pathogens, such as *Aspergillus sydowii* (a fungus of terrestrial origin) that causes infection and mortality in sea fans and other gorgonians, and is postulated to have entered the marine environment via terrestrial runoff or clouds of dust from West Africa (Harvell et al., 1999; Richardson and Aronson, 2002). White pox, a disease only known to affect *Acropora palmata* in Florida, is caused by a common fecal enterobacterium *Serratia marcescens*, which may enter the marine environment via sewage discharge (Patterson et al., 2002). Other diseases are thought to be caused by known microorganisms that have changed hosts or exhibited increased virulence in response to environmental stresses and reduced resistance of the host coral (Santavy and Peters, 1997; Harvell et al., 1999; Sutherland et al., 2004). At least four coral diseases (black-band disease, white plague, dark-spots disease, and Aspergillosis) are associated with high water temperatures (Kuta and Richardson, 1996; Bruckner et al., 1997; Richardson et al., 1998; Gill-Agudelo and Garzon-Ferriera, 2001; Alker et al., 2001). Nutrient input, sedimentation, and runoff have also been implicated as potential contributing factors in the initiation and elevated virulence of a disease, although few quantitative data have been published (Bruckner et al., 1997; Harvell et al., 1999; Kim and Harvell, 2001; Richardson and Aronson, 2002).

It appears that the ability of corals and other organisms to withstand infection has been compromised by climate change, eutrophication, sedimentation (Rogers, 1990), and other human-induced ecosystem perturbations (Knowlton, 2001). The vulnerability of tropical coral reef ecosystems is related to the fact that many warm water corals grow slowly and persist only within a narrow range of light, temperature, dissolved oxygen and salinity fluctuations, and, in an evolutionary sense, they are thought to have a limited ability to recover from disease (Knowlton, 2001). However, the relative importance of anthropogenic influences is still unclear, especially since disease outbreaks are being reported with increasing frequency on reefs that exist in areas relatively far from the direct effects of human activity (Bruckner and Bruckner, 2004).

A decline in the health of many coral reefs worldwide has created an urgent need for multidisciplinary studies of coral health and disease, with emphases on coral physiology, biology, and disease etiology, including mechanisms of resistance and susceptibility to disease, factors affecting the transmission, spread and virulence of pathogens, and relationships between environmental factors and disease. By better understanding causative agents and factors responsible for the emergence and proliferation of diseases, scientists will be able to contribute to the development of strategies that can be used by resource managers to mitigate disease impacts.

Tropical Storms

Most coral reef environments are found in tropical climates and periodically experience cyclonic storm events. Cyclonic storms are an important process in the structure and dynamics of coral reef ecosystems (Hughes and Connell, 1999). They are classified as “pulse disturbances” since they are typically intense and of relatively short duration, yet are a powerful mechanism for change and can dramatically disrupt ecosystems, communities, population structure, resource availability, and the physical environment (Pickett and White, 1985). Coral reefs, however, are often located in dynamic regions of the ocean and have clearly shown resilience to historical bouts of disturbance. In fact, such disturbances are thought to maintain high species diversity, particularly when the disturbance alters the structure of the reef by opening up bare substratum, thereby creating space available for the settlement of new coral recruits (planulae). The influence of disturbance in community structure and dynamics has been illustrated by the intermediate disturbance hypothesis, which states that the highest number of species in a community will occur at intermediate levels (frequency and size) of natural disturbance. Lower diversity will exist where disturbances are either very large or very small, or very frequent or very infrequent (Connell, 1978, 1979). The size of the new space also influences the type of recruitment. Small patches are usually colonized by the nearest dominant species, while larger areas provide an opportunity for less dominant species to establish. Interestingly, many Caribbean corals release planulae in late summer/early fall, which coincides with the hurricane season in the Atlantic, and this may enhance recolonization (Rogers, 1993).

The effect of storms is strongly dependent on the ecology and geology of a specific area and the characteristics of the storm. For instance, a wide range of reef-specific variables influence the magnitude of the impact including spatial location, community structure, coral age, size, morphology, and reef depth. Variables associated with the storm itself include the path of the storm and its strength (measures of wind velocity and wave height), and heavy rain can cause excessive runoff as well as localized decreases in salinity which have been linked to a reduction in the planulae production (Figure 3.4; Jokiel, 1985). Some species of corals exhibit a growth form that is more robust to storm energy than others (e.g., boulder shapes). In contrast, corals with fragile skeletons and typically those with branching morphology will be more easily damaged by extreme wave action. In the Caribbean, *Acropora palmata* and *Acropora cervicornis* are very susceptible to storm damage (Brown, 1997). Breakages may be advantageous to these species since they produce relatively few larvae and instead are thought to rely primarily on asexual reproduction through fragmentation to produce new colonies (Bak and Engel, 1979; Hughes, 1985). Furthermore, delayed mortality from outbreaks of disease among injured corals, bioerosion of damaged skeleton, and altered predator-prey relationships may occur for years after a hurricane has struck (Knowlton et al., 1990).

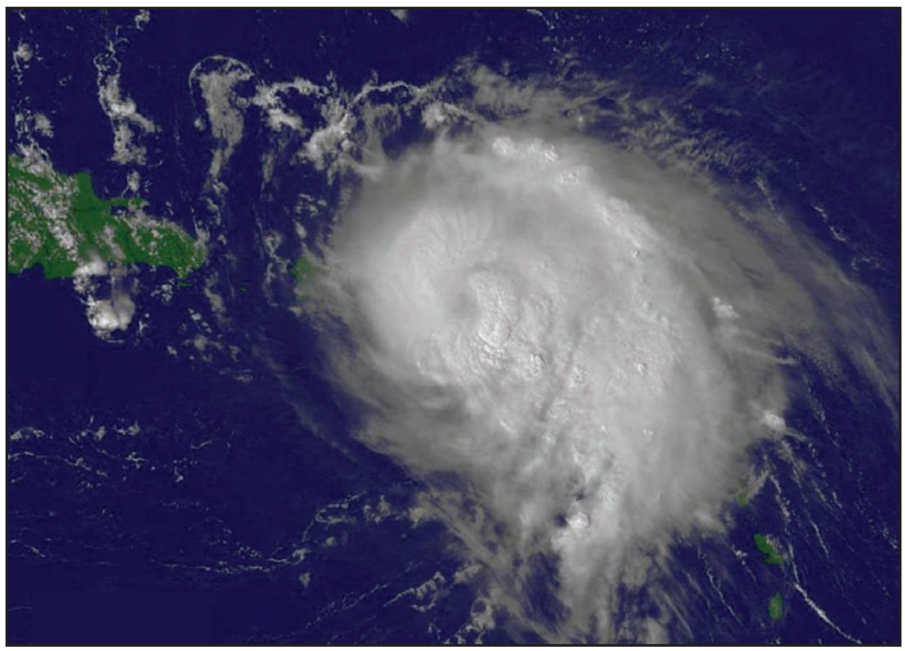

Figure 3.4. Hurricane Georges, a category 3-4 storm hit the USVI, Puerto Rico, and the Florida Keys in September, 1998. Damage included the physical breakage of corals and a massive pulse of sediment and nutrients that were discharged into nearshore waters. Georges was one of four hurricanes in progress in the Caribbean at the time. Photo: NASA and NOAA, http://rsd.gsfc.nasa.gov/rsd/images/Georges.html, Accessed 2/10/05.

Age is another factor that influences the ability of a coral colony to withstand the mechanical stresses of large storms. As corals grow, they become more vulnerable to breakage and dislocation (Brown, 1997). The majority of wave impacts occur in the shallowest (0-20 m) depth range, so corals at greater depths are generally less directly impacted. Deeper corals, however, can be significantly damaged indirectly by large blocks that tumble down from shallower waters (Brown, 1997). Damage to corals can indirectly impact other reef-associated organisms through the reduction of coral cover and topographic complexity which influence biological interactions such as predation, succession, and competition. As coral cover is reduced, the refuge function for many fish and invertebrates is diminished. Also the removal of organisms from substrate via scouring reduces the abundance of food available for some species. In addition, increases in turbidity and sedimentation that often accompany storms can affect the emergent community by impairing photosynthesis and feeding, and limiting sexual reproduction (Kojis and Quinn, 1985).

The direct effects of cyclones on fish are size-specific. Lassig (1983) noted that during the final stages of Cyclone Peter, many fish that were normally associated closely to the benthos were found in the water column and some had fresh wounds. This suggests that fish try to weather the storm in the water column, where they are less likely to be injured. It was also noted that after the storm, overall fish abundance decreased significantly, with juveniles sustaining higher mortality than adults due to strong storm-driven currents.

To understand how a cyclonic storm affects a reef requires examination of the recovery patterns and processes. Detailed comparative investigations of pre- and post-hurricane coral reef ecosystems that include variables such as amount of coral, number of species, settlement characteristics and growth rates, and nutrient cycling may provide valuable insights. Multiple year trends using continuous monitoring data, however, are likely to provide the most accurate assessment of both short- and longer-term impact and recovery (Hughes and Connell, 1999). The trajectory and rate of recovery will be influenced by a number of interacting factors including the rates of recruitment, species involved, and sequence of colonization (Brown, 1997). Research also suggests that anthropogenic impacts can interfere with the recovery process. Finally, separating storm effects from those caused by direct human activity and phenomena such as coral bleaching and competition with algae, is problematic due to the level of degradation of some reef systems (Brown, 1997).

The terms "hurricane" and "typhoon" are regionally specific names for a strong tropical cyclone. This report follows the geographically-specific naming convention recognized by NOAA (i.e., NOAA Research's Hurricane Research Division, http://www.aoml.noaa.gov/hrd/tcfaq/A1.html, Accessed 01/07/05) whereby the term "hurricane" applies to the North Atlantic Ocean, Northeast Pacific Ocean east of the dateline, and South Pacific Ocean east of 160E; "typhoon" applies to the Northwest Pacific Ocean west of the dateline; "cyclone" applies to the Southwest Pacific Ocean west of 160E and Southeast Indian Ocean east of 90E. The characteristics of storm and hurricane categories are given in Table 3.3.

