CORAL TRANSPLANTATION AND RESTOCKING TO ACCELERATE THE RECOVERY OF CORAL REEF HABITATS AND FISHERIES RESOURCES WITHIN NO-TAKE MARINE PROTECTED AREAS: HANDS-ON APPROACHES TO SUPPORT COMMUNITY-BASED CORAL REEF MANAGEMENT

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INTRODUCTION

Of the planet's 600,000 km^2 of coral reefs (Jameson et al., 1995), roughly 70-80% are located in developing countries. Many of these reefs are owned or controlled by indigenous fishing communities rather than national or state governments. These rural fishing communities are a primary force of destruction to coral reefs on a global scale (Wilkinson, 1998), therefore their involvement in the management and conservation of coral reefs will be an essential part of reversing coral reef decline.

Rural fishing communities are implicated in routine practices that break and kill corals, leading to serious coral reef decline (Wilkinson, 1998). Among these problems are: blast fishing (McManus, 1997; Nzali et al., 1998), fishing net damage (Edward, 1999), anchor damage (Rogers et al., 1988), dredging and sand mining (Maragos, 1974; Clark and Edwards, 1995; Edward, 1999), and coral harvesting for lime production (Brown and Dunne, 1988; Dulvy et al., 1995; Berg et al., 1998), for use as building materials (Brown and Dunne, 1988; Dulvy et al., 1995), and for commercial sale as curios or for the aquarium trade (Oliver and McGinnity, 1985; Franklin et al., 1998; Green and Shirley, 1999). All of these destructive practices convert rocky reef substrata into unconsolidated rubble beds that may not recover coral populations on biological time scales (Lindahl, 1998; Riegl and Luke, 1998; Fox et al., 1999; Bowden-Kerby 1997, 2001a). Rubble beds thus become "alternate steady state" environments with very little hope for natural recovery.

Even where reefs are left relatively intact, over-fishing alone can cause basic shifts in ecological functioning, resulting in decreased coral cover and lower biodiversity (Hughes, 1989, 1994; Done, 1992; Jackson, 1997; Szmant, 1997). Management plans addressing over-fishing must be implemented as part of coral reef rehabilitation, restoring the ecological balance required to reverse coral reef decline.

A promising development in coral reef management in recent years has been the widespread establishment of no-take marine protected areas or "MPAs" (Biodiversity Conservation Network, 1999; World Bank, 1999). The concept of no-fishing areas is part of the traditional culture of many Pacific Islands. Particular *tabu* reef areas (pronounced tambu, taboo, tapoo, or kapu) were closing to fishing by the chief for a specific time period to honor a death, or in some cases reef areas were closed permanently due to customary religious beliefs. However, this practice of establishing tabu areas mostly died out during the colonial era. In recent years, widespread coral reef decline has inspired various governmental and NGO initiatives to conserve reefs, and tabu areas have begun to be reestablished in several areas by chiefs and communities, often facilitated by these agencies.

Reef Restoration in Community-Based MPA Management

Perhaps the greatest threat to the success of community-based no-take marine protected areas is the delay in resource recovery. The closure of reefs to fishing activities deprives communities of the use of portions of their fishing grounds. Severely degraded reefs low in fish habitat due to low coral cover and dominated by recruitment-inhibiting substrata, and missing breeding populations of formerly abundant organisms, may not respond effectively to closure, even after the conditions that lead to decline are discontinued. If an MPA is established by a community, and if the MPA takes many years to respond positively to closure, the delay would likely erode support for the project and cause the collapse of local management plans (World Bank, 1999). With a history of failed development in many countries, failed MPAs run the risk of becoming yet another example of an empty development promise.

Active interventions such as coral transplanting to increase fisheries habitat and restocking key shellfish species within the no-fishing MPAs could potentially shorten the lag time in fisheries recovery, helping ensure the success of community-based management and thus contributing significantly to coral reef conservation. Simple, community-appropriate, and low-cost restoration methods have recently been developed for use as workable tools for low-energy environments. While more work in this area is needed, reef managers and reef owning communities now have the option of intervening to restore some of their problematic non-recovering reef areas (Lindahl, 1998; Fox et al., 1999; McClanahan and Muthiga, 1999; Bowden-Kerby, 1997, 2001a,b).

