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Atlantic coral reefs: the transplantation alternative

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Abstract Although all of the world's coral reef regions have suffered degradation due to direct and indirect human influences, only the Western Atlantic reefs have declined to the extent that their continued existence appears to be in jeopardy. Of a once flourishing reef system, only about 10% is still alive and it is depauperate in terms of the food web diversity necessary to maintain a stable and productive ecosystem. The large carnivores and herbivores have become so scarce that they have failed to control the populations of smaller animals and plants, so that almost all the trophic levels have been disrupted. At the same time, the impacts of many other factors have contributed to the process of degradation resulting in a true crisis. The extent of damage, and the general impairment of regeneration, is such that a hope for natural recovery appears to be unrealistic. It is suggested that a recovery could be achieved through transplantation of corals and other reef species from areas where more diverse, relatively stable ecosystems still exist. Available data on the introduction of exotic species into marine ecosystems indicate that such species are generally accommodated and do not cause extinctions among the native species. Although some introduced species are considered to be pests, others have proved to be beneficial, and all have apparently increased the biodiversity of the invaded areas.

Keywords Coral reefs · Tropical Atlantic · Ecology · Biodiversity · Transplantation

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Introduction

Within the Western Atlantic, coral reefs extend southward from Bermuda and Florida through the Caribbean to reach their southern limit on the Brazilian coast near Rio de Janeiro. In the Eastern Atlantic there is minor reef development, primarily around the offshore islands of Principe and Sao Tome. Historically, all of the reefs supported a broad diversity of marine life. Early accounts by explorers in the 17th and 18th centuries (Jackson et al. 2001) indicated that the West Indies reefs supported an abundance of large animals including turtles, manatees, and large predatory fishes. Now, the large animals are almost gone, the great majority of the once-living reefs are dead, and most of what remains are bleached coral skeletons and piles of rubble covered with seaweed.

Although there had been a long term decline (Pandolfi et al. 2003), a sudden degradation has taken

place recently. A broad-scale survey of the Western Atlantic (WA) reefs conducted in 2003, concluded that the average amount of living coral cover had been reduced by 80 % over the past 30 years (Gardner et al. 2003). The 2004 report, *Status of the Coral Reefs of the World* (Wilkinson 2004), described a major decline in the reefs of the wider Caribbean (including those of the nearby Atlantic) when it called attention to the 50% cover of live coral that existed on the reefs 25 years ago compared to about 10% now. At about the same time, a detailed evaluation *Reefs at Risk in the Caribbean* (Burke and Maidens 2004) emphasized the reality of a coral reef crisis.

In response to the crisis, the U.S. Geological Survey published its plan for an enhanced long-term, research effort (USGS 2007). While this plan was being written, a combination of high sea-surface temperatures and strong hurricanes in 2005 resulted in extensive coral bleaching and widespread physical damage (Wilkinson and Souter 2008). At some locations, over 90% of the reefs died and at many others 20–50% were destroyed. Finally, there has appeared a comprehensive study *The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States* (Waddell and Clarke 2008). This work confirmed that few reefs in the U.S. Caribbean and Atlantic currently have a percentage of mean live coral cover greater than 10%. Furthermore, it noted that the reefs were once dominated by branching corals in the genus *Acropora* and that these corals had undergone population declines of about 90% since the 1970s.

The book edited by Waddell and Clarke (2008), and the other major works that have been cited, have made obvious two important aspects of the coral reef crisis. First, the present condition and future prospects of the reefs are matters of utmost concern, not only to the countries that border the Caribbean, but to the public in general. The concern is due to the enormous economic significance of the reefs as well as to their esthetic, recreational, and biological values. Second, this concern about the reefs has resulted in a multitude of research and monitoring activities being carried out by governmental and private conservation organizations. So far, in terms of reef management, the results have been centered on the establishment of Marine Protected Areas (MPAs) and the repair of local damage when it occurs.