Table 3.3. The Saffir-Simpson scale for tropical storm and hurricane classification and associated storm characteristics provide a consistent way to characterize major storm events. na=not applicable. Source: NOAA National Hurricane Center.

SAFFIR-SIMPSON SCALE FOR HURRICANE CLASSIFICATION					
Storm Type	**Category**	**Wind Speed (kts)**	**Wind Speed (mph)**	**Pressure (millibars)**	**Damage Potential**
Tropical Depression	na	20-34 kts	23-39 mph	1007 mb	na
Tropical Storm	na	35-64 kts	39-74 mph	1006-1000 mb	na
Hurricane	1	65- 82 kts	74- 95 mph	980-999 mb	minimal
Hurricane	2	83- 95 kts	96-110 mph	965-979 mb	moderate
Hurricane	3	96-113 kts	111-130 mph	945-964 mb	extensive
Hurricane	4	114-135 kts	131-155 mph	920-944 mb	extreme
Hurricane	5	>135 kts	>155 mph	919 mb	catastrophic

Coastal Development and Runoff

In the past several decades, there has been a well-documented demographic shift toward higher concentrations of human settlement in the coastal zones of many countries including the U.S. (Culliton et al., 1990; Figure 3.5). More than half of the U.S. population now lives in coastal counties, a trend that is expected to continue to increase (Pew Oceans Commission, 2003; Cicin-Sain et al., 1999). This trend has increased the frequency and magnitude of impacts from activities such as the construction of residential developments, hotels and resorts, recreational facilities, and infrastructure such as roads and wastewater treatment plants (WWTPs).

Terriginous sediments in runoff from construction sites and roads are often a major threat to nearshore areas. Dredging of nearshore sediments for marina facilities, ship access and navigation, beach nourishment, and building materials can introduce significant quantities of particulate matter into the water column. While strong currents tend to dissipate some of the added sediments, nearshore areas with gentle slopes and low flushing rates tend to accumulate sediments, which can have detrimental effects on sessile invertebrates like corals (Rogers, 1990). Physical smothering may be the most obvious effect of sedimentation. Although most corals have some ability to rid themselves of foreign particles, the removal of sediments requires the diversion of energy from vital activities such as reproduction and feeding. The negative effects of the accumulation of

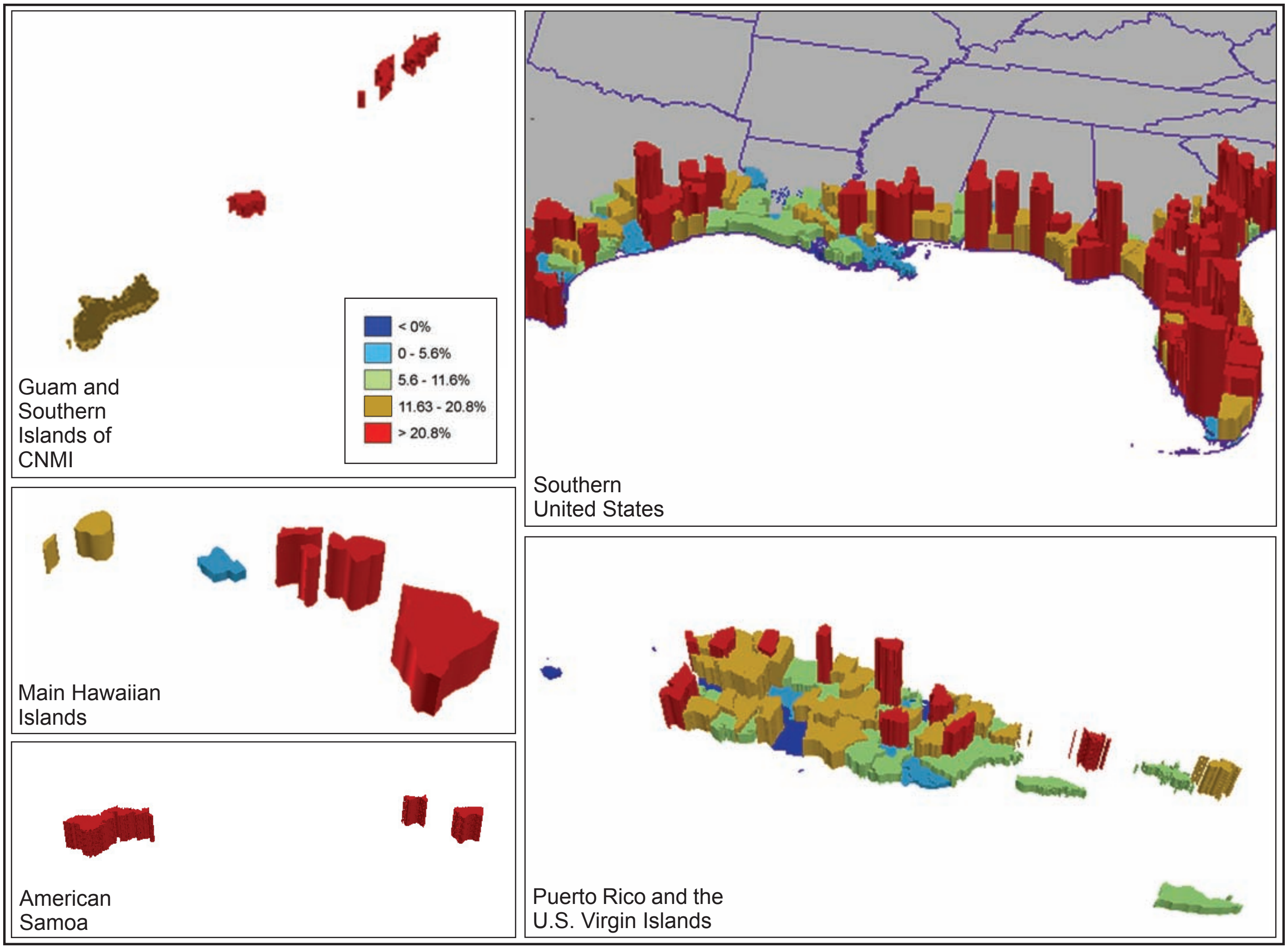

Figure 3.5. Coastal population change between 1990 and 2000 and associated development pressure pose a significant threat to coral reef ecosystems, particularly in island jurisdictions with limited land area. Maps not drawn to scale. Maps: K. Buja. Data: U.S. Census, 1990, 2000; Secretariat of the Pacific Community, http://www.spc.org.nc/prism, Accessed 2/15/05.

sediments on corals can be exacerbated by wave action that repeatedly resuspends sediments into the water column (Rogers, 1990). Increased turbidity in the water column, whether episodic or chronic, reduces light availability for photosynthesis and growth. Increases in nearshore sediment loads have been shown to affect morphology of corals and gorgonians as well as inhibit the development and recruitment of coral larvae (Rogers, 1990). Coral species react differently to this stressor, and coral reefs in waters experiencing increased turbidity may exhibit a shift in community composition toward greater dominance of corals that are more tolerant of lower light levels and better adapted to remove sediments.

Alteration of watersheds and associated changes in vegetative cover often decrease the ability of the land to absorb rainfall, which flows through streams and channels, carrying sediments and pollutants into nearshore areas. Generally, runoff from developed watersheds carries higher sediment loads than from undeveloped areas, and this is more pronounced in areas where the topography is characterized by steep slopes. Removal of mangrove forests that normally trap sediments may allow a greater proportion of terriginous sediments to reach reef ares.

In addition to sediments, runoff from developed watersheds tends to have higher concentrations of waste products. Increased freshwater inputs are actually considered pollutants as they can decrease the salinity levels in some nearshore areas. Other contaminants derived from human use of nearshore areas include oil leaking from vehicles, pesticides and lawn fertilizers applied to yards, parks and golf courses, chemicals in asphalt that wash off roads, excrement from livestock and domesticated animals, and litter.

The development of infrastructure is also a major concern. In many areas, coastal development often occurs without a commensurate improvement in the wastewater infrastructure, and existing systems cannot adequately accommodate the added burden. As a result, untreated or partially-treated sewage overflows into nearshore areas. Outside of urban areas, many homeowners are not able to access WWTPs and often must rely on septic tanks, which are subject to corrosion and leakage. The hard-to-detect leaks often allow untreated sewage to seep into groundwater and nearshore waters. A recent report (Carter and Burgess, Inc., 2002) assessing the sustainability of tourism in Hawaii noted that many of the island's municipal wastewater systems are nearing capacity. While most new developments have private WWTPs to satisfy permit conditions, many residents still rely on private systems, such as septic tanks, which are in various stages of disrepair. Though they considered myriad aspects of tourism, the authors of the study contend that such nonpoint source pollution is "one of Hawaii's greatest environmental threats" (Carter and Burgess, Inc., 2002).

Other infrastructural issues include the problems of adequate waste disposal and the construction of docks and piers that can result in habitat loss. In summary, coastal development presents a wide range of challenges for coastal areas, especially in terms of the number and scale of construction projects, capabilities of infrastructure, intensity and type of land use, and increases in sedimentation and pollution levels.

Coastal Pollution

Worldwide, the threat to coral reef ecosystems from pollution is surpassed in severity only by coral bleaching and fishing (Spalding et al., 2001). Model estimates indicate 22% of the world's coral reef ecosystems are threatened by land-based pollution and soil erosion (Bryant et al., 1998). Pollution often desensitizes the ecosystem, so that it becomes more susceptible to other stressors such as climate change, disease, and invasive species. The primary stressors from land-based sources are nutrient and chemical pollution from fertilizers, herbicides, pesticides, human-derived sewage, and increased amounts of sediment from coastal development and storm water runoff. Other pollutants, such as heavy metals and oil, can also be prominent at specific locations.

