A Literature Review of Sources and Effects of Non-extractive Stressors to Coral Reef Ecosystems

Southeast Florida Coral Reef Initiative
Fishing, Diving and Other Uses Focus Team
Local Action Strategy Project 19 Phase I

Southeast Florida Coral Reef Initiative
Acting above to protect what's below.

A Literature Review of Sources and Effects of Non-extractive Stressors to Coral Reef Ecosystems

Final Report

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Introduction

This report describes the collection of previously published, peer reviewed, and other technical literature that addresses the impacts of non-extractive stressors to coral reefs. While the deleterious effects of over-harvesting and related, direct stressors (see, for instance, Bellwood et al., 2004; Pandolfi et al., 2003; and Knowlton, 2003, among others) have been well documented, it has become increasingly clear that factors affecting coral reef resilience often consist of a number of chronic abuses (Bellwood et al., 2004; Hughes and Connell, 1999) that contribute to elevated levels of coral reef stress. Whether this is manifested as a decreased ability to compete with macroalgae under nutrient enriched conditions (Szmant, 2002), a prevalence of coral diseases (Knowlton, 2003), or susceptibility to bleaching Bellwood et al., 2004), the net result has been a steady decline (or, as described by Pandolfi et al., 2003, a trajectory towards extinction) of coral reefs. For example, hard coral cover in the Caribbean, has declined on average between 10-50%, with up to 80% loss at locations suffering severe decline, during the last three decades (Gardner et al., 2003).

Coastal and marine tourism has increased considerably in coral reef areas across the world (US Ocean Commissions Report, 2004, Uyarra et al., 2005; Harriot, 2002; van't Hof, 2001; Leeworthy and Wiley, 1996; and Davis and Tisdell, 1996, among others). Tourism represents a spectrum of uses, many of which compromise the health and sustainability of the very resources that attract visitors. While many of the activities in which visitors (and residents, alike, in many developed nations) participate are often described as non-extractive, the activities have been shown to have impacts. When these impacts are isolated and occur over a short period of time (Hughes and Connell, 1997), they may not be perceived as being particularly insidious; however, when coupled with a system undergoing multiple and chronic stressors, their effects become increasingly deleterious. Thus, in utilizing the literature compiled for this report, it is recommended that the non-extractive stressors be considered as part of a compendium of impacting activities that are collectively undermining the health and sustainability of coral reefs.

This report is organized as a series of descriptive summaries on each use type, ranging from direct impacts, which result from anthropogenic interactions in coral reef environments, to indirect impacts, which are comprised of the effects of sundry, off-site activities. Due in part to the purpose of the study, summaries on direct impacts are represented by all available literature, whereas references describing indirect impacts are less complete. Therefore, literature on the impacts to coral reefs associated with diving and snorkeling are more completely represented than the literature on nutrient enrichment (which is included most often when it is associated with non-extractive stressors, e.g. coastal tourism development).

Finally, as this is solely a literature review, the material presented (and collected in the accompanying database) represents a majority of the non-extractive stressor literature produced on the topic in the past two decades. As such, it is not necessarily prioritized in any particular manner or for any singular purpose. Where there exist disagreements over impacts, both sides of the argument have been provided in the form of the original works (to the extent possible, in that no opinions or essays are included, and the focus is on primary research literature).

Methodology

The methodology of the project consisted of a literature compilation and review, which was facilitated first by contacting the SEFCRI Teams, the SEFCRI LBSP TAC, the Coral List-serve, and other groups/sources to solicit information on sources for the literature review. This step was completed in the late summer of 2006.

The second step involved a literature survey on various databases (such as the Aquatic Science and Fisheries Abstracts, Web of Science, and others) to compile a preliminary list of abstracts, authors, and references. Another search was conducted on United Nations (e.g. Regional Seas Programme, UNESCO, and other websites) and non-governmental organizations' resources on the Internet to obtain information on ongoing and recently completed activities, including project updates, reports, and bulletins, among others.

The third step involved the organization of literature both according to geography and to the type of impact that the research analyzed, such as stressors described from a biological perspective, discussion on the socioeconomic limits of acceptable change, and economic opportunities and challenges in implementing management to reduce non-extractive stressors. This separation, based on geography and major theme, would allow for expanded analyses, including (but not limited to) the comparison of regional studies, regional-thematic congruence (or lack thereof), differences in non-extractive impact type (e.g. boating vs. diving), and inter-site evaluations, in terms of types of recommendations provided to ameliorate damages and/or improve management.

The fourth and final step was the development of the draft report to be submitted for all comments, along with the accompanying literature database.

Non-extractive stressors and their impacts on coral reefs

Non-extractive stressors can be defined as the (mostly unintended) effects of mainly (but not exclusively) recreational activities on coral reefs; such stressors do not include the direct impacts resulting from coastal construction activities, which are considered as extractive impacts (in that the activities are generally undertaken with an understanding on the impacts that they will have on affected resources), but the report does address the resulting, indirect impacts, such as sedimentation and turbidity. For the purposes of this report, the stressors are described under the following headings: Direct impacts; indirect impacts; and synergistic or multiple stressors. Direct impacts include the effects of trampling, snorkeling, and diving, the effects resulting from structures over or near coral reefs, impacts from fish feeding and other wildlife interactions, and damages caused by vessels, including anchoring, propeller damage and scarring, groundings, and chemical damage from the release of anti-foulant paints, petrochemicals and crude oil. Indirect impacts refer to those effects that are associated with land-based activities, including nutrient enrichment and sedimentation. Finally, synergistic or multiple stressors include those literature sources that address how non-extractive stressors act in combination with extractive and other stressors in compromising coral reef health. It should be noted that several of the literature sources discuss more than one of these topics, and thus such sources may be utilized in multiple sections.

