



Reef Restoration

Concepts & Guidelines:

Making sensible management choices in the face of uncertainty.



by Alasdair Edwards and Edgardo Gomez

www.gefcoral.org



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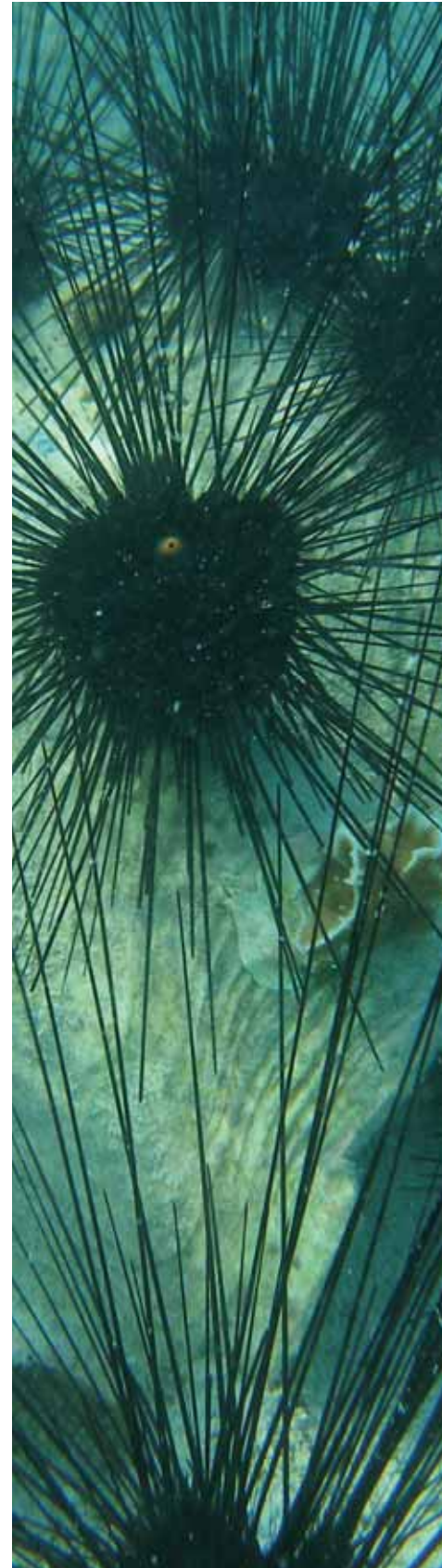
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Two important caveats:

“Although restoration can enhance conservation efforts, restoration is always a poor second to the preservation of original habitats.

The use of ex situ ‘restoration’ (mitigation) as an equal replacement for habitat and population destruction or degradation (‘take’) is at best often unsupported by hard evidence, and is at worst an irresponsible degradative force in its own right.”

Young, T.P. (2000). Restoration ecology and conservation biology. *Biological Conservation*, 92: 73-83.

How to use these guidelines



These guidelines contain simple advice on coral reef restoration for coastal managers, decision makers, technical advisers and others who may be involved in community-based reef restoration efforts. Those attempting reef restoration need to be aware that there is still much uncertainty in the science underpinning restoration, not least due to the great complexity of reef ecosystems.

Much scientific research is currently underway around the world to address these gaps in our knowledge and improve our understanding of what reef restoration interventions can and cannot achieve. Despite these uncertainties there are many useful lessons which can be learned from previous work both in terms of what works and what doesn't work.

The following guidelines seek to summarise these lessons in a succinct form for practitioners so that they may have a clearer idea of what can and cannot be achieved by reef restoration and can set goals and expectations accordingly.

Much of the available literature details the plethora of methods that can or have been applied in active restoration projects, but does not consider their use in a management context or offer advice on technical know-how needed, chance of success, risks or likely costs. There is also a reluctance to disseminate information on restoration failures, analyse the causes, and pass on the lessons learnt. Often advice on what doesn't work can be almost as valuable as advice on what does work and can save people repeating past mistakes. Sometimes this may be all the advice that can be given. Despite all the uncertainties, we stick our necks out and attempt to offer broad brush advice where we can so that managers can at least have some idea of where their actions may lead them.

These guidelines are not intended to provide detailed practical advice on how to carry out reef restoration but we are intending to prepare a companion *Reef Restoration Manual* which will cover these aspects, build on the various manuals already available (e.g., Clark, 2002; Harriott and Fisk, 1995; Heeger and Sotto, 2000; Job *et al.*, 2003; Miller *et al.*, 1993; Omori and Fujiwara, 2004 – see Bibliography for details), and synthesise the results of several major international projects currently carrying out research on reef restoration.

Meanwhile, for more detailed information practitioners are referred to both the manuals above and the 363 page *Coral Reef Restoration Handbook* edited by William F. Precht and published in 2006 by CRC Press (ISBN 0-8493-2073-9). This is the first book devoted to the science of coral reef restoration and its 20 chapters by many of the leaders in the

field summarise much of the scientific literature available to date. The book is designed to guide scientists and resource managers in the decision-making process from initial assessment through conceptual restoration design, implementation and monitoring, and is an essential resource for those wishing to delve deeper into the scientific, legal and socioeconomic background to reef restoration. About one third of chapters have a strong focus on the US perspective, but the broader international issues are also covered.

For a general overview of ecological restoration the practitioner is referred to *The SER International Primer on Ecological Restoration* (version 2: October 2004) which is available on the website of the Society for Ecological Restoration International at www.ser.org/content/ecological_restoration_primer.asp. This gives a useful and succinct overview of the conceptual basis of restoration with a strong practical focus.

These guidelines are for dipping into rather than reading from cover to cover. Sections 1, 2, 3.1 and 4 provide important advice to coastal managers and decision makers who are considering coral reef restoration, whereas sections 3.2 to 3.9 and section 5 are aimed more at technical advisers (that is, professional marine biologists who have a good background in the subject, but may not have specialised in reef restoration ecology). Any reef restoration project needs at least one such person to guide it!

For those needing just a quick overview, key points in the text are summarised as "Message Boards" and "Good Practice Checklists".

Key Messages



Ecological restoration is the process of *assisting* the recovery of an ecosystem that has been degraded, damaged, or destroyed.



Coral reef restoration is in its infancy. We cannot create fully functional reefs.



Although restoration can enhance conservation efforts, restoration is always a poor second to the preservation of original habitats.



Coral reefs that are relatively unstressed by anthropogenic impacts can often recover naturally from disturbances without human intervention.



Active coral reef restoration has been carried out with some success at scales of up to a few hectares only.



Restoration includes passive or indirect management measures to remove impediments to natural recovery, as well as active or direct interventions such as transplantation.



Active restoration is not a magic bullet. Improved management of reef areas is the key.



The aims of reef restoration are likely to be dictated by economic, legal, social and political constraints as well as ecological realities. However, ignoring the latter means a high risk of failure.



The goals of restoration projects should be formulated at the outset as precisely as possible and potential ways of achieving them considered within a wider coastal management planning context.



Targets or measurable indicators should be set that allow both the progress towards restoration goals to be assessed over time and adaptive management of the restoration project.



Monitoring of progress towards targets should be undertaken at regular intervals over several years.



Successes, failures and lessons learnt should be widely disseminated so that others can benefit from your experiences.



Major physical restoration of reefs is for experts only. Seek expert civil engineering advice.



Some physical restoration may be a prerequisite for any chance of successful biological restoration.



There are at least 300,000 km² of coral reefs in the world. Lack of hard substrate is not a critical issue. Management of degradation of natural reefs is the critical issue.



Use of artificial reefs in restoration needs to be considered carefully and critically in terms of need, ecological impact, cost-effectiveness and aesthetics.



Consider restoration not as a one-off event but as an ongoing process over a time-scale of years which is likely to need adaptive management.



Major physical restoration of reefs costs in the order of US\$100,000 –1,000,000's per hectare.



Low-cost transplantation appears to cost about US\$2000 –13,000 per hectare. With more ambitious goals this rises to about \$40,000 per hectare.



For comparison, a global ball-park estimate of the average total *annual* value of coral reef goods and services is US\$6,075 per hectare.

1. Background

The purpose of this section is to provide a management context to reef restoration. We assume some familiarity with what coral reefs are. A key point we make is that reef restoration should be treated as just one option within an integrated coastal management (ICM) planning agenda for a stretch of coast. Too often, enthusiastic proponents of active restoration omit to consider the wider context and factors outside their control which may jeopardise their efforts.

1.1 Why are coral reefs important?

As well as preventing coastal erosion, coral reefs provide food and livelihoods for hundreds of millions of coastal people in over 100 countries via the harvestable marine resources that they generate, and through tourists attracted by their beauty, biodiversity and the white sand beaches that they support and protect. At least half a billion people around the world are thought to be partially or wholly reliant on coral reef resources for their livelihoods. These livelihoods include fishing, gleaning, mariculture, the marine aquarium trade, and a wide range of employment and commercial opportunities associated with tourism. They are also a promising source of novel pharmaceuticals treating diseases such as cancer and AIDS. In terms of biodiversity, about 100,000 described species, representing some 94% of the planet's phyla, have been recorded on coral reefs and some scientists estimate that there could be five or more times that number still undescribed.

On a global scale, the value of the total economic goods and services provided by coral reefs have been estimated at roughly US\$375 billion per year with most of this coming from recreation, sea defence services and food production. This equates to an average value of around US\$6,075 per hectare of coral reef per year. In the Philippines, which has an estimated 27,000 km² of coral reef (though with only about 5% in excellent condition), the reefs are thought to contribute at least US\$1.35 billion per year to the national economy from the combined values of fisheries, tourism and coastal protection.

Degradation of reefs means the loss of these economic goods and services, and the loss of food security and employment for coastal peoples, many of them in developing countries and many of them living in poverty.

1.2 What are the threats to coral reefs?

The *Status of Coral Reefs of the World: 2004* report estimates that 20% of the world's coral reefs have been effectively destroyed and show no immediate prospects of recovery, that 24% of the world's reefs are under imminent risk of collapse through human pressures, and that a further 26% are under a longer term threat of collapse.

Until about 20 years ago it seemed that the biggest threats to coral reefs were from chronic human disturbances such as increased sedimentation resulting from land-use change and poor watershed management, sewage discharges, nutrient loading and eutrophication from changing agricultural practices, coral mining, and overfishing (Figure 1). However, in recent years global climate change – with on the one hand, mass bleaching events and subsequent coral mortality, and on the other ocean acidification – has emerged as probably the biggest threat to the survival of coral reefs. Undoubtedly, the ability of reefs to recover from anomalous warming events, tropical storms and other acute disturbances is profoundly affected by the level of chronic anthropogenic disturbance. Where reefs are healthy and unstressed, they can often recover quickly (sometimes in as little as 5-10 years). Such reefs can be described as “resilient” in that they “bounce back” to something close to their pre-disturbance state following an impact. Whereas reefs that are already stressed by human activities, often show poor ability to recover (i.e. they lack resilience). Natural disturbances have impacted coral reefs for millennia prior to human induced impacts and reefs recovered naturally from these impacts. Even now, healthy reefs can and do recover from major perturbations. It is estimated that approximately 40% of the 16% of the world's reefs that were seriously damaged by the unusually warm seawater during the 1998 El Niño Southern Oscillation (ENSO) event are either recovering well or have recovered.

In the context of restoration it is important to distinguish between acute and chronic disturbances. Restoration interventions are unlikely to succeed on reefs that are chronically stressed. Management measures must be undertaken first to ameliorate or remove the chronic anthropogenic stressors (e.g., sediment run-off, sewage, overfishing). On the other hand, there is little that managers can do in the face of the large-scale “natural” drivers of degradation such as climate change related mass-bleaching, storms, tsunamis, and disease outbreaks. However, these stochastic factors should not be ignored during restoration and should be taken into account during the design of restoration projects with efforts being made to minimise the risks posed by such events.

The economic case for better management is strong. For example, in Indonesia it is estimated that the net benefit to individuals derived from blast fishing is US\$15,000 per km², whereas the quantifiable net loss to society from this activity is US\$98,000-761,000 per km². Examples from Indonesia for this and other threats are shown in Table 1. Using mid-range figures one finds that on average net losses to society are nearly ten times the net benefits to individuals.

Table 1. Total net benefits and quantifiable losses due to threats to coral reefs in Indonesia (present value; 10% discount rate; 25 year time-span). Adapted from Cesar (2000).

Threat	Total net benefits to individuals	Total net losses to society
Poison fishing	\$33,000 per km ²	\$43,000-476,000 per km ²
Blast fishing	\$15,000 per km ²	\$98,000-761,000 per km ²
Coral mining	\$121,000 per km ²	\$176,000-903,000 per km ²
Sedimentation from logging	\$98,000 per km ²	\$273,000 per km ²
Overfishing	\$39,000 per km ²	\$109,000 per km ²

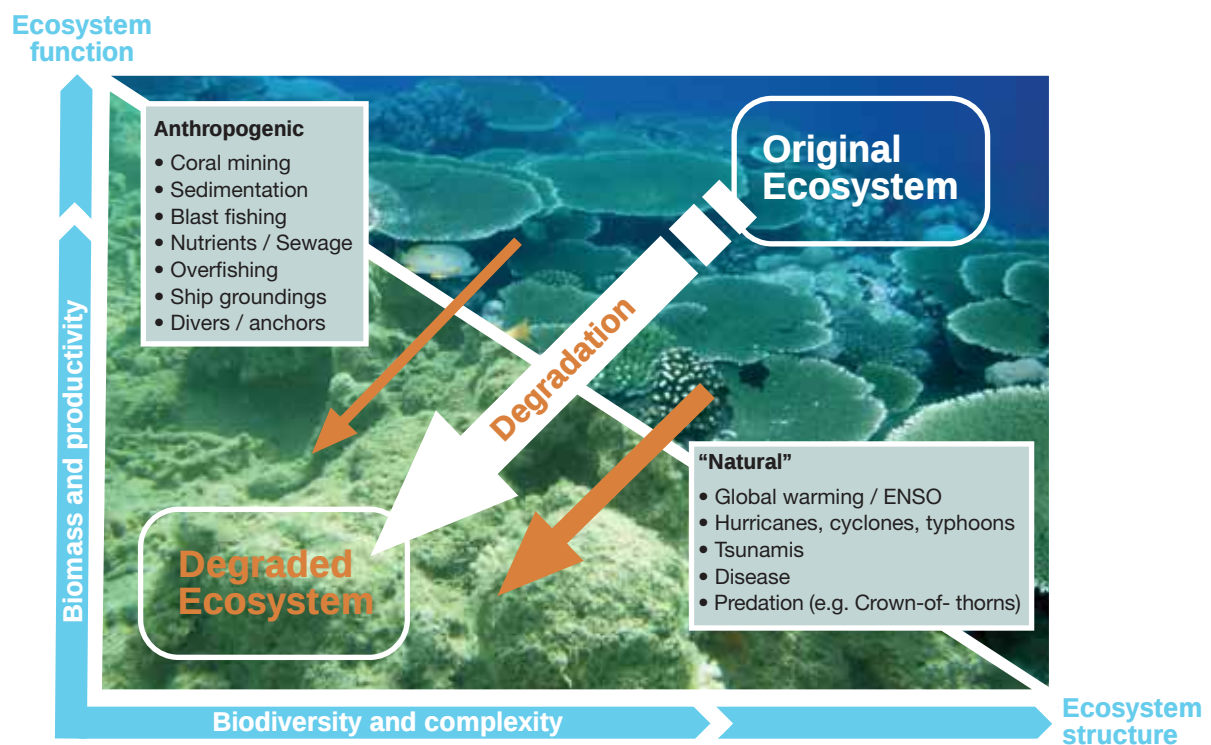


Figure 1. Drivers of reef ecosystem degradation. Degradation will tend to reduce biodiversity and complexity on the one hand and biomass and productivity on the other, with the knock-on effect of reducing the flow of economic benefits from the reef in terms of both goods (e.g., fish) and services such as sea defence. Direct anthropogenic and “natural” impacts are separated with the thickness of the orange arrows indicating the relative scale of the impacts. Although direct anthropogenic impacts may act at smaller scales they can build up cumulatively over decades to degrade reefs at scales of 100s to 1000s of km². Mankind’s activities have been implicated in several of the “natural” drivers of degradation.

The scale on which the various drivers of coral reef degradation act is important in terms of what restoration might achieve (see section 1.5). Large scale disturbances such as ENSO-induced mass coral mortality, tropical cyclones (hurricanes, typhoons), and Crown-of-thorns starfish (*Acanthaster planci*) outbreaks can cause damage at scales which are several orders of magnitude larger than those at which restoration can be attempted. However, the areas that are typically damaged as a result of ship groundings, discrete sewage discharges, blast fishing, SCUBA divers or boat anchors are of a similar size to those at which restoration has been tried with some success.

In summary, if reefs are stressed by anthropogenic activities (e.g., overfishing, sediment and nutrient run-off), they are less likely to be able to recover from large scale disturbances. Active restoration is highly unlikely to be able to assist such recovery due to the huge scale-mismatch, but good coastal management (referred to by some as “passive restoration”) may give them a fighting chance. If mankind attempts to manage those threats to reefs that are potentially manageable, then restoration at small scales can assist management.

1.3 What are the aims of restoration?

Before thinking about the aims of specific reef restoration projects, it is worthwhile considering what is meant by *ecological restoration*. The Society for Ecological Restoration International offers the following definition:

“Ecological restoration is the process of *assisting* the recovery of an ecosystem that has been degraded, damaged, or destroyed.”

