Reef Restoration for Coastal Defense: A Review
by

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Executive Summary

Coral reefs provide numerous economic benefits; the highest valued of these are tourism, fisheries, and coastal defense. Worldwide, reefs provide US $9 billion yearly in coastal defense provisions. Coral mortality that results in loss of structure decreases the coastal defense function provided by reefs. In low-lying areas, the loss of this function greatly increases the vulnerability of coastal areas to inundation from flooding; this risk is likely to become more serious as sea level continues to rise.

Active restoration of degraded reefs is a growing pursuit, but there has not been a review of coastal defense benefits provided by restoration projects. This study seeks to close this information gap through a literature review and global survey designed to gather information about reef restoration projects, particularly those designed for coastal defense benefits. The literature review identified costs and benefits of active restoration projects designed to deliver coastal defense. We also provide case studies describing restoration projects using different restoration techniques.

There are about a half dozen structural reef restoration approaches (e.g., restoration approaches that aim to rebuild habitat structures); these include Reef Balls; concrete structures such as mattresses and blocks; BioRock®; EcoReefs; and rock and rubble piling. There are thousands of structural reef restoration projects, but very few of these projects have clearly accessible information on size, cost, or measured benefits. The vast majority of these projects were designed to deliver coral recovery and fishery benefits, and these benefits are the primary benefits measured; many projects we report on had no systematic measurement of benefits other than a few coral and fish surveys lacking control treatments.

Reef Balls represent the most commonly used structures in reef restoration projects. The Reef Ball Foundation reports that more than half a million Reef Balls have been deployed in 4,000 projects in 70 countries. Many (likely most) of these projects were designed for fishery and tourism benefits. The Foundation specifically identifies the projects used for coastal defense purposes (at www.reefbeach.org); there are 10 of these projects in tropical waters (two additional projects were in temperate waters); five projects were at hotel beaches, and three included reef species plantings on the modules. Three of these projects provided enough information on costs and benefits to be included in our analysis.

Overall we could find only nine reef structural restoration projects that included sufficient details on costs, measured benefits, and methodology for our analysis. Of these only four projects provided any direct information on coastal defense benefits, usually in terms of widening of hotel beaches; severe storms usually negated these effects, as they often do for shoreline breakwaters. Costs ranged from US $10,560 for a small-scale pilot project to $1.8 million for the largest Reef Ball breakwater. Ship grounding restoration efforts have cost several millions of dollars, but these projects were not presented in this report as they did not explicitly address coastal defense.

Fifty-three restoration practitioners responded to our reef restoration survey. Most respondents had worked on greater than four restoration projects (55%). The vast majority of projects (90%) were designed for coral recovery benefits; this likely underestimates the actual percentage as we specifically focused on survey towards practitioners working on coastal defense projects. After coral recovery, the highest-prioritized benefits included in projects design were fish production and recreation (43% each). Coral recovery and fish production were the most commonly measured benefits (84% and 42%, respectively). Few respondents (20%) reported designing projects to deliver coastal defense benefits. Even when practitioners planned projects for coastal defense benefits, they rarely measured if these benefits were delivered. Only 10% of respondents reported measuring coastal defense benefits as compared to 84% for coral recovery.

Costs of projects ranged widely, with 60% projects costing US $100,000 or less. The greatest challenge to project success was lack of funding for maintenance, reported by 63% of respondents. The second greatest challenge was lack of effective management or political problems (44%). “Direct threats” such as sedimentation, bleaching, and water quality were of lesser concern than maintenance and political issues but still significant challenges (42%, 34%, and 32% of respectively).

Overall, it was surprising how few of the thousands of reef restoration projects were designed to deliver coastal protection benefits. Nonetheless we were able to identify the range of methods that have been used and to provide estimates of their costs. Consolidated rock and rubble piles are among lowest-cost methods for providing coastal defense; these methods have been used with success, although coastal defense benefits cannot be meaningfully assessed without data from more studies. Little quantitative information is available on coastal defense benefits provided by restoration projects because these benefits are rarely measured. Projects designed for beach protection were largely successful in reducing erosion under normal conditions, but not under wave conditions and storm surges associated with larger storms. There is evidence that reef restoration can be used for coastal defense, but clear and compelling field demonstrations are missing. This is a critical gap that needs to be filled.

Introduction

Coral reefs protect coasts from erosion and flooding by attenuating wave energy and sediment trapping; reefs are also sediment sources for adjacent beaches (Koch et al. 2009). As waves approach the shore, they break at the reef edge, creating wave set-up over the reef flat. Swell waves tend to break into groups of shorter waves, which are attenuated over the width of the reef flat due to frictional drag. Estimates of wave attenuation of coral reefs in the literature have ranged from 72% to 97% (references within Lugo-Fernandez et al. 1998). At a given site, wave attenuation is controlled by the amount of material obstructing waves, as well as the water depth over the reef; i.e. how much of the water column is occupied by reef structure, which is set by tidal level and height of the reef (Koch et al. 2009). Changes in tidally controlled water depth have been found to be a primary factor affecting wave attenuation in field studies, even at small tidal ranges. At low water, short-period waves tend to dominate across a reef flat, while longer periods dominate at higher water levels (Brander et al. 2004, Lugo-Fernandez et al. 1998). Bottom friction plays a critical role in dissipating wave energy; frictional drag over a reef flat may be ~10 times greater than drag over a sandy bottom, so coral structure at both the reef crest and across the reef flat play an important role in attenuating wave energy (Lugo-Fernandez et al. 1998). Gourlay's (1996b) semi-empirical model identified offshore wave height and period, in addition to depth over the reef top, as important determinants of wave energy reaching the shore. The model provides a basis for predicting whether a coast will be flooded during storm events under specific wave and tide predictions and reef characteristics (presented in Gourlay 1997).