As a large part of coral reef decline on a global scale is caused by overuse and misuse by fishing communities, a large part of the solution to coral reef decline will be the sustainable management and use of coral reefs by the communities of the developing world that own and utilize much of the planet's reefs (McManus, 1997; Biodiversity Conservation Network, 1999; World Bank, 1999). Involving fishing communities in low-tech methods of coral transplantation could help to educate reef users about the importance of corals a fish habitat, how coral grow, and various environmental sensitivities of corals. Low-tech coral transplanting could thus potentially serve as a powerful hands-on educational tool in support of community-based management, even if implemented only on a relatively small scale and in association with community managed marine reserves.

Natural Processes of Coral Reef Recovery

Attempting to restore degraded coral reefs requires a basic understanding of the natural recovery process, as well as knowledge of the conditions under which these natural processes succeed or fail. Coral reefs can take as long as 20-50 years to recover from severe damage (Grigg and Maragos, 1974; Stoddart, 1974; Pearson, 1981; Dulvy et al., 1995). However, reefs often recover in 5-10 years or less when numerous corals and coral fragments survive (Endean, 1973; Shinn, 1976; Highsmith et al., 1980). The recovery of coral reefs from disturbances such as hurricanes depends on the survival of coral fragments (Shinn, 1976; Highsmith, 1982; Edmunds and Whitman, 1991), as well as the settlement of coral larvae (Connell, 1973; Harrison and Wallace, 1990; Nzali et al., 1998). Moderate storms often serve as major reproductive events for branching corals (Highsmith, 1982), while non-branching coral species rely mostly on larval recruitment.

The availability of suitable substrata for larval recruitment can limit coral reef recovery and restrict reef development, as coral larvae require specific types of rocky settlement substrata (Connell, 1973; Highsmith, 1982; Harrison and Wallace, 1990). Recruitment of coral larvae is inhibited where substrata have become unstable (Brown and Dunne, 1988; Lindahl, 1998; Fox et al., 1999), overgrown by algae (Birkeland, 1988; Wittenberg and Hunte, 1992; Gleason, 1999), or covered with a fine layer of sand or silt (Hodgson, 1990; Maida et al., 1994; Babcock and Mundy, 1996). Even where the substratum is ideal for larval settlement, poor larval supply may sometimes limit coral reef recovery (Lindahl, 1998; Nzali et al., 1998; Quinn and Kojis, 1999).

Coral Transplantation to Accelerate Natural Recovery Processes

Transplanting coral fragments has been suggested as a means to rehabilitate reefs by bypassing the critical early stages of coral recruitment, especially on substrata not favorable to larval recruitment or to post-recruitment survival, (Lindahl, 1998; Bowden-Kerby, 2001a). Coral fragments have a distinct advantage over newly settled larval recruits due to their considerably larger size, having increased survival and growth rates (Sousa, 1984), increased ability to compete for space (Bothwell, 1981; Tunnicliffe, 1981), and greater stability on unconsolidated substrata (Gilmore and Hall, 1976; Highsmith, 1980; Lindahl, 1998). Transplanted coral fragments would also potentially fare better than storm-generated ones, which sometimes have high mortality (Bak and Criens, 1981; Knowlton et al., 1981), presumably due to abrasion and tissue damage incurred during storm transport.

Various transplantation methods have been attempted with the goal of restoring coral cover to reefs. Much of the restoration efforts to date have been focused on responding to acute episodes of damage; in particular the repair of reefs subsequent to ship groundings. Most of these efforts are located in high-energy reef front areas, using expensive methods, and requiring hundreds of hours underwater to secure dislodged coral colonies. Apparently little consideration has been given to the fact that the high-energy environments most often affected are normally dominated by stable sediment-free substrata where natural recruitment and recovery processes are most active, potentially making restoration efforts in these habitats unnecessary. The overall positive impact of expensive coral reef restoration projects has recently been questioned (Harriott and Fisk, 1988b; Hatcher et al., 1989; Maragos, 1992; Edwards and Clark, 1998; Birkeland, 1999; Challenger, 1999). Rather than abandoning the work, Maragos (1992) suggested that less costly methods should be developed. A recent reevaluation (Edwards and Clark, 1998), detailed the conditions where transplantation was most appropriate, and concluded that transplantation should be viewed as a tool of last resort, for use only where natural recruitment and recovery processes are failing. We concur with these sentiments.

