Conservation of Coral Reefs through Active Restoration Measures: Recent Approaches and Last Decade Progress

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The scientific discipline of active restoration of denuded coral reef areas has drawn much attention in the past decade as it became evident that this ecosystem does not often recover naturally from anthropogenic stress without manipulation. Essentially, the choices are either the continuous degradation of the reefs or active restoration to encourage reef development. As a result, worldwide restoration operations during the past decade have been recognized as being a major tool for reef rehabilitation. This situation has also stirred discussions and debates on the various restoration measures suggested as management options, supplementary to the traditional conservation acts. The present essay reviews past decade’s (1994–2004) approaches and advances in coral reef restoration. While direct coral transplantation is still the primer vehicle of operations used, the concept of in situ and ex situ coral nurseries (the gardening concept), where coral materials (nubbins, branches, spats) are maricultured to a size suitable for transplantation, has been gaining recognition. The use of nubbins (down to the size of a single or few polyps) has been suggested and employed as a unique technique for mass production of coral colonies. Restoration of ship grounding sites and the use of artificial reefs have become common tools for specific restoration needs. Substrate stabilization, 3-D structural consideration of developing colonies, and the use of molecular/biochemical tools are part of novel technology approaches developed in the past decade. Economic considerations for reef restoration have become an important avenue for evaluating success of restoration activities. It has been suggested that landscape restoration and restoration genetics are important issues to be studied. In the future, as coral reef restoration may become the dominant conservation act, there would be the need not only to develop improved protocols but also to define the conceptual bases.

Introduction

The coral reef, of the most productive biological ecosystems on earth, has been degrading over the past 2–3 decades in an alarming rate throughout the world (1–4; and literature therein). This global decline in reef status and health (5) reveals that natural recruitment may be prolonged or prevented because of permanent shifts in reef communities and changes in physical/environmental conditions (6, 7).

Consequently, varying levels of coral reef degradation were documented around the world showing that many reefs have already been partly or completely destroyed. This situation stirred the discussion on the development of various active restoration measures to be applied as supplementary management options to the traditional operations of conservation (1, 2, 8–14).

The terms “restoration” and “conservation” are often used in the context of efforts to “preserve” the original habitats or to “replace equivalent” lost habitats or destroyed populations. However, restoration ecology suggests the notion of active measures, while conservation biology focuses on “passive” measures, allowing natural processes to mitigate impacts without or with only minimal human interference (2). Restoration ecology, therefore, at its core, the supposition that at least some proportion of habitat loss is recoverable through artificial manipulations that facilitate reef development. Consequently, Hobbs and Harris (15) have predicted that restoration ecology will eventually become the dominant discipline in environmental science during the 21st century. However, the technical rehabilitation activities in the natural environments have already progressed faster than the conceptual scientific assays (16), and in various theoretical aspects restoration ecology has not yet matured to a well-established discipline. The literature (17) further reveals that while conservation biology has been far more biased toward zoological aspects (by nearly 3 to 1 in the number of manuscripts published), restoration ecology, in comparison, has been far more biased toward the botanical aspects (by more than 4 to 1;17). It is therefore not surprising that principles of forest restoration, concepts, and theories are currently being applied in coral reef rehabilitation (2, 18).

Reef restoration has been employed in various situations created by fundamentally different causes, and restoration activities must consider the actual (or perceived) cause of coral degradation. In reality, restoring any degraded reef area is complex, because degradation involves various paths of change in species abundances and ecosystem functions. However, until recently, restoration of coral reefs “has not been widely applied as a management option” (19), and most studies were based on small-scale, short-term experimental protocols and on a few sampling episodes, analyzing only some ecological/biological attributes. During the past decade, worldwide coral reef restoration operations have been more frequently employed and tested in various reef localities (14), and the concept of active restoration measures has been acknowledged as an important approach for reef rehabilitation (1, 2, 8, 9, 18, 20). Various methodologies for coral reef restoration have been proposed to enhance reef recovery. The present review analyzes the most common approaches used recently for reef rehabilitation (addressing primarily...
biological/ecological impacts) and presents the advances documented in scientific articles published in the past decade (1994–2004).

**Coral Transplantation**

Most reclamation measures of denuded reef areas are based on the concept that the new established coral colonies should originate from external source populations. Only in areas of ship grounding (see below) or hurricane damage (21), restricted stocks of local whole colonies and colony fragments are used.

During the past decade, coral transplantation measures have frequently been employed and have gained recognition as the prime management tool for reef restoration (2). The rationale behind this practice is replacement of dead coral colonies with new ones to accelerate natural recovery. Several types of source material are available for transplantation. These include: transplantation of small or large, whole coral colonies (8, 13, 19, 22–28), enhanced by deliberate seeding of planula larvae (8, 13, 29–32), transplantation of coral branches or fragments (4, 9, 13, 33–40), and the transplantation of nubbins (41–48).

Several studies discussed different facets and parameters for the rationale of coral transplantation: (a) choice of coral species (1, 8, 10, 11, 49); (b) transplantation efficacy as compared to natural recruitment processes and rates (2, 10, 50); (c) colony pattern formation (20); (d) major goals for impacts of specific restoration acts on donor reef areas (1, 2, 8, 10, 50); (e) survival of transplanted coral material (8, 11, 28, 49); (f) growth rates of transplanted colonies as compared to naturally recruited colonies (10, 11, 28, 50); (g) the type of substrate (2, 11, 28); and (h) the drawback of reduced fecundity in transplanted and/or donor colonies (1, 8).