Direct impacts

Trampling, diving, and snorkeling

Direct impacts of non-extractive stressors and coral reefs have been well studied for the past three decades. Among the first studies conducted on the effects on recreational activities on coral reefs was a reef trampling experiment by Woodland and Hooper (1977). By walking on transects along a reef in the Great Barrier Reef, it was determined that trampling led to precipitous declines in coral cover (from 41 to 8%), and that only the most robust coral colonies withstood the damage. Woodland and Hooper concluded, in a pair of now prescient statements, that "coral reef marine parks will not long remain in a virgin state unless considerable caution is exercised in their uses…(and) increasing human population pressures on coral reefs will lead to deleterious changes to the reefs" (p. 4).

Two additional studies (Liddle and Kay, 1987; Kay and Liddle, 1989) in the subsequent decade continued research on the effects of trampling on coral reefs. The earlier study described the differential effects of trampling on branching and massive corals. It demonstrated that the response and rate of recovery of individual corals to trampling varied by species. The latter study showed that the outer reef flat coral species were more susceptible to coral reef damage (up to 16 times more vulnerable) than were reef crest species and recommended appropriate management measures. Neil (1990) considered the impacts of sedimentation by reef walkers and how resuspension and particle size affects corals. The study found that chronic resuspension can lead to elevated coral stress and that the grain size of the sediment resuspended may exacerbate sedimentation loads that corals (at least some species) can tolerate.

Since then, Rogers et al. (2003a, 2003b) have evaluated the effects of trampling on Hawaiian corals. Their studies determined the corals most likely to break under trampling were those adapted to low energy environments, and that all coral forms broke during in situ experiments suggesting no corals are entirely resistant to trampling. From a stress perspective, Rogers et al. (2003a) concluded that while trampling may not result in high mortality rates, it does induce a sub-lethal stress which may result in lower reproductive output. Rogers et al. (2003b) also found an inverse relationship between coral cover and use levels. At sites where they tested in situ survivorship of transplanted corals, they demonstrated coral restoration should not be considered in high use areas unless the impacts leading to the decline are stopped.

Several other studies (Casu et al., 2006; Brown and Taylor, 1999; Keough and Quinn, 1998; and Povey and Keough, 1991) have evaluated the effects of trampling on algal mats and coralline algal turfs in highly visited areas. As in coral reef trampling experiments, such research has found that trampling on algal mats, especially if it is chronic, may lead to declines in algal mats and epifaunal densities and, in other cases, may promote herbivory.

Similar to trampling, swimming (namely, snorkeling) and diving activities can result in direct breakage of corals. Whereas several earlier studies (Tilmant and Schmahl, 1981; Tilmant, 1987; and Talge, 1991) suggested that the then present rates of diver use generated minor damage compared to hurricanes and other storms on coral reefs, later studies (Rogers, 1998a and 1998b) in areas where use was more prevalent, and anthropogenic damage could be ascertained and differentiated from natural damage, reported deleterious levels of uses and activities. Tilmant and Schmahl (1981) completed their study in Biscayne National Park, Florida, over a three year period. Although they concluded natural damages were more prevalent than anthropogenic ones, the authors did find that there was a significant correlation between reef use and incidence of physical damage; suggesting increasing use would undoubtedly result in greater damage.

Talge, in a pair of studies (1991 and 1992), reported on diver interactions with corals in Looe Key National Marine Sanctuary, located off Big Pine Key in the lower Florida Keys. In the 1991 paper, she noted that the most frequent interactions divers had with corals were kicking corals with their fins and using corals to push off. She also found that divers without gloves had fewer interactions than those using gloves, and that the average number of interactions was 10 per dive (without gloves). Talge also reported that touching had no long-term impact on the 11 species of corals she studied, and that based on an average 10 interactions per diver, it would lead to 4% of the live coral in heavy dive areas being impacted per week. A key discovery by Talge (1992), later supported by other studies (Harriot et al., 1994, and Rouphael and Inglis, 1999) was that only a small percentage of divers were responsible for a majority of the interactions.

Two Red Sea studies conducted in the 1990s assessed visitor impacts upon to the coral reef ecosystem in Egypt and Israel. By this time, both countries had developed into major tourist destinations, receiving several hundred thousand coastal visitors per year. Riegl and Velimirov (1991) found breakage as the most common form of damage at coral reef sites in Eilat (Israel) and Hurghada (Egypt). Both areas were considered high use areas (diver and snorkeler sites) and showed significantly higher rates of damage than low use areas. In 1992, Hawkins and Roberts reported finding significantly more damaged corals, especially branching corals, at heavily dived sites compared to lightly used ones near Sharm-el-Sheikh, a dive resort in Egypt's Red Sea. They suggested that as new dive sites are opened, they suffer an initial high rate of damage, followed by a consistent level of deterioration, that eventually stabilizes, leaving the reef in an asymptomatic state of decline. A later paper by Riegl and Riegl (1996) used the concept of estimating damage to coral reefs in South African marine protected areas in order to suggest which areas most vulnerable and should be closed altogether from all uses (much like the Eilat no-use zone established in the 1990s (Epstein et al., 1999).