This section focuses on point source pollution. Point sources of pollution originate from confined or discrete conveyances, such as a pipe, tunnel, ditch, channel, well, or fissure. Examples of point source pollution include sewage outfalls, factory wastewater, and dumping of chemicals. Household chemicals and untreated industrial wastewater may also be discharged into the domestic wastewater stream. Finally, short outfalls contribute to the pollution of nearshore waters. Other point sources include vessels without holding tanks that discharge their wastes in marinas and nearshore coastal areas. Dredging for shipping lanes, marinas, and coastal construction projects resuspends sediments that increase turbidity and decrease coral reef ecosystem productivity. Industrial point sources include manufacturing operations, effluent discharges, accidental oil spills and the release of contaminants discharged as a byproduct of oil-drilling (e.g., toxic poly-aromatic hydrocarbons (PAHs), benzene, ethylbenzene, xylene) and heavy metals, such as lead, copper, nickel and mercury.

Direct impacts of pollutants include reduced recruitment, loss of biodiversity, altered species composition (a shift from predominantly phototrophic to heterotropic fauna), and shallower depth distribution limits. Sewage pollution causes nutrient enrichment around population centers, treatment facilities, and sewage outfalls. Increased nutrient concentrations promote increased algal and bacterial growth, can degrade seagrass and coral reef ecosystems, and ultimately may decrease fisheries production. Sediments smother benthic organisms, which can become diseased when exposed to dredged sediments contaminated with toxic heavy metals and organic pollutants. Toxic chemicals can decrease coral reef ecosystem productivity and biodiversity and increase human health risks through food contamination.

Management actions by NOAA to address water quality concerns are taken in partnership with the Environmental Protection Agency (EPA), the Department of Agriculture, and local or state governments. Research is needed to understand how coral reef ecosystems respond to poor water quality, and to provide mangers with tools to detect, assess, and remedy negative impacts from pollution. Therefore, the sources of the substances that adversely affect water quality must be identified, and relevant policies and control strategies for limiting pollutants must be developed and validated. Monitoring pollutants in highly polluted or "at risk" areas can alert managers to changes in pollutant inputs and impacts. To be most useful, results from pollution monitoring programs should be integrated into modeling efforts that quantify the relative amounts of natural and anthropogenic inputs to ecosystems Additionally, monitoring results should be used to develop models and indicators that assess threats or identify stressors causing coral reef ecosystem decline.

Tourism and Recreation

Tourism and recreation are by far the fastest growing sector of coastal area economies. This growth is predicted to continue as incomes rise, more Americans retire, leisure time expands and accessibility to the coasts and oceans increases (U.S. Commission on Ocean Policy, 2004). Coral reefs, in particular, have a major economic value. Cesar et al. (2002) calculated that the greatest contribution to the annual value of coral reefs in Hawaii is tourism and recreation, which brings in $304 million per year. Coastal tourism contributes $9.9 billion to the Californian economy annually and is considered the largest sector of the “ocean industry” compared with $6 billion/year for ports, $860 million/year for offshore oil and gas development, and $550 million/year for fisheries and mariculture (Wilson and Wheeler, 1997; Cicin-Sain and Knecht, 2000). Travel and tourism are estimated to have provided $746 billion annually to the U.S. gross domestic product (GDP), making travel and tourism the second largest contributor to GDP (Houston, 1995). Tourism is particularly significant in many Caribbean and Pacific islands surrounded by coral reef ecosystems. In the Florida Keys alone, over four million tourists purchase about $1.2 billion in services annually. Over three million tourists visit at least one of Hawaii’s coral reef sites per year, and approximately 90% of new economic development in Guam and the CNMI is related to coastal tourism (NOAA, 1997). The vast demand for tourism and recreational services associated with coral reefs generates considerable income for many local communities. Those who engage in reef-related recreational activities purchase goods and services, such as charter boats and diving trips via dive centers. In addition, they spend money on lodging, travel, food and beverages, etc. English et al. (1996) estimate an annual economic impact of $1.2 billion in visitor spending in the Florida Keys which results in a total sales impact of $1.3 billion, $506 million in income, and over 33,000 jobs. Leeworthy and Wiley (1997) estimate an annual economic impact of $94.3 million in resident spending in the Florida Keys, resulting in a total sales impact of $105.6 million and supporting over 2,400 jobs. Cesar et al. (2002) estimated that recreational use values in Hawaii represent 85% of annual benefits accrued from coral reefs (the others being amenity/property values, biodiversity, fisheries, and educational spillover), which amount to $304.16 million/year. In southeast Florida, the annual use value accrued from coral reefs is estimated at $229.3 million (Johns et al., 2003).

Human uses of coral reefs are both direct and indirect, with recreation and tourism among the most prominent uses. Recreational activities on U.S. coral reefs include snorkeling, scuba diving, boating, fishing, and shell-collecting. The intensity of each activity varies widely from region to region, but can be considerable in some areas. In southeast Florida, residents and visitors spent 28 million person-days using artificial and natural reefs during a 12 month period (June 2000 to May 2001) and 4.94 million person-days snorkeling and scuba diving (Johns et al., 2003; Figure 3.6). Water-based activities such as scuba diving are increasing in popularity, and over 3 million people are currently certified to dive in the U.S. Scientific studies have now shown that divers and snorkelers can have a significant negative impact on coral reefs in terms of physical damage and a concomitant reduction in their aesthetic appeal (Hawkins and Roberts, 1993; Hawkins et al., 1999; Rouphael and Inglis, 2001). For example, a snorkeling trail created in the Virgin Islands National Park’s Trunk Bay in the 1960s had deteriorated substantially when observed in

Figure 3.6. Some reef areas in the Florida Keys may have hundreds of visitors per day. Photo: Bill Harrigan.

1986 with visitor numbers estimated at over 170,000 per year. Only 10 of 50 tagged Elkhorn coral colonies remained undisturbed during a seven-month period of observation (Rogers et al., 1988). Plathong et al. (2000) examined the effects of snorkelers using self-guided interpretative trails around a reef within the Great Barrier Reef Marine Park, Australia and found that despite comparatively low levels of use (approximately 15 snorkelers per trail per week), snorkelers caused significant damage to corals along the trails. Hawkins et al. (1999) examined the impacts of diving on a reef off the Caribbean island of Bonaire and concluded that impacts would be minimized by maintaining a site carrying capacity of between 4,000 and 6,000 dives per year. In contrast, Rouphael and Inglis (2002) suggested that management actions should focus on identifying and mitigating the causes of damaging behavior rather than setting numerical limits to site use.

Concern has also been directed at the activity of fish-feeding. Feeding fishes negatively impacts both fishes and habitat in several ways including: (1) fish consume food that is very different to their normal diets; 2) the concentration of fish at feeding stations disrupts normal distribution/abundance patterns; (3) fish behavior changes with some individuals or aggregations exhibiting abnormal aggression; and (4) inputs of nutrients and incidental damage to benthic structure can result in an increase of macroalgae (Perrine, 1989; Alevizon, 2004).

In addition to these direct threats, indirect threats can be equally, if not more devastating to coral reefs. Indirect threats include development of hotels and resorts, construction of the infrastructure needed to support such resorts, seafood consumption, beach replenishment, construction of airports and marinas, as well as the operation of cruise ships. The impacts resulting from these activities include increased sedimentation, nutrient enrichment, pollution, exploitation of endangered species, and increased litter and waste (UNEP, 2002). Mitigation of the impacts of tourism often involves education and raising awareness with the goal of behavioral change (UNEP, 2002). In Hawaii, a strategy for both defining a carrying capacity and influencing visitor behavior through education has been implemented. Oahu's Hanauma Bay Nature Preserve in Hawaii has an estimated three million visitors annually and 13,000 per day in the high season. Impacts at Hanauma Bay, including widespread trampling of reefs and resuspension of sediments, fish-feeding, littering, and other pollution, prompted a management strategy to limit visitor numbers (NOAA CSC, 2004). Determining the carrying capacity for this area was critical to its long-term sustainability and was supported by the development of an education center aimed at influencing visitor behavior (Cesar et al., 2002).

Clearly, tourism is a major source of economic welfare and livelihood for many coastal communities. Unfortunately, detrimental side effects and physical damage often result from direct visitor activity and the development of facilities to support tourism. Without long-term planning for tourist activities at these fragile sites, both resourcse and revenues are at risk. Sites such as Hanauma Bay Nature Preserve have had to make operational adjustments and offer education and instruction to visitors. Managers are increasingly challenged to develop strategies that mitigate unsustainable usage, while continuing to support the tourism industry.

Fishing

Coral reefs and associated habitats support important commercial and recreational fisheries. Over 4,000 species of fishes (>25% of all marine fishes) inhabit shallow coral reefs (Spalding et al., 2001), along with a large number of marine plants and invertebrates – many of which are exploited for human use. Coral reef fisheries support and sustain communities by providing food and sources of income. Fishing also plays a central social and cultural role in many island communities. Coral reef fisheries are generally small-scale, but coral reef fishers exploit hundreds of species of fishes and invertebrates using a wide variety of fishing gear. In a number of U.S. reef areas, recreational fishery catch now equals or exceeds the commercial catch. The rich biodiversity of coral reefs also supports a valuable marine aquarium industry, especially in Hawaii and Florida, and provides materials for a range of natural products developed by the biotechnology and pharmaceutical industries.