Simple, low-tech methods of coral transplantation have been investigated for restoring coral cover to damaged lower-energy reefs, using unattached coral fragments to mimic and accelerate asexual fragment-driven reef recovery processes (Guzman, 1991,1993; Lindahl, 1998; Fox et al., 1999; Bowden-Kerby, 1997, 2001a,b). Transplanting corals into lower energy areas precludes the necessity of securing coral transplants, considerably lowering cost and effort. A high survival rate for unattached coral transplants has been demonstrated for such sheltered areas (Maragos, 1974; Harriot and Fisk, 1988a; Guzman, 1991,1993; Lindahl, 1998; Bowden-Kerby 2001a,b), particularly for rubble environments and for larger fragment sizes.

Transplanting corals directly onto sand has also been done successfully (Bowden-Kerby, 1997; 2001b), establishing that entirely new patch reefs can be created on barren sand-flat “deserts”, providing for increased fish habitat. This patch reef creation process is modeled on the natural process of coral colonization of sand dominated reef areas (Bowden-Kerby 2001b), whereby storm currents sweep detached coral colonies into sandy back reef environments where larval recruitment is not possible, but where conditions for coral growth are ideal. The key factor in coral survival on sand is the large size of coral colonies, as small fragments always perish (Bowden-Kerby 2001b).

Coral Transplants as Fish Habitat and Fish as Vital to Coral Reef Recovery

From the perspective of coral transplantation as a potential fisheries management tool, the impact of coral transplantation on fish populations is of fundamental importance, particularly for reefs suffering from algal overgrowth or other ecological imbalances related to low fish numbers. Living coral cover has been shown to positively influence fish abundance (Bell and Galzin, 1984; Bell et al., 1985; Jones, 1988; Sale, 1991). Certain species of reef fish are obligatory live-coral dwellers for life (Jones, 1988), while other species of reef fish require highly complex nocturnal or diurnal shelter provided by living corals, habitat that is sometimes in short supply on a particular reef (Brock, 1979). If the lack of grazing fish is related to a lack of habitat, coral transplantation could potentially be important in reestablishing these fish populations, that would in turn clean the substratum and help reestablish a broader ecological balance leading to conditions more conducive to larval-based recovery (Bowden-Kerby, 2001b). If such positive reinforcement were to occur, a limited amount of transplantation could begin a cascading effect leading to wider reef recovery. If enough fish habitat remains on moderately degraded reefs, transplantation may not be required to restore the natural balance of fish to a reef, and fisheries management alone may lead to restoration.

Other types of ecological imbalances can inhibit reef recovery, and low-tech approaches at restoration are beginning to be investigated. Sea urchin removal has proven effective in restoring corals to reefs with high post-recruitment mortality due to an overabundance of sea urchins (McClanahan et al., 1999). Crown-of-thorns starfish have also been removed from many reefs, (including almost 5,000 from the Cuvu site), where COT over-abundance threatens coral population recovery. Reefs overgrown with macroalgae have also been restored by removing the algae, re-exposing coral recruitment surfaces (McClanahan and Muthiga, 1999). Restocking invertebrates may be an important ecological contribution as well, particularly where the particular species was abundant in the past on the particular reef area, but where over-fishing has been so severe on up-current reefs that a lack of larvae prevents the natural recovery of the population for many years.

OBJECTIVES

The primary objective of the ongoing work described here is to address the problem of delayed coral reef resource recovery, exploring the potential for utilizing low-tech coral transplanting methods and restocking of severely depleted invertebrate species to accelerate the recovery of coral reef fisheries resources within no-fishing MPAs. A secondary objective of the work is to increase community involvement and to raise awareness among the fishing communities for corals and other important reef species through hands-on restoration and restocking activities.

WHAT WAS DONE

These methods are being used as part of a community-based coral reef management project, The Coral Gardens Initiative, being implemented in the Pacific region by the Foundation for the Peoples of the South Pacific International, in partnership with local FSPI affiliated NGOs. The initial site (chosen as an ICRAN model site) is the eight coastal villages of Cuvu and Tuva Districts in Fiji. The project is being implemented in Fiji by FSP-Fiji, recently renamed Partners in Community Development Fiji. An ICRAN extension site is being established in the Solomon Islands, with the FSPI affiliate the Solomon Islands Development Trust. Recently obtained EC funding will allow for expansion to several other countries in the Pacific, and Counterpart International (the FSP-USA affiliate) is developing a planned Caribbean extension of the work.