Dixon et al.'s (1993) paper introduced the concept of a diver carrying capacity, based on a study conducted of divers on Bonaire's coral reefs. The study determined that areas of high dive use showed lower coral percent cover and that species diversity was higher at low use sites. The study also documented a direct relationship between coral damage and distance from the mooring buoy (where divers often enter and leave the site), with coral cover and species diversity increasing with distance from the mooring buoy. Based on these findings, the authors developed a threshold for dive use, predicting that diver impacts would become apparent once use exceeds a threshold of 4,000 – 6,000 dives per site, per year. This total was the first estimate for diver carrying capacity in coral reef dive sites. Jameson et al. (1999), Hawkins and Roberts (1994) and Davis and Tisdell (1995, 1996) utilized the carrying capacity concept to derive estimates and make recommendations for other areas (including

Egypt and Australia), and the latter suggested combining ecological goals with economic objectives to determine capacity totals for individual sites.

A 1997 paper by Medio et al., based on research conducted with divers in the Egypt's Ras Mohammed National Park in the Red Sea, found that pre-dive briefings made a significant, positive difference in terms of diver-related damage. The authors suggested more educational strategies should be implemented across dive sites (with dive operations as the conduit to provide the information); and by doing so, the briefings may raise diver carrying capacity for individual sites. Other studies, by Zakai et al. (2002), Tratalos and Austin (2001), and Hawkins and Roberts (1997) also reported similar results, although other experiments (Barker and Roberts, 2004; Rouphael and Inglis, 2001) did not have the same findings. In fact, certain research suggests that instead of pre-dive briefings, certification professionals should focus on improving diver skills and promote specialization, the latter of which was shown to be related directly to environmentally responsible behavior (Thapa et al., 2006).

How activities such as underwater photography and wildlife viewing changes diver behavior, has also been evaluated. Some studies (Serour, 2004; Walters and Samways, 2001) found that photography led to divers damaging coral more frequently, however, Rouphael and Inglis (2001) determined that novice photographers did not damage corals more frequently than non-photographers. A more recent study, by Uyarra et al. (2007), determined that divers tend to make more contact with coral when viewing cryptofauna in Bonaire, and contact increased among divers who participated in underwater photography. Wildlife interactions and viewing represent another form of non-extractive impact, which is considered further in this report, but it bares emphasizing here that specialized dive trips (for viewing charismatic megafauana, and cryptic and rare species) are an important and growing component of coastal and marine tourism (Orams, 2002; Shackley, 1998; Davis et al.,1997).

A few studies have evaluated the effects of intensive snorkeling on coral reefs. Generally, snorkelers are reported to have few to no impacts on coral reefs (Zubillaga et al., 2003; van't Hof, 2001), due mainly to most snorkelers floating above the corals on the surface; with damage usually being limited to shallow water areas where snorkelers can either stand directly on or kick corals (Rogers et al., 2003; van't Hof, 2001; Plathong et al., 2000; Stepath, 2000). Rogers (1998) reported high visitation rates to Trunk Bay, a self-guided snorkeling trail set up in the Virgin Islands National Park, and found that the trail had been impacted substantially by snorkelers standing on, or breaking corals, or removing corals for souvenirs. Plathong et al. (2000) found that opening up snorkeling trails in the Great Barrier Reef led to considerable damage, especially around the interpretation areas. In this and several of the other studies, the authors recommend managing snorkeler impacts by keeping snorkelers in designated, high-intensity zones (Roman et al., 2007), by establishing ecotourism zones that host fewer users (Roman et al., 2007) and by rotating sites (Plathong et al., 2000). Others (Hawkins and Roberts, 1997; Medio et al., 1997; Allison, 1996) emphasize the need for snorkeler and tour operator education and proper instruction as a means by which to reduce coral damage.

Finally, diving activities in ecosystems other than coral reefs tend to result in damage as well, especially in areas that contain fauna sensitive to touching or other forms of contact. Sala et al. (1996) reported that population densities and colony sizes, among other parameters, of bryozoans in frequently dived sites in a Mediterranean marine protected area were significantly lower and located in more cryptic areas than in less frequently dived sites. The authors considered direct contact by divers with the substrate to be the primary cause. In another diver-related study in the Mediterranean, a nine-year experiment determined that mortality rates of gorgonians were 300% higher in heavily dived sites than at lightly dived ones (Coma et al., 2004). Lloret et al. (2006) developed a topographic map for a Mediterranean MPA to identify areas that may result in the highest diver-related damage and concluded that management is necessary as divers tend to concentrate in shallow, sensitive areas.

Often, the socioeconomic conditions, derived from user attitudes, perceptions, and beliefs (APB) are as important as the actual impacts that diving, snorkeling, and related recreational activities have on coral reefs; users (and other stakeholders) can often establish other baselines on resource conditions than those determined by biophysical research. As shown by Leujak and Ormond (2007), the nationality of visitors to the Red Sea coral reef resorts can establish a baseline coral reef quality as much as could other socio-demographic indicators, such as diving experience and capability. Moreover, as shown as by that research, the baseline can shift significantly as the nationality of tourists, among other socio-demographic and socioeconomic factors, changes at a coral reef location. Inglis et al. (1999) found that diver experience influences views on crowding, and Letson et al. (2004) reported that diver views on resource quality may be related in part to a coral reef site's reputation.