Unfortunately, these fishery resources and the ecosystems that support them are under increasing threat from overfishing and fishery-associated impacts on habitats and ecosystems. Fishery-related impacts include: 1) direct overexploitation of fish, invertebrates, and algae for food and the aquarium trade; 2) removal of a species or group of species which can impact multiple trophic levels; 3) by-catch and mortality of non-target species; and 4) physical impacts to reef environments associated with fishing techniques, fishing gear, and anchoring of fishing vessels.

Overfishing

Overfishing, along with pollution and global climate change, is generally considered to be one of the greatest threats to the health of coral reefs. It is also the most widespread threat, estimated to be of medium or high threat to over 35% of the world's reefs (Bryant et al., 1998). In many cases, significant depletion of reef resources (especially large fishes and sea turtles) had already occurred before 1900 (Jackson et al., 2001; Pandolfi et al., 2003). Since then, increases in coastal population, improved fishing technology, and overcapitalization of fishing fleets driven by demand from rapidly growing export markets have greatly accelerated resource depletion. Many reef fishes have relatively slow growth rates, late maturity, and irregular recruitment - characteristics that make overexploitation more likely. The trend is for high-value or vulnerable resources – generally large predators such as groupers, jacks and sharks – to be removed first, and then target species further down the food chain are subsequently fished (Pauly et al., 1998).

Overfishing has been identified as a major concern in all U.S. states and territories with coral reefs and has been identified by the USCRTF as a priority reason for the development of local action strategies. In most cases, the large number of species in these multi-gear, small-scale fisheries has made it impractical to conduct standard stock assessments for more than a fraction of the species (see Table 3.4), and such data-intensive, single-species approaches have been criticized as unrealistic for most reef fish systems (Sale, 2002). There is evidence of serial depletion of reef resources in Florida and around all populated U.S. islands. In Hawaii, long-term catch rates suggest that stocks of nearshore fishes have declined by nearly 80% between 1900 and the mid-1980s (Shomura, 1987). Catch per unit effort (CPUE) of reef fishes in Guam fell by more than 50% between 1985 and 2000 (Birkeland et al., 2000), while the CPUE fell 70% in the American Samoan reef fishery, accompanied by a shift in species composition, over a period of 15 years between 1979 and 1994 (Birkeland, 1997). The Nassau grouper fishery, the highest value commercial fishery in Puerto Rico and the U.S. Virgin Islands (USVI), collapsed in the 1980s due to overexploitation of spawning aggregation sites and the species was identified as a candidate to be listed as threatened or endangered under the Federal Endangered Species Act (16 U.S.C. § 460 et seq.) in 1991. In the Florida Keys, the nation's most extensive and long-term reef fish monitoring program has revealed that 77% of the 35 individual stocks that could be analyzed in Biscayne Bay are overfished (Ault et al., 2001).

Table 3.4. Overfished Coral Reef Species in Federal Fishery Management Plans (FMPs). Source: 2003 Status of U.S. Fisheries Report (NOAA, http://www.nmfs.noaa.gov/sfa/reports/html, Accessed 2/14/05) and Western Pacific Coral Reef Ecosystem Fishery Management Plan (NOAA, http://www.wpcouncil.org/coralreef.htm, Accessed 2/14/05).

Table: Overfished Coral Reef Species in Federal Fishery Management Plans[1]				
Region	Total Number of Federally Managed Coral Reef Species	Number of Species Overfished or Approaching Overfished	Number of Species Not Overfished	Species with Insufficient Data
South Atlantic[2]	62	8	12	42
Gulf of Mexico[2]	44	5	4	35
Caribbean[2]	154	3	1	150
Western Pacific[3]	28	0	0	28
Total	422	16	16	389
Notes:				

1 Overfished analysis includes only stocks in Federal waters–most reefs and fishing pressure occur in state and territorial waters.
2 Excludes coral species for which the fishery is closed.
3 From the Bottomfish, Precious Coral and Crustacean FMPs only–does not include the hundreds of species covered by the new Coral Reef Ecosystem FMP.

Because of long-term trends in the exploitation of mixed reef fisheries, there are few places that maintain relatively intact fish populations to serve as experimental controls. The Northwestern Hawaiian Islands (NWHI) and some of the uninhabited U.S. Pacific Remote Island Areas probably represent the closest approximation to unexploited coral reef ecosystems in U.S. waters. The average fish biomass in the NWHI is 2.6 times greater than in the Main Hawaiian Islands (MHI). More than 54% of the total fish biomass in the NWHI is composed of apex predators, compared to less than 3% in the MHI. These differences have been attributed to overfishing in the MHI (Friedlander and DeMartini, 2002).

Ecosystem Shifts

There is increasing evidence that overfishing on reefs results not just in shifts in fish size, abundance, and species composition, but that it is also a major driver altering the ecological balance and contributing to the degradation of coral reef ecosystems (Bellwood et al., 2004). In particular, overfishing of herbivorous fishes has been linked to phase-shifts from high-diversity coral-dominated systems to low-productivity algal-dominated communities (Hughes, 1994). U.S. reefs, especially in the Atlantic, are increasingly facing coral declines, though uncertainty remains about the processes and links to fishing levels, especially in the Pacific (Jennings and Polunin, 1997). Herbivores comprise a significant component of the catch in the MHI, Guam, CNMI, and American Samoa. Parrotfishes and surgeonfishes are increasingly important in Puerto Rico and in St. Croix, where they represent the predominant catch. In nearly all areas except Florida, declines in the abundance of these species have been observed. There is also evidence that heavy fishing pressure on certain invertebrate-feeding fishes has played a key role in outbreaks of crown-of-thorns (COTS) starfish, snails, and herbivorous sea urchins (Hay, 1984; McClanahan, 2000; Dulvy et al., 2004). There is no clear evidence of the extent to which this has been an important factor in bioerosion on U.S. reefs, nor is there a clear understanding of the ecosystem effects due to the removal of top predators. Overfishing can also compound the impact of other threats. For example, overfishing of herbivorous fishes and enhanced nutrient flows to reefs may lead to reef overgrowth by macroalgae. Likewise, reefs devoid of herbivores may be less likely to recover from coral bleaching events (Westmacott et al., 2000).

Impacts from Fishing Gear

A number of protected species, such as hawksbill and green sea turtles as well as a number of seabird species are untargeted victims of fishing activity and are especially vulnerable to longline fishing and shrimp trawling. Traps and gill nets also result in mortality of non-target species.

Physical damage to the benthos from certain fishing techniques is well-documented. Traps set for fishes or lobsters can cause physical damage to corals, gorgonians, and sponges. They may also result in by-catch and "ghost fishing" if they are lost or not regularly checked. Trap fisheries are most common in Florida (lobster and stone crab) and the U.S. Caribbean (fish and lobster), and are generally less prevalent in the U.S. Pacific. Large gill and trammel nets have also been identified as a growing concern, particularly in St. Croix (USVI) and Hawaii. Large gill nets are set on reefs and their lead-lines can cause extensive damage when the nets are hauled into the boats. In addition to legal fishing activities, illegal techniques can cause severe damage to reefs. Use of chlorine bleach has been reported in Hawaii, Guam, and Puerto Rico (USCRTF, 1999), and traditional plant-derived poisons are still used occasionally in the subsistence fishery in American Samoa. The use of cyanide for fishing has not been reported on U.S. reefs, although the expansion of the live food fish trade to the Marshall Islands has raised concerns about its potential use there. Blast fishing, probably the most destructive technique, has rarely been reported on U.S. reefs.

Other indirect impacts to coral reefs associated with fisheries include anchor damage from fishing boats, which has been identified as a problem in Florida and the U.S. Caribbean. Trawling damage to coral areas has been identified as a problem in deeper coral areas in the Gulf of Mexico. It was also a major cause of destruction of the deep water *Oculina* coral banks off the east coast of Florida before the development of the Experimental Oculina Research Reserve. In general, such damage is inadvertent rather than due to directed fishing, but trawls can cause tremendous damage when hauled over hard bottoms with coral. Furthermore, groundings of fishing vessels have had major, albeit localized, impacts on certain reefs.

Trade in Coral and Live Reef Species

Many coral reef species are harvested domestically and internationally to supply a growing international demand for seafood, aquarium pets, live food fish, construction materials, jewelry, pharmaceuticals, traditional medicines and other products. In many locations, collection is occurring at unsustainable levels, and overharvesting may lead to reductions in the abundance and biomass of target species, shifts in species composition, and large-scale ecosystem shifts including population explosions of non-target species or the replacement of thriving, coral-dominated systems with low-productivity algal reefs (Hughes, 1994; McClanahan, 1995; Jennings and Polunin, 1996). In addition to overfishing, there is widespread use of destructive techniques such as cyanide poisoning of fishes and coral colony breakage. Cyanide is used illegally in Southeast Asia and other parts of the Indo-Pacific to capture live reef fish for the aquarium trade and live fish markets, and has been found to: 1) kill many non-target species, 2) cause habitat damage, and 3) pose human health risks (Barber and Pratt, 1997). High levels of mortality associated with cyanide and inadequate handling and transport practices pose significant challenges to achieving sustainability. The use of cyanide has not been reported or observed in the U.S., with the possible exception of limited use in some of the Freely Associated States (e.g., Marshall Islands) associated with the live reef fish food trade. In addition, unsafe diving practices resulting from the collection of corals, sea cucumbers, fish, and other species in deep water are causing a high incidence of illness, paralysis, and even death of collectors in some regions (Johannes and Riepen, 1995; Barber and Pratt, 1997).