In addition to the major publications, many marine biologists have called attention to the condition of the reefs. Among the most notable articles were those that decried the terrible condition of the U.S. coral reefs (Pandolfi et al. 2005), urged a reassessment of current management practices (Bellwood et al. 2004), advocated the adoption of a common goal to reverse the degradation (Pandolfi et al. 2003), and expressed concern about the resilience of the reefs under their present impacts (Hughes et al. 2003). Research conducted in various parts of the Caribbean reinforced the general impression of a region-wide decline. In the U.S. Virgin Islands, the loss of coral was followed from 1989 to 2003 with no evidence of recovery (Rogers and Miller 2006). In the Virgin Islands as well, a 25 year decline of the coral *Monastraea annularis* indicated a strong likelihood of extirpation within the next 50 years (Edmunds and Elahi 2007). In Quintana Roo, Mexico, live stony coral cover decreased 40–50% between 1990 and 2000 (Roy 2004). In Curacao (Netherlands Antilles) hard coral cover decreased from 52% in 1973 to 22% in 2003 (Nagelkerken et al. 2005). In the Florida Keys National Marine Sanctuary, between 1996 and 2004, coral cover declined from 11.9% to 6.6% (Beaver et al. 2004). But, in a Jamaican area, some recovery of urchins and coral seemed to be taking place (Carpenter and Edmunds 2006).

Causes of decline

The loss of the reefs and of the biota they support is usually attributed to overfishing, chemical pollution, and sedimentation due to soil erosion. But other factors are important as well. Human induced nutrification has become recognized as a major threat (Hallock 2005). Sources of nutrients range from local sewage to regional urban and agricultural runoff, atmospheric washout of nitrous oxides, and dust fallout due to desertification. Global warming, as the result of carbon dioxide buildup in the atmosphere, together with increased ultraviolet radiation, may be responsible for episodes of coral bleaching and disease (Goreau et al. 1998). Another effect of the carbon dioxide buildup detrimental to corals is ocean acidification (Caldiera and Wickett 2003). Hurricanes are considered to be important because, in the year following, coral cover is reduced by about 17% on

average (Gardner et al. 2005). Studies conducted between 1980 and 2001 showed that coral cover at sites impacted by a hurricane declined at a significantly faster rate (6% per annum) than non-impacted sites (2% per annum). All these factors are important, but the most conspicuous declines in the reef ecosystems are evidently due to overfishing (Pandolfi et al. 2003).

Overfishing of the large herbivorous and carnivorous vertebrates reduces population control on smaller species. When some smaller species become too crowded, they are vulnerable to disease. In the mid-1980s, more than 99% of the *Diadema antillarum* populations were eliminated by an infectious disease. The removal of this important herbivore and the large parrotfishes allowed many of the reefs to be overwhelmed by algae to the extent that they have not been able to recover. The macroalgae grow on dead coral but, if not consumed by herbivores, they prevent settlement and recolonization by coral larvae (Aronson and Precht 2006). As a result, the ability of Caribbean reefs to cope with local and global changes may be irretrievably compromised (Gardner et al. 2003). Today, the most degraded reefs are little more than rubble, seaweed and slime (Pandolfi et al. 2005).

In Jamaica, until about 25 years ago, the shallow and intermediate depths were dominated by large colonies of *Acropora palmata* and *A. cervicornis*. Now in that area, and in other parts of the Caribbean, populations of these species are represented only by small, scattered colonies. The failure of the *Acropora* recovery is most likely associated with the low production of planktonic recruits (Quinn and Kojis 2006). In addition to the conspicuous decline of the *Acropora* species, there is evidence of a dramatic decrease in overall coral recruitment suggesting a generalized loss of resilience (Miller and Szmant 2006).

Reef restoration

Habitat repair

Restoration usually refers to the process of repairing accidental injuries to the reef so that a natural recovery of the biota is promoted. The goal is to restore the damaged reefs until they are functionally equivalent to their uninjured counterparts (Precht and Robbart 2006). A new *Coral Reef Restoration Handbook* (Precht 2006) discusses the technique of repairing damage caused by ship groundings, construction, coral mining and other activities. It has been pointed out that there are two major approaches to reef restoration: artificial reefs and coral transplantation (Abelson 2006). Over the years, a variety of materials have been dumped into the ocean in the hope of providing a suitable substrate for corals and their associated biota. Experience had shown that cast concrete objects or limestone boulders work best (Kaufman 2006). The idea is to present surfaces that would be attractive to coral planulae (larvae). But in the Caribbean, such coral propagules have become less common so that the attachment of live coral fragments to artificial reefs has become a standard procedure.

Recently, substantial progress has been made in the design of habitats for small aquaria and for aquatic farms that do large-scale propagation of marine invertebrates. To date, over 150 species of Scleractinia, 100 species of symbiotic Octocorallia, and virtually all available Zoantharia are being propagated (Borneman and Lowrie 2001). It has been suggested that significant losses, typical of efforts to directly transplant coral fragments, can be avoided by using farm raised corals for reef replenishment. Live coral species are frequently advertised on the internet.