When comparing different transplantation measures employed on diverse coral species at distant reef sites, it is evident that each transplantation parameter tested is characterized by high variation. For example, Raymundo et al. (30) recorded <5% mortality of transplants (size >10 mm diameter) over 6 weeks, while on a high energy reef flat in the Maldives, Edwards and Clark (10) recorded, after 2 years, mortality of 5–50% for nine coral species transplanted onto concrete mats. Clark and Edwards (22) transplanted entire coral colonies (primarily *Acropora*, *Porites*, *Porites*, *Porites*, and *Porites*) onto three, approximately 20 m² areas of Armolflex concrete mats, on a reef-flat, which had been severely degraded by coral mining. Loss of most coral colonies occurred during the first 7 months. At the end of the experiment, 51% survivorship was recorded on 530 transplants that were monitored over 28 months. Contrarily, only 3% mortality was recorded 1 month after transplanting of 2150 scleractinian colonies, 428 octocoral colonies, and 488 sponges onto a small area off Cozumel Island, Mexico (24). A 8.2% mortality was observed after 27 months following the transplantation of 271 coral colonies in southwest Florida (27), and in a study at La Parguera, Puerto Rico, only 7% mortality (3 of 42 transplanted colonies of 6 coral species) was recorded after 1 year (26).

Ecologically sound management of coral transplantation should consider the growth and mortality rates of transplants in relation to their sizes. Although studies have yielded variable results, possibly due to the use of different species, divergent protocols employed, and various habitats restored, most outcomes confirmed size-dependent survivorship (4, 10, 30, 34, 42, 44, 51). However, several studies (33, 35) found no evidence of size-dependent mortality in *Acropora palifera* fragments transplanted onto reef flat rubble, or *Porites* transplants deployed into several reef habitats, respectively. The generalization of size-dependent growth rate and size-dependent pattern formation had also been studied and has already guided several transplantation operations (4, 20, 30).

Coral transplantation measures might be more effective when combined with other management measures such as substrate stabilization, either by conventional protocols (34, 37, 40, 48, 52) or by electrochemical deposition of calcium carbonate (41, 46, 47, 53, 54). Those studies (see below) clearly revealed that dispersing coral branches or attaching coral fragments onto unstable rubble or coarse sand were ineffective for restoring coral sites (except in specific cases of sheltered areas; [34]). In those unstable substrates, natural-based recruitment processes were inhibited because settled coral colonies were either buried or smothered by the loose substrate (36, 40).

Transplantation of coral colonies/fragments onto denuded reef areas is, therefore, best employed when: (a) disturbed reef area undergoes a “phase shift” to communities dominated by soft corals or macroalgae (that limit recovery of hard coral colonies; [46]); (b) when natural recruitment is unlikely or limited, resulting from diminution of planula larvae as a source material, or from the existence of unconsolidated substrate (22, 40, 50, 53); (c) when enough material of donor coral colonies is available (10, 25, 26); and (d) where water quality is good enough to sustain growth and survival of coral colonies (10).

**In Situ and Ex Situ Coral Nurseries**

Transplantation of coral colonies/fragments has been employed as an essential methodology for expediting the recovery of denuded reef areas. However, while the techniques used for removal of coral materials, their transportation, and reattachment are straightforward, varying degrees of success have been reported, stemming from transplantation stress, insufficient donor colonies, and too small fragments, as the major source for transplantation (10, 38, 64). In other cases, the failure of corals to recover denuded reef areas also reflects post settlement mortalities (55). To minimize or circumvent these problems, Rinkevich (8) suggested the strategy of “gardening coral reefs”, a two-step protocol in which the central concept is the mariculture of coral recruits (spats, nubbins, coral fragments, and small coral colonies) in nurseries. At first, large in situ or ex situ pools of farmed corals and/or spats can be constructed. In situ nurseries are installed in sheltered zones, where the different types of coral recruits are maricultured to an adequate size. In the second step, nursery-grown coral colonies are transplanted to degraded reef sites. This strategy is theoretically associated with ideas of terrestrial forest plantation (18). Silviculture is a core strategy in forest restoration programs (56). The concept of nursery installed on the sea floor has already been applied to corals (1, 4) and to other reef invertebrates, such as the mariculture of *Tridacna* juveniles within plastic cages (57). Recently, the approach of “mid water coral nursery” has been successfully tested in the Gulf of Elat, Red Sea (18, 58). This study found that a floating coral nursery (at 6 m depth, 14 m above sea floor) provides improved environmental conditions to the growing colonies (enhanced water flow, optimal light conditions, lack of sedimentation, and elimination of corallivorous organisms).

During the years since the gardening concept had first been suggested (8), several studies tested the applicability, the feasibility, and the detailed developed protocols for holding corals in nurseries under in situ conditions (1, 4, 29, 33, 58), in open-seawater ex situ systems (44), or in closed-seawater ex situ facilities (aquaria, holding tanks; [30, 31, 44, 59–62]).