The Marine Aquarium Trade

The marine aquarium trade has an estimated value of $200-300 million per year (Larkin and Degner, 2001). The global trade in coral has increased by 500% over the last 10 years, with over one million live corals and 1.87 million kg of live rock traded in 2002 (Bruckner, 2003). In addition, an estimated 20-24 million reef fishes are traded annually, representing 1,450 species in 50 families (Balboa, 2002; Wabnitz et al., 2003). The U.S. is the world's largest consumer of ornamental coral reef species, importing 60-80% of the live coral, over 50% of the curio coral, 95% of live rock, and 50-60% of the marine aquarium fishes each year (Wood, 2001; Bruckner, 2003). The most important sources of coral are currently Indonesia, Fiji, and Vietnam (Bruckner, 2001). Indonesia and the Philippines each supply about 30% of the total global trade in reef fishes, with another 30% exported from five locations (Brazil, the Maldives, Hawaii, Sri Lanka, and Vietnam); Florida and Puerto Rico are currently the largest exporters from the wider Caribbean (Wood, 2001; Balboa, 2002).

Although it is illegal to harvest stony corals and live rock in U.S. waters, ornamental reef fishes and many motile invertebrates are collected in U.S. waters both for domestic use and export. In Florida, 318 marine species (181 fishes and 137 invertebrates) have been collected for commercial purposes, with a total annual value of up to $4.2 million. Over 200,000 ornamental reef fishes are landed in Florida each year, with a maximum of 425,781 fishes in 1994 (Larkin, 2003). Annual reported harvest of ornamentals from West Hawaii rose from 90,000 in 1973 to 422,823 in 1995 (Tissot and Hallacher, 1999).

The Live Reef Food Fish Trade

Groupers, humphead wrasse, coral trout, and other large fishes that use coral reefs are harvested live to supply restaurants in Hong Kong. Exports increased rapidly during the 1990s and peaked at 32,000 metric tons (mt) in 1997, with a slight decline between 1998 and 2000 due to the Asian economic crisis (Lau and Parry-Jones, 1999). More recently an estimated 22,000 to 28,000 mt of live reef fishes have been imported by Hong Kong, China, Taiwan, and other Asian markets, with Hong Kong imports comprising 65-80% of the total regional trade (Graham et al., 2001). In addition to widespread use of cyanide to capture the fish live, fishers target spawning aggregations and have been reported to eliminate entire breeding populations relatively rapidly (Lau and Parry-Jones, 1999). In addition to concerns regarding the use of destructive fishing techniques, most of these species are vulnerable to heavy fishing pressure due to their longevity, late sexual maturation, aggregation spawning, and sex change habits (Sadovy et al., 2004).

Curios and Jewelry Trade

Coral reef species harvested for curios and jewelry include mollusk shells; stony coral skeletons; and black, pink, gold, bamboo, and other precious corals (Figure 3.7). Of these species, only stony corals, black coral, and giant clams are internationally regulated through the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). International trade in shells involves as many as 5,000 species of an unknown volume primarily supplied by the Philippines, Indonesia, Thailand, Singapore, Taiwan, Mexico, India, Africa, and Haiti (Wells, 1989). Shells are used for construction materials; shell craft; mother of pearl and other collectors items; as well as additives to floor tiles, toothpaste, pottery, and poultry feed (Marshall et al., 2001). The volume of trade in coral skeletons has varied over the years, with the Philippines being the major supplier in the 1970s and 1980s; exports from the Philippines were prohibited in the late 1980s, with a temporary lifting of trade bans in 1992 during which over three million kg were exported. Fiji and Vietnam are currently the major source countries for coral skeleton (Bruckner, 2001).

Figure 3.7. The shells of reef organisms are often sold at curio shops, such as this one in Palau. Although many of the shells were probably imported from Southeast Asia, some local collection is thought to occur as well. Photo: J. Waddell.

International trade in black coral, according to the CITES trade database, has averaged 430,000 items per year since 1983, with the maximum trade in 1994, and 320,000 items traded in 1998 (CITES Trade Database, http://www.cites.org/eng/resources/trade.shtml, Accessed 02/16/05). The world's largest supplier of worked black coral is Taiwan (>90% of the total), with most reported to be harvested in the Philippines. Commercial harvest occurs in U.S. waters in Hawaii, with annual landings averaging 1,014 kg/year; about 90% of this is for domestic use.

International Protection

CITES is an international agreement among the governments of 165 countries to protect wildlife by ensuring that international trade does not threaten the survival of a species in the wild. CITES regulates international trade in wildlife according to three levels, or appendices, of threat. Species listed in Appendix I, which includes marine turtles and most whales, are believed to be threatened with extinction and thus, commercial trade of these species is generally prohibited. Most species are listed in Appendix II which includes organisms that are not presently threatened or endangered, but may become so if trade is not regulated. These species can still be commercially traded with export permits which require the exporting country to ensure that the species was legally harvested and its export will not be detrimental. Coral reef species currently listed in Appendix II include about 2,000 species of stony corals (including all scleractinian corals), black coral, giant clams, queen conch, and seahorses. Trade of Appendix III species requires an export permit ensuring that the organism was harvested legally and prepared and shipped so as to minimize damage, injury or cruel treatment.

Ships, Boats, and Groundings

Of all physical damage caused to coral reefs by human activity, ship groundings and the impacts of boats and anchors are perhaps the most destructive. The U.S. Coast Guard (USCG) reports that over 2,100 grounding accidents are reported annually, with about 440 vessels sinking each year. In addition, over 800 abandoned barges litter the inland and coastal waters of the U.S., many still loaded with hazardous cargo (Helton, 2003). As recreational and commercial boating traffic increases in nearshore ocean waters, these shipwrecks pose a threat to coral reef habitat. When anchors, especially the enormous anchors of cruise ships, are carelessly dropped and dragged on fragile reef, hundreds of meters of habitat can be destroyed. Recent studies demonstrate the extensive impacts of groundings when hazardous cargo is released. However, once cargo and fuel are spilled, the vessel may continue to cause repeated physical damage to the reef due to movement by wind and waves. Furthermore, abandoned barges can often become illegal dump sites for other hazardous materials, trap wildlife, and become public safety hazards (Helton and Zelo, 2003).

Initially many considered the impacts of grounded vessels to be significant only at a local level, but the widespread effects of these events have recently been the subject of closer examination (Precht et al., 2001; Ebersole, 2001). Damage resulting from ship groundings often continues well beyond the initial event of impact as a result of slow recovery and fragmentation of keystone species essential to reef structure and function. In particular, spur and groove reefs do not seem to recover their diverse fish assemblages following a ship grounding incident (Ebersole, 2001). The potential threats of grounded vessels became the subject of increased political attention in 1999 when nine vessels were cleaned, cut apart, and removed from a reef in Pago Pago, American Samoa and the grounding sites were restored by the USCG, NOAA, and American Samoan government. The increasing frequency of vessel groundings in coral reef environments led to the development of the National Action Plan to Conserve Coral Reefs (USCRTF, 2000) which recognizes the impact of grounded vessels to coral reefs and their associated habitats (Helton and Zelo, 2003). In response, NOAA initiated the Abandoned Vessel Project, which seeks to increase awareness of abandoned vessels, particularly where they occur in coral reef systems, as well as provide the technical assistance necessary to remove the vessels (NOAA OR&R, http://response.restoration.noaa.gov/dac.vessels/overview.html, Accessed 6/2/04).

A study conducted on the site of the 1984 grounding of the M/V *Wellwood* in the Florida Keys National Marine Sanctuary suggested that damaged spur and groove habitat will take decades to recover without substantial restoration efforts (Smith et al., 1998). A reduction of topographic complexity also influences local hydrodynamics and the structure of reef fish and invertebrate communities (Miller et al., 1993; Szmant, 1997).

The damage caused to a coral reef habitat by boat anchors is an additional threat resulting from frequent boat traffic. A study conducted in a 220 ha area of coral reef in Fort Jefferson National Monument, Dry Tortugas, Florida documented the extensive damage that can be caused by anchors (Davis, 1977). Cruise ship anchors present a significant and increasing threat to coral reefs. In Grand Cayman, an estimated 1.2 million m^2 of coral reef have been destroyed by cruise ship anchors (Smith, 1998), while cruise ships in the Cancun National Park in Mexico, are thought to have impacted over 80% of the coral reefs there (Schultz, 1998). Designation of anchorages in less sensitive areas, installation of mooring buoys, and identification of areas sensitive to anchor damage are necessary to reduce the destructive practice of unregulated anchor dropping and dragging.

Major vessel groundings in the FKNMS such as the M/V *Alec Owen Maitland* and M/V *Elpis* in 1989 and the R/V *Iselin* in 1994 are examples of events in which waves and currents occuring between the grounding and restoration resulted in further injury to the reef. Loose coral rubble threatened adjacent undisturbed coral habitat, and restoration efforts involved removing broken pieces of coral from the seafloor and re-attaching them before the arrival of winter storms. The extent of the broken coral can be extensive. For example, the 325-foot M/V *Fortuna Reefer* container ship ran aground near Mona Island, Puerto Rico in July 1997 and damaged over 6,400 m^2 of elkhorn coral (Figure 3.8; Zobrist, 1998).

Figure 3.8. The *M/V Fortuna Reefer*, a container ship that ran aground near Mona Island, Puerto Rico, damaged a large area of reef including stands of *Acropora palmata*. NOAA scientists have undertaken restoration efforts at the site and have monitored the recovery of the coral community there since 1998. Photo: NOAA Fisheries.

An additional impact of ship groundings involves contamination from Tributyltin (TBT), a component of antifouling paint. TBT-based paints have been banned for use on small craft, but TBT-based paints are still widely used on large ships which navigate routes that pass through coral reef habitat. The effects of this paint on a reef were examined following the grounding of a 184 m cargo ship *Bunga Teratai Satu* on Sudbury Reef, Australia in 2000. Results demonstrated that this kind of contamination can significantly reduce coral recruitment in the area of the grounding and may consequently hinder recovery of the community (Negri et al., 2002).