Other studies have found that visitors may be unwilling to return to certain sites based upon the perception that non-extractive impacts, such as physical damage and crowding, are leading to negative tourism experiences. A study conducted with liveaboard and single day divers in Phuket, Thailand (Dearden et al., 2007) determined that divers who witnessed diver-related damage were less willing to return for another visit. Similarly, snorkelers from a study in a marine park in Thailand (Roman et al., 2007) showed a lower satisfaction for sites that exhibited high coral mortality. While these results suggest that divers and snorkelers may prefer pristine (or less impacted) coral reef sites, it should be emphasized that such preferences are often a function of demand. For example, as described by Uyarra et al., (2005) the majority of visitors traveling to Bonaire to snorkel and/or dive believed it was pristine and that incidence of coral bleaching was among the most important factors influencing their decision on whether to return for a another visit or experience (Inglis et al., 1999).

Another non-extractive impact that may often afflict coral reefs is created by structures, either moored on or around coral reefs, that can result in excessive shading and diving pressure (especially where the moored structure may promote other uses). As shown by Dixon et al. (1993), coral reef damage tends to be concentrated around mooring buoys. Divers often commence and terminate their dives near they mooring buoy and the study documented a direct relationship in which diver damaged decreased with distance away from the mooring buoy. Rogers (1979) reported that the shading of corals in a five-week shading experiment led to an alteration in coral reef community and structure, which included the bleaching of some coral species and even the mortality of others. Management for the Great Barrier Reef Marine Park has long allowed the use of permanent structures such as pontoons. The pontoons are situated along popular dive and snorkel sites and help facilitate tourism and reduce use on the sensitive reef islands (Harriot, 2002; Kapitze et al., 2002).

Although shading may produce localized effects on coral reef community, in general, it is accepted that mooring buoys and other structures that minimize the use of anchors and other vessel-based mooring systems actually result in lower, overall impacts and may help concentrate use and thus prevent diffused impacts across the reef (Kapitze et al., 2002; Salm et al., 2000; NOAA, 1995; van Breda and Gjerde, 1992).

Wildlife interactions, such as fish feeding and encounters with charismatic or rare megafauna are increasingly popular activities that often co-occur often with diving and snorkeling. Orams (1999) summarizes the series of interactions and their potential impacts on both the visitors undertaking these activities and the fauna that are the subjects of these interactions. Within coral reefs, it should be noted that wildlife interactions have the potential to alter trophic dynamics (Sweatman, 1996), suppress spawning aggregations (Martin, 2001), and otherwise lead to reef damage that reduces coral reef fish populations and/or compositions (Ebersole, 2001; Hawkins et al., 1999). Other studies have considered how diving and snorkeling may affect transient (or migratory) species, such as whale sharks (Quiros, 2007) and marine mammals, and how creating shark feeding frenzies may enhance shark aggressiveness towards humans (Avelizon, 2000).

The results on fish feeding have been reported by Sweatman (1996) in a study conducted at dive sites off tourist pontoons in the Great Barrier Reef. Overall, the study concluded that fish feeding on two carnivorous species presents very few impacts, if any, and the two main concerns (that the fish were forming aggregations around the pontoons and that the fish were changing their feeding behaviors) were not realized. Also, the research findings suggested that any reproductive advantage gained by the tourist-fed fish may be minimized by the large population of the species in the region, and that management measures should focus on limits on food quantity and quality. Martin (2001) reported that while divers and snorkelers have the ability to impact fishery spawning aggregations (e.g. through fish feeding, which may attract larger predatory fishes to such sites), that management measures in the Great Barrier Reef Marine Park should include the placement of moorings at a safe distance away from fishery spawning aggregation sites, and the effects of fish feeding on aggregations should be monitored. The effects of diver-related damage on coral reef fishes were evaluated in high-use dive sites in Bonaire by Hawkins et al. (1999). While the study found higher rates of damage in such sites compared to other, low use sites, it did not observe any effects on coral reef fish communities. Ebersole (2001), conversely, reported significant and almost permanent shifts in coral reef fish complexes on coral reefs impacted by vessel groundings, which result in more severe forms of coral reef damage.

The impacts of diving and snorkeling (and viewing, in general) of megafauna that are either conditioned to frequent a site (such as the stingrays of Stingray City in the Cayman Islands (Shackley, 1998) or shark feeding locations in the Bahamas) or are opportunistically intercepted during their movements through coral reef or other dive destinations (whale sharks in the Philippines (Quiros, 2007), dwarf minke whales in the northern Great Barrier Reef Marine Park (Valentine et al., 2002), and sperm whales in New Zealand (Dawson and Slooten, 2006)) are less well understood. Shackley (1998) reports that the concentration of several generations of stingrays in dive and snorkel sites in the Cayman Islands are most likely a result of conditioned behavior, but the overall effects of the potential attenuation of stingrays from other, more natural environments is not well studied. In the case of shark feeding frenzies in which carnivorous species are attracted by chum, some areas (such as

Florida, U.S.) have prohibited the practice, arguing that it may lead to more aggression among sharks towards humans (FWC, 2001). Dive operators in other locations, such as the Bahamas, still offer shark feeding frenzy trips. As for larger, often transient species, such as whale sharks, operators have developed codes of conduct (Quiros, 2007) aimed at minimizing unacceptable diver and snorkeler behavior towards the fish and promote cooperation within the operators (Davis et al., 1997).

Vessel-based impacts: Anchoring, scarring, grounding, alien species, discharges and dumping

Direct contact by vessels is an, important non-extractive impact. Vessels ranging from small shallow-draft, aritisanal boats to enormous, deep-hull cruise ships can directly impact the environment through chronic scarring (mainly in seagrasses), anchoring, catastrophic groundings, and the associated release of oil and anti-foulants. Coral reefs, due to the fact that they are often located in very shallow water and because of their generally slow growth rates, are highly susceptible to vessel-based damage.