One of the major ex situ restoration approaches is the collection, settlement, and maintenance of planula larvae and spats under optimal conditions (13). For example, Raymundo et al. (30) used planulæ collected from wild gravid
colonies of *Porites damicornis*. The planulae were allowed to settle and then were reared for up to 6 months (to sizes > 1 cm in diameter) to seven reefs in the Philippines. Within the first 6 weeks of transplantation, over 95% survival was recorded. Richmond (59) collected gravid colonies from reefs at Guam and maintained the corals in the laboratory until they spawned. Larvae were nurtured ex situ until they matured and were ready to settle and then returned to the reef. A similar approach was also tested with broadcast spawners. Szmat (61) reared colonies of *Montastrea annularis* and *Acropora palmate*, following the successful collection and settlement of spawn larvae in the laboratory and in the field. Sammarco et al. (63) cultured coral larvae in laboratory tanks until fully developed and competent to settle, then seeded into the center of eddies associated with target reefs. They considered that larvae could be retained in eddies for 1–3 weeks, promoting enhanced local settlement. At least in one case, larvae of *Acropora florida* were collected in Okinawa, Japan, and then air-transported to Germany where they were settled and maintained in a closed ex situ system (31).

The in situ nursery approach sustains the mariculture of nubbins, coral fragments, and small colonies. A protected nursery phase (gardening) provides the transplanted material with an acclimation period, essential for increasing post transplantation survivorship and growth to size suitable for transplantation. The transplantation of nursery-grown “propagules” back into their natal reef helps in preventing genet and species extinctions in degrading sites, thus exercising the “rescue effect” (64) on a local scale. It also preserves the genetic heterogeneity of coral populations (2). A coral nursery may also be considered as a pool for local species that supply reef-managers with unlimited coral colonies for sustainable management (1, 2, 8, 9, 10, 58). The two-step “coral gardening” methodology concept was also suggested (33) for back reef and reef flat rehabilitation. In this study, reef flat rubble areas were initially used as natural nursery sites to culture unattached *Acropora* fragments into 25–50 cm³ size colonies (over 2–3 years). Those colonies then either were used as a new source of fragments for transplantation or were transplanted to back-reef lagoon sites. Reef flat rubble areas, however, are some of the most troubled environments and less than ideal for nursery areas. Oren and Benayahu (29) used a nursery reef that was composed of PVC plates, attached to a wire, both vertically and horizontally. Propagules of two coral species, the stony coral *Stylophora pistillata* and the soft coral *Dendronephthya hemprichi*, were transplanted to this nursery. The results indicated that vertical artificial surfaces offer the optimal biotic and abiotic conditions for the survival of the two examined corals. The vertical plates were exposed to low sedimentation rates, revealed low coverage of turf-algae, minimal grazing by sea urchins, and absence of the competitor tunicate *Didemnum* sp.

In a detailed study, Epstein et al. (1) established in situ nurseries in Eilat’s (northern Red Sea) shallow waters, sheltering three types of coral materials that were taken from the branching species *Stylophora pistillata* (small colonies, branch fragments, and spat). Nurseries were monitored for up to 2 years. Pruning more than 10% of donor colonies’ branches increased mortality, whereas surviving colonies displayed reduced reproductive activity. Maricultured branches, however, exceeded donor colony’s life span and reproductive activity and added 0.5–45% skeletal mass per year. Forty-four percent of the small colonies survived after 1.5-year mariculture, revealing average yearly growth of 7% ± 32%. Three months ex situ maintenance of coral spat (sexual recruits) prior to the in situ nursery phase increased survivorship. Within the next 1.5 years, they developed into colonies at sizes 3–4 cm diameter. Nursery periods of 2 years, 4–5 years, and more than >5 years were estimated for small colonies, spat, and isolated branches, respectively (1), although much reduced in nursery periods were recorded for mid-water floating nurseries (18, 58).

Soong and Chen (4) performed similar studies on branch fragments taken from *Acropora pulchra* colonies in southern Taiwan. Branches were maintained in a shallow water nursery to test, in situ, the possible effects of different factors on the generation of new branches and on the initial skeletal extension rates of transplants. The variables studied were the origin and length of the fragments, their new orientation, presence of tissue injury, and position of the fragment. All of these factors were found to make a difference in either one or both aspects of coral growth (i.e., branching frequency and skeletal extension rate). These variables clearly determined the success rate of a small fragment developing into a large colony that had a much higher probability to survive and grow on its own. This study revealed that very small fragments of the *Acropora* species tested (e.g., 1 cm) were unsuitable for coral generation in nurseries, because they tend to be smothered by algae or simply get lost, perhaps due to predation. Furthermore, thick branches of *Acropora* tend to develop cores without live tissues; fouling organisms, such as algae, once growing and establishing themselves on these exposed surfaces inhibit further growth of the corals. Additionally, from the perspective of the donor colonies, removal of long branches may render the original colonies infertile or result in low fecundities for some years (1). This study also revealed that about 4 cm long *Acropora* fragments were the most suitable for use in coral colony generation. As new branches were being generated on the cut edges of the fragments, it was recommended that longer branches should be broken into 4 cm fragments to produce more cut edges per unit length of a branch fragment.