With boat traffic rapidly growing, it is crucial to better understand the ecological implications of vessel groundings and anchor damage, and to take steps to limit or prevent damage through education and guidance supported by strong legislation. Severe physical damage to coral reefs by vessels requires a rapid response and carefully designed methods of removal and restoration to limit the extent of the impact (NOAA, 2002b).

Marine Debris

Globally, marine debris presents a continuous threat to the marine environment. Marine debris adversely impacts marine life through the destruction of essential habitat as well as entanglement and ingestion by marine organisms and seabirds. Typically, the majority of marine debris comes from land-based sources, particularly urban centers, but a significant proportion comes from ships.

All U.S. jurisdictions with coral reefs participate in the International Coastal Cleanup to remove marine debris from their shorelines and nearshore waters. Additional community-based cleanup efforts have been conducted at many locations, including South Point and Kahoolawe in Hawaii. Typical debris collected from the shorelines includes beverage cans and bottles, cigarettes, disposable lighters, plastic utensils, food wrappers, and fishing line (Figure 3.9). Underwater cleanups conducted by snorkelers and divers have found similar materials beneath the surface.

Figure 3.9. Tons of marine debris wash up on the shores of the NWHI every year. Though NOAA's Pacific Islands Fisheries Science Center, Coral Reef Ecosystem Division has removed 401,055 kg of debris from the shallow waters of the NWHI since 2001, resource limitations prevent debris removal on land. Photo: S. Holst.

The most notable impacts of marine debris on coral reef ecosystems come from derelict fishing gear including nets, fishing line, and traps. Prior to the 1950s, fishing gear was composed of natural fibers, such as cotton and linen, and was susceptible to environmental degradation. Since the 1950s, fishing gear has primarily been constructed with synthetic materials, such as nylon and polyethylene, which is less susceptible to environmental degradation. Synthetic nets and fishing line can persist in the ocean for decades and can be transported for thousands of kilometers.

The NWHI has been a focal point for the removal of abandoned fishing gear comprised of conglomerates of netting and fishing line that roll across coral reef habitats, crushing corals and dislodging sessile organisms (Figure 3.10). Fishing gear frequently becomes snagged on corals and continues to trap fish ("ghost fishing") and endangered monk seals and sea turtles (Boland and Donohue, 2003; Donohue et al., 2001; Henderson, 2001; Balazs, 1985). Since 2001, NOAA's Pacific Islands Fisheries Science Center, Coral Reef Ecosystem Division (PIFSC-CRED) has led a large-scale interagency partnership to study and remove derelict fishing gear from the NWHI. NOAA collaborates with the State of Hawaii, City and County of Honolulu, U.S. Fish and Wildlife Service (USFWS), USCG, U.S. Navy, University of Hawaii, Hawaii Sea Grant, Hawaii Metals and Recycling, Honolulu Waste Disposal, and other partners from local agencies, businesses, and non-governmental organizations. From 2001 to 2004, this large-scale effort removed 401,055 kg of fishing gear from these remote islands and atolls (R. Brainard, pers. comm.). Types of fishing gear removed included monofilament gillnet, seine net, and trawl nets, the majority of which was thought to have originated from fisheries operating around the continental shelves of the North Pacific Rim which are located thousands of kilometers from the NWHI.

Figure 3.10. A tangle of abandoned fishing gear removed from Pearl and Hermes Atoll in the NWHI by a team of divers from PIFSC-CRED and the Joint Institute for Marine and Atmospheric Research (JIMAR). The net had to be freed from the reef, lifted to the surface, and towed to shallow water before debris team members could cut it into smaller pieces and remove it. Photo: A. Hall.

Derelict fishing gear has also been a concern in other U.S. coral reef ecosystems. Chiappone et al. (2002) surveyed the Florida Keys for fishing gear and other marine debris and concluded that lobster trap debris was often found in offshore and mid-channel patch reefs, while hook and line gear was more common in shallow and deep forereef areas. Since 1994, the FKNMS, The Nature Conservancy, The Bacardi Foundation, and local dive operators have supported an annual effort to clean the reefs around the Florida Keys. In 2002, divers removed over 1,800 kg of marine debris including fishing line from the Keys. In 2003 and 2004, Amigos de Amoná, Inc. and other partners removed 3,235 kg of marine debris from the islands in Puerto Rico's Mona Channel. The debris consisted of fishing gear (48%), plastics (13%), glass (14%), metal (8%), and miscellaneous items such as refrigerator doors, rubber shoes, packing and insulation materials, and washing machines (17%; Amigos de Amoná, Inc., 2004).

Aquatic Invasive Species

Aquatic invasive species are aquatic organisms that have been introduced, either intentionally or unintentionally, into new ecosystems which result in harmful ecological, economic, and human health impacts (USDA, http://www.invasivespecies.gov, Accessed 2/11/05). Aquatic invasive species have been reported in all U.S. reions and probably exist in every region of the world. Invasive species are generally second only to habitat destruction in causing declines in biodiversity and are thought to impact nearly half of the species currently listed as threatened or endangered under the Federal Endangered Species Act (Wilcove et al., 1998).

The impacts are not only ecological. Damages to infrastructure, such as clogged intake pipes, and environmental losses due to terrestrial and aquatic invasive species cost over $120 billion per year in the U.S. alone (Pimentel et al., in press). The cumulative effects and costs of aquatic invasive species are difficult to quantify, but evidence clearly indicates that the impacts will continue to increase. In fact, the frequency of aquatic invasions has increased exponentially since the late 1700s and shows no signs of diminishing (Ruiz et al., 2000).

Although there have not been many studies that focus specifically on the impacts of aquatic invasive species on shallow-water coral reef ecosystems as a whole, there have been a handful of smaller studies. In Hawaii, it has been determined that the number of marine and estuarine invasive species is approximately 343, including 287 invertebrates, 24 algae, 20 fish, and 12 flowering plants (Bishop Museum, http://www2.bishopmuseum.org/HBS/invertguide, Accessed 02/14/05). Pearl Harbor alone contains more than 100 invasive species. Additionally, some of Hawaii's worst invaders have been intentionally introduced, such as algal species, *Kappaphycus alvarezii* and *K. striatum*, which smothered large tracts of coral reefs in Kaneohe Bay, thus diminishing the ecological and economic value of the area (Carlton, 2001).

Shallow-water coral reef ecosystems are particularly sensitive to a number of non-native species introduction pathways, including ships (due to ballast water discharges and hull fouling), aquaculture of non-native species, releases by aquarium hobbyists, and marine debris.

Introductions from Ballast Water

By 1996, 80% of all commercial goods were being transported aboard ocean-going vessels (NRC, 1996). That percentage is likely to increase as global trade increases. In addition to greater movement of goods across the world's oceans, the speed and size of ships have greatly increased, resulting in faster voyages and larger volumes of ballast water. Because most marine species have planktonic stages as part of their life cycle, they are subject to entrainment during the uptake and discharge of ballast water. Furthermore, because voyage times have greatly decreased, the chances of survival are greater. Ballast tanks have been shown to carry bacteria, protists, dinoflagellates, diatoms, zooplankton, algae, benthic invertebrates (e.g., mollusks, corals, sea anemones, and crustaceans), and fish (LaVoie et al., 1999; NRC, 1996).

Releases by Aquarium Hobbyists

Although there are relatively few documented marine fish invasions, 94 of the 241 documented invasions involved tropical marine species. Additionally, a link has been identified between invasions and marine aquarium imports. Such findings highlight the susceptibility of warm water coral reef ecosystems to intentional introductions by hobbyists and the need for public education. For example, a species of lionfish (*Pterois volitans*) common to the Indo-Pacific regions that was thought to have been introduced from a home aquarium in 1992 has established viable populations all along the southeastern coast of the U.S., with juveniles recently found as far north as Long Island (Figure 3.11; Whitfield et al., 2002).

Figure 3.11. The Red Lionfish, *Pterois volitans*, is native to the Indo-Pacific but has established viable populations along the southeastern coast of the U.S. This fish was photographed off the coast of Beaufort Inlet, North Carolina in about 40m of water. Photo: P. Whitfield.

Introductions from Marine Debris

The amount of marine debris generated as waste from society has increased at a rapid rate in recent years (Silvia-Iniguez and Fischer, 2003; Moore, 2003). For instance, the amount of marine debris in the waters around Great Britain doubled from 1994 to 1998 (Barnes, 2002). Much of the debris is fisheries related, comprised mostly of netting. Floating material provides habitat for many organisms and can result in the transportation of species into new areas, often many thousands of kilometers from their existing species range (Barnes and Fraser, 2003). Problems occur when newly arrived alien species successfully colonize and overwhelm local marine ecosystems. Barnes (2002) found that marine debris was typically colonized by bryozoans, barnacles, polychaetes, hydroids and mollusks.

Security Training Activities

U.S. military installations near coral reefs include operations in Hawaii (Hickam Air Force Base, Pearl Harbor, and Kaneohe Bay); Johnston Atoll (PRIAs); Wake Atoll (PRIAs); Kwajelein Atoll (Republic of the Marshall Islands); Guam; CNMI; Key West and Panama City, Florida; Puerto Rico; USVI; Cuba; and Diego Garcia in the Indian Ocean. Military bases and associated activities including exercises, training, and operational procedures (i.e., construction, dredging, and sewage discharge) have the potential for adverse ecological impacts on coral reefs such as excessive noise, explosives and munitions disposal, oil and fuel spillage, wreckage and debris, breakage of reef structure, and non-native species introductions from ship bilge water or aircraft cargo (Coral Reef Conservation Guide for the Military, https://www.denix.osd.mil/denix/Public/ES-Programs/Conservation/Legacy/Coral/coral.html, Accessed 12/6/04).