Anchoring, is the most common and best studied effect of vessel damage to coral reefs. . Anchoring can result in coral breakage and fragmentation when anchors are dropped and weighed, and coral reef sites that host large concentrations of vessels can often suffer degradation from anchor damage. Rogers (1988) reported that a large percentage (14%) of vessels in the Virgin Islands National Park anchored on coral reef habitat and over a quarter of these vessels have some impact on corals. Davis (1977) found that up to 20% of an *Acropora cervicornis* (staghorn coral) zone had been destroyed by anchors in the Dry Tortugas, located at the southwestern edge of the Florida Reef Tract. While anchoring impacts have been partially alleviated due to user and/or agency funded mooring buoys in developed nations (NOAA, 1996; van Breda and Gjrede, 1992; Halas, 1983), anchoring by small vessels on coral reefs remains a chronic problem in developing countries (Wilkinson, 2002). This is especially problematic because a majority of the world's coral reefs are located in such countries (e.g.. Indonesia contains the world's largest percentage of coral reefs (Wilkinson, 2002)).

More recently, with the advent of cruise tourism in areas such as the Caribbean over the past two decades, and of vessel traffic in general (Thomas J. Murray and Associates, 2005; Hall, 2002), the impacts of large anchors on coral reefs have become an important issue. In 1988, the Florida Keys National Marine Sanctuary banned anchoring by vessels larger than 50 meters in length (which can have anchors and chains weighing up to 8 tons) over the Tortugas Bank, an area in the western section of the Sanctuary known to contain a rich diversity of deepwater (>30 meters) corals (NOAA, 1998). Rogers and Garrison (2001) evaluated the long-term impacts of a cruise ship anchor that scarred a coral reef in Virgin Islands National Park. The study determined that the reef had not recovered after a decade following the incident, and that both coral cover and the average size of coral colonies were significantly smaller in the scar area compared to adjacent, control sites. The authors concluded that because the 1-ton anchor had gouged out part of the coral reef's framework, it had left the site less complex and more instable, leading to conditions that inhibit recruitment.

Anchoring can also affect ecosystems located near, and associated with, coral reefs, namely seagrass communities. As seagrasses (and mangroves) are frequently found in conjunction with coral reef ecosystems and often serve as nursery and juvenile habitats for many coral reef fauna, impacts that affect the integrity and functionality of these ecosystems inevitably affect coral reefs. Rogers (1988) noted that a majority of anchoring in the Virgin Islands National Park occurred in seagrasses and Rogers and Beets (2001) reported that anchoring on seagrasses is among the reasons for marine degradation in the U.S. Virgin Islands. A study conducted on seagrasses in the Albrolhos Marine National Park in Brazil (Creed and Amado Filho, 1999) found extensive anchoring damage to seagrasses in concentrated tourism areas in the park and also reported that while the seagrasses themselves may recover over a short time span, the epiphytic algal communities show more seasonal and species-dependent recovery rates. Milazzo et al. (2003) evaluated the effects of different anchor types on seagrasses in a marine protected area in the western Mediterranean. They determined that more environmentally friendly anchors could decrease anchor damage on seagrasses, most damage tends to occur during the weighing stage. The authors also found that the use of anchor chains or ropes did not significantly affect the amount of damage that resulted. The authors concluded that while mooring buoys may represent a solution, boater education on the type of anchor and anchoring techniques would greatly minimize anchor damage in seagrasses (and other vulnerable areas). Apart from anchoring damage on seagrasses, vessels are responsible for propeller-related impacts that result in habitat degradation, fragmentation, and even complete destruction (in the case of blow-outs, where propeller scarring can lead to the total removal of seagrass patches, leaving behind unstable substrate). Sargent et al. (1995) calculated that 173,000 acres of Florida's 2.7 million acres of seagrasses show some level of propeller scarring; also, areas that show the highest concentrations of moderate to severe scarring as those that host the highest vessel traffic. Other studies examining the associated motile and infaunal species have determined mainly that seagrasses can withstand significant levels of scarring (greater than 30%), but that system-wide evaluations are necessary to determine overall impacts (Burfeind and Stunz, 2006; Bell et al., 2002; Bell et al., 2001).

The other, major category of vessel impacts on coral reefs is that of vessel groundings. As with anchoring, vessel groundings can be subdivided into large and small vessel groundings. The former occasionally result in catastrophic effects and can have associated impacts, associated with cargo and fuel spills, leaching of toxins, and even sinkings (where the vessel cannot be salvaged). Small vessel groundings generally have lesser impacts but almost inevitably present a chronic problem, in that due to the higher number of smaller vessels, these groundings are by contrast common in many coral reef environments (Lutz, 2006). The immediate and most obvious impact of large vessel groundings is the physical destruction of the coral reef framework. A review by van't Hof (2001) described the varying extent of physical damage that large vessels can have, based in part on the coral cover in the collision area, the capacity of the region to be able to address the collision, and weather conditions following the grounding. Dependent upon the magnitude of these factors, groundings may result in an impact of a few hundred meters to several hundred thousand meters of coral reef. The long-term impacts of large vessel groundings depend on the type and amount of damage incurred to the coral reef affected, as well as other, ambient conditions (e.g. level of recruitment, amount of competition that prevents effective restoration, etc.). Ebersole (2001) found that coral reefs in the Florida Keys affected by ship groundings resulted in flattened impact areas that then attract small-mouthed opportunist fishes, rather than grazers. This in turn favors algal growth over coral recruitment and development and without the recovery of a complex coral reef structure, the grounding sites are not effectively restored to their natural state. In a self sustaining cycle, the assemblages of coral reef fishes that shape and rely on coral reefs never return to the grounding sites, even after as much as 100-200 years following the collision. By contrast another study conducted on a vessel grounding (and sinking) off the Galapagos Islands in a rocky seafloor habitat found no shift in fish assemblages, at least over the short-term, or 15 weeks after the incident (Edgar et al., 2003). Thus, as previously stated, vessel grounding impacts vary considerably, based on the extent of the initial damage and the prevailing biophysical conditions.