Ex situ and in situ coral reef maricultures (the “gardening” concept) are improved practices to the common, but potentially harmful, protocols of direct coral transplantation. This concept has already been tested for its applicability and use in various coral reefs worldwide. However, site-specific considerations and the use of different local coral species as donors require the development of different specific protocols, tailored to the conditions at diverse reef areas. More importantly, part of the coral material reared in nurseries can then be fragmented again to generate additional coral colonies, reducing restoration impacts on native coral populations. Moreover, in situ coral nurseries can supply transplantation operations with corals adapted to natural reef conditions (1, 18), while ex situ coral nurseries may facilitate the yield from coral planulae, increasing genetic variability of transplanted colonies (8). Both ex situ and in situ approaches can also provide ample material for the coral trade, thus reducing collections of coral colonies from the wild (31, 62).

**Restoration of Ship Grounding Sites**

Ever since humans have first begun building boats, ships have grounded on coral reefs. Impacts of ship aground include dislodging and fracturing corals, pulverizing coral skeletons, displacing sediments from ground, and destroying the 3-D structural complexity of the reef (65). Salvage operations usually add to reef damages, without counting damages by fuel and cargo slicking from the ruptured hull. This causes acute and long-lasting effects on regenerative processes of coral communities (66). Resulting from low natural fusion levels of fragments to substrates, virtually complete mortality of living coral colonies fragments may occur during subsequent storms (67). In Florida, boating is the primary cause for reef damage, mainly by ships colliding into reefs (67–70; and literature therein; but see also cases from the Red Sea...
(71)). In the Florida Keys alone, >500 small vessel groundings are reported annually with, at least, 2–3 times that number unreported (63).

Efforts to restore reef sites damaged by ship grounding include activities such as salvaging coral colonies, coral fragments, and sponges, removing loose debris from the reef floor, reconstructing 3-D structural complexity of the reef, and reattaching detached corals and sponges to cleared reef substrates or specially designed artificial reef structures (65, 71, 72). A major difference in this type of reef restoration, as compared to other restoration measures, is that most, if not all, coral material for restoration comes from the damaged sites and not from adjacent coral reef populations. Another challenge to tackle is the massive amounts of rubble. The options include: (a) leaving it in place and stabilizing it with cements; (b) moving it far from the site and dumping it in deep water; or (c) reconfiguring it by moving it off reef and building piles where it can do no harm. After removing the debris from the reef platform, corals and other sessile benthic organisms can be transplanted on the damaged area (65, 69).

Not all restoration activities yielded similar outcomes. For example, following two major ship groundings that damaged reef habitats in the northern Florida Keys National Marine sanctuary, the M/V Alec Owen Maitland and the M/V Elpis, Miller and Barimo (68) assessed the restoration activities performed 6 years earlier. The structural restoration entailed emplacement of exogenous structures to stabilize loose rubble and fill-in lost reef framework. These two projects of the same age and similar location, but different in depth and structure design, provided an opportunity to evaluate restoration success in terms of the reestablishment of coral population via in situ recruitment. In fact, coral assemblages vary markedly in density, size distribution, and diversity between the two sites. At the Maitland site, coral recruitment was positively associated with rough lime rocks embedded in the concrete restoration structure, but the coral assemblage was dominated by a single species, Porites astreoides. At the Elpis site, the juvenile coral assemblages had substantial representation by four taxa associated with benthic algal communities. Miller and Barimo (68) hypothesized that differences in algal assemblages resulted from varying structural designs (differing material, surface texture, and/or surface orientation) that mediated differential coral recruitment success at the two sites. Ironically, the less ecologically successful restoration structure at Maitland was the more costly to construct (almost double). Therefore, despite the abundant resources spent on ship grounding restoration projects, the lack of clear scientific goals, the need for hypothesis-driven monitoring efforts, and a general lack of tools to assess restoration success have hindered progress in the reef restoration field (69, 70, 73).

The practices of coral reef restoration by the U.S. government often involve expensive construction projects (at least hundreds of thousand of USS per case), for example, repairing reef framework damaged by acute impacts such as ship grounding (www.darp.noaa.gov/seregion/elpis.htm). Often, the stated goals of such projects are described as “providing structural stability to reduce future incidental damages from loose material and increasing structural complexity of the site to enhance recovery of benthic and fish assemblages” (65, 70). However, the results do not always meet the expectations. The capability to foresee outcomes is therefore an important parameter for assessing costs and finding the best protocols for specific site restoration. Restoration “success criteria” such as those suggested by Jaap (65) and Precht et al. (69) should be set up and evaluated following monitoring programs and within specific time frames. It may be postulated, therefore, that adequate in situ nurseries would provide additional coral material for transplant, adding to the restoration activities that, up to date, rely on local broken colonies.

Use of Artificial Reefs

In many cases, the literature refers to artificial structures introduced onto the seabed as artificial reefs. While various definitions for artificial reefs are summarized by Schultma-

cher (74), within the context of this review, only artificial reefs aiming to the rehabilitation of denuded reef areas are considered, especially those focusing on establishment of corals and other reef invertebrates rather than on the development of fish communities.

Many types of disturbances, separately and in combination, are changing the face of the reefs, worldwide. In many cases (such as ship grounding, hurricanes, mining, coral blasting, etc.), the 3-D structure of the reef is damaged so that regular restoration procedures of transplantation cannot repair the damages or reverse the situation to conditions similar to those prior to the disturbances. In other cases, new reef constructions are needed to reduce the pressure of tourist activities (diving, snorkeling, and fishing) on popular reef sites. These needs have been evaluated during the past decade in various studies (22, 26, 53, 74–80).