Three basic types of coral cover enhancement interventions are being used in the sites, each targeting a different habitat type: shallow water high energy reef flat areas, rubble-dominated lagoon areas resulting from dynamite fishing, coral harvesting, or severe storms, and sand-dominated “lagoon deserts” where coral larvae can not settle but where corals grow well once they are established.

In the main Fiji sites of Cuvu and Tuva districts, five MPAs were established in mid 2001 as part of a community-based plan to restore the fisheries resources on the rather severely degraded and over-fished reefs. The use of Derris plant poisons, although now effectively banned, was rampant at the start of the project. Nutrient pollution and siltation are also a problem in these sites due to proximity to land in a sugarcane growing and tourism development area. Chronic COT outbreaks and overgrowth by macroalgae appear to be related to land-based nutrification. Extreme temperatures and periodic storm wave assault are problematic to these fringing reef sites, so the restoration methods used for such degraded reef flats must resist waves and high temperatures. In these challenging sites, early coral transplanting experiments were mostly failures, being destroyed by storm waves, killed from temperature-induced bleaching, or killed by COT predation. However, in recent months, a major breakthrough in the methods has occurred, and the restoration work is now focused on first constructing hollow, igloo-shaped, stone and cement “fish houses’ about 40-50cm high x 40-50cm wide at the base. These multi-windowed fish houses are cemented into tide pools on the reef flats and serve as stable bases for planting temperature tolerant corals and restocking tridacnid clams, situated above the often shifting reef debris. In these sites, increased fish numbers due to increased habitat (and MPA establishment) appears to be having the effect of reducing algal overgrowth in the sites.

Rubble beds dominate the lagoons of Malaita, Solomon Islands, and are the end result of generations of coral harvesting for burning to produce betel nut lime and for use as fill material to construct “artificial islands” in the lagoons, as well as a recent upsurge in dynamite fishing. Where reefs have been converted into shifting gravel-sized rubble, coral larvae can still recruit, however, the rapid turn-over of the rubble kills the tiny corals, so reefs converted into rubble tend to remain in a degraded state. For these low-energy rubble sites, restoration can be a rather simple process, scattering coral branches of various sizes into small test patches, and expanding the work (or not) based on the results. “Staghorn” *Acropora* corals have work well due to their rapid growth and ability to reattach to and re-cement the rubble. *Porites* corals, although they grow considerably slower, tend to work better in silty areas or areas with periodic freshwater runoff. Coral branches >15cm tend to work better than smaller sizes. The next phase of the Solomon Islands lagoon restoration work will involve training the coral harvesters in sustainable coral farming to replace the wild coral harvesting.

Sheltered lagoon areas of barren sand are being enhanced as well, particularly at Marau Sound, Solomon Islands, transplanting coral colonies directly onto the sand. Smaller and less 3-dimensional fragments often die due to close contact with the sand, so only highly branched and larger colonies are used, often taken from corals grown in the rubble restoration sites. Isolated patch reefs created in this way serve as nursery habitat for fish recruiting from the larvae, particularly the coral colonies planted further (>50m) from the reef (Bowden-Kerby 2001b). Patch reef creation shows promise for use as a tool for reef fisheries management, particularly in sandy atoll lagoons.

Ephemeral populations of juvenile corals often occur in extremely shallow areas where corals recruit well from the larvae, but where long-term survival is jeopardized by impending and recurring disturbance (Connell, 1973, 1997; Glynn and Mate, 1997). To avoid negative impacts to healthy coral populations on the reefs, corals for the various types of restoration work were obtained from populations of such jeopardized corals: taken from extreme shallows where coral colonies are exposed to air and rainfall during extreme low tides. Other corals were obtained by pruning back fast growing corals from situations of overgrowth, where they were killing slow-growing massive corals. For future work, coral colonies for use in transplanting on sand can be grown in about 2-3 years from smaller fragments scattered on rubble beds (Bowden-Kerby, 1997a).

Within the Fiji MPAs, 500 tridacnid clams of three species have been restocked, obtained from the Department of Fisheries clam hatchery. Close to 1,000 *Trochus*, 1,500 *Turbo*, 2,000 chitons, 2,000 *Anadara* clams, and 50 *Lambis* spider conch were also restocked into appropriate habitats, obtained from women fishers in Rewa province. Predatory snails have caused mortality among the *Tridacna* clams and have thus been a problem in the sites. Storm waves also smashed the clam cages and many clams were lost. However, clams placed directly onto the stone and concrete fish houses only five days before the storm were not swept away, and these clams, being elevated above the substratum, appear to have a lower snail predation rate as well. Juvenile *Trochus* have been observed in abundance inside fish house structures and appear to prefer such cryptic habitat (fish houses that had not been cemented to the reef allowed for close inspection). While these sorts of results can at best be considered preliminary, they indicate a potential for enhancement of reef restocking areas to conform to the conditions of the particular reef area.