Small vessel groundings and their cumulative impacts on coral reefs are well referenced in the literature and are often cited as the primary source of damage (along with associated anchor damage) resulting from anthropogenic activities. Tilmant and Schmahl (1981) expressed greater concern over vessel groundings in Biscayne National Park than they did over diving impacts (which they considered to be minor at that level of use). Tilmant (1987) predicted that vessel impacts would become a growing source of coral reef impacts as coastal populations increased. Ginsburg et al. (2001) also noted the impact of small vessel groundings on coral reefs in the Florida Keys but concluded that for massive corals in the upper Florida Keys section of the Florida Reef tract, the amount of dead coral tissue was more a result of the natural, marginal conditions in which the corals live than anthropogenic impacts. Another study (Lutz, 2006) evaluated the condition of 315 shallow water, coral colonies from 49 sites in the upper Florida Keys and determined that almost 60% of the sites and 80% of the coral heads showed vessel-based damage. Moreover, the study found that mooring buoys did not affect the frequency of damage incidence, and that sites near metropolitan areas and high vessel use were the most impacted. Most studies concerning vessels and vessel-based damage recommend boater education (Lutz, 2006; Milazzo et al., 2003; Sargent et al., 1995) as the only effective way by which to reduce small vessel groundings and related vessel impacts as coastal populations continue to grow in the future.

Another type of vessel impact in coral reef environments is noise pollution generated by motorized vessels and direct collisions with marine animals. Coastal, resident populations of dolphins have become largely accustomed to vessels and have been shown to change behaviors (that may be detrimental to their overall fitness) as vessels approach or pass within a certain distance (Lemon et al., 2006; Allen and Read, 2000). Similarly, dugongs in eastern Australia have been observed to stop feeding with approaching boats, resulting in a calculated maximum loss of 6% in the amount of time spent foraging (Hodgson and Marsh, 2007). These changes in behaviors, as well as mortalities resulting from collisions, are not directly related to coral reefs; but, where such organisms are part of a coral reef ecosystem, it is clear that increasing vessel traffic may in part alter the region's trophodynamics.

In addition to the potential damage caused by vessels directly to coral reefs, their associated habitats (such as seagrasses), and coral reef organisms, oil spills, the release of vessel by-products such as antifoulant paints and chemicals, and the leaching of metals can add to that damage. The effects of oil spills on coral reefs is well reviewed in the NOAA Office of Response and Restoration's 2001 technical document, "Oil spills and coral reefs", which describes the immediate and long-term impacts of oil pollution and presents a series of case studies. While oil can kill corals, oil spill impacts depend on the oil type and quantity, the species composition, and the nature of oil exposure. Chronic oil toxicity affects coral growth and development. The time of year when a spill occurs is also critical, as coral reproduction and settlement are both negatively affected by oil. Also, not all coral species have the same response to oil pollution. For example, branching corals are more sensitive to hydrocarbons than are massive corals. Finally, recovery rates vary considerably, based in part on the type and amount of oil spilled and the biophysical characteristics of the region.

Eliminating oil from coral reefs can be difficult, and options such as skimmers and booms and other mechanical methods are preferred over burning or chemical means (NOAA, 2001). Dispersants, chemicals containing surfactants that are used to disperse oil throughout the water column, can have deleterious effects on corals and especially larvae. A study from the Red Sea area (Epstein et al., 2000) determined that dispersants used to counter crude oil in the region are extremely toxic to coral larvae and concluded that the dispersants should not be used in the case of an oil spill near coral reefs.

Vessel groundings can also result in the release of chemicals other than oil, including anti-foulant paints. In a pair of studies conducted after the grounding of a freighter on a reef in the Great Barrier Reef Marine Park (Haynes et al., 2002; Negri et al., 2000), it was determined that a large section of the reef was contaminated by tributyl tin (TBT) laden anti-foulant paint from the vessel's hull, both during the grounding and then in the refloating operation. As this event occurred a few days prior to a coral spawn, researchers conducted an experiment in which they exposed coral larvae to TBT-contaminated sediment. The results showed that larvae contracted to a spherical shape and stopped swimming at lower concentrations, and at higher TBT concentrations, all larvae perished (Negri et al., 2000). Another study conducted recently settled corals and branchlets from two coral species with TBT contaminated sediments found that newly settled corals exposed to the sediments showed significantly higher rates of mortality than those in control groups (Smith et al., 2003). Given that the anti-foulant compounds may take up to six years to deteriorate, the authors recommended remediation and monitoring at the impact site to determine the long-term effects of the grounding.