It has often been noted that artificial reef structures were chosen for a restoration project simply because they were used elsewhere. However, it is questionable whether the results obtained at one restoration site can be extrapolated onto another. For example, Pamintuan et al. (75) examined 32 concrete tent-like structures in clear versus silty areas in the Philippines. The reefs at the silty site had greater cover percentage and higher numbers of sessile species and mobile invertebrate species. The authors attributed the difference in initial colonization to negative phototactic behavior of settling larvae of some species and concluded that the divergent development at the two site types was primarily due to physical factors. Likewise, Abelson and Shlesinger (80) tried to follow the effects of morphology of artificial reefs, substrate types, and location on the succession of reef organisms. They constructed two types of limestone-rock aggregates: a randomly aggregated (RA) reef, comprised of relatively small rocks, and an orderly aggregated (OA) reef, composed of relatively big rocks. Communities were tallied every 4–6 months for more than 4 years, with a final coral census taken after 100 months. While the OA reef was colonized by significantly higher numbers of reef fish than the RA reef, and reached its carrying capacity faster (30 months versus 50 months), the number of reef-building corals on the RA reef was significantly higher (in terms of both species and colonies) than on the OA reef. Its plateau had not even been reached after 100 months. By testing a different approach, Lam (78) used coal waste concrete blocks as reef units for reef restoration and followed coral colonies transplanted on these structures. One year later, only 62% survivorship was recorded. Ortiz-Proser et al. (26) employed a similar approach of planting coral colonies on artificial modules (Reef Balls). After 1 year, only a very small number of natural recruited were recorded.

Clark and Edwards (76) investigated the feasibility of using artificial reef structures to promote rehabilitation of a severely degraded reef flat. In their study, 360 tons of concrete structures were deployed over a 4-ha experimental site on a 1–2 m deep reef flat in the Maldives, which had been mined for corals 20 years earlier and still showed less than 2.5% live coral cover. Colonization of coral colonies was monitored on four sets, each of three, 50 m² artificial reef structures of varying topographic complexity, and on one set of three 50 m² replicate mining control areas. In a follow-up study, Clark and Edwards (81) found about 500 coral recruits on the larger structures, some of which were approaching 25 cm in diameter after 3.5 years. Those coral numbers resembled the
census of Wilhelmsson et al. (77) done on larger artificial reefs; some were scuttled vessels submerged about 10 years earlier, which yielded up to ca. 17 coral colonies per m² were recorded. Using the same rationale, Bachthar (82) introduced small concrete blocks to promote scleractinian corals recruitment in Gili Islands, Indonesia, and Thongham and Chansang (83) recorded that coral recruitment and growth on cylinders of concrete modules (Thailand) were higher than on natural substrates.

To date, artificial reefs have not proven to be an efficient restoration tool, neither when used for transplantation measures nor when left for natural recruitment. For over 20 years, Schuhmacher (74) followed several artificial reefs in Elat, Red Sea, established some 30 years ago. That long-term study revealed that none of the artificial reefs developed coral communities on the 3-D profile. The artificial reefs had not been covered by corals during the long period and were limited in coral recruits as compared to adjacent natural habitats. Similar conclusions can be drawn from other long-term studies on artificial reefs measures aiming to restore denuded reef areas (76, 81).

**Substrate Stabilization**

In many coral reefs (such as Florida Keys; [44]), the hard substrates are just veneered by cemented live stone overlaying unconsolidated loose reefal material. Blast fishing (52, 84, 85) and other human interventions such as ship groundings and natural catastrophes such as hurricanes (12, 44) shatter reef structures and substrates into rubble and sand, weakening the reef’s ability to recover from natural processes of recruitment (19, 48, 65, 69, 84). Substrate turned into rubble is dynamic, easily shifted by storms and currents, and its fine fractions are continuously resuspended. Settled corals may therefore endure higher sedimentation and increased mortality than corals damaged by overturning and abrasion. Moreover, this type of habitat is unfavorable for some coral aggregations that successfully compete with new hard coral settlers (46). In the past decade, substrate stabilization as an important restoration tools has been widely accepted and continuously used for both enhancing natural recruitment and articulating transplanted branches and coral colonies (12, 19, 22, 40, 41, 44, 46, 48, 51, 53, 65, 69, 71, 74, 84).

A variety of material and methodologies has been used for substrate stabilization. Much of the work has been done with artificial material laying on top or attached to the reef substrate, preferably concrete (usually reinforced with steel) or natural rock (22, 48, 52, 74, 76, 81, 85). Others claim success with cheaper and less laborious methods, such as lashing corals to seabed to form a grid.