The italics are ours and emphasise that restoration interventions are designed to assist natural recovery processes. If these processes are severely impaired, other management measures are likely to be needed before restoration interventions can have any chance of success. Our “assistance” to natural recovery may be either in the form of passive or indirect measures, or in the form of active or direct interventions. The former generally involve improving the management of anthropogenic activities that are impeding natural recovery processes: the latter generally involve active physical restoration and/or biological restoration interventions (e.g., transplantation of corals and other biota to degraded areas).

Coral reef restoration is still in its infancy and it is unwise to overstate what restoration can achieve. If decision-makers are led to believe that functioning reefs can be created by

restoration interventions (e.g., transplanting reef organisms from a sacrificial site, wanted for development, to an area outside the impact zone), they will act accordingly. It should be emphasised to decision-makers that we are a long way from being able to recreate fully functional reef ecosystems (and possibly will never be able to!) and thus decisions which rely on compensatory mitigation are effectively promoting net reef loss.

It is perhaps useful to define what we mean by restoration, rehabilitation and remediation.

- Restoration: the act of bringing a degraded ecosystem back into, as nearly as possible, its original condition.
- Rehabilitation: the act of partially or, more rarely, fully replacing structural or functional characteristics of an ecosystem that have been diminished or lost, or the substitution of alternative qualities or characteristics than those originally present with the proviso that they have more social, economic or ecological value than existed in the disturbed or degraded state.
- Remediation: the act or process of remedying or repairing damage to an ecosystem.

With reefs we are usually aiming for restoration but may be pleased if we can just achieve some form of rehabilitation.

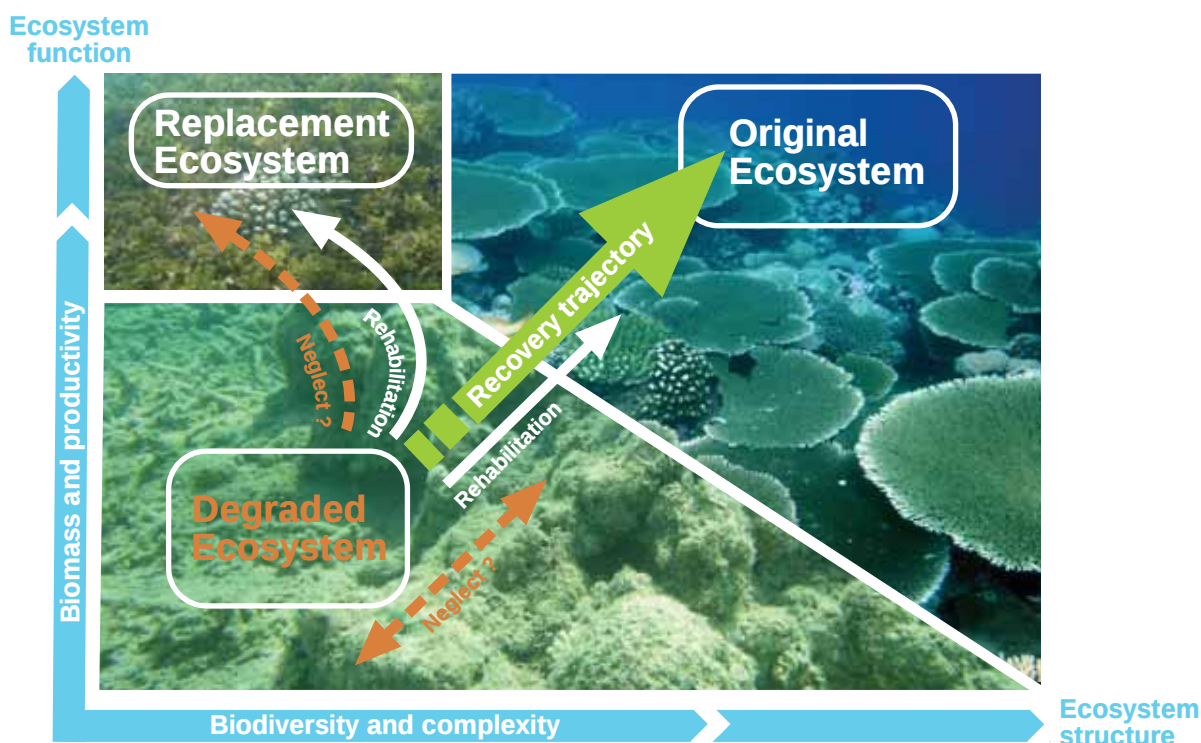


Figure 2. Possible paths of recovery or state change for a degraded ecosystem with and without active restoration interventions. See text below for an explanation. (Diagram based on Fig. 5.2 in Bradshaw, A.D. (1987). The reclamation of derelict land and the ecology of ecosystems. In: Jordan III, W.R., Gilpin, M.E. and Aber, J.D.(eds). *Restoration Ecology: A Synthetic Approach to Ecological Research*. Cambridge University Press.)

The primary aim of restoration is to improve the degraded reef in terms ecosystem structure and function. Attributes to be considered might be biodiversity and complexity on the one hand and biomass and productivity on the other (Figure 2). In a healthy reef system which has not been physically damaged, an impacted area might be expected to recover naturally to its pre-disturbance state along a successional trajectory (thick green arrow). In such a case, benign “neglect” (letting nature take its course) and patience may achieve restoration. However, if degradation is sufficiently severe or spatially extensive, or the reef system is subject to additional chronic human-induced stresses (e.g., overfishing, nutrient loading, sedimentation) then “neglect” (doing nothing) may see further decline, or possibly a switch to an alternate (perhaps undesirable for local resource users) stable state (e.g., a reef dominated by macroalgae). In such cases, active restoration, if necessary, in combination with management actions to reduce anthropogenic stress, is likely to be needed if the reef is to have any chance of recovery to a desirable state. Even with active restoration measures, recovery may progress to some state different from the original ecosystem. This may be a broadly similar state (e.g., coral dominated but with different dominant species) in which case “rehabilitation” (improvement of the ecosystem’s function and structure) has been achieved, but not full restoration. Alternatively, the active restoration may disappoint and lead to a rather different ecosystem state (“replacement” system), the perceived desirability of which will depend on the goals of the restoration intervention. Deciding whether active measures are needed and what these should be is perhaps the hardest issue to resolve. We will try to give some guidance as to how to approach this issue in the following few sections.

Above we have concentrated on the biological aims of reef restoration and possible outcomes. However, in the real world, the aims of restoration are likely to be dictated by economic, legal, social and political constraints. These constraints may drive the ecological aims of a project and at worst conflict with ecological best-practice advice. Projects which ignore the ecological realities are likely to be at high risk of failure, have poor cost-effectiveness and may do more harm than good.

Not all reef restoration projects fit into the scheme above. In the tourism sector, there is often a desire to promote easy access to patches of coral habitat so that anyone at a resort can see the corals and brightly coloured reef fish for themselves in a shallow, safe and sheltered environment. To do this, patches of reef may be (re-) created in a sandy lagoon either on natural or artificial substrates. Usually coral transplantation and other “restoration” techniques are involved. Such projects may also occur in marine park areas and can clearly have a valuable educational and public awareness role. These projects may not be reef restoration in the strict sense, but rather habitat substitution or habitat creation, nonetheless they are often considered

as restoration activities and are subject to the same ecological constraints. Here the aims are simple, to create some easily accessible, aesthetically pleasing, (and hopefully self-sustaining) coral reef habitat for tourists or park visitors who are not accomplished snorkellers or SCUBA divers.

A second type of restoration project that does not really fit in the scheme, is where an area of reef is being destroyed by a development (e.g., land reclamation, a power plant, a port development) and living coral and other reef organisms – which will die if left *in situ* – are transplanted to an area of reef out of harm’s way. The management decision has already been taken that there will be net habitat loss; the main aim of the mitigation project is to save as many of the sessile organisms as possible from the impact site. As a by-product, the receiving area is likely to benefit if the project is well planned and executed. Again transplantation and other reef restoration techniques are involved, and such projects can usefully be considered in a restoration context even if the prime driver is mitigation and not a perceived need for restoration.

1.3.1 Setting goals and success criteria for restoration projects

Before any restoration project is undertaken the aims of the restoration work should be carefully considered and described as precisely as possible. Surprisingly, this is seldom done, with the result that aims are often poorly defined, or not thought-out and may be ecologically unrealistic, such that the project is doomed from the start. Without aims, it is also not possible to evaluate success and it is difficult to learn lessons. Once the aims are agreed and clear to all stakeholders, then a set of objectively verifiable and measurable indicators (or targets) needs to be established that will allow the success (or failure!) of the restoration project to be evaluated. The indicators should match the aims so that, if the targets are attained, then the aims will have been successfully achieved. The targets need to be realistic and fairly easily assessed, and the timeframe in which they are to be achieved should be defined. An explicit timeframe with milestones allows the progress of the restoration to be monitored over time and corrective actions (adaptive management) to be undertaken if appropriate, such as when indicators fail to perform within the predicted timeframe. Indicators may be endpoints such as percentage live coral cover or evidence of restoration of key ecosystem processes such as coral recruitment or fish grazing.

Deciding on criteria which demonstrate successful restoration and picking indicators and target values for these is not easy. The expected timescale of recovery may be unclear and the “reference ecosystem state” to aim for may not be obvious unless the degraded area is small and a comparable reef that is in good condition exists nearby and can serve as a “yardstick”. Historical data or data from quite distant sites of similar aspect, depth, exposure, etc. may

have to be sought to provide clues as to what state you are trying to restore the reef. In the face of global climate change the “reference ecosystem state” is also likely to be in flux, so a pragmatic approach is needed. Given this uncertainty, you may be wise to set aims and indicators, which will show whether recovery is on the right trajectory in terms of direction of change, but are not very explicit about the amount of change expected.

For active restoration, measuring success can be made easier if you set up a number of “control” areas at your degraded site where no active interventions are carried out (Figure 3). You can then compare what happens over time in areas where you have actively assisted natural recovery processes, and what happens in adjacent areas where you

have just let natural recovery (if any) take its course. The costs are what you’ve paid out; the benefits are any improvements of indicators (e.g., % live coral cover, numbers of fish grazers, rates of coral recruitment) in restored areas over and above those in the control areas. Given the increasing amount of reef degradation, the high costs of active restoration, and the potential benefits in terms of learning lessons from projects that include an element of experimental design, such an approach is strongly recommended wherever possible. The time-span over which changes are evaluated should be at least several years to match the expected time-course of recovery. Studies show that natural recovery takes at least 5-10 years. Long-term (5-10 years +) restoration is the goal, not short-term, often ephemeral, improvements in indicators.

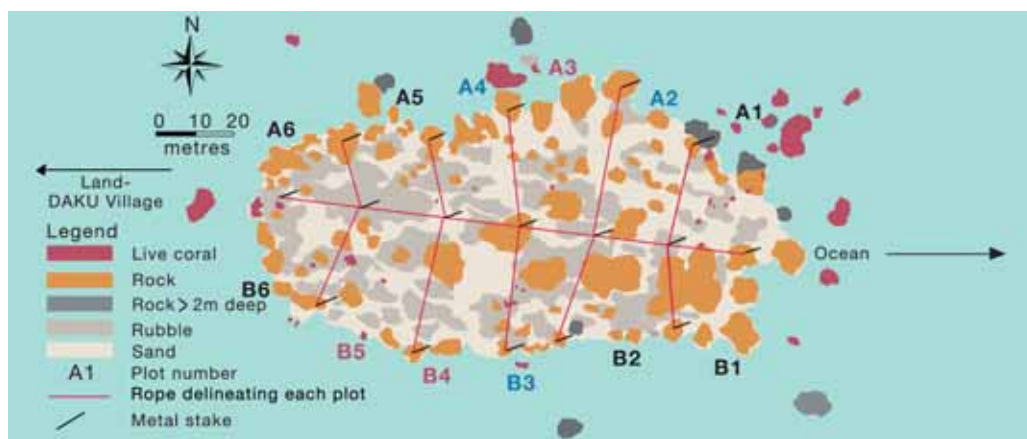


Figure 3. Map of a restoration site in Fiji showing how the degraded reef area was divided into 12 plots, three of which were selected for restoration (A3, B4 and B5) and three of which were selected for monitoring as controls (A2, A4 and B3). (From: Job, S., Bowden-Kerby, A., Fisk, D., Khan, Z and Nainoca, F. (2006). Progress report on restoration work and monitoring. Moturiki Island, Fiji. Technical Report. Coral Reef Initiative for the South Pacific.)

Message Board

-  Ecological restoration is the process of *assisting* the recovery of an ecosystem that has been degraded, damaged, or destroyed.
-  Restoration includes passive or indirect management measures to remove impediments to natural recovery, as well as active or direct interventions such as transplantation.
-  The aims of reef restoration are likely to be dictated by economic, legal, social and political constraints as well as ecological realities. However, ignoring the latter means a high risk of failure.
-  The goals of restoration projects should be formulated at the outset as precisely as possible and potential ways of achieving them considered within an integrated coastal management and planning context.
-  Targets or measurable indicators should be set that allow both the progress towards goals to be assessed over time and adaptive management of the restoration project.
-  Monitoring of progress towards targets should be undertaken at regular intervals over several years.
-  Successes and failures, and lessons learnt should be disseminated so others can benefit from your experiences. Little is known; every little bit of knowledge helps.

1.4 Why carry out reef restoration?

Coral reef systems have evolved to cope with natural disturbances and indeed these disturbances may be important in structuring reef communities. On the whole (if healthy and unstressed by man's activities) they tend to recover well from such acute disturbances but full recovery of areas may take decades – a short time on an ecological or evolutionary time scale, but a long time on our time scale.

Anthropogenic impacts are often chronic (long-term) and even when acute, like ship-groundings, can cause physical damage that compromises natural recovery processes. Where there are chronic human impacts, to allow any chance of recovery, passive or indirect restoration measures such as sewage treatment, watershed management, fisheries enforcement, etc. may be needed to allow natural recovery processes to operate, followed by active or direct restoration interventions such as coral transplantation or substrate stabilisation to assist. Where recovery is impeded because of physical damage, then active physical restoration may be a pre-requisite for recovery. Thus it is mainly where humans impact reefs that restoration (passive or active) is needed. The main socio-economic reason to restore, is to bring back the flow of goods and services provided by healthy reefs (see section 1.1).

Again, decisions on reef restoration are often likely to be driven by the local economic, legal, social and political environment. Thus one finds that much reef restoration has been associated with repairing injury to reefs caused by ship groundings. In such cases, insurance that covers shipping companies (referred to as “responsible parties” in legal jargon) from liability provides a source of funding. In countries such as the USA there is also a legal framework to support compensatory restoration to replace the lost reef resources and services. The scale of damage (in the order of 100–1000 m² per grounding incident) matches well onto the scale of what can be attempted by current restoration techniques. As a result, ship-grounding restoration projects in areas such as the Florida Keys National Marine Sanctuary

have provided many useful lessons. One important lesson from studies of ship-groundings is that even such localised anthropogenic impacts may not recover to a pre-disturbance state but may “flip” into algal dominated or hard-ground communities quite unlike the pre-impacted reef.

A very encouraging development in reef restoration is the increasing interest by local communities in developing countries in improving the quality and productivity of reef resources which have been degraded by blast fishing, long-term overfishing, sedimentation, nutrient loading or other impacts. In such cases the communities tend to use a combination of management measures (e.g. declaration of marine protected areas or no-take zones) and localised restoration to attempt to restore the flow of marine resources (especially fish) on which the community used to subsist. In such cases, active restoration is just one tool in the coastal manager's armoury and should be seen as just one component of a larger integrated management plan, not as a “magic bullet”. Such activities may also have tourism related spin-offs (see Heeger and Sotto, 2000).

Two types of project which involve reef restoration techniques and may result in restoration of areas of reef, are the creation of easily accessible reef habitat patches for tourism and education, and the saving by translocation of reef organisms which will otherwise be killed due to a development. The motives are clear in both cases.

Another argument for restoration relates to the risk of coral dominated systems being “flipped” into alternative stable states by disturbances (see Box 1). Reef restoration is very expensive, more so than seagrass or mangrove restoration. Trying to restore habitat patches which have flipped into an alternative stable state will be even more costly and perhaps prohibitively so. However, a combination of management measures (to reduce the chronic anthropogenic stressors) and active restoration on a degraded reef system, may improve resilience and reduce the risk of the ecosystem sliding into an alternative state.

Box 1: Jamaican case-history

The dangers posed by a combination of chronic anthropogenic impacts and natural disturbances to reefs are exemplified by what has happened in Jamaica over the last several decades. A somewhat simplified account follows. In the 1970s the reefs of Jamaica were coral dominated ecosystems with around 45–75% live coral cover depending on depth and location. Fishing was already intense with clear evidence of overfishing on the reefs since the 1960s. On the more accessible reefs it was estimated that fish biomass had been reduced by up to 80%. Thus large predators such as sharks and large snappers, jacks, triggerfish and groupers had been virtually fished out followed by large herbivores such as big parrotfishes. As fishing pressure continued down the food web, other herbivorous fish were reduced in abundance and size but the ecosystem had some redundancy in the form of grazing sea urchins (*Diadema antillarum*) and these took over much of the grazing service

provided by fish. Fishing reduced the abundance of both fish which preyed upon the urchin (e.g., triggerfishes) and herbivorous fishes which competed with them for algal resources. As a result the *Diadema* urchin populations boomed.