While TBT is no longer commonly applied as an anti-foulant on recreational and other small vessel hulls, it is still used in larger, commercial tankers and freighters. An alternate, non-TBT containing anti-foulant that has been increasingly used since the 1990s (and introduced into the US since 1998) contains herbicidal product called Irgarol 1051 (Carberry et al., 2006). It has been shown to inhibit photosynthesis at low concentrations and reacts against many coastal flora, including mangroves, seagrasses, and coral symbionts; moreover, it is persistent in seawater (24-100 day half life) and occurred in a dissolved state (Owen et al., 2002). Recent studies in south Florida and the Caribbean have shown mainly low concentrations of the herbicide near marinas and boatyards (Carberry et al., 2006; Zamora-Ley et al., 2006; Gardinali et al., 2002); however, due to its ability to inhibit photosynthesis across a broad spectrum of marine flora, states such as Bermuda have banned its use and at least one set of authors recommend that other Caribbean states consider similar prohibitions (Carberry et al., 2006).

Finally, while not extensively studied within coral reef environments, vessels are responsible for transporting alien, or nonindigeneous, species. In a study conducted in Guam's nearshore coral reefs, it was determined that Guam contained as many as 85 nonindigeneous marine species; of this total, 41 could be described as introduced (Paulay et al., 2002). While most of the species are sessile and the introductions have been largely unsuccessful in colonizing the coral reefs (and are more established within artificial structures), the authors note that this may be primarily due to the fact that Guam's nearshore coral reefs remain intact. In other areas, such as Hawaii's Pearl Harbor, where there have been extensive anthropogenic impacts on coral reefs, nonindigeneous species have been more successful in colonizing reef habitats. This suggests, when in stressed conditions, coral reefs may be less resilient and thus more vulnerable to invasive species (further discussed in the next section).

Indirect impacts

As previously stated, the focus of this report is on the direct impacts of non-extractive stressors and, as such, this section provides more of an overview of indirect impacts and on how these may synergistically affect coral reef resilience. The indirect impacts considered are nutrient enrichment, sedimentation, and coral disease.

The ability of nutrient enrichment to affect coral reefs is a complex issue because considerable debate exists on whether enrichment by itself is sufficient in producing the phase shifts observed in degraded coral reef systems; that is from coral reef to macroalgal dominance (Precht and Arnason, 2006; Fabricius, 2005; Lapointe et al., 2003; Szmant, 2002; Lapointe et al., 1997). Proponents of enrichment suggest a bottom-up approach to understanding the phase shift; opponents argue that such a phase shift occurs only in conditions where other changes in addition to enrichment have occurred. Szmant (2002) points out that not all coral reefs exposed to nutrient rich conditions necessarily are outpaced by macrolgae, and Precht and Arnason (2006) provide examples of coral reefs located far away from nutrient sources that nevertheless have been overtaken by macroalgae. Models in contrast to the bottom-up, enrichment theory are the top-down, grazer model (Bellwood et al., 2004; Gardner et al., 2003) and the side-in model (Precht and Arnoson, 2006; Pandolfi et al., 2003). In the former model, concerning grazers, macroalgae competition is effectively curtailed by grazers, including fish species. With overfishing, grazers are removed from the coral reefs, thereby leaving coral reefs more vulnerable to macroalgal dominance (especially in the case where other, unfished grazers (such as the 1983-84 *Diadema antillarum* die-off) suffer catastrophic or regional declines (i.e. lack of functional substitutes (Bellwood et al., 2003)). The side-in model argues that it is the increase in coral and other grazer diseases, coupled with bleaching and storm events, that have led to the demise of (especially Caribbean) coral reefs. Regardless of which model is the most accurate in describing either the chain of events or key mechanism that result in the current coral reef decline, anthropogenic activities – and especially as a multiplicity of stressors – are increasingly affecting the ecological balance on and overall health of coral reefs (Knowlton, 2001; Hughes and Connell, 1999; Brown, 1997).

As with nutrient enrichment, the effects of sedimentation vary considerably, based on the type of sediment, the nature of input (whether it is chronic or a single episode), and on other, ambient conditions (Fabricius, 2005; Brown, 1997). Sedimentation does result in suboptimal conditions for coral reefs and has been shown to cause adverse effects in various studies. Rogers (1979) reported bleaching and mortality as coral responses to shading experiments that mimicked the effects of sediment-related turbidity. An in-vitro study by

Telesnicki and Goldberg (1995) using two coral species from south Florida exposed to conditions of elevated levels of turbidity (corresponding to the highest levels allowed during construction in Florida coastal waters) for three weeks resulted in stress responses, including significant increases in respiration and mucus production. A monitoring project off Saint Thomas, US Virgin Islands (Nemeth and Sladek Nowlis, 2001) found that bleaching rates in corals exposed to sedimentation resulting from coastal development were significantly higher than in unexposed corals and that sedimentation may act as a synergistic stressor that weakens corals to other impacts (such as bleaching). Torres (2001) determined that corals adjacent to developed areas are subject to higher rates of sedimentation in Puerto Rico and exhibited lower growth rates than corals found off less developed coasts. Finally, Dikou et al. (2006) examined the effects of sedimentation on the spatial patterns of coral reef communities in the upper reef slope off Singapore, determining that corals adapted for turbid conditions and deeper waters dominated coral cover at the sites closest to the main island. They also documented a clear relationship between sedimentation and water quality with coral recruitment rates and that overall coral cover had declined at half the sites since monitoring had commenced in the 1970s.