The overfishing problem

The pervasive threat of overfishing can theoretically be counteracted by the establishment of MPAs that are well managed, i.e. those that consist of no-take areas in which all species in the ecosystem are protected. Unfortunately, the existing network of MPAs is largely ineffective (Mora et al. 2006). Although 285 MPAs have been established in the Caribbean, only 6% were rated as effectively managed and another 13% were judged to have partially effect management (Burke and Maidens 2004). And it has been found that, although efficient MPAs increased the fish biomass, they had no effect on the regional mortality of corals and the abundance of macroalgae (Mora 2008). But, if a sufficient number of large herbivorous fishes are present, they can reduce the algal cover and clear a substrate for coral recruits (Mumby et al. 2006, 2007). However, in the Caribbean, large herbivorous fishes have virtually disappeared. At Little Cayman Island, the existence

of a no-take MPA had no effect on the decline of live coral cover (Coelho and Manfrino 2007).

Considering that reef restoration projects are expensive and generally practical only for small localities, and that MPAs in the Caribbean apparently cannot prevent the decline of the reefs, what other options are available? Artificial propagation of *Diadema* urchins and herbivorous fishes could be attempted but there are no indications that such species could be efficiently produced in hatcheries. Must we depend on managing fishing, pollution, and coastal development and hope that the reefs will recover (Wilkinson and Souter 2008), or is there another alternative?

Proactive management

Proactive management means the addition of species to low diversity ecosystems in order to improve resistance to environmental disturbance, to increase biodiversity, and to augment organic production. This procedure has been recommended in order to rehabilitate the temperate North Atlantic fisheries (Briggs 2008). For the tropical Atlantic, such a solution may not be agreeable to some biologists who adhere to the "competitive exclusion principle." This principle, based on the work of Lotka and Volterra in the 1920s and Gause in 1934, is still influential although not as strong as in the 1970s (Begon et al. 1996). That influence, when combined with the effects of the extinctions that have been caused by invasive species in restricted terrestrial and freshwater habitats (Lockwood 2004), has resulted in an ecological mind-set that is often opposed to the introductions of almost all exotic species. But, despite some extinctions, investigations of island ecosystems have shown that diversity generally rises as invading species are assimilated into recipient communities (Sax and Brown 2000; Sax et al. 2002).

If the prospect of marine transplantations can be viewed objectively, and in the light of current and historical information about such introductions, perhaps it can be considered to have practical value. A recent review of studies devoted to marine invasions and introductions has found no evidence that native species have been driven to extinction (Briggs 2007). Instead, almost all of the invasions have apparently resulted in a gain of biodiversity. It has been noted that the greater temporal stability of more diverse communities is apparently due to higher productivity, competitive interactions, and statistical averaging (Lehman and Tilman 2000). This "portfolio effect" has been confirmed by results from harvesting in an eight-species ecosystem (Finnoff and Tschirhart 2003). Comparison of records from coastal ecosystems has shown that those with higher species richness demonstrated lower rates of collapse of commercially important fish and invertebrate taxa over time (Worm et al. 2006).

Contemporary invasions

Thousands of marine introductions have taken place, almost all of them accidental, in various parts of the world. Although some invaders have turned out to be pests, others have been beneficial. For example, in northern Patagonia, the introduced Pacific oyster *Crassostrea gigas* increased the population densities of epifaunal and infaunal organisms, and also the feeding rates of local and migratory bird species (Escapa et al. 2004). In Chile, invasive bio-engineer species, such as the ascidian *Pyura praeputialis* and the seagrass *Heterozostera tasmanica* have created new habitats for nursery grounds and biodiversity enhancement (Casilla et al. 2005). In the northeastern Pacific, the Asian hornsnail *Batillaria attramentaria* provided a substrate that was utilized by two other exotic and two native species (Wonham et al. 2005).

The effects of exotic species invasions were studied in estuarine strandline plant communities in Narragansette Bay, Rhode Island, USA (Bruno et al. 2004). Established exotics comprised 79 out of a total of 147 species. Information from 24 sites in the bay indicated that native and exotic species richness was positively correlated and that the invaders appeared to pose little threat to the native plant diversity. The authors concluded that their results did not support earlier predictions of ecosystem damage.