Grazing of algae is important because if macroalgae (seaweeds) become dominant, they can occupy most of the available space on the reef and prevent settlement of corals and other invertebrates. Normally there is a balance, with macroalgal biomass held in check by grazers, which by their feeding continuously create small patches of bare substrate where invertebrates can settle. However, in the absence of sufficient grazing, macroalgae (which when well-grown may be unpalatable to most herbivores) can take over. When this happens you can get a dramatic shift to an alternative, macroalgal dominated, ecosystem state.

Coupled with overfishing were land-use changes, which probably led to increased nutrients and sedimentation on some inshore reefs, and also increased prevalence of coral disease. Then in 1980 Hurricane Allen struck. This major disturbance caused a major loss of shallow water coral cover and a short-lived algal bloom. However, the reefs appeared resilient with the *Diadema* urchins able to control the algal growth such that there was substantial coral recruitment and coral cover began to recover slowly. Then three years later in 1983 there was a mass die-off of the *Diadema* urchins from disease with densities being reduced by 99%. At this point the last bastion of herbivorous control was breached and firstly shallow reefs and then deeper reefs were taken over by macroalgae. By the late 1980s the reefs had largely shifted to an alternative stable state with 70-90% algal cover.

From a restoration point of view, this alternative state is probably an order of magnitude harder to restore than the various degraded versions of the coral dominated system that persisted before the *Diadema* die-off. To regain the original

state, not only is there a need for management measures (passive restoration) to shift conditions from C2 towards C1 in Figure 4 (fisheries management and/or urchin culture to restore herbivory), but there is likely to be a need for some large active restoration disturbance to remove macroalgae and add corals before the system is likely to have any chance of flipping back.

The lessons learnt are that chronic anthropogenic impacts over decades cumulatively chip away at the resilience of the ecosystem with little sign that the system is at risk. After Hurricane Allen it still appeared resilient and showed signs of bouncing back. Then, eventually, one disturbance too far becomes the straw that breaks the camel's back and the system collapses into an alternate state.

With global climate change, the disturbances appear to be coming thick and fast and unless we can manage those reefs under anthropogenic stress better, it seems increasingly likely that we shall see reefs in many locations toppling like dominoes into alternate states.

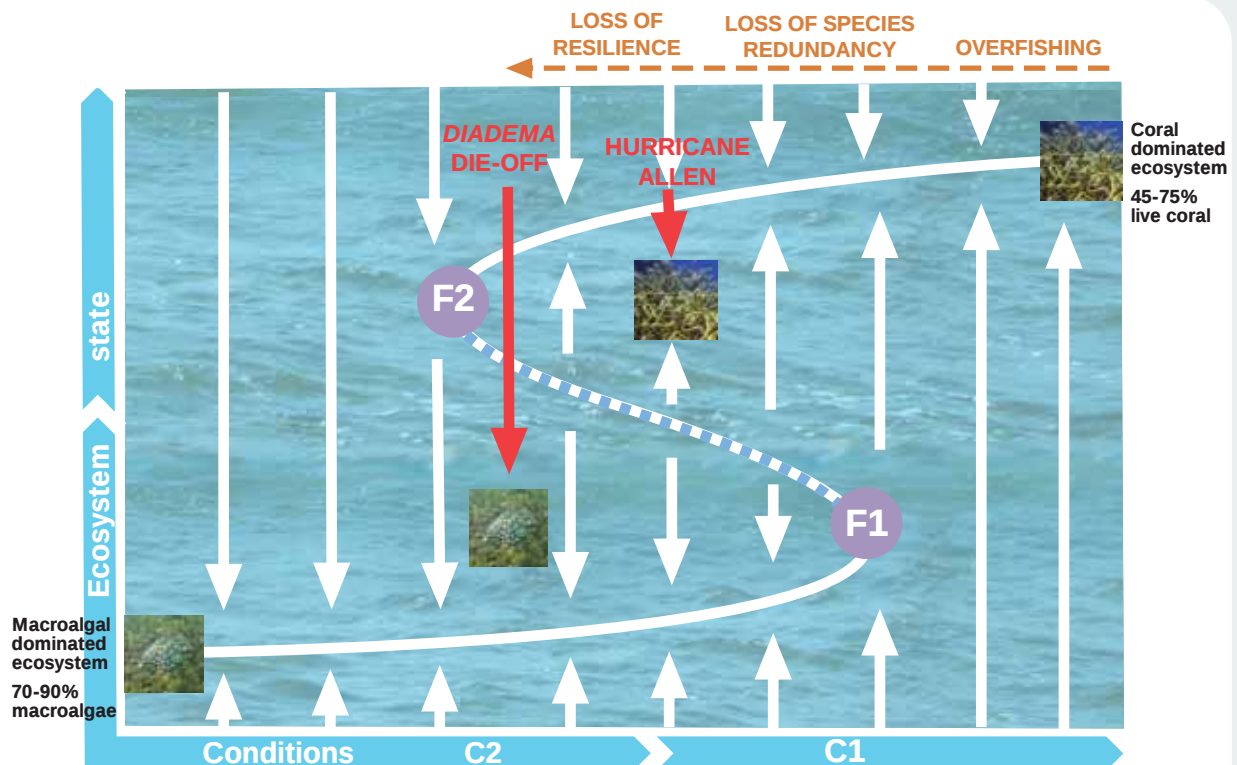


Figure 4. Shifting to an alternative state. The solid white curves represent “attractors” for two different stable states, one coral dominated (top right) and one macroalgal dominated (bottom left). When the ecosystem state is near each attractor various feedback processes will tend to maintain stability, pulling it back towards the attractor. As conditions deteriorate for the coral dominated ecosystem from C1 towards C2, its state drifts towards the bifurcation point F2 and its resilience (difficulty with which disturbances can move it into an unstable or alternate stable state) decreases. The dashed white curve between F2 and F1 is a “repeller” where the ecosystem state is unstable and may flip into either stable state.

As conditions change there may be little obvious change in ecosystem state but the system may become less and less able to respond to large disturbances. In the case of Jamaica, the reefs appeared to be recovering from Hurricane Allen and moving back towards the attractor, but then mass die-off of *Diadema* flipped the system to an alternate stable state. To restore the system, not only is management needed to move conditions back towards C1 but some major disturbance or active restoration intervention will be needed to overcome the resilience of the macroalgal state attractor.

(Hughes, T.P. (1994). Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. *Science*, 265: 1547-1551; Suding, K.N., Gross, K.L. and Houseman, G.R. (2004). Alternative states and positive feedbacks in restoration ecology. *Trends in Ecology and Evolution*, 19 (1): 46-53.).

1.5 What can reef restoration interventions realistically achieve?

As should be clear from earlier sections, coral reef restoration is still in its infancy. The system we are trying to restore is very complex and it is not well-enough understood for us to be confident of the outcomes of restoration attempts. We are still learning what works and what doesn't work in a largely empirical way.

As we emphasised earlier, this means that the limited potential for restoration should not be used as justification by decision-makers for approving projects which will degrade healthy reefs.

Reef restoration should never be oversold and its limitations clearly understood (Richmond, 2005). It is humbling and somewhat depressing to compare the relative scale of restoration attempts to date and the scale of reef

Clearly there is a mismatch (of several orders of magnitude) between the scale at which reef restoration can currently be attempted and the scale at which major impacts can degrade reefs. In the case of large scale, natural (but perhaps exacerbated by man) acute disturbances, this is not necessarily a problem as healthy reefs are resilient and should largely recover of their own accord if not otherwise stressed.

One key area for research, is to find out whether localised restoration at scales of hectares can cascade benefits to down-current areas at scales of tens of hectares or square kilometres. Another, is to find out whether small community-based reef restoration projects can produce viable and sustainable functioning reef areas and whether there is a minimum size needed for sustainability. This relates to the wider issue of the minimum size needed for marine protected areas to be effective.

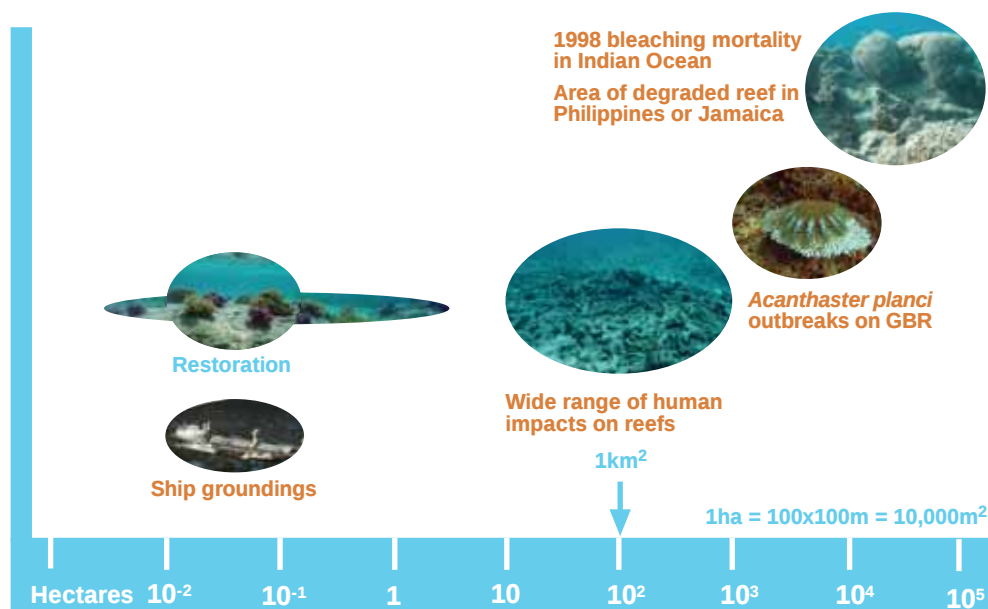


Figure 5. The scale of degradation versus that of restoration. A comparison of the approximate scales of degradation resulting from various causes with the scale at which reef restoration has been carried out with a degree of success. The extent of the “wide range of human impacts” is perhaps conservative and these can build up cumulatively to encompass huge areas as seen in Philippines and Jamaica.

degradation (Figure 5). Restoration has been carried out with some success on scales of tens of square metres to several hectares. However, a wide range of local human impacts on reefs act at scales of several square kilometres and cumulative human impact over decades has led to estimates of 100 -1000s of square kilometres of degraded reef in countries such as Jamaica and the Philippines. At a similar scale was the area of reefs in the Indian Ocean affected by mass post-bleaching coral mortality during the 1998 El Niño Southern Oscillation event. In between in scale, are events such as major Crown-of-thorns (*Acanthaster planci*) outbreaks on the Great Barrier Reef, which in a bad year might severely impact 100s of square kilometres of reef.

Message Board

- 📌 Coral reef restoration is in its infancy. We cannot create fully functional reefs.
- 📌 Active restoration has been carried out with some success at scales of up to a few hectares only.
- 📌 Natural disturbances and human impacts on reefs can affect reefs on scales of 10s to 1000s of km².
- 📌 Active restoration is not a magic bullet. Improved management of reef areas is the key.



Severely degraded reef with little live coral remaining. Algal turf and sediment cover dead coral colonies.

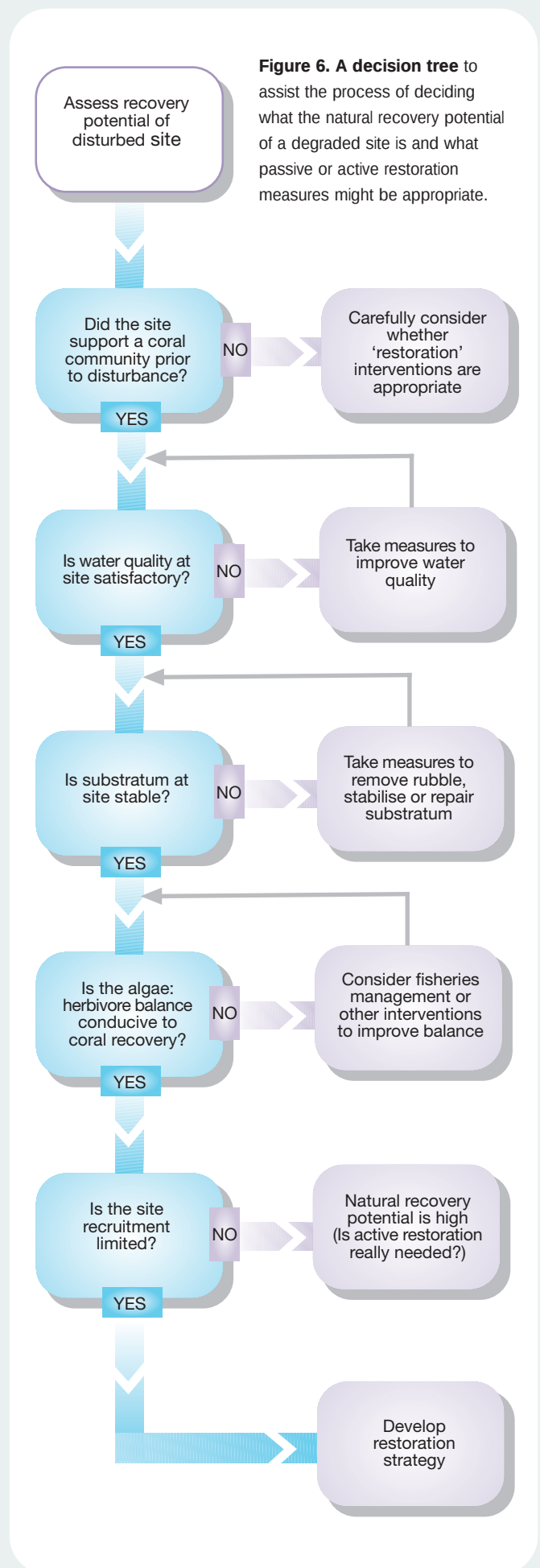
1.6 Is active restoration the right choice?

Restoration needs to be viewed as one option within a broader integrated coastal management context. A key factor in determining whether active restoration should be attempted is the state of the local environment. At one extreme, if local environmental conditions are good, the degraded area small, and there are no physical impediments to recovery (e.g., loose rubble), the degraded patch may recover naturally within 5-10 years. In such a case, active restoration may have very limited benefits. At the other extreme, if local environmental conditions are very poor (high nutrient inputs, sedimentation, overfishing, etc.), the chances of establishing a sustainable coral population may be negligible. In such a case, major management initiatives (passive or indirect restoration) will be needed before any active restoration should be attempted. It is somewhat of an art deciding at what point along the continuum between these two extremes, active restoration is likely to be effective and what other management actions need to be taken before attempting restoration.

To assist in this process a decision tree, which addresses many of the key questions that should be asked, is shown in Figure 6. We look at these questions in more detail below.

For true restoration projects, the first question ("Did the site support a coral community prior to disturbance?") should not need to be asked, but for some tourism developments where there is a wish to create coral patches in safe sheltered lagoon areas, this may be pertinent. What corals can survive where the resort owner wants them? Ultimately ecological constraints will determine this; not money and human wishes.

Even though a site may have supported a healthy coral reef community in the past, water quality may have deteriorated and it may now only be able to support a few tolerant species. If you aim to restore to some more diverse



previous state, then you need to improve the water quality first by management measures. Otherwise active restoration attempts are unlikely to be successful.

The next question relates to whether some physical restoration is needed first. If it is, this may be very expensive. If it cannot be afforded but is necessary, then attempts at active biological restoration are likely to fail. In such a situation, perhaps part of a site can be restored for the funding available.

The next question is perhaps the hardest to answer and relates to the likely sustainability of corals that may be transplanted to the site. The aim of restoration is to restore a self-sustaining community. If there is insufficient grazing due to overfishing and/or loss of invertebrate grazers through disease and macroalgae are dominating, then there is little chance of recruitment to establish the next generation. Transplants may survive but if the ecological processes which allow them to produce future generations of young corals are compromised, the population is ultimately not sustainable. Without some management measures to restore ecological functioning, active restoration may be futile. At present we do not know what level of herbivory may be needed, but a survey can reveal whether there are many herbivores (e.g., parrotfish, surgeonfish, rabbitfish, urchins), the percentage cover of

macro-algae, and whether there are any small corals (say < 5 cm) present. For example, if herbivores are rare, macro-algae are rampant and there's no sign of juvenile corals, this suggests that transplantation by itself will achieve little in the long term. Some other management measures (e.g., fisheries regulation, reduction of nutrient inputs) are needed first.

Finally, comes the question of whether the site is "recruitment limited", that is, does it lack an adequate supply of coral larvae? Even on healthy reefs, some areas may receive few coral and other invertebrate larvae in the currents and recover much more slowly from disturbances than those areas with a better supply. In such cases, using transplants to establish a viable local coral population may greatly accelerate recovery. However, on healthy reefs with a good natural supply of larvae, (particularly in the Indo-Pacific) there is likely to be little ecological need for active restoration. Despite this, there may be other drivers pushing active restoration, such as mitigation compliance, a political need for a restoration effort to be attempted (e.g. public outcry, concern, or insistence that an environmental injustice is corrected), or just human impatience with the rate of natural recovery. In such cases, given the large costs, the money made available for active restoration could probably be better spent on prevention of human impacts or on passive restoration measures (i.e. better coastal management).

2. Physical restoration

It is sometimes useful to distinguish between "physical restoration", which centres on repairing the reef environment with an engineering focus, and "biological restoration", which focuses on restoring the biota and ecological processes. The former can be orders of magnitude more expensive than the latter. Corals, giant clams and large sponges can provide both structural and biotic components, so the distinction is sometimes blurred. For some impacts, only biological restoration (either passive or active) may be needed; for others, a combination of physical and active biological restoration may be required. This is sometimes dubbed "dual restoration". When planning ecological restoration you should always consider both components together.