Depositing CaCO₃/Mg(OH)₂ onto steel frames or onto other conductive material and administering low voltage direct current (<24 V) was employed by several working groups (41, 46, 47, 53, 54, 71, 74, 86). Hiltbert and Goreau (86) also suggested three hypotheses for enhancing growth of corals that are under this low current state: (a) electric field enables carbonate accretion and may cause the precipitated carbonates to attach directly to the skeletons of coral transplants; (b) low current induces CaCO₃ enrichment of water in the immediate vicinity of the coral, thereby enhancing natural calcification; and (c) excess production and release of electrons, due to the electrochemical processes occurring in the coral vicinity, might affect the electron-transport chain for ATP production, where the excess energy could be used for growth enhancement. Recent studies that had tested these hypotheses (46, 47, 71, 74), although documenting high survivorship of attached coral branches, did not reveal an ubiquitous accelerated calcification by live corals, or resulted in lateral growth at the base of branches instead of a longitudinal increase. Goreau et al. (54) have further revealed that corals growing on electrically stimulated substrates had higher densities of zooxanthellae in their tissues and higher rates of symbiotic algal division, coincident with the claim (86) for higher coral skeletal growth rates. Results also showed clearly that electrolysis in seawater induces cathodic accretion of calcium and magnesium minerals on structured framed cathode (made of steel mesh such as chicken wire). As a result, new hard substrate designated to fit local needs and 3-D structures can be generated within short in situ periods. This provides a natural substrate with limestone character, available for transplantation of new coral colonies (41, 53, 71). This is probably one of the best, cheapest, fastest, and environmental friendly substrate stabilization methodologies available today.

**Novel Technology Approaches**

During the past decade, novel approaches for reef restoration have been tested or suggested in laboratory and field experiments. The four most discussed approaches are: (a) the use of coral nubbins; (b) the consideration of initial and final 3-D structures of fragments employed for transplantation; (c) the application of molecular biology tools for evaluating coral health state/stress; and (d) the employment of chemical attractants for in situ and ex situ manipulations of larval settlement (surface chemistry).

**Use of Coral Nubbins.** An important prerequisite for improved ex situ and in situ cultivation methodologies for coral reef restoration is the availability of many ramets from specific coral genets (45, 58, 87). Various in situ and ex situ studies (literature in 4, 8, 33, 35, 37, 44, 60, 88, 89) that evaluated the merits of fragmenting coral colonies into several or many subclones (ramets, nubbins, microcolonies, etc.) revealed that isolated ramets may well survive and grow independently from original genets, even in species for which natural fragmentation is not part of their life history portrait. A valuable and novel approach was the attempt to develop numerous genetically identical colonies from a single genet, by isolating coral fragments to sizes no larger than a single or few polyps (4, 42, 87, 90). Subcloning minute fragments from a coral colony may reduce the stress inflicted on it and yield a high number of ramets amenable for various purposes (91, 92). These isolated minute fragments were named “nubbins”, a term used earlier to characterize isolated tips of whole coral branch (93–95). In those studies, the term “nubbins” referred sometimes to the upper 3 cm-long branch tips as well as to isolated single corallite (94), while in other cases, the 0.6–1 cm-long branch tips were named microcolonies (89). Shafir et al. (45) have suggested using the term “nubbins” literally to characterize minute portions of coral colonies (a single isolated polyp or a coral fragment containing only a few polyps) and to term larger subclones of coral colonies as microcolonies, fragments, or ramets. Shafir et al. (45) further suggested that the successful production of coral colonies by nubbing minute sections from donor colonies could be the preferred methodology for establishing large numbers of genetically identical colonies (ramets) to be used as a valuable substance for different approaches, including restoration practices (1, 58). Not only are nubbins easily pruned from any part of a coral colony, but working with them is quick and yields significant quantities of source material within a short period. A set of 50 nubbins pruned from a 5 cm-long branch can be established within 30 min (58, 92).

**3-D Structural Considerations.** Coral fragments, as small as a single branch or a nubbin (87), initiate after breakage a series of developmental processes to regain the species-specific 3-D colony structure. Loya (96), who examined regeneration in broken colonies of the Indo-Pacific branching coral *Stylophora pistillata*, suggested that an injured colony would allocate resources from its healthy parts to damaged branches to restore its former shape. An isolated ramet,
however, may reconstitute a whole colony configuration from even a minute, nonbranching portion. This mode of regeneration has been characterized by the growth of new, correctly patterned branches and leads to the reformation of whole colonies via an asexual propagation pathway. The idea that the structural complexity of ramets may significantly influence its growth and regenerative ability received little attention, and, when discussed, its importance was attributed to ecological rather than to developmental concepts. One example is Highsmith (88), who briefly remarked that a 3-D ramet shape could prevent branch burial under sediment. Epstein and Rinkevich (20) have recently explored the conception that an architectural complexity threshold of isolated ramet is required for faster regeneration to a full colony structure. It was suggested that the developmental biology concept of colonial pattern formation should be considered as an additional criterion (to the well-discussed criteria of survivorship and growth) in the applied discipline of coral transplantation for reef restoration. This is particularly relevant to branching forms. For example, when evaluating aspects of coral reef restoration, Rinkevich (9) and Epstein and Rinkevich (20) found that the original architecture complexity of isolated ramets might influence their ability to instigate promptly these specific 3-D growth patterns. Similar outcomes were recorded by Soong and Chen (4) working on Acropora, another branching genus. They found that fragments collected from the proximal part of Acropora colonies “were better” (in relation to transplantation measures) from the perspectives of fragment growth and branching.