Self-assessment of Success in Achieving the Objectives to Date

The work demonstrates that simple and low-cost restoration methods are already available for use by reef managers and communities in support of no-fishing MPA functioning, intervening to restore degraded, non-recovering coral reef areas and invertebrate brood stocks. A diversity of colorful fish have moved into the corals at all Solomons and Fiji sites, and experimental areas have become popular tourist attractions with guests at the Fiji's Shangri-La Resort and Marau Sound's Tavanipupu Resort. This is an added benefit to the work. These resorts have been major financial and in-kind contributors in both countries.

LESSONS LEARNED

Involving members of the reef owning communities actively in the coral planting work is helping build a deeper understanding and appreciation for corals as fish habitat as well as the requirements of corals for survival and health. Involvement of communities in restocking trials is also helping captivate, encourage, and educate the communities, regardless of its less definitive biological impacts. Particular individuals have been especially captivated by the work, and these active individuals need more intense follow-up and interaction with the reef experts, in order that the work be monitored closely and proceed in a cautionary manner. Small trials are needed for each potential new site, and adequate time is required to determine the best way to proceed in each site, and this likely requires more experience than presently exists at the community level. Whatever is attempted, it is important to conduct small trials first, to see what is appropriate for a particular site, and to expand, modify, or abort the work based on these results. Scientific and community-based monitoring of the restocking and restoration sites is required. The results from one site do not always apply exactly to another site, and there is a great variability among corals and environmental conditions, so it is important to incorporate diversity into the restoration sites and to adapt the methods through trial and error.

In addition to the restoration methods described above, other types of community-based interventions may need to be implemented, particularly those addressing the underlying causes of reef decline at a particular site: COT starfish removal, mangrove replanting where streams enter the reef systems, watershed and waste management activities. Traditional coral harvesting and destructive fishing practices must come under sustainable management or alternatives found, otherwise restoring reefs outside the MPA areas is useless. Restocking should obviously be implemented only in MPA areas effectively and permanently closed to fishing.

Restocking, while encouraging to the community, and contributing to the overall ecological balance of the reef, may or may not accelerate the recovery of the particular restocked species in the direct reef management area, particularly if larvae are swept away quickly by currents. Species with short-lived larvae might be more practical for restocking that species with weeks or months-long larval cycles. However, there are some indications that for particular species like *Tridacna* clams (Hugh Govan, personal communication), the presence of adults may serve as a settlement signal for larvae coming from elsewhere, aiding in their recruitment to the reef. The presence of oceanic gyres in many areas may also return the developing larvae to the source reefs over time.

RECOMMENDATIONS

Before widespread transplantation is attempted at a specific site, test plantings using diverse clones and species should be carried out and observed for at least a year, to determine feasibility: site suitability, relative fragment mortality, and possible methods modifications required to increase success. Considerable mortality in the initial months following transplantation can be acceptable if particular clones increase rapidly and compensate for losses (Bowden-Kerby, 2001b). Subsequent transplants could be obtained from the newly established and most rapidly growing colonies, and survival would be expected to be greater in the next generation, as the surviving corals would likely those best adapted to the particular site. Once transplants become adult size at about six months to one year, colonies could again be sub-divided to increase the number of transplants and to prevent spawning and the associated decrease in growth rates that accompany adulthood.

It is important to include as diverse an assortment of coral transplants as is practical in the sites, understanding that corals less suited to the particular site will eventually die out. Coral reef restoration sites including as much species diversity and within-species clonal diversity as possible would help ensure resilience of the coral population to changing environmental conditions, and would provide for greater disease resistance as well as greater spawning compatibility.

Because massive coral species grow considerably slower than branching corals, they generally have been neglected in coral reef restoration research. The establishment of massive corals in transplantation sites where their survival over time is likely could have long-lasting positive impacts, as these corals live for centuries and survive severe storms, while branching species are more ephemeral, being easily killed or swept away.