In recent years, the military has decommissioned several properties and transferred management responsibility to other agencies. In June 1997, the U.S. Navy officially turned over the management of Midway Atoll (NWHI) to the USFWS for use as a national wildlife refuge. Parts of the island required major remediation to mitigate contamination by lead-based paints, asbestos, fuels and chemicals, but the refuge soon offered fishing, diving, and eco-tour opportunities. When the military decommissioned Kaho'olawe, a former naval bombing range in the MHI, they established a framework for cleanup that included government-appropriated funds and a transfer of the island to a native Hawaiian organization with a state-appointed council to oversee the cleanup process. In June 1995, an evaluation of the nearshore coral reef resources of Kaho'olawe documented the continued presence of metal debris, but reported that relatively few pieces of ordnance were found despite many years of bombing exercises on the island (Naughton, 1995). The 10-year, $460 million cleanup on Kaho'olawe ended November 11, 2003. At that time, the Navy ceased active remediation and access control was returned to the State of Hawaii. The Navy continued surface clearance as a further risk reduction measure until April 2004 when final demobilization occurred. At that point, full-time management of the island shifted to the state. In May 2003, the U.S. Navy ceased military training on the eastern side of Vieques Island, Puerto Rico and transferred management of all remaining Navy property on Vieques, including the bombing training range on the easternmost parcel, to the USFWS. According to the statute governing such transfers, the property can only be used as a wilderness area. Vieques and the surrounding waters have been proposed by the U.S. Environmental Protection Agency (EPA) for listing on the National Priorities List, which EPA uses to determine which uncontrolled waste sites warrant further investigation. As such, the Navy, EPA, and Puerto Rico Environmental Quality Board will work cooperatively on conducting investigations required by the Comprehensive Environmental Response, Compensation and Liability Act (42 U.S.C. § 9601 et seq.). The investigation may conclude the need for the Navy to complete hazardous substances remediation and/or munitions clearance in some areas. Baseline assessments of 24 permanent coral reef monitoring sites at Vieques Island were commissioned by the U.S. Navy and completed in 2001-2002 in an effort to comply with Executive Order 13089 and the U.S. Department of Defense (DoD) Initiative for Coral Reef Protection at the Roosevelt Roads Naval Station in Puerto Rico (Deslares et al., 2004).

According to the DoD Coral Reef Implementation Plan (2000), U.S. military services (i.e., the Air Force, Army, Navy, and Marine Corps) "generally avoid coral reef areas in their normal operations except for some mission-essential ashore and afloat activities." DoD policy is to avoid adversely impacting coral reefs during military operations and ensure safe and environmentally responsible action in and around coral reef ecosystems, to the maximum extent practicable. However, exceptions to this policy can be made during wars; national emergencies; and threats to national security, human health, and the safety of vessels, aircraft, and platforms (Executive Order 13089, 1998). DoD has implemented a number of actions to comply with natural resource and environmental protection laws, and has developed programs to protect and enhance coral reef ecosystems. These efforts include developing geographic information system (GIS) planning tools, coral surveys to evaluate impacts from bombing exercises, assessments to determine the impact of amphibious training exercises on reef ecosystems, pollution and oil spill prevention programs, and invasive species management and effective land management programs (Defense Environmental Network and Information Exchange, https://www.denix.osd.mil, Accessed 2/14/05).

Oil and Gas in Coral Reef Ecosystems

The introduction of oil and other hydrocarbons into the marine environment can have serious consequences for coral reef ecosystems. Whether from chronic or episodic oil spills or from activities related to the exploration, production or transport of energy resources, oil can impact reefs through physical breakage, sedimentation and smothering, toxic contamination by heavy metals, and by inhibition of growth and recruitment. Sources of oil entering the marine environment vary. Summary information for North America is provided in Figure 3.12.

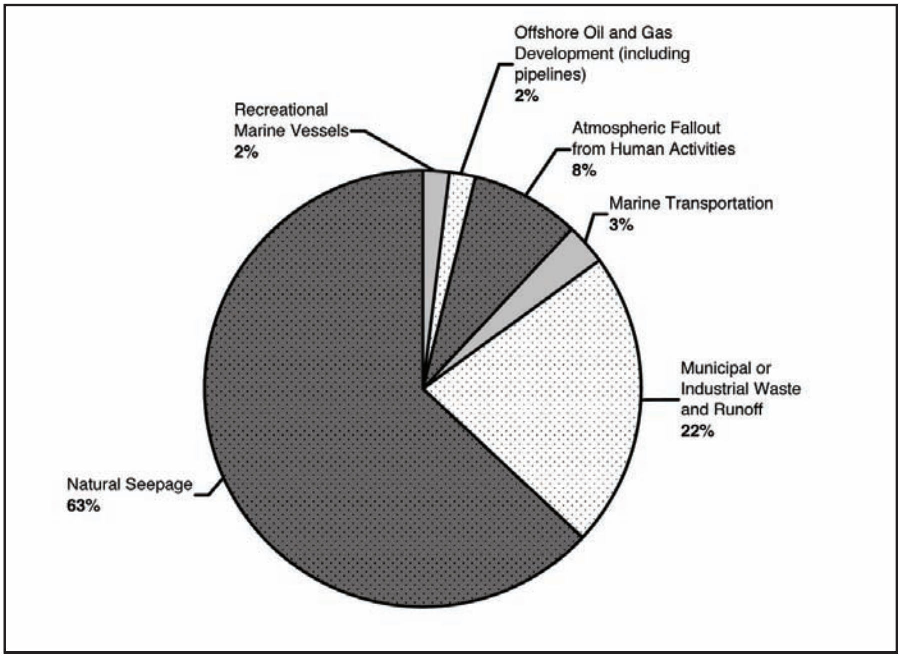

Figure 3.12. Sources of oil entering the marine environment of North America. Source: Minerals Management Service, 2002.

Once introduced, oil tends to persist in sheltered tropical coastal environments. Because of the difficulty of navigation in shallow-water coral reef environments, cleanup following a spill is often extremely difficult. Booms and skimmers can be used in lagoon areas when the oil is on the surface, but these responses become less useful over time as the oil combines with mineral particles in the water and sinks or is churned into the water column during inclement weather. The use of dispersants is often discouraged in shallow-water areas because they cause the oil to sink to the bottom where it comes into contact with sensitive reef habitats. Reduced water circulation in nearshore areas hinders natural dissipation by currents. When spills occur in shallow-water coral reef ecosystems, the best option may be to let natural processes handle the task of removing oil from the fine sediments of mangrove forests, seagrass meadows, and complex reef frameworks (Corredor et al., 1990; Guzman et al., 1994). Oil spill recovery in shallow-water reef ecosystems can require decades. Five years after a major oil spill on a Panamanian reef (April 1986), scientists found that surviving colonies of the four most massive species of reef-building corals were still experiencing extensive, chronic effects on vital processes (Guzman et al., 1994).

Several studies have been undertaken to determine the impact of oil on the physiology of coral reef organisms (reviews in Shigenaka, 2001). Laboratory experiments have demonstrated that exposure of coral species to oil can result in decreased growth, reproduction, and colonization capacity, as well as other negative effects on feeding, behavior, and mucous cell function (IPIECA, 1992). A field study in the Gulf of Eilat, Red Sea demonstrated that repeated discharges of oil onto a coral reef caused many changes to the reef system as a whole, and in particular damaged the reproductive system of scleractinian corals (Rinkevich and Loya, 1979).

In southern Florida, Dustan et al. (1991) evaluated the impacts of drilling wells on reef building corals, gorgonians, sea grasses, macroalgae, and reef fishes. Primary impacts included physical destruction by drilling machinery and the accumulation of drilling debris, although no organisms appeared to be damaged by drilling fluids or cuttings. The results implied that exploratory drilling, in light of present technology and stringent dumping regulations, may be achieved without leaving lasting impacts; however, no conclusions could be drawn from this study relative to the drilling production wells (Dustan et al., 1991).

In the North Sea, Olsgard and Gray (1995) assessed the spatial and temporal effects of production discharges on benthic fauna along contamination gradients. Results suggested that discharges reduced abundance of benthic fauna, many of which were key prey species for bottom-living fish. The fauna that became established in the contaminated sediments was considered less valuable as a food source for fish populations.

In addition to spills, exploration for offshore oil and gas reserves has the potential to have major impacts on marine ecosystems. Petroleum resources are difficult to find, and the process of locating, recovering, transferring and transporting them can pose a significant potential hazard to species living in the surrounding area. In the early stages, exploration for oil and gas involves seismic testing which involves emitting loud booming shock waves in order to determine what lies under the seafloor. The impacts of seismic testing on marine organisms are not well understood (The Ocean Conservancy, 2003). Once oil and gas reserves are located, energy exploration and production requires platform installation; dredging; drilling; the discharge of liquid, solid, and gaseous wastes and drill cuttings; noise and light pollution; and polluted air emissions. These impacts, in addition to the physical effects related to the movement of ships and equipment, can all present significant threats to the environment where the activity is taking place (http://earthsci.org/energy/gasexpl/exproil.html, Accessed 6/25/04).

The primary drilling areas in the U.S. Exclusive Economic Zone that occur near reef ecosystems are in the Gulf of Mexico, where major development has resulted in the installation of 6,500 production platforms and over a 160,900 km of pipelines and other infrastructure. Numerous wells, platforms and pipelines surround the Flower Garden Banks National Marine Sanctuary (FGBNMS) in the northwestern Gulf of Mexico (see Chapter 8), and one oil production platform even lies within the boundaries of the sanctuary, less than 1.6 km from the East Flower Garden coral cap. Fortunately, FGBNMS managers report that no major spills or impacts have occurred to date within sanctuary waters.