Certain impacts such as ship-groundings, coral mining and blast fishing can cause major physical damage to the coral reef framework or create substantial areas of unstable coral rubble and sand that are unlikely to recover even over many decades unless some physical restoration is carried out. Major physical restoration is generally a very expensive engineering exercise (costing in order of US\$100,000-1,000,000's per hectare) that requires expert advice. For this reason most of these

guidelines concentrate on biological restoration. Minor reef repair and emergency triage is, however, within the scope of community-based projects.

2.1 Triage and repair of damaged reefs

Where acute impacts have cracked coral boulders, overturned massive corals, dislodged and fragmented coral colonies and other sessile organisms, or deposited foreign objects on the reef, emergency triage in the short term can greatly assist recovery. This may involve cementing or epoxying large cracks in the reef framework, righting and reattaching corals, sponges and other reef organisms



Thai diver righting an overturned Porites colony after the 2004 tsunami.

or at least storing detached organisms in a safe environment until they can be reattached. Tasks should be prioritised with criteria such as size, age, difficulty of replacement and contribution to topographic diversity determining which reef components receive first aid. Foreign objects that threaten intact areas if moved around by wave action (e.g. tree trunks) or contain pollutants (e.g., cars, such as those deposited on reefs after the 2004 tsunami disaster) should be removed from the reef.

Following a ship-grounding, the structural integrity of the reef framework is often under threat – with large craters, gouges and fractures of the reef limestone, which are likely to expand in the event of storms. Physical restoration is called for under these circumstances, and it is essential to seek expert advice. Where there is major loss of topographic complexity, then there may be a risk that unless this complexity is restored the area will recover to some alternative state. To restore topographic complexity, major physical restoration is likely to be needed; again, expert advice should be sought.

Unstable rubble fields, unless very small, are unlikely to show recovery for many decades, with any corals settling on them being overturned, abraded, smothered, or buried. Survival is very low and such mobile rubble areas have been called “killing fields” for corals. Further, rubble and sediment patches created by disturbances may be spread across the reef during storms and cause damage to neighbouring unimpacted areas. The rubble can either be removed or stabilised. Stabilising rubble fields in high energy environments is both expensive and difficult. Partial success has been achieved using flexible concrete mats, or by pouring concrete onto the rubble, but at great expense and with evidence of scour and undermining following storms. Such work should be considered as major physical restoration and expert engineering advice should be sought.

In somewhat less exposed situations, promising results have been achieved (and for lower cost) by covering loose

rubble with patches of large limestone boulders. Boulders should be of sufficient size to remain stable in the environmental setting, even during storms. Impact-related fine sediment lying on reef surfaces may inhibit coral settlement and impair coral growth and should be removed if natural processes do not do the job. If sand has buried areas of coral and other reef biota during a disturbance, then it will need to be removed within several days if there is to be much chance of the buried organisms surviving. Rubble fields in low energy environments (e.g., lagoons or deeper water) may be sufficiently stable to be recolonised by corals and other sessile biota and may be consolidated over time by sponges, coralline algae and other organisms which bind rubble fragments together.

It should be remembered that coral reefs are a patchwork of habitats which may include sand areas, rubble areas, coralline algal reef, macroalgal dominated areas, gorgonian plain as well as areas with high live coral cover. If sand and rubble patches created by an impact are not threatening healthy coral on adjacent areas and substantial funding for physical restoration is not available then leaving them alone and concentrating efforts elsewhere is likely to be the better use of limited funds.

Before biological restoration is attempted, the need for physical restoration should be assessed (see section 1.6). If major physical restoration is needed at a site but funds are not available, then attempts at biological restoration of the site are likely to be unsuccessful.



Tree trunk swept onto a Thai reef by the 2004 tsunami.



Acropora colony damaged by debris swept onto a reef in Thailand by the 2004 tsunami.

Message Board



Physical restoration of reefs is likely to cost US\$100,000-1,000,000's per hectare.



Major physical restoration is for experts only. Seek expert civil engineering advice.



Some physical restoration may be a prerequisite for any chance of successful biological restoration.



Rapid triage of a reef after a disturbance can be very cost-effective and can be carried out by any competent divers under informed supervision.



Large limestone boulders can provide an effective and relatively low-cost way of restoring stability and topographic complexity to rubble fields in less exposed environments.

2.2 Artificial reef creation

Within the scope of physical restoration is the use of artificial reefs, which may range from limestone boulders, to designed concrete (e.g., ReefBalls™) or ceramic (e.g. EcoReefs™) modules, to minerals (brucite and aragonite) electrolytically deposited on shaped wire mesh templates (e.g. BioRock™). Use of such structures in restoration projects should be considered carefully and critically. There is a danger that introducing artificial substrate becomes a displacement activity, which avoids the real issue of managing natural reefs, whilst suggesting that useful action is being taken in a restoration attempt. Use in some countries of artificial reefs as fish aggregation devices (FADs) to create managed “fishing reefs”, following a failure to manage gross overfishing on natural reefs is an example. There is also the question of relative scales. There are estimated to be in excess of 500,000 “reef balls” of varying size deployed worldwide. These will provide at most a couple of square kilometres of topographically complex substrata at a cost of US\$ tens of millions. There are an estimated 300,000 km² of shallow coral reefs in the world so there is plenty of reef substrate available. The main problem is that much of it is poorly managed or degraded.

Bearing these caveats in mind, there are clearly special instances where artificial reefs have a useful role in restoration. Introducing artificial reef structures provides: (1) an instant increase in topographic complexity, (2) stable substrate for coral and other invertebrate settlement (or for coral transplantation), (3) hard structures that discourage various forms of net based fishing (including trawling and seine net fishing) which cause reef damage, (4) alternative dive sites for SCUBA divers in areas with high diving pressure on the natural reefs, and (5) they are likely to attract fish. This assumes that the artificial structures are well-constructed and deployed so that they remain stable in storm conditions. For restoration, the aesthetics and “natural look” of the artificial structures, both initially and after colonisation by corals and other reef organisms, needs to be considered. The various trade marked systems listed above all claim some level of aesthetics and naturalness, and

managers intending to utilise such structures can judge for themselves via the websites of the companies involved. Use of tyres¹ and other man-made junk for artificial reef creation for restoration is not recommended for both structural and aesthetic reasons.

Potential roles for artificial reefs in reef restoration are:

1. Stabilising and restoring topographic complexity to degraded rubble areas such as those produced by blast fishing and thus bringing back fish and corals to areas with little chance of recovery.
2. Tourism or marine park education and public awareness, where easy and safe access to bits of “reef” habitat are required. Several resorts around the world have utilised artificial structures as platforms for coral transplantation in this way.
3. Reducing diver pressure on natural reefs in areas with large numbers of tourist divers. A few resorts have created artificial reefs attractive to divers with a view to focusing early dives by trainees with poor buoyancy control on these structures and reducing overall diving pressure on natural reefs (by perhaps 10% if each diver visits the site at least once in a one week vacation).

Appropriate (specially designed for sea defence) artificial reef modules may also be useful where sea defence services of reef flats are being lost. Such services may cost from US\$1 million –10 million per kilometre to replace depending on the shoreline.

The standard and regular artificial surfaces provided by some artificial reef modules are also used by biologists carrying out restoration research as a way of standardising their experiments. This does not mean that they are endorsing them for use in real restoration projects. Beware also that although in some places almost any artificial substrate (concrete, PVC, tyre, or ship) will be rapidly colonised by corals, in other places, artificial reef structures may remain stubbornly devoid of coral recruits and serve little purpose.

Message Board



There are at least 300,000 km² of coral reefs in the world. Lack of hard substrate is not a critical issue. Management of degradation of natural reefs is the critical issue.



Use of artificial reefs in restoration needs to be considered carefully and critically in terms of need, cost-effectiveness and aesthetics.



Artificial reefs, if well-designed and constructed, can provide (1) an instant increase in topographic complexity, (2) stable substrate for coral settlement or transplantation, (3) fish aggregation, (4) sea-defence services, (5) hard structures to discourage net-based fishing (trawling, seining) in coral areas, (6) dive sites to reduce diver impacts on natural reefs in areas with high concentrations of diving tourists.

¹ Tires (U.S.)

3. Biological restoration

Biological restoration should always be considered in the context of the overall environment of the site being restored, both the physical and biotic environment and the human and management environment. As noted in section 1.3, "Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed." This assistance may be in the form of indirect management measures that remove impediments to natural recovery, or direct, active biological restoration such as transplantation of corals and other organisms. Examples of the former would be management actions to reduce fishing pressure, sediment run-off or sewage discharges. Thus passive biological restoration may be realised through a range of coastal management actions that reduce anthropogenic pressures on coral reef systems.

The most frequent active restoration intervention is to transplant corals (and other biota) to a degraded site. It is very important to minimise any damage to healthy (or

less degraded) "donor" reef areas from where transplants may be obtained, and to maximise the survival of the transplants on the reef being restored. Ultimately, a restoration project will only be successful in the long term if a self-sustaining and functioning coral reef community is established.

The following sections look at aspects of active biological restoration and discuss major issues. Given the prevalence of coral transplantation in restoration projects, we devote most of the discussion to this activity. There are now a range of options that show promise in allowing practitioners to minimise the collateral damage involved in sourcing transplants and maximise the effectiveness of the coral material used. These range from care in how transplants are sourced to sexual and asexual propagation of corals in either *ex situ* (in aquaria) or *in situ* (in the sea) culture (Figure 7). These options for managers are discussed more fully below.

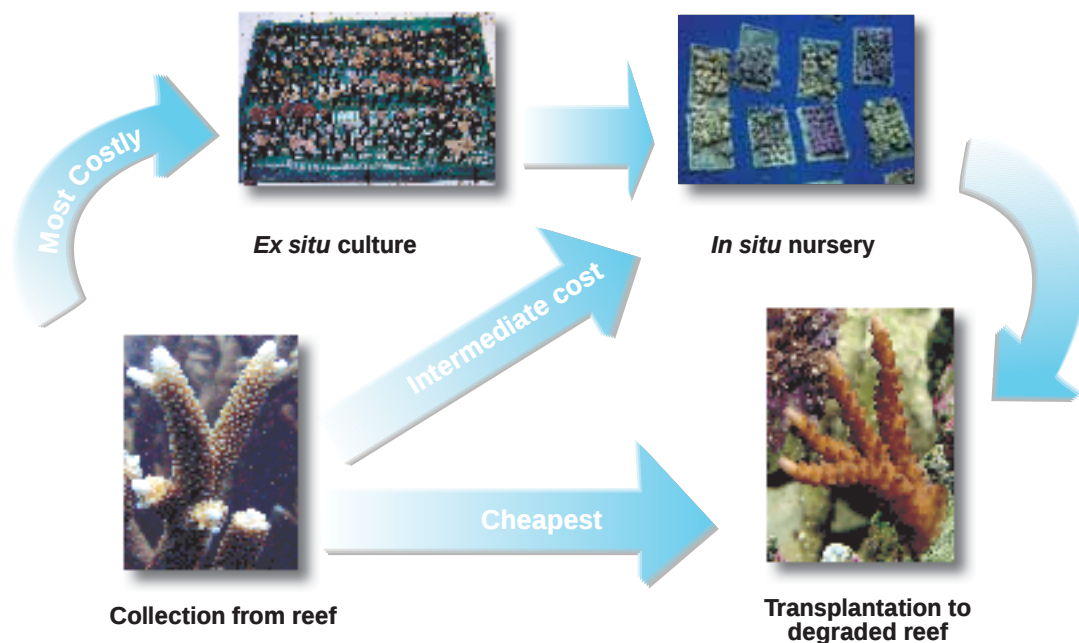


Figure 7. Direct versus indirect propagation of corals. The cheapest route is to collect corals directly from the reef and transplant to the degraded area. However, to obtain good survival, individual transplants need to be quite large (say, >5-10 cm). Smaller fragments (say, 2-3 cm) can be successfully cultured in the sea in mid-water or benthic nurseries until large enough to survive well. This has costs but makes better use of coral material. Very small fragments do not survive well in *in-situ* culture but can survive and grow in *ex situ* culture. Thus for even greater cost and a longer two-stage culture process, there is the potential to create tens of thousands of small colonies from similar numbers of tiny fragments (say, 10 mm in size). The longer the period in culture the greater the cost of producing each transplant. *Ex situ* culture has much higher set up costs than *in situ* culture. Planktonic coral larvae can also be cultured, settled onto pieces of substrate, and grown up in mid-water cages for 6-12 months until large enough to have a reasonable chance of surviving on the reef.

3.1 Why focus on corals?

A criticism sometimes levelled at coral reef restoration projects is their focus on corals. The critics make the valid point that transplanting corals and ignoring the diverse other major groups of living organisms, does not restore the complex reef ecosystem. However, as discussed in section 1.3, the restoration practitioner is not trying to rebuild an ecosystem piece by piece, but is attempting to assist natural recovery processes. At present, the structure, assembly rules and functioning of reef ecosystems are too poorly understood for restoration to attempt anything more ambitious. Also, restoration is expensive and resources must be focused where needed most.

Corals are keystone species of the reef ecosystem in the same way that trees are keystone species of forest ecosystems. Corals appear to be essential to reef restoration just as trees are essential to reforestation. They are also particularly at risk from a range of impacts (section 1.2), in part because of their intimate symbiosis with zooxanthellae which makes them sensitive to small rises in seawater temperature above normal yearly maxima.

- Corals provide the major constructional and accreting element for the sea-defence service provided by reefs.
- Corals provide structural complexity (usually correlated with biodiversity) and shelter for both fishes and invertebrates.
- Coral habitats provide shelter for herbivores which can help control algal overgrowth.
- Living corals are attractive and representative of healthy reefs in the minds of tourists.

When corals are lost then fish biodiversity and abundance may decline too, along with revenues from both diving tourists and fishing. If a sustainable coral population and some structural complexity can be established, then it is more likely that other elements of the system will re-establish naturally, along with functioning and feedbacks. Most transplantation studies have focused on stony corals with symbiotic algae which are the main reef builders (zooxanthellate scleractinian corals), but other hard corals such as the blue coral *Heliopora*, organ-pipe coral *Tubipora* (related to soft corals in the subclass Octocorallia), and fire coral *Millepora* (class Hydrozoa) can be important in certain habitats and can be successfully transplanted.

Other components of the reef ecosystem should not be ignored in restoration. On the contrary, soft corals, sponges, giant clams, *Trochus* shells and urchins among other groups have featured strongly in both culture and transplantation projects. Soft corals, sponges and giant clams can all provide topographic complexity and individuals or individual colonies may be decades old. In restoration projects such as ship-groundings they should be rescued, and reattached if necessary. Grazing urchins such

as *Diadema* and snails such as *Trochus* may have a role in assisting recovery of herbivory processes in areas where fish herbivores are rare due to overfishing.



A beautiful and topographically diverse reef in the Similan Islands, Thailand with huge *Porites* colonies providing shelter for fishes and invertebrates.

3.2 Sourcing coral transplants

To obtain a transplant you have to remove some coral from a reef (unless you've grown the coral from scratch). Thus, for every asexually produced transplant, there is some collateral damage. You can minimize this damage in a number of ways. The first rule is to make the best use of the live coral material available to you. Note that local legislation may require that you obtain a permit before you can source transplants or indeed introduce them to a degraded area.

In some cases where damage is being repaired immediately after an impact such as a ship grounding, there may be whole coral colonies which have been detached and which can have their survivorship enhanced by being reattached *in situ* as whole colonies. This is more physical restoration than biological as no new living material is being introduced. In cases where a reef is threatened by "reclamation" or a high impact industrial development (e.g. a power plant), whole areas of reef may be transplanted and whole colonies translocated to a refuge site. However, this use of whole colonies tends to be the exception. Given the increased likelihood of mortality from transplantation, if whole colonies are used there is likely to be a net loss of coral. Although whole colonies are thought to be less susceptible to transplantation stress than fragments, for some sensitive species 50% of colonies transplanted have died within two years. Thus, even in such cases, some fragmentation of colonies being translocated may be advisable, in an attempt to balance likely losses. Even in the same species, different genotypes can show differing susceptibility to transplantation stress.

Normally coral transplants will be sourced as fragments. Small fragments may then be reared for a period of time in nurseries (see section 3.3) where they can be grown into small colonies which are then transplanted, but they must originally be sourced from somewhere.

On most reefs one can find coral fragments (often broken-off branches) which have been become detached and

which, apart from species that naturally reproduce by fragmentation, tend to have a low chance of survival unless they can be reattached. Often parts of these fragments may already be dead or dying. Such coral fragments have been called “corals of opportunity” and represent a generally non-controversial source of transplants. The logic being that most would die anyway if not utilised for transplantation (except in those species which naturally reproduce by fragmentation). Even partly dead branch fragments have been shown, once the dead and dying parts have been cut away with pliers, to provide healthy transplants with good survival. Branching species tend to supply most “corals of opportunity”, with more fragile species providing more fragments, and more robust species providing less fragments. Thus these “corals of opportunity” may not provide a cross-section of the common species and other sources may also be needed.