Molecular Biology Tools. The application of molecular biology tools for evaluating coral health state and stress levels may challenge our understanding of reef ecosystems. Most studies to date have been dominated by morphological-ecological considerations and ecological indices, rooted in our knowledge on coral reef biology. Studying the biochemical response to stressors would improve our ability to analyze the effects of environmental perturbations and their combinatorial outcomes and impacts on restoration activities. It will also improve our understanding of coral-diseases potential impacts on reef rehabilitation, an issue that has yet to be studied or evaluated. Transplant stress may lower natural resistance of coral colonies, leading to their increased susceptibility to pathogens (37). Diseases appearing in transplanted colonies and coral fragments have already been documented in at least two recent publications (37, 67). Many of the stress responses entail rapid synthesis of highly conserved proteins, the heat shock proteins (HSP70s), which are involved in diverse cellular functions in bacteria, mammals, in every species in which they have been sought, including aquatic organisms. Tom et al. (97) characterized the first coral heat shock protein 70 gene, cloned from the branching Indo-Pacific coral Stylophora pistillata (SP-HSP70), to be used as a tool for studying coral stress responses. RT-PCR studies confirmed SP-HSP70 mRNA expression in corals grown within their normal physiological conditions. Using a different approach, Ammar et al. (43) used the metabolic enzymes fructose-1,6-bisphosphatase, and succinate-dehydrogenase from the azooxantellate soft coral Dendronephthya klunzingeri for assessing health status of colonies sampled from two localities at the Gulf of Suez. Low levels of cDNAs were measured at a site characterized by high sedimentation rates. It is expected that extensive use of molecular tools for evaluating stress syndromes in corals will revolutionize our knowledge of the state of health of the coral source material used for restoration practices.

Chemical Signals and Surface Chemistry. It is well known that chemical signals and communications may direct intracolony growth patterns (98) and intraspecific interactions. More recently, Koh (99) has documented antimicrobial chemical defense in 100 scleractinian species, whereas Koh and Sweetman (100) revealed that extracts of Tubastrea coccinea might reduce competition between existing corals and recruits by killing coral larvae. However, only very little attention has been given to this phenomenon in transplantation measures (28, 37).

Still, the topics of surface chemistry (12, 65, 101–104) and the possible inputs of toxic leachate from artificial substrates (12, 105) were discussed and considered as important factors for enhancing natural recruitment. At least one artificial reef manufacturer (Reef Ball) recommended the addition of microslica to concrete to provide a neutral pH surface. In addition, the organic and microbial biofilm that is quickly formed on any clear substrate that is immersed in seawater may provide negative settling cues (104). It is also well documented that initial colonizing microbial algal and invertebrate assemblages may affect settlement of coral larvae (12). Morse et al. (103) and Morse and Morse (101) isolated the chemical glycosaminoglycan from a coraline alga (Hydrolithon boergesenii) that signals Agaricia agaricites humulis larvae to settle. The synthesized material, called “coral flypaper”, proved effective for attracting larvae. Moreover, Morse et al. (102) revealed in all abundant coral genera they tested, a common chemosensory mechanism that is required to bring larvae out of the plankton and onto the reef. There does not yet appear to be extensive work on species-specific responses of invertebrate larvae to differential organic or microbial makeup of biofilms (12), and the field is, therefore, open to studies that will result in manufactured biofilms that promote selective recruitment of specific coral species.

Economic Considerations

Two approaches lead to the various economic considerations for coral reef restoration. The first focuses on the priorities of the underdeveloped world by adopting “low-tech” methodologies (19, 34) that address the type of reef damages most common in underdeveloped countries. These methodologies are designed to be executed by local rural communities in the most accessible localities, lowering cost and effort considerably. The major activity made by these efforts is transplantation of whole coral-colonies or unattached coral fragments (33).

The rationale behind the second economic approach is that the “cost of restoration need to be offset by the benefits gained” (12). This is not an easy task. While, in the past, efforts were made to describe and to quantify costs for reef restoration (literature cited in 12, 73, 81, 106), numbers differ from area to area and from one restoration measure to another. Dominating this scientific discipline are the economic figures from the USA, where “coral restoration has become a mandatory and generally accepted part of the damage assessment and compensation claim process for addressing coral-related ship-grounding incidents” (106).

The U.S. government has been increasingly relying on a method, known as habitat-equivalent analysis (HEA; 72, 73, 106, 107), for assessing the extent to which damaged habitats should be restored or compensated. This approach combines biological and economic information, particularly relating to the timing of lost biological functions, to determine the scale of a suitable compensatory habitat replacement project. A major limitation of this approach is that the ecological, social, and economic beneficiaries of replacement habitats are not necessarily the same as those suffering losses from the damaged habitat. There are numerous other associated problems (73). However, the HEA analysis, as other techniques used, is incapable of addressing whether coral restoration schemes are money well spent (106).
Coral reefs, in most places, are held in public trust by local or governmental agencies. Therefore, placing values on anthropogenic losses (like for coral reef fish, coral colonies, etc.) is not easy to quantify. Any economic estimation is also challenged by fast dynamics (loss of coral colonies/invertebrates/fish, recruitment, natural events) of the coral reef environment that blur evaluations. In the U.S., owners of grounded ships have been paying from $1200 to $11,000 m⁻² for restoration of totally destroyed coral reef (12). Costs of restoration in other places, like in the Maldives (81) or in Tanzania (19), are much less than $1000 m⁻². However, analyses revealed that more expensive restoration measures do not always result in improved situation as compared to less costly cases (68). Therefore, recent studies have asked for better economic assessment of the justification for restoring coral reefs and improving the overall effectiveness of such initiatives (106). Due to the high expenses involved in restoration and the variety of practices that should be studied, we also need innovative economic templates that will guide us in restoring damaged habitats (9).