Of the world's oceans, the Mediterranean is evidently the most invaded with 558 non-indigenous species reported to date (Galil 2008). Beginning with the opening of the Suez Canal in 1869, an almost continuous stream of successful (colonizing) invasions has been taking place. These are virtually all one-way migrations from the Red Sea into the Mediterranean. Very few reciprocal migrations are

known. The rich tropical fauna of the Red Sea, a part of the Indo-West Pacific Region (IWP), is much more diverse than that of the warm-temperate Mediterranean (Briggs 1995). The results of the invasions into the Eastern Mediterranean provide instructive information about accommodations that take place between invaders and equivalent native species: Along the Mediterranean coast of Israel, two invading Red Sea goatfish (Mullidae) species *Upeneus moluccensis* and *U. Pori* were accommodated by the native Mediterranean species *Mullus barbatus* and *M. surmuletus* by means of niche sharing (Golani 1994). The invading goatfishes appear to dominate the shallow waters while the natives are most abundant below about 55 m.

Many of the invaders from the Red sea are fishes that have commercial value and their relative abundance over time has been reported. Nearly half the trawl catches along the Israeli coast, about half the fish biomass in rocky shore habitats, and 50–90% of the fishes in shallow sandy habitats, are now of Red Sea origin (Goren and Galil 2005). The invasions of the barracuda *Sphyraena chrysotaenia* and two clupeids *Dussumieria elopsoides* and *Herklotsichthys punctatus* were primarily responsible for a shift in trawling activity from deep to nearshore waters of less than 50 m. Two invasive, herbivorous rabbitfish (Siganidae) species *Siganus rivulatus* and *S. luridus* comprise about one third of the total fish biomass along Israeli rocky shores. Before the arrival of the rabbitfishes, the algae entered the food chain by the slow process of decomposition. The rabbitfishes grazed on the algae, grew rapidly in numbers, and began to provide the major food source for three species of grouper (Serranidae) (Aranov 2002). The more rapid turnover of algal resources augmented the production of the large carnivorous fishes and altered the community structure to resemble a more tropical ecosystem.

An invertebrate example of niche sharing in the Eastern Mediterranean is the snapping shrimp *Alpheus*. The immigrant snapping shrimp *A. rapacida* occurs between 15 and 50 m, apparently having replaced the native *A. glaber* that now occurs at depths of 45–145 m (Galil 1993). It has been noted that the littoral and infra-littoral of the Levantine Basin are dominated by invader species and that the Eastern Mediterranean is gradually becoming a Red Sea Province (Goren and Galil 2005). In his book on biological invasions, Williamson (1996) stated that Suez Canal invaders have not obviously affected the native species. But, the siganid invasion has had beneficial economic results in terms of an increased production of native carnivorous fishes, and other invaders have also augmented the fishery resource. Altogether, more than 300 species from the Red Sea have invaded the Mediterranean but there are no records of subsequent extinctions in the native biota.

On the other hand, the authors of some invader reports have expressed alarm because a few exotics have proven to be pests that cause economic damage and others have had adverse effects on native populations. For example, the American slipper limpet *Crepidula fornicata* was introduced to Europe in the 1870s and has significantly reduced the growth rate and survival of the native blue mussel (Thieltges 2005). San Francisco Bay has been invaded by an alien green crab *Carcinus maenas* that has reduced the abundances of several native invertebrates, including a shore crab and two clam species (Grosholz et al. 2000). But, at the same time, there were significant increases in the populations of tanaid crustaceans and two polychaete taxa. None of the recent studies devoted to the effects of invaders into the marine environment have indicated any species extinctions or loss of biodiversity. In fact, it appears that invasions have generally increased biodiversity within the affected areas.

Historical invasions

Admittedly, it is difficult to forecast the eventual results of invasions that have taken place in recent years. However, the fossil record does provide information about historical invasions and their long-term effects. The opening of the Bering Strait permitted the Great Trans-Arctic Biotic Interchange that apparently began about 5.4 Ma (million years ago) (Gladenkov et al. 2002), but with most of the migrations taking place about 3.5 Ma. The North Pacific Ocean is much richer in species than the North Atlantic. The former has about twice as many molluscan species (Vermeij 1991) and possibly three times the number of fish species (Briggs 1974). As might be expected, the interchange consisted primarily of invasions into the Atlantic with few reciprocal migrations. Recent information on the Pacific molluscan species that invaded the North Atlantic (Vermeij 2005a) indicates

that at least 143 species colonized European shores while 176 settled in eastern North America. On the eastern American rocky shores, the invaders now comprise the majority of common species. The invaders have replaced incumbents in some of the preferred habitats but have not driven the natives to extinction.