If intact donor colonies are used as a source of fragments for either direct transplantation or a period of culture followed by transplantation, then the limited research suggests that only a small part of the colony (less than c.10%) should be excised in order to minimise stress to the donor colony. Until more research is done and we have a better understanding of the impact of pruning coral colonies, we suggest that it is best to apply the precautionary principle and not excise more than 10% of donor colonies. For massive coral colonies it would appear best to remove fragments from the edge of the colony.

Good Practice Checklist

- ✓ Check local legislation to ascertain whether you require a permit before you can collect transplants or indeed introduce them to a degraded area.
- ✓ Source transplants from areas as similar as possible to the site that is to be restored (same depth, same exposure, same sedimentation regime, same salinity, same substrate, same range of water temperature)
- ✓ Carefully consider how to make the best use of the coral transplant source material available to you.
- ✓ Try to use “corals of opportunity”, that is, naturally generated fragments on the reef that have a poor chance of survival unless reattached.
- ✓ If intact donor coral colonies are used to source transplants, then try to use not more than 10% of the donor coral to minimise stress.
- ✓ Do not core massive colonies to obtain transplants but take fragments from around the edge of colonies.

3.3 Coral culture

Methods for both asexual and sexual propagation of large numbers of corals have now been successfully demonstrated. As will be seen from the discussions below, the main scientific unknown is whether the cultured corals can be successfully deployed on degraded reefs and will survive well there. The cheapest option for transplantation is to transplant directly; culture may make better use of coral material but it does so at a financial cost. The more sophisticated the culture, the greater the costs; also, the longer the time in culture, the greater the costs (Figure 7). Reef restoration is already expensive compared to seagrass or mangrove restoration. Thus the drive is towards low-cost methods, and maximising the efficiency and cost-effectiveness of coral culture is a key challenge. *Ex situ* culture in aquaria is generally more expensive than *in situ* culture in the sea in either mid-water or benthic nurseries. However, survivorship of very early stages or very small transplants (e.g. nubbins of <5-10 mm diameter) is generally only satisfactory in *ex situ* aquaria. There are thus a range of trade-offs between survival, type of culture, and costs, which are as yet not well quantified.

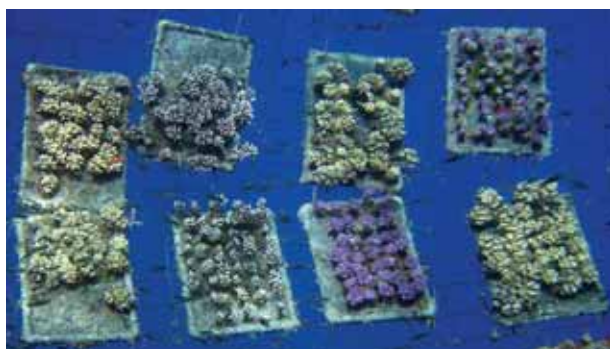
3.3.1 Asexual propagation of corals

Corals can be grown asexually from fragments (known as ramets when derived from the same colony (clones), and when very small, often called “nubbins”) and this is the most common form of culture. Colonies have even been grown experimentally from single excised polyps in *ex situ* culture. However, for most projects larger fragments (3-10 cm in size) are more likely to be used as these can be cultured *in situ* in benthic or mid-water nurseries at reasonable cost. The technologies involved are within the reach of small community-based projects that have access to scientific advice and have been used successfully by such projects. The aims of asexual culture are: 1) to maximise benefits from a given amount of source material and thus minimise damage to donor areas, 2) to grow fragments into small colonies which should survive better than the fragments would have done if just transplanted directly to the reef, and 3) to have banks of small corals readily available for transplant in the event of an impact such as a ship-grounding.



In situ culture of *Acropora muricata* nubbins in a shallow water nursery in Philippines

The potential benefits of nursery culture are that hundreds of small colonies may be produced from fragments of a single colony. The costs are those involved in setting up the nursery areas, collecting the fragments, attaching these to some substrate and then looking after them until deemed ready for transplantation. This husbandry, which may involve removing algae and other fouling organisms that threaten to overgrow the cultured fragments, or removing predators such as the coral-eating snail *Drupella*, can be quite time consuming. The smaller the fragments, the longer they are likely to need culturing before they can be transplanted and the more benign the nursery environment will need to be if there is to be good survival. For branching species, “nubbins” of about 3 cm in size may require 9-12 months to develop into substantial fist-sized colonies. As yet, too little is known about the trade-offs between size and survival (see section 3.6) to know how long to culture. This is likely to vary between species and also depend on the state of the degraded site.



Trays of coral nubbins being cultured in a mid-water coral nursery

In the same way that the donor site should match the transplant site with respect to environmental conditions, so the intermediate nursery site should be as similar as possible to the donor and transplant sites in terms of conditions. Experience shows that if the nursery site environment differs significantly from that of the source site for the culture material, you may get poor survival in culture. The exception appears to be that if the nursery conditions are better (e.g., less sedimentation, better water clarity, etc.) than those of the source site then the corals may thrive in nursery culture. However, it is unclear what happens when these cultured colonies are returned to a harsher environment on a degraded reef. Nursery sites require shelter from strong currents, surge and wave action – conditions that are typical on coral reefs – thus the nursery site is often removed from the coral reef site itself but must still have suitable conditions for coral survival and growth.

Producing hundreds of cloned colonies from a single colony can be very useful for experimental work but for actual restoration projects the genetic diversity of the nursery needs to be considered. Sourcing the fragments to be cultured from “corals of opportunity” (i.e. loose coral fragments lying around on the reef) or taking 10% or less of colony mass from a variety of appropriate donor colonies

is one way of ensuring reasonable genetic diversity among the prospective transplants. Should it become possible to identify bleaching resistant or otherwise tolerant genotypes, then asexual culture presents a promising way of propagating large numbers of these strains.



Acropora nubbins which have grown into small colonies in a mid-water coral nursery

To date there are several examples of both mid-water and benthic nurseries where asexual culture of many thousands of small colonies has been achieved with generally good survival (often 90% plus over 6 months). As such, asexual culture of corals appears to have great potential in reef restoration in an analogous way to how silviculture in land-based nurseries supports reforestation projects on land. However, the next step, which is the successful transplantation of nursery-reared colonies to degraded reef areas and their long-term survival there, has yet to be demonstrated on a large scale (0.1-1.0 ha) and is the subject of considerable ongoing research.

Estimates from *in situ* mid-water and benthic nursery culture suggest that in the order of 5–10 transplants can be reared per US dollar. At a spacing of 0.5 m on a degraded reef this would suggest culture costs alone of US\$4,000-8,000 per hectare (for the 40,000 transplants/ha that would be needed).

3.3.2 Sexual propagation of corals for seeding reefs

Corals reproduce sexually either by broadcast spawning or by internal brooding of planular larvae followed by “planulation” (release of planulae into the surrounding sea-water). Corals often produce very large numbers of eggs and/or larvae. In nature the vast majority of these do not survive, however if larvae produced by planulating or broadcast spawning corals can be collected and artificially reared, mortality rates can be dramatically lowered and the larvae can be a potentially valuable source of corals for reef restoration projects. Sexual propagation of corals has two



Tracking a coral spawning slick with a drogue.



Rearing of a coral spawning slick (embryos and planular larvae) in a floating pond.

main advantages over asexual propagation techniques. Firstly, there is less need to fragment donor colonies, thus reducing collateral damage to source reefs; secondly, sexually produced coral colonies are not clonal and therefore have considerably greater genetic diversity. This method may require a few colonies or large fragments to be removed from the reef and brought in to aquarium tanks to spawn. Although colonies can be replaced on the reef following spawning, the stress of removal and subsequent transplantation may occasionally cause the colony to die.

The larvae produced by planulating or broadcast spawning corals can be collected and reared for varying periods of time before either settling them directly onto the reef or settling them on substrates in aquaria. Once settled in aquaria, the tiny corals can be grown until of sufficient size to be better able to survive transplantation to the reef. The methods are still being tested by scientists and these technologies require more technical expertise to apply them successfully than the more widely applied asexual culture and transplantation methods above.



Montastraea colony in the process of spawning in Philippines.



Simple and inexpensive floating ponds can provide a means for controlled culture of coral spawn in remote areas.

Although some planulating corals may produce planular larvae on a monthly basis, many broadcast spawners may release eggs and sperm only once or twice a year. The broadcast spawning is generally synchronised, with mature colonies of a species tending to release gametes on the same few nights. This leads to mass spawning events when many species spawn at around the same time producing large slicks of coral larvae. Broadcast spawning is the most common mode of coral reproduction and the timing of major spawning events is reasonably predictable for given locations. But this does mean that some knowledge of the reproductive patterns of the local coral assemblages is required and that for most species supplies of larvae are available for only a few weeks a year.

There are two approaches to obtaining supplies of larvae. Either mature colonies of planulating or broadcast spawning species can be collected and held in aquaria until they release planulae or gametes respectively, or, for broadcasters, slicks of millions of coral larvae can be collected from the sea surface at one or two well-defined times of year. These slicks can either be held *in situ* or transferred to *ex situ* aquaria.

In the first case, slicks can be stored in floating culture ponds in the sea (even plastic paddling pools appear adequate) for about a week by which time most larvae are

ready and able to settle onto the reef (this condition is referred to as “competent”). At this point, they can be pumped down into mesh tents on the degraded reef and allowed to settle in high densities. The mesh tents are to prevent them being washed off the reef by currents. With such techniques you can achieve around 100 times the amount of coral settlement that you might expect naturally. However, a large unknown is whether this makes any significant difference in the long-term because so very few of these newly settled corals will survive to become mature reproductive colonies and mortality may be density-dependent.

By transferring to *ex situ* aquaria, the newly settled corals can be carefully reared away from the perils of the natural reef environment and only transplanted to the reef when they have a reasonable chance of survival. Survival on the reef increases dramatically with size/age. For example, a study with the planulating coral *Pocillopora damicornis* showed almost 8 times better survival of newly settled corals in aquaria over one week (69%) compared to the natural reef (9%) and negligible survivorship in the wild over 3 weeks. The same study also showed that if corals were cultured for about 6 months until they were >10 mm diameter, they had around a 25-30 times better chance of surviving for 5 months when transplanted to the natural reef, as those transplanted when <3 mm (about 1 month old).

Using slicks from broadcast spawners, many thousands of coral polyps can be settled on tiles (preconditioned in seawater for about 2 months) in *ex situ* aquaria. This settlement can be assisted by using attractants derived from certain species of coralline red algae (sometimes referred to as larval “flypapers”) which stimulate the settlement and metamorphosis of coral larvae into juvenile corals.



Acropora tenuis planulae and newly settled and metamorphosed juveniles attached to live crustose coralline algae.

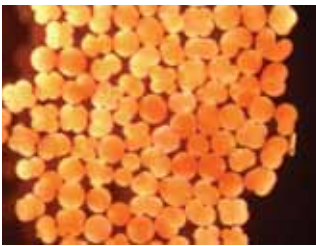


Juvenile herbivorous snails such as 5-7.5 mm *Trochus* can be used to graze away algae that may otherwise smother young corals.

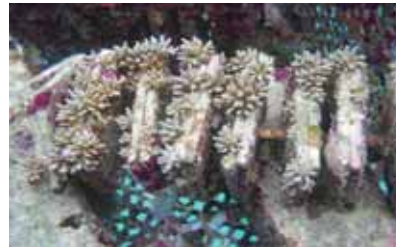
After a few weeks the tiles and juvenile corals can be co-cultured in mid-water cages with small herbivorous snails (such as 5–7.5 mm *Trochus*) which graze away algae that may otherwise smother the young corals. Using these methods thousands of *Acropora* colonies of about 4 cm diameter can be raised from coral spat within 12 months. Again, the step which is yet to be demonstrated is the transplantation and successful growth of these small colonies on degraded reefs. Since culture carries a cost, we need to find out details of the trade-offs between the time spent in culture (costs) and the increase in survival subsequent to transplantation as a result of this (benefits). This is the subject of ongoing research.



Monitoring growth and survival of corals and cleaning algae and other fouling organisms off mid-water cages in which juvenile *Acropora* are being raised on tiles in Palau.



Acropora eggs and embryos in early stages of development, two hours after coral spawning.



Nine month old *Acropora* (3-4 cm diameter) settled as spat on tiles in tanks and then reared in mid-water cages in co-culture with *Trochus* snails which graze on algae.

Message Board



Corals can be successfully cultured from asexually produced fragments or from larvae produced by sexual reproduction.



The main reason to culture asexual fragments is to maximise benefits from a given amount of source material and thus minimise damage to donor areas. Culturing fragments can generate hundredfold gains in transplantable material.



Care should be taken to ensure reasonable genetic diversity of cultured transplants.



Culture from larvae has been carried out experimentally but requires more technical expertise than asexual culture. For many species, spawning is very seasonal which restricts when larvae are available. However, sexual propagation in culture does have the potential to produce huge numbers of small corals.



The biggest question-mark over culture as a source of coral material for restoration is how well the nursery reared corals will survive when transplanted onto degraded reefs.

3.4 Attaching coral transplants

Transplants should generally be securely attached to the reef unless they are in such sheltered conditions that fragments will remain in place without assistance. This can be done with cement, a range of epoxy adhesives, nails, stainless steel wire, insulated wire, and cable-ties. Nails or long staples hammered into the reef may provide attachment points for cable-ties or wire where otherwise difficult to attach. Small nubbins of coral have even been successfully attached to plastic pins (for mid-water nursery culture) and other substrates (e.g., giant clam shells) using cyanoacrylate glues (Superglue). Species which naturally reproduce by fragmentation are usually able to self-attach

within weeks, if stable. On exposed reefs, detachment of transplants can be the main cause of death and can decimate the transplant population.

The most effective method will depend on: (1) the size and growth-form of the transplants, (2) the exposure of the habitat to currents and wave action, and (3) the nature of reef substrate itself. In various projects, acceptably low rates of loss (detachment) from the reef have been achieved successfully with epoxy compounds, cement, and wire. Methods of attachment which allow any movement of the fragment may cause abrasion and tissue loss and are not recommended. This sometimes occurs when fragments are tied to the reef rather than cemented.

Coral fragments are often able to grow over the wires or cable-ties attaching them within months. However, generally you should try to minimise introductions of man-made materials into the reef environment. Where living coral tissue is in stable close contact with a reasonably clean (e.g. not with thick sediment or thick algal turf) surface, the coral can self-attach by growing onto the surface. Once the coral fragment has grown onto the substrate then the risk of detachment is much reduced. This self-attachment process can occur within a few weeks to a few months and methods which encourage the process are recommended.

One low-cost method which has been used successfully for transplanting branches to coral rock areas is to find natural holes which are about the same diameter as the base of the branch, or to make holes in the reef with a chisel or broad screwdriver to this size. The area around the hole is scraped back to bare substrate and the branch inserted, being fixed in place with epoxy-putty on one side but with

live tissue pressed against the bare substrate on the other side. This promotes self-attachment on that side and appears to work well.

Cultured coral fragments are usually already attached to some substrate. These may range from plastic pins used in mid-water nurseries to 20 cm x 5 cm pieces of limestone, which have been used in some benthic nurseries. Fragments or small colonies from nurseries are likely to have already self-attached to the substrates on which they have been cultured. Plastic pins can be fixed into natural or man-made holes in the reef, with epoxy if necessary. The area surrounding the hole should be scraped clean and the growing base of the coral should be given every encouragement to extend onto the reef substrate itself. Where fragments have been grown on pieces of limestone, these have been wedged onto the reef between the branches of dead corals and additional attachment points encouraged with branches pressed against the substrate.



An Acropora colony one month after transplantation, showing the rapid self-attachment to the substratum around the base (bluish growth).



Auger for boring a hole in soft coral rock and two-part epoxy putty for fixing transplant in hole.



Transplant inserted into hole in coral rock with one side of base anchored with epoxy putty and other side in contact with substrate.

Good Practice Checklist

- ✓ Transplants should in general be securely attached to the reef at the site being restored.
- ✓ A range of epoxy adhesives, cement, wires and cable-ties have all been used to successfully attach transplants to degraded reef areas.
- ✓ The most effective method of attachment will depend on: (1) the size and growth-form the transplants, (2) the exposure of the habitat to currents and wave action, and (3) the nature of reef substrate itself.
- ✓ Where feasible, try to avoid introducing man-made materials e.g. nails and staples into the reef environment.
- ✓ Try to encourage self-attachment by transplants by juxtaposing living coral tissue to bare substrate. Once colonies have self-cemented the chance of detachment is dramatically reduced.

3.5 Which species?

At present there is limited information on which coral species are suitable or unsuitable for transplantation. For some species, the results of studies by different researchers are apparently contradictory. This could be a result of misidentification, differences in handling, or differences in the transplant sites. The dearth of controlled experimentation in reef restoration has meant that few specific recommendations can be given. However, there is some general guidance that we can provide.

The first priority must be to find out which species would be expected to survive at the site that is being restored. Surveys of what still survives at the degraded site or at nearby, similar, less impacted sites (potential “reference ecosystem” sites), or historical data from the area can give some idea which species may be appropriate. For example, if only sediment tolerant species appear to be surviving at a degraded site, then it is unlikely that introducing non sediment-tolerant species will be successful unless the source of sedimentation is reduced or removed. Candidate species for transplantation would be those that persist at undegraded (or less degraded) sites in the same environmental setting. They should be transplanted only if any chronic adverse anthropogenic impacts which are likely to cause their death are being addressed by management measures. Otherwise transplantation is likely to be futile.