Because oil and gas development is such a major activity on the outer continental shelf in the Gulf of Mexico, the U.S. Department of the Interior's Minerals Management Service (MMS) has supported mapping and study programs of the Flower Garden Banks since the early 1970s to determine how to mitigate environmental impacts of oil and gas exploration. Information from these studies has supported MMS's belief that lease stipulations can minimize the potential impact of discharged contaminants to reef communities in the area. One such important stipulation requires shunting of drill cuttings so that they are deposited within 10 m of the bottom and not further up in the water column (MMS, http://www.mms.gov/eppd/compliance/13089/banks.htm, accessed 6/25/04).

Furthermore, removal of the enormous platforms, which weigh thousands of tons, is nearly impossible without the use of explosive materials. Gitschlag and Herczeg (1994) conducted one of the few known observations of fish mortality following such explosive activity. They reported that one event killed as many as 51,000 fish (larvae and juveniles were not counted). Removal of structures may also decrease the availability of habitat for fish that utilize the sites as artificial reefs (Patin, 2004, http://www.offshore-environment.com/abandonment.html, accessed 6/24/04).

Other Threats

Crown-of-Thorns Starfish Outbreaks

The COTS (*Acanthaster planci*) is a species of echinoderm found throughout the Indo-Pacific region (Figure 3.13). COTS feeds on several common species of hard coral, particularly *Acropora* spp., showing a clear preference for tabular forms and those corals that are least well defended (De'ath and Moran, 1998; Pratchett, 2001). They reproduce sexually with synchronized release of gametes and have a remarkable ability to regenerate damaged parts. COTS is preyed upon by several species of fish including triggerfish (Balistidae), and pufferfish (Tetradontidae), and a few large crustaceans and mollusks. At relatively low densities, the starfish are considered to play an important role in maintaining high diversity on coral reefs (Aronson and Precht, 1995). At many locations, however, populations periodically increase to levels that result in the degradation of coral reefs. Aggregations of hundreds of thousands of individuals have been reported across the Indo-Pacific, including Australia's Great Barrier Reef, Fiji, Micronesia, American Samoa, the Cook Islands, the Society Islands, the Ryukyu Islands (Japan), Hawaii, Malaysia, the Maldives, and the Red Sea. The rate of recovery after a major outbreak is highly variable, with full recovery estimated to take decades or even many hundreds of years (Sano, 2000; Lourey, 2000).

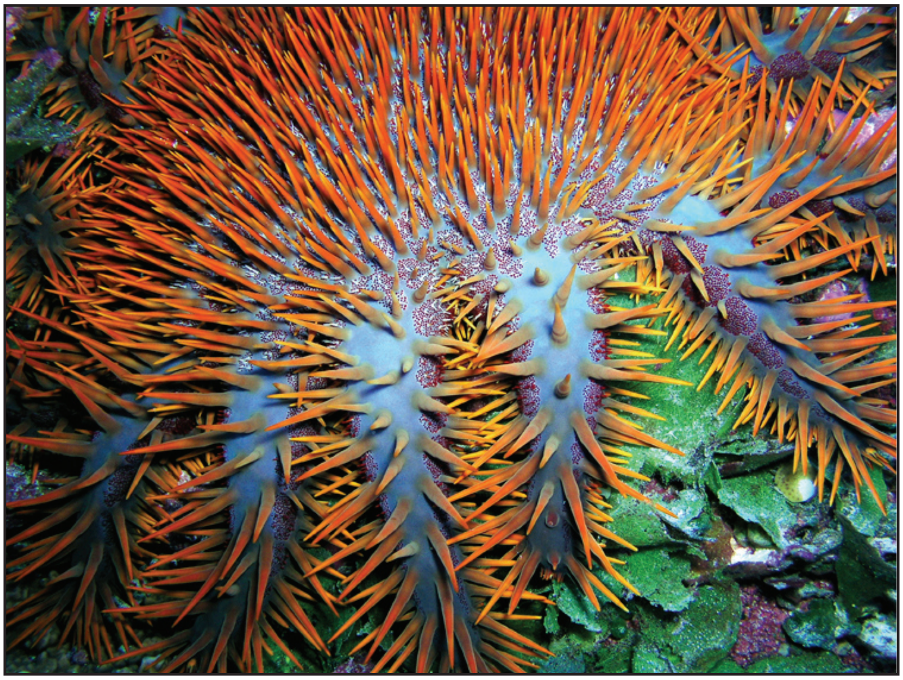

Figure 3.13. A closeup of a crown-of-thorns starfish, *Acanthaster planci*, on a reef in the PRIAs. Photo: J. Maragos.

A number of environmental factors have been considered causative in COTS outbreaks, including hurricanes, nutrient input, and overfishing (Birkeland, 1982; Ormond et al., 1991). The level of impact from human activity is still unclear since outbreaks have also been reported in remote areas with very little human activity. Nevertheless, stressors generated through human activity are likely to influence the trajectory and rate of post-outbreak recovery.

Outbreaks of other echinoderms, such as spiny sea urchins (Echinoidea), can also adversely impact coral reef ecosystems through excessive erosion of coral substratum, removal of newly settled corals, and intense herbivory (Sammarco, 1982; Carreiro-Silva and McClanahan, 2001). Damage to coral reefs due to high density populations (12-100 urchins/m^2) of urchins have been occasionally reported in U.S. waters including Hawaii, USVI, and the Marshall Islands.

Earthquakes and Volcanoes

Many islands in the Pacific and Caribbean were formed and transformed through tectonic and volcanic activity. In fact, coral reef atolls are formed through the erosion and subsidence of volcanoes and the subsequent gradual upward growth of coral reefs (Darwin, 1842). Volcanic eruptions can have important direct and indirect consequences for coral reefs. The eruption of Mt. Pagan, CNMI in 1981 resulted in extensive damage to coral communities due to scouring by lava and smothering by volcanic ash, although observation of new coral recruits indicated recovery occurring within two years of the eruption (Eldredge and Kropp, 1985). Similarly, rapid recovery was observed after high coral mortality as a result of burial by ash after the 1994 eruption of Rabaul Caldera in Papua New Guinea (Maniwavie et al., 2001). Major eruptions can also impact coral reefs many thousands of kilometers away through a complex sequence of events (Figure 3.14). For example, the 1991 eruption of Mount Pinatubo in the Philippines led to a short-term atmospheric cooling throughout the Middle East during the winter of 1992. This abnormal cooling resulted in deep vertical mixing in the Gulf of Eilat and excessive nutrient upwelling, which in turn, triggered algal blooms causing widespread coral death (Genin et al., 1995). However, cooled larva flow can also create new habitat suitable for the settlement and growth of corals and other organisms.

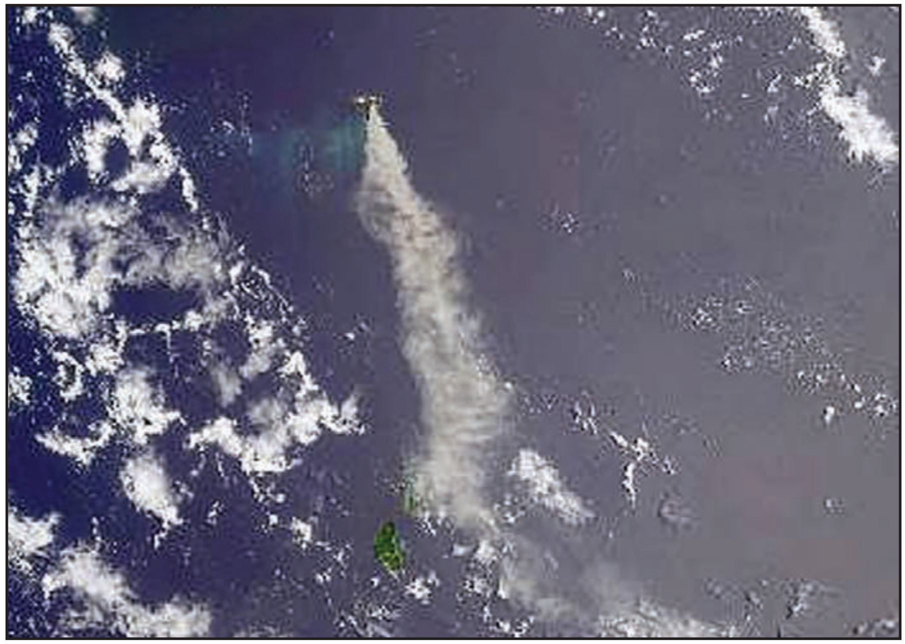

Figure 3.14. In the past few years, eruptions of the volcanic island Anatahan in CNMI have deposited tons of ash on nearby reefs and temporarily closed international airports in Saipan and Guam. The latest major cluster of eruptions occurred in April 2005. Photo: NASA, MODIS sensor.

In 1993, an earthquake measuring 8.2 on the Richter scale caused collapse of some coral reefs around Guam and also destroyed some large coral colonies that had formed on unstable substrata (Birkeland, 1997). Earthquakes that uplift some areas while subsiding others, or even triggering catastrophic sedimentary events, are thought to be important factors in the present spatial patterns of fringing reefs in the Gulf of Aqaba, Red Sea (Shaked et al., 2004). In the Hawaiian archipelago, a high frequency of deep earthquakes combined with submergence and rising sea-level may explain the absence of coral reefs in some locations around the island of Hawaii.

Cable-laying Operations

There has been a rapid increase in the need for submarine cables, particularly fiber optic cables, to support the telecommunications industry. Cable-laying operations and the movements of unsecured cables have been found to disrupt and destabilize benthic structure (Sultzman, 2002). The impact of laying a cable on benthic habitats will depend on the location of landing points, route chosen, and installation process. In some instances, sand channels through reefs have been used, but damage has occurred where cables have been laid directly over corals. Coral transplants and artificial reef modules have been used to replace lost hard coral, yet little is known about the effectiveness of these methods. Furthermore, few restoration efforts have considered damage to non-scleractinian components of the biota.

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