Over geological time, numerous other faunal invasions or interchanges have taken place. An aspect of such invasions that deserves emphasis is the eventual fate of most of the invaders. An example in the WA has been provided (Vermeij 2005b). During the early Pliocene to early Pleistocene, 40 southern molluscan species invaded northward into Florida and the surrounding area. All, with the exception of four or five, eventually evolved into endemic species resulting in a rise in diversity for the region as a whole. It was observed that community relationships change as immigrants become assimilated, but most members survive and adapt. This is so because interactions among species are numerous and flexible, and because adaptations arising from these interactions are often of a very general nature and thus applicable under a wide variety of circumstances.

Discussion

Although it has been pointed out that the human degradation of coral reefs began hundreds of years ago (Pandolfi et al. 2003), the Cenozoic history of Caribbean reefs (Budd 2000) indicated an earlier episode of extinction during the latest Pliocene. Previous to that extinction, 38 genera and 133 species of reef corals were present but today's fauna contains only 25 genera and 61 species. The losses may have been due to the tectonic rise of the Panamanian Isthmus that caused oceanic circulation changes in the WA. The changes may have resulted in a slowing of upwelling currents causing a decline in primary productivity (Vermeij 2004). Also, there were temperature drops associated with the onset of northern hemisphere glaciation. The combination of low productivity and low temperature may have caused the extinction of many corals and their associated organisms. In earlier times, the losses might have been repaired by an influx of propagules from the Pacific. But, the rise of the isthmus occurred at about the same time as the extinction and prevented access from the Eastern Pacific. Therefore, it is apparent that, by the beginning of human exploitation, the diversity of the Atlantic corals and their commensals had already been severely reduced.

The apparent fragility of the Atlantic corals may be due to their generic age. In the East Indies Triangle of the IWP, the average generic age of the corals is about 30 million years, but in the Atlantic the genera are about twice as old (Veron 1995). In addition, the Atlantic genera are represented by relatively few species. Geographic subspecies, such as are common on the high latitudes of the Indo-Pacific, are generally absent in the WA. Another indication is the slow pace of post-hurricane recovery pointing to a general impairment of regeneration potential (Gardner et al. 2005). So the age of the genera, the apparent low level of evolutionary innovation, and lack of resilience may be characteristics of a ancient fauna that has received no input from its evolutionary source for several million years. A major diversity recovery, without introduction of exotic species, does not seem possible within the time span of the next few dozen human generations.

Compared to the Atlantic, the tropical IWP contains enormous numbers of species. So many of them have yet to be named and described that a total diversity figure is not yet possible. It has been estimated that the IWP contains five to ten times the number of species found in the Caribbean (Gosliner et al. 1996). As noted, this influx of young, dominant species has been unable to reach the WA for at least three million years due to the blockage of the Panamanian Isthmus. For the reef corals, there are more than 450 species in the central IWP (Veron 2000) compared to 61 in the Caribbean (Budd 2000). Along the shores of the tropical Eastern Pacific, about 95% of the coral species, many other invertebrates, and about 80 fish species are recent arrivals from the IWP (Robertson et al. 2004).

In regard to reef conservation, it has been suggested that first priority be given to the establishment of a series of MPAs within the bounds of the two tropical centres of origin, the East Indies Triangle and the Southern Caribbean (Briggs 2005). Within the Atlantic, the southern part of the Caribbean has the greatest species diversity and has evidently been exporting species to other parts of the ocean. There are indications that the reef fauna of the Caribbean began to separate from that of the Eastern Pacific in

the late Miocene about 7 Ma (Muss et al. 2000). Therefore, the barrier between the two regions may be much older than the complete emergence of the isthmus. This may help to explain the relict nature of some of the biota associated with the reefs. The ages and historical distribution of the coral genera themselves indicate that they once existed in the Tethys Sea of the early Cenozoic (Veron 2000).