Branching species such as those in the families Acroporidae and Pocilloporidae tend to be fast-growing and easy to fragment (or find natural fragments of). As such they have been much favoured in transplantation as they can produce a rapid increase in % live coral cover in a relatively short time. On the downside they tend: 1) to be somewhat more sensitive to transplantation than slower growing sub-massive and massive corals, such that survival rates can be much lower, 2) to be more susceptible to warming associated with El Niño Southern Oscillation events and thus more likely to be subject to mass-bleaching and subsequent mass mortality (if the warming event is prolonged), and 3) to be more susceptible to disease than some other families. Thus there are significant risks

associated with restoration projects which rely on such species. In the Indo-Pacific where these families are very prevalent, it is also the case that these species are in many places the first to recruit and may dominate natural recruitment. In sites that are not recruitment limited, their populations are thus likely to recover relatively quickly. For example, in seven years in the Maldives one can expect tabulate *Acropora* colonies around 1.3 m in diameter to grow from naturally settled coral spat.

Other growth forms (massive, submassive, foliaceous) and branching species in other families such as the Poritidae and Merulinidae, which tend to be slower growing, have been less studied in terms of restoration potential. Although there is considerable variation between genera and even species within these other families, it is clear that at least some of these less favoured species (*Porites lutea*, *P. lobata*, some *Pavona* species) are less sensitive both to transplantation and to warming anomalies and are thus likely to survive better in the long term despite growing more slowly. The drawback for these slower growers is that the desired topographic complexity (which provides shelter and tends to attract fish and other fauna) is achieved far more slowly with these species.

A sensible compromise is to transplant a good mix of species and not to put all your eggs into one high-risk basket by concentrating on acroporids and pocilloporids. In environments dominated by these families, the key question is whether the site is recruitment limited. If it isn't, then there is a risk that money spent on restoration may not be well spent. If it is, then the risks are probably worth taking.

There is ongoing research that aims to provide an index of relative susceptibility to bleaching for common coral species; this will be a useful guide when choosing species to transplant. Even within coral species, colonies with particular clades of symbiotic zooxanthellae have been shown to be more resistant to bleaching than colonies with other clades. Whether such resistant colonies can be readily identified in the field and then selected for transplantation or propagated asexually in nurseries (see section 3.3.1) is an interesting area for research.



View of transplant site in Fiji with recently transplanted *Acropora*.



Snorkel diver placing a transplant on a degraded site in Fiji.

Good Practice Checklist

- ✓ Only transplant species appropriate to the reef environment being restored. That is, species which are surviving at nearby sites in the same or a very similar environmental setting ("reference ecosystem" sites).
- ✓ If trying to restore to a prior ecosystem state, ensure that any chronic anthropogenic impacts which may have contributed to degradation, are first either ameliorated or removed.
- ✓ Attempt to transplant a mix of common species using surveys of reference ecosystem sites as guidance.
- ✓ Consider that although fast-growing branching species can provide a rapid increase in coral cover and topographic complexity, they also tend to be more susceptible to coral bleaching, stresses of transplantation, and disease.
- ✓ Consider that although slower growing submassive and massive coral species may provide a slower increase in coral cover and effect on topographic complexity, they tend to survive coral bleaching episodes better and tend to be less susceptible to the stresses of transplantation and to disease.

3.6 Size of transplants

There is evidence that the size of transplant matters, with better survival being achieved at larger sizes. The benefits in terms of survival may operate over a wide range of sizes from 1 mm to 10 cm. Work with very small coral transplants suggests a marked improvement in survival above about 10 mm (1 cm) in diameter (see section 3.3.2), whereas some experiments working with larger transplants have shown better survival of transplants over a size of about 10 cm compared to smaller ones. The critical sizes may vary with both species and site, being dependent on both the amount and type of algae (and other organisms) competing for space and the abundance and size of potential coral grazers like parrotfish. If a transplant is just one mouthful then one bite from a grazer might destroy it. If it is several

mouthfuls then it may survive. If there is a lot of macroalgae then a small coral may easily be shaded and overgrown, whereas a larger one may be able to persist.

At present we do not know enough about how size and survival vary from species to species or the trade-offs between size and survival, or indeed whether there really is a critical size at which survival dramatically improves, or a continuum of improved survival with size. However, it seems likely that transplanting asexually derived fragments at a minimum size of 5-10 cm will promote better survival and do more to enhance topographic diversity. Given the time and labour involved in transplantation it seems more cost-effective to err on the side of larger and less vulnerable transplants until better information becomes available.

Good Practice Checklist

- ✓ Consider that larger transplants may survive better.
- ✓ Consider that for asexually derived transplants a minimum transplant size of about 5-10 cm may promote better survival.
- ✓ If rearing sexually produced transplants from larvae, consider culturing until at least 1 cm in size before attempting to transplant to the reef. Trade-offs between cost and survival above this size are unclear.

3.7 Diversity and density of transplants

Since the aim of restoration is to restore a site to its pre-disturbance state, then the "reference ecosystem" state (see section 1.3.1) should provide a reasonable indication of the diversity of species present and the approximate densities in which the main species occur on healthy reefs in similar environmental settings. Line-intercept-transect or

quadrat surveys (see English *et al.*, 1997) of potential source areas for transplant material (which should be in a comparable environmental setting to the reef to be restored) ought to provide information on relative abundances of the main species and their densities. These can be used to guide transplantation or at least provide long-term targets.

This emphasises the importance of restoration goals and defining a “reference ecosystem” state to which you are trying to restore a site. The difficulties of defining this state in the face of both global climate change and the widespread decline of coral reefs from anthropogenic impacts have been mentioned earlier. However, the perils of embarking on a restoration project without any goals and without any idea of the state you expect the restored reef to achieve, seem far greater, with little likelihood of a successful outcome. Without some reference state, you have no idea which species to transplant or what numbers to transplant or what kind of fish, coral, algal and invertebrate community you might eventually expect to see. By at least thinking about what an appropriate reference ecosystem state might be, you may avoid pitfalls such as transplanting reef crest corals into a lagoon and then watching them die.



Experimental transplants arranged on a degraded bommy in Philippines.

As densities rise, so do costs, and very fast. Transplanting corals one metre apart would require about 10,000 transplants per hectare (ha). However, transplanting corals on average every 0.5 m over one hectare would require over 40,000 transplants/ha. Reports of reef restoration projects indicate that different groups have suggested aiming to restore anything from 2 corals per m² on reefs which already had around 20% coral cover to c. 25 corals per m² on totally degraded reef. The latter target density was based on the densities of corals at a “reference ecosystem” and calculations showed that the cost of restoration would be well over US\$400,000 per hectare. From a cost perspective a “planting ratio” of 10% target density was considered feasible in that case. Others have opted to increase coral cover by a fixed amount, for example, from an initial 10% on a degraded site to 20% post-transplantation. Defining an optimum transplant density is at present clearly more art than science. Returning to the aims of restoration, we reiterate that these are to assist natural recovery not rebuild the reef

piece by piece. The important thing is to assist the reef to get on a positive trajectory (see Figure 2) heading towards improved functionality. Thus the density of corals at a reference ecosystem is only a guide to a long-term goal, not a transplantation aim. If resources are limited it is better to attempt to restore a relatively small area well, than a larger area poorly.

Using the density of all corals on the reference ecosystem as a guide is also a somewhat crude measure. Some corals might be 1 cm across, others 1 m across. If size-frequency distributions were available from surveys of the reference ecosystem then densities of corals at the average transplant size or larger would be a more justifiable target. Whether this target should be something that is the immediate aim of the transplantation, or the ultimate aim after say 5-10 years of natural recovery, assisted by some initial transplantation, will make a considerable difference to the transplant density attempted. An alternative approach would be to set a goal that – within say 5-10 years – the restored site should aim for say 75% of the coral cover (or better) of the reference ecosystem. Knowing existing coral cover, starting sizes of transplants and average growth rates, one could then estimate the number of transplants that would be appropriate to achieving the goal. This is clearly an area where modelling can assist and where modelling is very much needed. Interestingly, a recent modelling study with fairly simple assumptions has suggested that greatest restoration benefit is obtained if transplants are arranged in regular grids. However, more sophisticated models with additional parameters are needed to investigate this subject further.

There are a range of constraints that can be considered. The aim is a self-sustaining coral population. Colonies of the same species will need to be near enough each other to be able to reproduce successfully. Perhaps some clumping might assist this, rather than spreading transplants thinly over the degraded area. In terms of topographic complexity gains, clumping may also be beneficial with clusters of coral transplants aggregating fish more effectively than small isolated transplants. At the other extreme, some species of coral are quite aggressive and may kill others if placed close to them. Incompatible species should not be placed close together. Like many other areas of reef restoration, the unanswered questions loom large.

Good Practice Checklist

- ✓ Make use of surveys of a “reference ecosystem” (healthy or less degraded reef in a similar environmental setting) to inform selection of appropriate species and provide estimates of the density of colonies (over 5-10 cm) that could be an eventual goal.
- ✓ Remember that you are not trying to create an “instant” reef but trying to assist its recovery.
- ✓ Better to restore a small area well than to try to restore a large area poorly, because of funding constraints.

3.8 When to transplant?

Transplantation causes stress to corals. Often transplants show “bleaching” for a month or two after transplantation before returning to a normal colour. If donor colonies are being used as sources of fragments for transplantation these will be stressed and the transplants themselves will be stressed. The key to successful transplanting is to minimize stress and so transplants should be kept at temperatures as near as possible to that of the sea, kept in the shade, exposed to the air as little as possible, handled as little as possible and transported for as short a time as possible. If corals are held in closed containers then try to exchange the seawater regularly. Some workers avoid transplanting in the middle of the day on hot sunny days. However, some corals have been found to be surprisingly tough (see case-studies in section 5). A key sign that a coral is stressed is when it starts to produce lots of mucus.

The main point of this section is to emphasise that at certain

times of year corals are normally under more stress and these times of year should be avoided for transplantation if possible. In general it is during the warmest months when bleaching tends to occur that the corals are likely to be under stress. It is also during these months that coral disease appears to be more prevalent. If you transplant then, you are likely to have greater mortality of transplants. Examine the annual sea surface temperature records for your area and try to transplant at least a few months before or after the annual peak in temperature (Figure 8). Bad weather at these times may also be another constraint. Another factor to consider is the reproductive state of the corals. Corals which are channelling lots of energy into egg production and are just about to spawn seem likely to be more susceptible to the additional stress of transplantation (as either donors or transplants) than colonies in between spawning seasons. For species with seasonal broadcast spawning, it may be wise to avoid transplantation around the time of spawning.

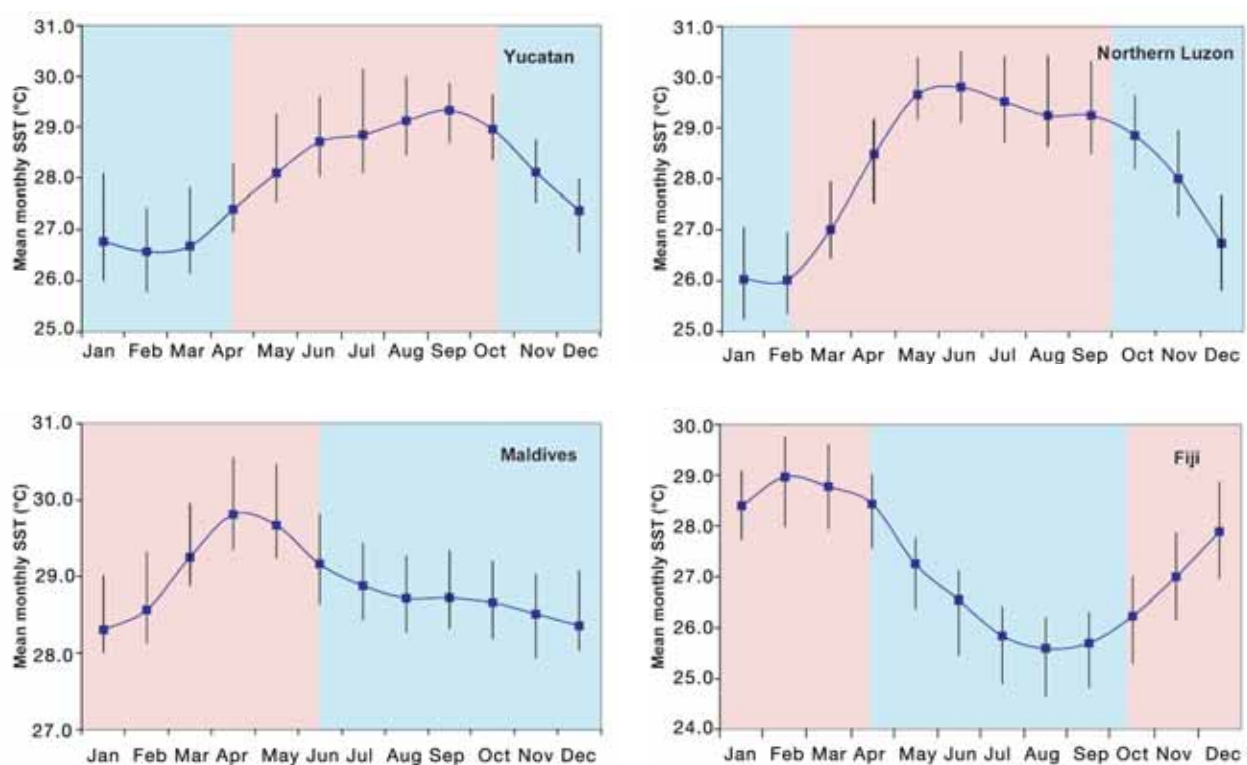


Figure 8. Comparisons of the average mean monthly sea surface temperatures (SST) for the east Yucatan coast of Belize and Mexico, Northern Luzon (Philippines), Maldives and Fiji based on the UK Met Office Hadley Centre's global sea ice and SST (HadISST1.1) data set from 1980-2005. The error bars show the range of the mean monthly SST over the period. Potentially poor times for transplantation in terms of SST are indicated in pale red. Note how SSTs in southern hemisphere Fiji are more or less the mirror image of those in the northern hemisphere Philippines and the relatively small seasonal change in SST in the equatorial Maldives compared to the other sites.

In a few parts of the world near the northern and southern limits of the distribution of coral reefs or in areas with seasonal cold-water upwelling, corals may also be stressed by winter cooling. It is unclear whether the coldest months should also be avoided.

Good Practice Checklist

- ✓ Transplant when corals are likely to be least stressed (i.e. a few months before or after the annual peak in sea temperatures; and not around main spawning time for seasonal spawners).
- ✓ Minimize exposure of coral transplants to air, sun and heat.
- ✓ If transplants are held in closed containers for significant periods (approaching 1 hour or more), then exchange water with fresh seawater at least every hour.
- ✓ Minimize handling (wear gloves).
- ✓ Beware when transplants start to produce a lot of mucus. This is a key sign that they are stressed.

3.9 Monitoring and maintenance

Unfortunately, because of the widespread lack of systematic monitoring of restoration activities we often do not know why there have been apparent successes or failures. Were failures due to chance external events or were they due to innate flaws in the methods used for restoration? Often we just know that certain restoration activities were carried out, but have no idea whether these worked or not. Sadly, without careful monitoring we learn little from either past mistakes or past good-practice. Restoration should not be considered as a one-off event but an ongoing process which will benefit from adaptive management over a period of several years.



Monitoring a transplant site at Mayotte. Note the yellow tags marking each transplanted colony.

If we are to learn from restoration interventions, we need to compare what we achieve with what would have happened anyway if nature had taken its course without any action from us. This means that we ought to leave alone some damaged patches of the same size as those we are trying to restore, and monitor what happens on these as well as what occurs on the patches on which we have carried out restoration work. This may not be appropriate for relatively small discrete damage such as that caused by a ship-grounding, but for community-based restoration projects where the degraded areas are usually far in excess of what can be restored, this should always be considered. Ideally, patches subjected to restoration actions should be

interspersed with comparable patches without interventions. Each restoration project is essentially an experiment and anything we can learn from each experiment will be useful to future restoration projects. Monitoring also gives you the information you need to carry out adaptive management of the project. The type of monitoring undertaken will depend on the precise goals of the restoration project but we offer some general advice below.

The more monitoring information you can get, the better in terms of learning from your restoration project and informing adaptive management. However, you need to be realistic; a little carefully collected data is more useful than a lot of poorly collected data. Scientific studies will often have full-time highly-trained personnel and significant funding to do monitoring. Community-based projects are likely to have much more limited resources. Monitoring normally focuses on the survival and growth of coral transplants or other transplanted organisms. In academic studies, the growth and survival of individual coral transplants may be followed through time but this is both very time-consuming and quite difficult to achieve. A more realistic monitoring goal may be to follow how the area of live coral cover (expressed as a percentage of the restored site's area) changes through time. This can be done using line intercept transect or quadrat methods (English *et al.*, 1997).

In addition, some attempt should be made to monitor changes in biodiversity at the restoration site. Corals, fish and other conspicuous or economically important and easily recognised species may be monitored. Identification to species can be difficult (especially for some corals!) and where this is the case, species groups, growth forms or functional groups can be used. The better the taxonomic resolution the more useful the data, but the more likely that people doing the monitoring will mix up similar species or disagree about identifications. It is better to have good reliable data at the genus or family level than unreliable data at the species level. Abundances of different taxa in the chosen groups can be monitored over time to see whether

the system is becoming more diverse and in particular whether it is becoming more like the reference healthy reef ecosystem chosen as a goal (see section 1.3.1).