More work is needed to refine the methods further, with more statistical verification that the work is helping with MPA recovery. Much of the work should therefore be considered preliminary in nature. There is a need for more in-depth study of all aspects of coral transplantation for reef restoration presented here, including:

1. time of year of transplantation,
2. air exposure times for various coral species and genotypes,
3. depth changes and transplantation,
4. effects of various depths and energy regimes on various methods,
5. sustainable sources for coral transplants on reefs,
6. procedures for effective repair of small-scale reef damage,
7. impacts of planting isolated coral colonies in sand flats,
8. possibility of facilitated larval settlement by the presence of corals,
9. coral recruitment subsequent to substratum stabilization by transplants,
10. impacts of coral transplantation on biodiversity,
11. climate change mitigation potentials of coral transplantation,
12. potential for transplantation to reduce ciguatoxic habitats,
13. more data on fisheries recovery of MPAs facilitated by coral planting

Obtaining Coral Transplants with Minimal Impact to Healthy Reefs

Rescuing jeopardized juvenile corals from extremely shallow reef areas before mortality events ensue and transplanting them into deeper reef areas allows corals to survive, and can provide a sustainable source for coral transplants (Ortiz-Prosper and Bowden-Kerby, 1999). Another situation where jeopardized corals can also be found is in situations where slow growing massive corals are being competitively overgrown by fast growing “weedy” coral species. Jackson (1991) suggests that coral cover is a good “proxy” for competition among corals. Liddell and Ohlhorst (1987) found a highly significant negative correlation between species diversity and coral cover, with maximum diversity occurring at about 30% cover, above which diversity of corals decreased progressively as coral density increased, becoming nearly monospecific in shallow water at 60-70% cover. This condition of low diversity at high coral densities appears to be widespread globally.

Intervention to prevent competitive overgrowth and coral mortality by “coral gardening”, trimming overgrowing coral branches or replanting juvenile massive corals to appropriate restoration sites, offers promise as a means for obtaining coral transplants sustainably while lowering the mortality of slower-growing corals on reefs, helping increase reef biodiversity.

Limitations and Potentials of Coral Transplanting for Reef Restoration

Severely degraded reefs chronically impacted by siltation, pollution, or ongoing destructive fishing will not recover coral populations naturally (Connell, 1997), and transplantation cannot be expected to restore corals to such chronically disturbed reefs, as long as conditions causing coral mortality continue. Only when the problems that led to the degradation of a particular reef have ceased can coral reef restoration be considered (Maragos, 1992). Long-term restoration of coral cover and associated organisms to reefs degraded by sedimentation or nutrient runoff may first require reforestation of watersheds and waste management measures (Hunter and Evans, 1995; Allard, 1999; Jameson et al., 1999).

If natural processes of larval recruitment and fragmentation lead to recovery of coral populations without intervention, restoration efforts involving transplantation are not required (Quinn and Kojis, 1999; Bowden-Kerby 2001b). Discontinuing negative impacts on coral reefs alone may be sufficient for the recovery of some reefs, and coral transplantation may be contraindicated on such reefs, as coral transplants could potentially overgrow and kill diverse natural coral recruits.

Perhaps the greatest potential of low-tech transplantation is for use on degraded coral reefs where corals can survive, but where altered substrata prevent larval recruitment or post-recruitment survival. Such conditions unfavorable to natural recovery prevail on reefs where blast fishing, reef flat dredging, or coral harvesting have left unconsolidated rubble, sand, and silt where coral thickets once prevailed (Lindahl, 1998; Fox et al., 1999).

A Cautionary Approach to Coral Manipulations

Coral reef restoration methods that involve species manipulations and transplanting corals could also have unforeseen consequences to the basic ecology of partially intact reef systems, or could degrade or alter donor reefs, and thus monitoring and a precautionary approach is required. The unwise application of coral transplantation might favor unnatural species mixes and distributions and could cause the demise of particular species. For example, stag horn species of *Acropora* have the ability to out-compete slower growing and long-lived massive corals, and these massive corals are more resistant to cyclones and might also be more tolerant of temperature and salinity extremes. Indiscriminate transplanting of *Acropora* could lower overall coral diversity on reefs and could make reefs more vulnerable to disturbance.

Transplanting corals for coral reef restoration should by no means be regarded as a universal solution for the dire position coral reefs are facing today. Prevention of coral reef decline is a considerably more effective management strategy than restoration. If the limited effectiveness of coral reef restoration is not fully appreciated, especially by the press, restoration efforts might give a false sense of hope, dissipating the sense of urgency for coral reef conservation.

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