Would it be better to import from the IWP a comprehensive sample of that biota in an attempt to establish an entire ecosystem de novo or should it be done in bits and pieces? Obviously, the corals are needed to provide structural habitat and they must be alive in order to support a variety of commensal organisms. In the 1960s, specimens of the mushroom coral *Fungia scutaria* were brought from the Indo-Pacific and liberated in Discovery Bay, Jamaica (Goreau et al. 1969). Soon after the coral colonies were established, several attempts were made to eradicate them. In those days, tropical shallow waters were thought to be ecologically saturated with species and that introduction of an exotic would likely do great harm to native species. But we now know that most ecosystems exist in a state far below species saturation (Vermeij 1991; Ricklefs 2004). Recently, a remnant population of *F. scutaria* was discovered in Jamaica (Bush et al. 2004). This Indo-Pacific coral showed remarkable resilience in enduring two hurricanes and environmental factors that resulted in the collapse of native coral populations.

In the IWP, the greatest variety, and the most competitive, of coral species are found in the East Indies Triangle. Corals from this area would probably be the most vigorous and the best candidates for transplantation. One other consideration may be important. Temperatures high enough to cause coral bleaching have been found to be detrimental, but there are areas such as the Arabian Gulf where the corals are acclimated to higher temperature fluctuations than elsewhere (Coles 1997, Coles and Brown 2003). Some corals on Ofu Island, American Samoa, have become adapted to temperature extremes (USGS 2007). Such corals would also appear to be promising.

Procedure

In the WA, it would appear advisable to introduce the exotics into reserves established in the Southern Caribbean. Its function as a centre of origin indicates a suitable habitat for reef organisms. At the same time, rather than depending entirely on natural dispersal from the centre, it may be wise to directly transplant exotics into those reefs that have been most severely damaged. A broad colonization program could result in a buildup of diversity over a greater area and a more rapid overall recovery. As diversity increases, the entire coral reef biota will become much larger. Larger biotas support dominant species whose competitive performance is higher than that of dominant species in smaller biotas (Vermeij 2005a). And larger biotas will eventually contribute to fishery and recreational resources outside the marine reserves.

It has been shown that marine reserves are effective in promoting increases in individual size, biomass, and the population density of native fishes (Mora 2008). But reserves in the WA cannot add new species to a biogeographic region that has been historically depleted. An increase in species diversity can be brought about only by evolution, often requiring a million years or more, or by the addition of species from other regions through natural invasion or transplantation. Reserves established in areas with good water quality, and fully protected, would be critical to the success of a transplantation program. Many existing reserves still permit fishing, so have limited conservation value.

One can predict that the timing of the transplants would be critical to the success of the operation. At present, The WA reefs are mainly occupied by foliose algae that have prevented colonization by native corals and would make transplantation difficult. This suggests that the first step should be the importation of algae-eating fishes that would clean the reefs and prepare them for occupation by corals. As was noted in the case of the invasions into the Eastern Mediterranean, two species of rabbitfish, *Siganus rivulatus* and *S. luridus*, were highly successful in controlling the algae of that region. These species would apparently be good candidates for introduction into the WA. The rabbitfishes would, in turn, provide a food source for the larger, carnivorous fishes.

The second step, the transplantation of IWP corals in order to form a basic reef structure, may not be a difficult task. About 75% of scleractinian corals reproduce by spawning and external fertilization (Frank and Mokady 2002) and, in the Great Barrier Reef of Australia, more than 100 species have been

observed to take part in mass spawning events (Babcock et al. 1986). Mass spawnings frequently produce slicks on the sea surface that contain billions of embryos. Millions of embryos from slicks have been confined and successfully maintained in floating ponds until competent to settle (Heyward et al. 2002). Another source, perhaps more suitable for small areas, is coral farms that now produce a large variety of species that are intended for transplantation (Borneman and Lowrie 2001).

Conclusions

In comparison to those of the IWP, the corals of the WA may be considered a relict, depauperate assemblage that has been isolated for several million years. By 25 years ago, half of the living reefs had already died and by 2004 only about 10% remained. The hurricanes and high temperatures that occurred in 2005 produced considerably more damage. The dead reefs are now mainly occupied by rubble covered with algae that prevent coral recolonization. After so much decline, and with an evident impairment of regeneration, the chances of a natural recovery of the native biota in less than several thousand years appear to be slight. In contrast to the terrestrial environment, empirical contemporary and fossil evidence demonstrates that exotic species can be assimilated into marine communities without causing extinctions in the native biota. Transplantation from the IWP could result in the establishment of a much more diverse ecosystem, one that would consist of many younger, more resilient genera and species. Such an undertaking could vastly increase the size, biodiversity, esthetic quality, and economic value of the existing reefs.

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