As well as systematic monitoring of the kind discussed above, a simple check on the status of the restoration site by a snorkeler or diver every few weeks can be very useful. Unfortunately, a lot can happen between 3, 6 or 12 monthly systematic monitoring visits, including mass mortality of transplants from disturbances (e.g. storms, terrestrial run-off, predators, various events causing coral bleaching). Brief checks of a site every 2-4 weeks should allow such events to be pin-pointed so that we can learn why corals have died and perhaps take some remedial action.

Maintenance – Given the expense and effort involved in any reef restoration project, it is sensible to attempt to maximize survivorship of transplants. Systematic monitoring may occur at intervals of several months, but it is beneficial to check transplants more frequently for predation, algal overgrowth or detachment and take remedial action, if

necessary. Some echinoderms (e.g. the Crown-of-thorns starfish *Acanthaster planci*), gastropod molluscs (e.g. *Coralliophila*, *Drupella*, *Phestilla*), and fish feed on live corals and there is some anecdotal evidence that transplants (particularly if stressed) may actually attract some predators (e.g., the cushion star, *Culcita*). There is little one can easily do about mobile fish grazers (many of which are on balance beneficial because they also have a key role in grazing down competing algae and creating space for invertebrate larval settlement) but the slower-moving starfish, cushion star and gastropod predators can be removed from the vicinity of transplants and deposited well away from the restoration sites. This routine husbandry can extend to removing excess algae (e.g. with a wire brush) that appears to be threatening transplants and reattaching any detached transplants. If there is excessive algal growth then there may be other management measures which need to be considered as well. If there is a significant outbreak of Crown-of-thorns then more drastic measures may be required.



Crown-of-thorns starfish feeding on Acropora.



Predation of Porites rus by the nudibranch Phestilla. Note the egg masses laid by the seaslug.



The predatory snail Coralliophila on a Porites rus colony.



Grazing scar left by Coralliophila.



The cushion star Culcita which is known to feed on corals.



The mouth area of Culcita.

Good Practice Checklist

- ✓ Consider restoration not as a one-off event but as ongoing process over a time-scale of years which is likely to need adaptive management.
- ✓ Monitoring of restoration projects is essential if we are to learn from past mistakes and past good-practice. Without it, you can evaluate neither the success nor cost-effectiveness of restoration, nor carry out adaptive management if needed.
- ✓ Setting up and monitoring of a few comparable “control” areas where no active restoration has been attempted is recommended. These provide a clear baseline against which you can evaluate the cost-effectiveness of your restoration interventions.
- ✓ Consider how much monitoring can be feasibly undertaken (both detail and frequency) but be realistic. Better a little carefully and regularly collected data than a lot of poorly and irregularly collected data.
- ✓ Routine maintenance visits to the restoration site are recommended. They are likely to be very cost-effective given the expense of active restoration and could prevent wholesale loss of transplants to predators.

4. What does reef restoration cost?

It is very difficult to find information on the true costs of restoration. In the rare instances where detailed cost information is available, this usually details the costs of carrying out restoration activities rather than that of achieving restoration goals. Carrying out an active restoration intervention, such as transplanting x numbers of corals to a reef, is not the same as successfully restoring an area of reef. Often it is unclear what the outcomes of the restoration activities have been since monitoring has not been undertaken for a sufficient period, if at all. Knowing the relative costs of different approaches to restoration used in different projects is useful but it is often very difficult to compare such costs in a meaningful way. Further, the costs need to be evaluated in the context of the restoration benefits generated. For active biological restoration, the benefits are presumably the improvements in target indicators achieved over and above what would have happened if natural recovery had not been assisted. Thus we include the recommendation in section 1.3.1 that where feasible some “control” sites be set up where no active restoration is carried out.

For the most frequent active restoration activity, that is, trying to restore corals to reefs, a cost-effectiveness

endpoint that could be readily compared between projects and different methodologies is the cost per coral colony surviving to maturity. Deriving such a cost is more difficult. Inputs may include consumables, equipment and labour and some costs may relate to one-off set-up expenses (which can benefit from economies of scale) whereas others may be running costs, which are proportional to throughput. In theory the various component costs could be reduced to \$ values and then adjusted for purchasing power parity between different countries. Working out a standard way of assessing cost-effectiveness is clearly a priority if low-cost methods are to be promoted and disseminated.

Having said this, we can give some guidance on likely costs of restoration activities. These need to be divided between restoration projects involving some physical restoration and pure biological restoration projects. Data from ship-grounding restoration costs in the Caribbean, which involved physical restoration of the sites, suggest costs of US\$2.0 million – 6.5 million per hectare. Data from low-cost active biological restoration projects in Tanzania, Fiji and Philippines suggest costs ranging from US\$2,000–13,000 per hectare, whereas a study in Australia suggested that transplantation to replace 10% of the target density of corals

would cost at least US\$40,000 per hectare. The two lowest estimates of restoration costs related to community-based projects. The first involved transplanting 2 corals per m² on reefs which already had about 20% coral cover (\$2,000/ha), and the second involved increasing coral cover on patches of reef from 10% to 20% by means of transplantation (\$4,590/ha). These are useful first steps towards restoration but clearly are optimistic estimates of the true costs of restoring reefs. The remaining estimates start at \$13,000/ha. These estimates can be compared to average global estimates of the total value of coral reef goods and services of US\$6,075 per hectare per year, and of potential sustainable economic benefits for Philippines reefs of US\$320–1,130 per hectare per year.

For small scale community-based restoration projects, this suggests that at least several years income stream from restored reef areas are likely to be needed to cover the costs. Any improvement in cost-effectiveness of biological restoration techniques can make a big difference to the economics. Clearly the same applies for physical restoration.

Comparison with estimates of restoration costs for other ecosystems, such as seagrasses, mangroves, saltmarshes, sand dunes and lagoons, is slightly reassuring. Costs of reef restoration tend to be higher at the low-cost end but not significantly so. It is only when ship-groundings and their physical restoration are considered that we find costs that are an order of magnitude greater than the upper estimates for other coastal ecosystems.

The realisation of the large costs of restoration per hectare focuses attention on an area of research that requires urgent attention but has been largely neglected; that is, how to scale up restoration to assist recovery of large areas (in the order of km²). Should relatively small “source” patches be actively restored in the hope that their recovery will kick-start recovery in larger down-current “sink” areas? If discrete patches are actively restored, will benefits spill over into surrounding areas? How can improved management, in concert with small-scale active restoration, generate larger scale benefits? At present we have little idea of the answers to these (and many other) questions. To tackle the huge mismatch in scale between what restoration can potentially achieve and reef degradation, these gaps in our knowledge need to be filled urgently. A combination of studies of local ecological processes, larger scale connectivity and oceanographic processes, and modelling offers a way forward.



Beautiful reef in the Similan Islands, Thailand.

Message Board



Restoration of ship-grounding sites that required major physical restoration of coral reefs indicate costs in the order of US\$2 million – 6.5 million per hectare.



Low-cost transplantation appears to cost about US\$2,000–13,000 per hectare. With more ambitious goals this rises to about \$40,000 per hectare.



A rough global estimate of the average total annual value of coral reef goods and services is US\$6,075 per hectare.



Annual potential sustainable economic benefits for Philippines reefs are estimated at US\$320–1,130 per hectare.



Economic comparison of benefits and costs of coral reef restoration using current methods suggests that the economic case for active restoration is not clear-cut. Improved cost-effectiveness of methods for restoration is essential if restoration is to be applied more widely.

5: Case studies

In the following section, five case-studies are presented by Sandrine Job of the Coral Reef Initiative for the South Pacific (CRISP) project's reef restoration programme. The project sites range from the western Indian Ocean to French Polynesia and illustrate some of the issues discussed in earlier sections. Two of the projects involved transplantation of corals in mitigation for developments that would destroy areas of reef, two sought to enhance coral reef habitat that had failed to recover from natural disturbances (cyclone and mass-bleaching respectively) – possibly compounded by anthropogenic impacts, and one aimed to reduce erosion and restore a sand mining site close to a tourist resort. For each case-study, the location, objective and methods used are briefly outlined and lessons learnt from the outcomes are presented, as perceived by those involved in the projects. In addition, information on the resources (staff, equipment, etc.) required for each project is summarised with actual budgets being presented where available. Staff costs vary greatly from place to place, therefore numbers of personnel and fieldwork duration is detailed so that the numbers of person-days required to perform various tasks can be calculated by those interested. Based on the information and guidance in the previous sections, you are encouraged to examine ways in which these projects could have been improved upon. A few comments in square brackets have been added to link some of the lessons learnt to appropriate sections of the Guidelines.

A key feature of the projects reported below has been the recognition of the need for monitoring (with from about 6 months to five years of post-transplantation monitoring scheduled in different projects). Without this monitoring, no lessons would have been learnt. In most of the case-studies, initial survival of transplants was good, but in a few, high mortalities occurred after about one year, emphasising the need to monitor for a minimum of at least a year and preferably for a time period which matches likely recovery (i.e. at least 5 years). In most of the case-studies, there was careful selection of transplant sites to ensure that these were as similar as possible, in terms of their environment, to the source sites. Where this precept was not carefully followed, high mortality of transplants eventually occurred. Important areas for further consideration are 1) the extent to which projects were considered within a broader coastal zone management and planning context, 2) the dominant focus in terms of monitoring on coral transplant survival and growth, 3) the need for clearer, more detailed, restoration goals, and 4) the need for *a priori* success criteria linked to these, against which progress towards recovery can be objectively evaluated [see section 1.3.1].

For further particulars of these case-studies please contact:

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View of the Bora Bora "coral garden" with transplants on artificial reefs in shallow water.

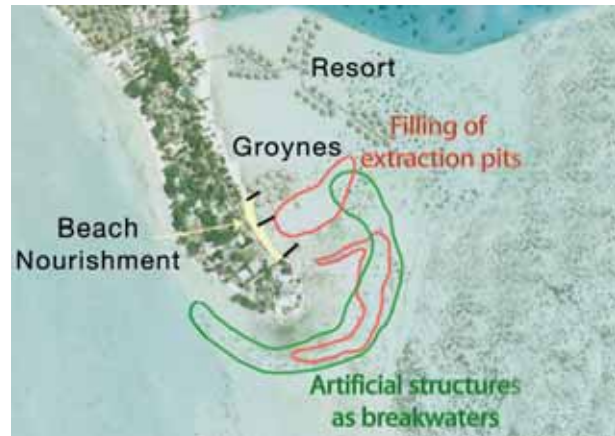
Case study 1: Restoration of a reef damaged by sand mining operations and creation of a coral garden, French Polynesia.

Location

Matira Point, Bora Bora, French Polynesia
(July 1996 – June 2000)

Objective

As a result of dredging operations to extract coral sand for construction works, the sand movement close to the coastline around Matira Point was altered leading to coastal erosion. In an attempt to rectify the problem, a two-step strategy was employed, using both physical and biological restoration techniques.



Methods

Physical restoration:

- Extraction pits created by dredging operations were refilled with 10,000 m³ of sand originating from the inner reef slope, to enable sediment transit to the coastline.
- Three 20 m long groynes² were installed and beach nourishment implemented in between the groynes. In addition the shoreline was remodelled and vegetation replanted.
- 125 artificial concrete structures (weighing between 1.6 and 17 tonnes) were deployed on the sandy shallow reef flat around Matira Point to act as breakwaters to protect the coast from lagoonal swells.

Biological restoration:

- A 7,200 m² "coral garden" was created by transplanting 311 coral colonies to 11 artificial structures and 200 large branching (*Acropora* spp.) and massive (*Porites* spp.) colonies to surrounding sand patches.
- Coral collection: donor sites (which included the extraction pits) were selected on the basis of (a) having similar characteristics to the transplantation site with respect to depth, water motion, exposure to waves and coral diversity, (b) proximity, and (c) accessibility. The 311 coral colonies were collected from a mix of different species and growth forms, to recreate the aesthetics of a natural reef.

- Corals were transported immersed in containers of seawater.
- Transplants were attached to the artificial structures using epoxy glue and quick drying cement.

Monitoring surveys were carried out at 1, 3, 6, 9, 13, 28 and 32 months after transplantation. Monitoring included:

- Survival and growth rates of coral transplants.
- Health assessment (observation of necroses on the living tissue, bleaching, predation on transplanted corals, etc.).
- Natural colonisation of the artificial structures by fish, algae, coral recruits and macro-invertebrates.



Transplants on artificial reefs in the shallow water of the Bora Bora "coral garden".

² Groins (U.S.)

Lessons learnt

- Overall survival rate of coral transplants after one year was 95% suggesting that selection of donor sites on the basis of their similarity to the transplantation site was effective.
- Mortality rate for sub-massive *Porites rus* was high, mostly as a result of smothering by sand. Colonies should have been placed higher above the seabed to reduce their exposure to resuspended sediment.
- Aesthetics and functionality was carefully considered when manufacturing the 11 artificial structures, with a view to creating something as natural-looking as possible. This involved simulating natural reefs in terms of shape, texture (made rough by incorporation of coral rubble and sand as aggregate in the concrete), and colour (colouring was added to the cement to obtain a substrate colour similar to that of natural reefs). Providing shelter is a critical function of reefs and thus artificial structures included holes, cracks, and void spaces as refuges for fish and invertebrates. Fish abundance and diversity was significantly higher after one year, with 30–50% of the fish being juveniles.
- Use of quick drying cement and epoxy glue to attach transplants was highly successful with no transplants becoming detached during the first year. Self-attachment of colonies at their bases via tissue expansion onto the substratum was widespread, providing secure long-term attachment and suggesting limited or only short-term adverse effects of either the cement or epoxy on the transplant bases.
- Owing to lack of awareness there was some localised destruction (2%) of coral transplants due to boat traffic and tourists visiting the coral garden. To avoid such issues, it is recommended that restoration projects be conducted in association with awareness initiatives with potential users.
- A mass mortality of the corals was recorded due to a bleaching event in January 2002 that affected both transplanted and natural corals on the shallow reef flat, whereas corals on the outer reef slope survived well. The risk of such mortality, particularly when transplanting in shallow lagoon areas with limited water exchange, needs to be considered in planning projects. Such risks should be carefully considered when selecting both transplant sites and the species to be transplanted.
- During the physical restoration of the excavation pits, large amounts of sand were deposited in the coral garden area. This had to be removed to avoid smothering and mortality of transplants. Physical and biological restoration activities should be carefully scheduled so that such impacts are avoided.

Contractor

French Agency of Development (AFD), Government of French Polynesia and National Scientific Program "Recreate Nature".

Costs and effort required

Physical restoration

Activity	# days	# people	Budget (US\$)
Construction of groynes	6	4	12,000
Filling of extraction pits and beach nourishment	75	?	445,000
Coastline profiling and vegetation planting activities	180	?	734,000
Construction and deployment of artificial structures as breakwaters	200	?	410,000
Total physical restoration			1,601,000

? = Number of people employed by external sub-contractors to carry out these aspects of physical restoration unknown.

Biological restoration

Activity	# days	# people	Budget (US\$)
Coral collection (mainly from extraction pits) and transplantation to artificial structures	19	3	40,000
Collection and transportation of 200 large massive and branching colonies	40	6	90,000
Creation of 7,200 m ² "coral garden", including 11 artificial structures	30	6	140,000
Coral garden monitoring (over one year)	21	3	80,000
Total biological restoration			350,000

Resources needed for creation of 7,200 m² coral garden with 11 artificial structures and >500 transplanted colonies:

Team of 3 to 6 people: 2 marine biologists + 1 boat driver + 3 field assistants (for part of the work); 1 boat; scuba-diving equipment; US\$350,000 or about \$50/m².

Reference

Salvat, B., Chancerelle, Y., Schrimm, M., Morancy, R., Porcher, M. and Aubanel, A. (2002). Restauration d'une zone corallienne dégradée et implantation d'un jardin corallien. *Rev. Ecol. Supp.* 9: 81-96.

Case study 2: Restoration of fringing reef impacted by a tropical cyclone, La Réunion

Location

Saint Leu, La Réunion Island (1997-2000).

Objective

During cyclone Firinga in 1989, many portions of the fringing reef of La Réunion Island were devastated, leading to 99% coral mortality in some places, particularly on the fringing reef of Saint Leu. The purpose of this study was to recreate habitat for fish to help replenish the fish population in the lagoon of La Réunion.



Methods

The project was conducted in 2 phases:

Phase 1 (June 1997-June 1999). Transplantation of branching corals (*Acropora muricata*, the dominant species on La Réunion fringing reefs) associated with larvae of the damselfish *Dascyllus aruanus* previously bred in tanks. Transplant survival and growth were assessed, as well as fish populations on control and experimental sites. Monitoring lasted one year.

Phase 2 (June 1999-June 2000). Deployment of locally made ReefBall-like artificial reefs and transplantation of coral fragments (5 cm long) onto them using quick-setting cement. Monitoring was conducted to assess the survival and growth of transplants and the natural colonisation of artificial structures by fish and invertebrates. Monitoring lasted 5 months.

Lessons learnt

- Quick-setting underwater cement can be used effectively to transplant coral fragments onto hard structures. There was 100% survival and no detachment after 2 months.
- Restoration activities should be located within areas where human activities can be managed. In this project, as a result of La Réunion lagoon being heavily used by fishermen and tourists, many transplanted corals (50% of transplants from phase 1 and 30% of transplants from phase 2) died from being trampled on.
- During phase 1, fish transplantation was not deemed to be successful as 1 month after their release, only 20% of original numbers were found inside the transplanted colonies. After one year, numbers were 30% of the original, suggesting

recruitment to the colonies. Thus, during phase 2, no fish transplantation was attempted but natural recruitment to the artificial structures was monitored. Juvenile fish were observed to recruit to branching coral colonies within a week.