The study of coral diseases is an emerging field and one which has contributed considerably to the understanding of coral reef decline and its potential relationship with anthropogenic activities (Bruckner, 2002; Knowlton, 2001). Altogether, there are 29 coral diseases that have been identified in Atlantic and Indo-Pacific corals, and it is suspected that the apparent rise of these diseases may be in part related to the general decline in coral reef health (which may in turn make them more compromised and thus vulnerable to infection) (Wilkinson, 2006). In the case of Caribbean acroporid corals, Precht and Arnoson (2006) point out that the mass mortality of framework building staghorn (*A. cervicornis*) and elkhorn (*A. palmata*) corals to white band disease may be a completely new event (over the past 30 years) or at least one that has not occurred for the past 3,000 years. While these authors do not attribute the die-off to specific human activities, other research examining coral diseases shows links between anthropogenic stressors and the incidences and rates of infection. Bruno et al. (2003) determined that corals infected with yellow band disease suffered twice the level of tissue loss after nutrient concentrations were doubled or increased five-fold, concluding that minimizing nutrient pollution could be important in reducing the severity of coral disease. Another study, by Winkler et al. (2004), identified a tourist area, subject to anchoring, groundings, and recreation, as the site that exhibited the highest incidence of skeleton eroding band disease in corals off Jordan's Red Sea coast; by contrast, corals in a marine protected area showed the lowest incidence of infection. The authors concluded that tourism impacts may have rendered the corals more susceptible to disease. Kaczmarsky et al. (2005) also compared coral reef sites for incidence of coral disease. Their study, conducted in two sites off Saint Croix, U.S. Virgin Islands, determined that incidences and morality rates from black band diseases and white plague type II were significantly higher in the impact site exposed to sewage effluent than the control site. The authors concluded that while recreational activities and runoff may play a role in the more popular impact site, sewage-based enrichment may exacerbate the effects of existing diseases.

Conclusions

When considering direct and indirect, non-extractive stressors on coral reefs, it is clear that the multiple stressors work synergistically to impact coral reefs to a state where outbreaks of bleaching, disease, and macroalgae blooms, etc. manifest these dire conditions. While some coral reefs, due either to adaptations (Lirman and Fong, 2007; Szmant, 2002), environmental conditions (West and Salm, 2004), or functional group diversity (Bellwood et al., 2005), have been able to withstand multiple stressors, others under less favorable conditions have declined and, in some cases (as in the Caribbean), the decline has been precipitous (Gardner et al., 2003). Thus, in determining the effects of non-extractive stressors, it is essential that all factors (including the effects of consumptive impacts and those of natural events) be considered together (Wilkinson, 2006; Hughes and Connell, 1999).

Related to the need to consider both direct and indirect stressors is the need to incorporate tourism models (such as the Butler Resort Cycle (Butler, 1980)), as used in the development of coastal and marine tourism (see Murray, 2007 and Jobbins, 2006 for case studies). As described by van't Hoff (2001), non-extractive stressors are often accompanied by land-based activities, including coastal development (e.g. hotels, resorts, restaurants, marinas, ports, etc.) and accompanying sedimentation (resulting from building material runoff, changes in drainage patterns and loads, etc.), enrichment (e.g. fertilizers, raw or treated sewage, vessel-based pollution), and solid waste (e.g. debris). Thus, when evaluating direct, non-extractive stressors, models should consider land-side effects as well, mainly because the former are abetted by and originate from the latter, and also because while the direct stressors may be controlled or minimized (e.g. diver training, mooring buoys, etc.), land-based, indirect stressors may persist.

Also, it is imperative that direct, non-extractive stressors be monitored over the long-term. Most of the studies considered examined the effects of such stressors over the short- (months to years) to medium- (one to five years) term. Long-term studies (such as those by Hawkins et al., 2005 and Epstein et al., 1999, among others) can provide more meaningful, trend-based information, which is otherwise lacking. This becomes of particular importance when past studies are evaluated critically, as by Rinkevich (2005), in his examination of previous research conducted at Eilat, Israel. He concludes that while considerable research has been conducted in what is perhaps the most heavily dived destination in the world, the reef has not been well enough characterized to establish baseline conditions prior to the advent of anthropogenic impacts. Similarly, long-term studies, such as the nine-year effort described in Hawkins et al.'s (2005) article on dive use stressors in Saba, can elucidate the combined effects of periodic, natural events and continuous, anthropogenic use. Without these long-term views at the chronic and often synergistic effects of human and natural effects, results can be often skewed towards static, or 'snapshot', findings.

Finally, in evaluating non-extractive stressors, it is important to consider the locations at which the studies have been undertaken. A review of the primary literature shows that while many coral reef regions have been represented in the studies, there are a few hotspots where much of the research has been conducted. These include the northern Red Sea, the Great Barrier Reef, and the U.S. Caribbean. Fewer studies have been conducted in Indonesia and the Philippines, which collectively contain over a quarter of the world's coral reefs (Wilkinson, 2002). There is considerable unevenness in terms of the amount of research completed within a region; thus, although several non-extractive stressor studies have been conducted in the Caribbean, most of these have been completed in the U.S. Virgin Islands, Puerto Rico, and the Netherlands West Indies. Fewer have focused on Caribbean community states such as Barbados, Jamaica, and Saint Lucia, and even fewer have considered the two states undergoing the largest growth rates in tourism in the Caribbean: Cuba and the Dominican Republic. Even in established tourist markets, like the Bahamas and the Mexican Riviera, the information available on non-extractive stressors is very general.

As tourism grows in the Caribbean region and other coral reefs across the globe, the threat from non-extractive stressors will likely outpace those currently presented by fishing and other extractive activities. While programs such as Reef Check (Hodgson, 1999) provide the means by which to assess current coral reef conditions, there remains the need to objectively and accurately quantify the status of non-extractive stressors, both as a means by which to monitor changes in use patterns and activities and their impacts on coral reef health and as a path towards comprehensive coral reef management.

References

Please refer to the accompanying database for all references in this report.