Future Considerations
Coral reef restoration is beginning to be viewed as requiring further research activities (1, 2, 8, 9, 14, 18, 34, 58). Principles underlining restoration measures consist of part of the plethora of the ill-defined state of the art in this discipline (9, 10). It is, however, increasingly evident that, while past actions for reef restoration were solely regarded as responses to pervasive efforts of human actions and not to natural disasters, degradation of so many coral reefs has come to a point where natural and manmade disasters are treated equally with regards to restoration (9). While restoration is a relatively new scientific discipline that is currently being employed on small scale, it will soon be necessary to be applied to larger scale (58, 67). Current research is still focused on methods to enhance coral recruitment, maintain the natural balance of herbivores (i.e., 108), and maintain coral nurseries, to rescue and rehabilitate fragments for use as transplants in degraded areas. However, the recognition of different needs expressed by various reef areas, the need for using different coral species in distant reefs, and the emergence of novel approaches challenge much of the accepted ideas and protocols for reef restoration. Moreover, the communication between the scientific community, the public aquaria, and the private sector of marine ornamental trade is inadequate. This is responsible for the under-exploitation of hundreds coral species already available under ex situ conditions as an accessible and nondestructive source material for reef restoration (31, 62).

Whereas ecological parameters may vary considerably from one site to another (11) and restoration should consider these variations (1, 8, 9), ecological constraints on restoration require the establishment of ubiquitous guiding rationale and protocols. Two such emerging principles and needs (landscape restoration and restoration genetics) are discussed below.

A central goal of restoration ecology is to anticipate the consequences of restoration measures. In fact, we have currently little ability to foresee the path that sites will follow when restored in alternative ways (one good example for two restored sites in Florida was discussed before; 68). Moreover, there is no guarantee that specific targets will be met (109). One reason is the naive approach that degradation and restoration of a coral reef area are two simple conflicting processes, along a straight line and not multifaceted, complex phenomena. Restoration by transplantation closely resembles natural succession, in which organisms colonize primarily from external source population, and recruitment may occur as a “nucleation” phenomenon. During this process, scattered colonization foci spread and coalesce (17, 110). The landscape restoration concept may use the impacts of restoration of denuded reef areas by nucleation and by the creation of mosaic recruitment, where edg-to-interior ratio of each restored focus has its own impact on restoration success (literature for terrestrial examples is cited in 17, 110). The influence of spatial arrangement of landscape elements on restoration success is therefore of special future interest, as it may directly influence the fate of a restoration project (111, 112). For example, the use of small “restored foci” designed to grow and spread may boost rates of natural recruitment as well as natural introduction of reef fish and mobile invertebrates.

Fragmentation of populations as well as restoration practices that use a limited number of donor colonies can reduce gene flow (113). Because gene flow is such a powerful evolutionary force, decreased gene flow could drastically alter a species genetic architecture and disrupt local adaptation, allowing higher rates of genetic drift or selection, depending on the population size. Gene flow among restored coral reef populations and genetically divergent restored population should become a major concern in coral reef restoration measures (1). Moreover, studies on gene pools in restored terrestrial habitats have already yielded interesting results. Much of the literature on ecological restoration pertains to the choice of species to be used (114). It is becoming increasingly apparent, however, that intraspecific genetic differentiation in relation to site ecology is also important. Many restoration sites, such as mine spoils, pose novel and severe challenges to vegetation, and only specific ecoregions may be able to survive. Then again, plants introduced from distant source populations may not be adapted to the local climatic, edaphic, or biotic environment (literature cited in 114, 115). Genetic oriented restoration strategies, adapted to coral reef biology, should therefore be developed in parallel to other concepts and protocols for reef restoration. It is therefore important that restoration principles frequently used in terrestrial habitats (such as for forest restoration; 2, 18) will be applied to coral reef rehabilitation measures and techniques.

It is now accepted that “the long-term future of conservation biology is restoration ecology” (17), and considerable effort is being invested in defining the conceptual bases for coral reef restoration (1, 2, 8–10, 18, 20, 65, 74; and literature therein). Pursuing the success of restoration activities, ecological criteria that evaluate the status of coral reefs (116) and biochemical/molecular markers that evaluate stress (43, 97) should be developed and applied. Additionally, deciding which habitat in the reef should be restored may be as important as the extent of its restoration (112) and should be defined and conceptually backed by experimental designs. Many restoration efforts are also limited on both financial grounds and available protocols and therefore should be highly focused.

The past decade of activities in coral reef restoration has been the first step in changing this scientific discipline from its previously descriptive level to practices fueled by ecological concepts and modern biological tools. Ecological restoration of the coral reef purports more than just simply replacing dead coral colonies or replanting denuded reef areas. Restoration of coral reef areas ranges from recreation of the original ecosystem (probably a naïve belief) to the opposite extreme, reconstruction of entirely alternative, new ecosystems (117). With more scientific background and concepts integrated with other biological disciplines (genetics, molecular biology, theoretical ecology, etc.), coral reef restoration measures will achieve far greater success than our present meager knowledge can.
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Literature Cited


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