- During phase 2, 5 months after their transplantation, approximately 50% of the transplanted corals were found to have died from either being smothered by filamentous algae or from grazing by corallivores. Two possible options to reduce mortalities would have been some husbandry (maintenance) of the transplants (e.g. removal of algae), or use of larger (e.g. >10 cm) fragments which might have been better able to survive partial grazing and out-compete algae.

Contractor

University of La Réunion and National Museum of Natural History (Paris).

Costs

The team was composed of an external consultant, 4 scientists, 8 students, 2 technicians and guards of La Réunion lagoon; the overall budget was US\$40,000; of this \$20,000 was spent on materials, including \$9,400 to construct and deploy the 6

artificial structures; \$20,000 was spent on salaries of external-consultant and technicians; scientist salaries were covered by the University and the students were unpaid volunteers.

Reference

Chabanet, P. and Naim, O. (2001). Restauration mixte d'un récif détruit par le passage d'un cyclone. Programme de recherche «Recréer la nature».

Case study 3: Transplantation of corals from the Longoni harbour, Mayotte

Location

Mayotte Island (Indian Ocean), April 2004.

Objective

This mitigation project aimed to compensate for degradation caused by the reclamation of a portion of the fringing reef in order to extend the main harbour. The objectives were: (1) the rescue of some 600 threatened coral colonies, and (2) a pilot scientific experiment on coral transplantation in the lagoon of Mayotte.



Methods

- Selection of 3 transplantation sites:
 - A fringing reef with very similar environmental conditions to the threatened site (Longoni Balise).
 - A patch reef in the lagoon further from the coast (Vaucluse).
 - A reef site located close to a pass through the barrier reef (Surprise).
- 600 colonies were selected from a range of genera and growth forms which were representative of the threatened fringing reef community.
- Small and medium sized corals were transported in large plastic containers filled with seawater but large colonies were placed in a submerged cage which was towed by boat. Time for transport to transplantation sites ranged from 30 minutes to 2 hours.
- Transplants were attached with cement to natural coral rock or to concrete slabs (50 cm x 50 cm x 10 cm).
- Transplants were marked with plastic cable-tie tags either nailed to the natural rock or fixed to the colony itself.
- Monitoring surveys were conducted 1 month after transplantation and thereafter every 3 months for one year. Monitoring included:
 - Survival rates.
 - Growth rates (greatest and least diameters measured).
 - Amount of partial mortality (% of the colony surface dead recorded).
 - Colonisation of the transplantation site by fish and invertebrates (assessed using 3 replicate belt transects per site of 50 m x 4 m and 20 m x 2 m respectively).



Transport of small to medium sized transplants submerged in seawater in large plastic containers.



Transport of large transplants in a submerged cage towed by a boat.

Lessons learnt

- The operation was broadly successful with an overall survival rate of 80% after 1 year, which implies that the methodology for collecting, transporting and attaching transplants was appropriate.
- The choice of the transplantation site was important: the site with environmental conditions most similar to the threatened source site, had highest survival. Survival rates were 90%, 65% and 80% respectively on the fringing reef (most similar), patch reef further from the coast, and reef located close to a pass through the barrier reef.
- Observation of partial mortality is useful to assess more precisely the behaviour of transplanted colonies through time. It allows one to determine whether the surviving transplants' health is declining or improving with time.
- Over half the transplanted colonies showed partial necrosis of tissue at one month but this did not increase subsequently. This suggests initial stress within the first month, which may be related to adaptation to the new environment and/or reaction to transplantation handling. It is therefore critical to minimize stress during transplantation.
- Regular cement was reasonably effective in attaching colonies. Even in environments with moderate water motion, less than 5% of transplanted colonies became detached.
- The flat concrete slabs placed on sand onto which some corals were attached were a failure: almost all transplants on these died from being smothered by sand. In sandy environments, it is essential that transplants are placed above the most significant sand movement, particularly in places where wave action and currents are resuspending sand particles and scouring surfaces.
- A few colonies which were spaced too close overgrew one another. Transplants should be placed sufficiently far apart from each other to avoid competition for space.
- Although branching colonies had significantly higher growth rates than massive forms, the latter appeared more resistant to stress and regenerated more quickly from tissue necrosis or partial mortality.
- Although tagging colonies was useful for monitoring, it was time consuming and required 6 person-hours to tag 100 colonies. The plastic tags required frequent checking and replacement about every 6 months. Stainless steel nails were found to be effective in fixing the tags to the substratum.
- About 5% of the transplants were damaged by fishermen with anchors, nets or rocks (in Mayotte, catching fish by throwing rocks in the water to stun them is a traditional technique of fishing). To enhance the survival rate, it is recommended that transplantation be conducted in marine protected areas, where human impacts can be better controlled.

Contractor

Direction de l'Équipement de Mayotte.

Resources required to transplant 600 colonies

Team of 3 divers (marine biologists) + 1 boat driver + 1 field assistant (preparing cement on the surface and helping with logistics); 2 boats (one speed boat to carry the team and small/medium size corals; one slow boat to pull the underwater cage carrying large coral colonies); scuba-diving equipment; fieldwork period of 25 days (site selection, coral collection, transplantation and initial monitoring); salary costs: US\$60,000 (including \$20,000 salary for external consultant); materials, transport and subsistence costs for transplantation work: \$25,000; costs of one year of monitoring (including salary of an external consultant who conducted the surveys): \$12,000.

Reference

Morancy, R., Job, S. and Thomassin, B. (2005). Transplantation des coraux du port de Longoni et suivi de l'opération. Rapport technique. Carex Environnement – GINGER.



A tagged Seriatopora transplant at Mayotte, with plastic tag nailed to the reef.



Sand encroachment onto concrete slabs used to provide stable substrate for coral attachment.

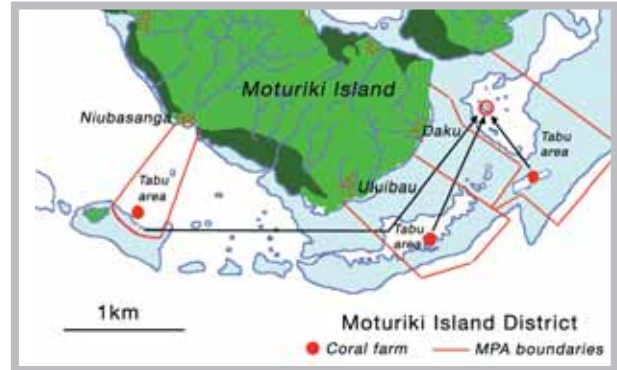
Case study 4: Restoration of reef degraded by bleaching events, Fiji

Location

Moturiki Island, Fiji (August 2005)

Objective

This was a community-based restoration project, using low-cost and low-tech techniques. The purpose was to restore a portion of reef degraded by bleaching events in 2000 and 2002. The specific goal of this work is the restoration of fisheries resources, and was more related to food security and community prosperity than to a biodiversity-driven rationale.



Methods

- Corals were sourced from *inter alia*: colonies threatened by sand smothering (where colonies or fragments had become detached and fallen onto sand), colonies very close to the sea surface that showed damage from exposure at low tide, fragments from colonies damaged by triggerfish, anchors, nets, etc., and farmed corals. (Four coral farms were established in Moturiki waters, owned and maintained by local communities.)
- Transplants were transported in the bottom of a boat, exposed to air, but with seawater continuously sprinkled over them during transfer. Duration of transport was approximately 30 minutes.
- Transplants were attached using 3 different methods:
 - (1) "Plug-in method": inserting coral fragments into small crevices and holes in the coral rock hard substrate taking care to find the right sized hole for each fragment, ensuring that they would be firmly held in place and thus attach faster;
 - (2) "Place-on method": larger colonies (usually *Acropora* species) were placed directly on rubble or on sand patches (the site was sheltered with little water movement) and subsequently stabilized by stones from the immediate vicinity placed around the base;
 - (3) "Cement method": farmed corals were attached to dead coral heads and rocks, using regular cement.
- Three sites covering a total area of about 2150 m² were transplanted and compared to three comparable control sites of similar area during monitoring.
- Monitoring was carried out at 1, 3, 6 and 9 months after transplantation. (Monitoring scheduled for 12, 15 and 18 months was abandoned following mass-bleaching and mortality.) Monitoring included:
 - Transplant survival rate.
 - Assessment of change in % coral cover using five 20-25 m line-intercept transects per site.
 - Assessment of fish and benthic macro-invertebrate populations using visual censuses and belt-transects respectively.



Transporting large branching corals to transplant sites in Fiji. Corals are splashed with seawater from a bucket.



Coral farm at Moturiki Island in Fiji.

Lessons learnt

Method of transportation: despite the relatively harsh conditions in which corals were transported, entangled and stacked on top of one other, exposed to air for 30-60 minutes, over 95% of transplants were surviving well at 6 months with branching species showing growth. Where time and budgets are limited, these simple methods can be successful. [See also Harriott and Fisk (1995). However, as recommended in section 3.8, it is advisable to minimise stress and where possible to keep corals shaded from direct sunlight and immersed in seawater when transporting.]

Coral planting method

Plug-in method

- The plug-in method was the easiest and quickest of the methods tested; little maintenance was needed, and the method appeared appropriate for restoring areas of coral reef dominated by dead colonies/coral rock into which branches could be “plugged”. However, the method is restricted to small branching corals.
- It is important to choose appropriately-sized holes for the transplanted fragments and ensure that living tissue is in direct contact with the substrate to maximize subsequent self-attachment. If available holes were too large, it was found that fragments could be successfully wedged in place with a piece of coral rubble. 60% self-attachment was recorded 6 months after transplantation.
- Based on case-study 3, spacing of coral transplants took into consideration potential competition between colonies and scarcity of resources, with coral colonies being planted at least 50 cm apart.

Placed-on method

- This method is only appropriate for low-energy environments [see section 3.4] in which the weight of the branching colony (or large fragment) is sufficient to keep the transplant stable until it can self-attach or its base can settle into sand.
- Where possible transplants should be positioned behind larger boulders and in depressions where they will be sheltered from current and wave action until they can self-attach. However, self-attachment took longer than the previous method with only 35% of transplants firmly self-

attached after 6 months. [This method carries the highest risk to transplant survival, and, if attempted, the risk needs to be carefully considered.]

- 30-40 cm rocks wedged around the bases of the transplanted coral colonies were found effective at giving them something to attach to even when on sandy substrata, increasing their overall weight and stability, and providing added insurance against potential storms.

Cement method

- This method was found effective for corals that could not be easily plugged into holes and that were too small and light to be placed on the substratum directly without attachment (small to medium sized rounded colonies, massive colonies, and farmed corals grown on cement discs). 95% of transplants showed self-attachment by tissue expansion over the cement within 6 months.
- Cement needs to be carefully contained in plastic bags and restricted to attachment site, with great care being taken not to damage adjacent living organisms (other sponges, molluscs, urchins, etc.).

Bleaching event

Due to a bleaching event that occurred 9 months after transplantation, two-thirds of the transplants died and one-third partially bleached. Natural coral communities on neighbouring patch reefs suffered considerably less. From this, a few lessons can be learnt:

- The donor and transplant sites should be as similar as possible with respect to environmental conditions (wave, current, depth, temperature, light, and disturbance regimes). In the study, corals were sourced from the outer lagoon and transplanted to an inner lagoon reef area. Although surviving well initially, they seemed poorly adapted to the more extreme conditions experienced in the inner lagoon. Transplants should be adapted to the prevailing environmental conditions at the restoration site [see Good Practice Checklists in section 3.2 and section 3.5].
- Monitoring should be undertaken for at least one full year to take account of seasonal changes in the environment at the transplant site. The critical question is whether the transplants can survive during the worst conditions at the site during each year.

Contractor

French Agency of Development (AFD) under Coral Reef Initiative for the South Pacific (CRISP) program.

Resources required for coral transplantation to

approximately 2000 m² of reef, increasing the coral cover by 10-15%: Team of 2 scientists + 2 field assistants + 1 boat driver; 1 boat; free-diving skills (no scuba used); fieldwork period of 10 days (60% of the time allocated for restoration

activities, 40% of the time allocated for scientific input (site selection, and baseline monitoring); material costs US\$1,300; salary costs \$10,100.

Reference

Job, S., Bowden-Kerby, A., Fisk, D., Khan, Z. and Nainoca, F. (2006). Progress report on restoration work and monitoring. Moturiki Island, Fiji. Technical report. Coral Reef Initiative for the South Pacific.

Case study 5: Transplantation of corals from the Goro Nickel harbour, New Caledonia

Location

Prony Bay, New Caledonia (December 2005 – January 2006).

Objective

This project was a mitigation measure imposed on a private nickel extraction firm, Goro Nickel, in relation to the construction of a harbour in a reef area. The objective of the project was twofold: to rescue coral colonies threatened by reclamation operations and to use them to restore 2000 m² of damaged reef. Their survival and growth is being assessed over 5 years.



Methods

- Selection of 3 fringing reef transplantation sites (one of 1000 m² and two of 500 m²) where environmental conditions were similar to the threatened source site.
- Collection of approximately 2000 coral colonies, from a diverse range of genera and growth forms, which were representative of the threatened reef area.



Sprinkling fresh seawater on transplants during transportation from donor site.



Rescued corals awaiting transplantation.

- Corals were transported exposed to air in plastic containers but regularly sprinkled with fresh seawater. Transport time was 20 to 30 minutes.
- Transplants were attached with underwater cement to natural coral rock.
- Monitoring surveys were conducted 1 month after transplantation and thereafter scheduled for about every 6 months; they will continue for 5 years. Monitoring includes:
 - Survival rates.
 - Assessment of coral cover through time using the line-intercept transect method with a 20 m long transect (10 replicates for the 1000 m² site, 5 replicates for the two 500 m² sites).
 - Colonisation of the transplantation site by fish and invertebrates (assessed using belt transects of 50 m x 4 m and 20 m x 2 m respectively) using 10 replicates for the 1000 m² site and 5 replicates for the two 500 m² sites.

Lessons learnt

- Overall survival rate after 9 months was almost 90% suggesting that selecting transplantation sites on the basis of similar depth, pH, salinity, turbidity, temperature and geomorphology was effective.
- It is possible to transport corals in air, at least for up to 30 minutes, provided they are sprinkled with clean seawater. Transplants did not show any obvious sign of stress (e.g. excessive mucus production or subsequent mortality) from being exposed to air for 30 minutes. [See also Harriott and Fisk (1995). However, as recommended in section 3.8, it is advisable to minimise stress and where possible to keep corals shaded from direct sunlight and immersed in seawater when transporting.]
- In one of the sites, half of the transplants suffered predation by the Crown-of-thorns starfish (*Acanthaster planci*) and the cushion star (*Culcita*) – 30% died, 20% suffered partial mortality. While monitoring, it is crucial to remove these predators to enhance survival of transplants.

- Underwater cement was appropriate to attach transplants with less than 5% of transplants detached or loose at the end of transplantation activities. Moreover, half of the colonies had grown onto their cement bases within 5 months.
- Transplanted branching *Acropora* colonies were colonised by many juvenile fish within a few months suggesting a useful role in fish recruitment.

Contractor: Goro Nickel, a nickel extraction firm.

Resources required to transplant 2000 colonies to 3 sites totalling 2000 m² of reef: Team of 3 divers (marine biologists) + 1 field assistant (preparing cement on the surface and helping with logistics); 1 boat; scuba-diving equipment; fieldwork period of 25 days (1/3 for fieldwork preparation, logistics and local transport; 2/3 for restoration activities – site selection, collection, transplantation and baseline monitoring); cost of materials: US\$17,000; salary costs: US\$45,000.

Reference: Job, S. (2006). Transplantation des coraux du port de Goro Nickel et suivi de l'opération. Rapport technique. SOPRONER – GINGER.

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 Tadashi Kimura: Back cover (*Porites* overturned by 2004 tsunami);
 Gideon Levy: p.15; Figure 7 (collection from reef, transplantation to degraded reef);
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The initiative for the protection and management of the coral reefs of the South Pacific (CRISP), championed by France, aims to develop a vision for the future for these unique ecosystems and the peoples who depend on them for their livelihood. It seeks to put in place strategies and projects to preserve the biodiversity of the reefs and for the future development of the economic and environmental services that they offer both locally and globally. Amongst many others, this programme addresses the issue of improving the skills of local communities in restoring and managing coral ecosystems through its component 2B, based on the joint venture between a French operator SPI-INFRA and a Pacific NGO FSPI (Foundation for the South Pacific People International).

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The research on coral reef restoration being carried out by the CRTR program in Philippines, Palau and Mexico is complemented by the community-based projects on restoring and managing coral ecosystems being carried out as part of the CRISP program in the South Pacific. Further, members of the CRISP team have wide experience in reef restoration projects across the Indo-Pacific from Mayotte to French Polynesia. The collaboration of CRTR and CRISP in section 5 on "learning lessons from restoration projects" adds valuable case-study experiences to the broad guidelines. This cooperation between the two programs will be extended in the preparation of the more detailed *Reef Restoration Manual* planned for late 2008. This practical manual will synthesise not only the reef restoration research outputs of the CRTR and CRISP programs but also those of the European Commission funded REEFRES project (entitled "Developing ubiquitous practices for restoration of Indo-Pacific reefs") as well as summarising previous knowledge.