When reef structure is lost, coastal defense provisions decrease due to 'pseudo-sea level rise,' as well as loss of frictional drag imparted by the three-dimensional reef structure, both of which reduce wave attenuation – this may occur instantly in the case of direct physical damage, such as a ship grounding, or more gradually, such as mass bleaching followed by unsuccessful recruitment. In low-lying areas, the loss of this function greatly increases the vulnerability of coastal areas to inundation from flooding. Sheppard et al. (2005) used Gourlay's (1997) model to determine changes in wave energy reaching the shoreline on 14 Seychelles fringing reefs from before the 1998 bleaching event to 2004, and for the next decade out. Surveys conducted in 2004 found negligible recovery, with >99% of reef flat corals dead, and a general rounding off of reef crests. Model results indicated that from pre-bleaching to 2004, more wave energy reached the shores, beaches experienced greater erosion, and there was more inundation during storms; changes for the decade following 2004 were predicted to be double that of the previous period (assuming no coral recovery).

Reefs are degraded worldwide (Reefs at Risk series, Souter 2008) due to overfishing, bleaching, biotic threats such as crown-of-thorns starfish outbreaks and disease, sedimentation and pollution, and destructive practices such as coral mining and blast fishing. Local and regional sources of reef degradation occur against a background of changing boundary conditions, particularly sea level rise. Global average sea level rose from 1950 to 2000 at a rate of 1.7-1.8mm/yr, and over the past 15 years satellite altimetry data has been used to calculate rates as high as 3.1-3.2mm/yr, though data is not sufficient to conclude definitively that sea level rise has indeed been accelerating

(references within Woodroffe and Murray-Wallace 2012). The IPCC Fourth Assessment Report predicts a sea level rise of 0.18-0.59 m from 1980-1999 to 2090-2099 with up to 0.17 m more due to uncertainties in ice dynamics, though these are likely to be underestimates (references within Rahmstorf 2010). It is important to note, however, the difficulty involved in predicting sea level rise at a given location, the rates of which can vary significantly from a constrained global average. This is mostly due to the complex nature of isostatic adjustments of land masses to the removal of glacial water and hydro-isostasy of oceanic crust from the increased weight of water in basins, which causes continental shelves to flex (Woodroffe and Murray-Wallace 2012).

Coastal defense functions provided by reefs will be sustained if reef accretion keeps pace with rising waters, but there is concern that this might be unlikely (Bruno and Selig 2007, Perry et al. 2013, Hubbard et al. 2013, Field 2011, references therein, for example), leaving reef-protected shorelines and islands vulnerable to erosion and flooding. Webb and Kench (2010) studied the responses of 27 Pacific atolls to 20th century sea level rise, and found that 86% remained stable or increased in area over 19 to 61 year periods with rates of sea level rise ~2mm/yr; they argued that erosion of reef islands is just one possibility, and that islands are more geomorphologically enduring features than is often appreciated. Woodroffe (2008) argued that erosion of a shoreline is much more likely to result from storms or other disturbance events than from sea level rise. However, erosion and flooding during storm events is likely to be more extreme where reef structure has been lost. Woodroffe (2008) concluded that on atoll oceanward shorelines, long-term accretion will overcome short-term erosion, provided there is sufficient sediment supply to maintain the morphological character of the shoreline.

Mitigating local stressors is essential for the recovery of degraded reefs, but active management may also be necessary to restore functions and services provided by reefs (Rinkevich 2008). For coastal defense provisions to be sustained, stability of vertical relief and three-dimensional structure is essential. Persistence of live corals and other calcifying organisms is crucial to maintaining sediment supply for reef-protected beaches. Active restoration efforts with shoreline protection in mind are essential to replace coastal defense benefits in low-lying regions that have already lost reef structure (Clark & Edwards 1995).

Large-scale coral restoration efforts started in the Indo-Pacific and the Red Sea in the 1990s (Young et al. 2012), and such efforts have become increasingly attractive to managers; hundreds to thousands of projects have been undertaken worldwide, and restoration techniques have improved as experience has been built and handbooks have been published. However, although restoration projects designed to aid coral recovery frequently cite the importance of coastal defense as a benefit provided by healthy reefs, few are explicitly designed to deliver coastal defense. Strategies for restoration will be dependent on site, but artificial reefs, in conjunction with coral gardening methods when appropriate, seem to hold the most promise for delivering coastal protection benefits. This report seeks to gather information on restoration projects that have delivered coastal defense benefits, and to review relevant literature on the costs, benefits, and design of projects that would deliver coastal defense benefits. We include results from a survey we

designed to learn more about costs and benefits of previous and ongoing restoration projects.

Methods

This study contains two parts: a survey designed to gather general information about relative successes, costs, and benefits of restoration projects; and a literature review and case study compilation of restoration methods that deliver coastal defense, and their costs and benefits. General input was also requested from some restoration experts.

The literature review drew upon published work, gray literature, websites, and email communications. Methods and projects that were most relevant to delivering coastal defense were the focus, but general information on the costs, benefits, and successes of projects was also sought. We developed a survey using the website surveymonkey.com, which was sent out to a handful of restoration practitioners, who provided feedback on the survey design. We then circulated the survey via NOAA Coral-list and the Reef Resilience Network listservs, as well as through personal email. Respondents had the option of answering a five question survey via email by placing an 'x' by their answer choices, or by clicking on the link to the survey. We also requested citations or URLs where we could find further information about projects. When respondents clicked on the link to complete the survey, they had the option to fill out an optional ten questions to provide additional information. The survey requested information about ecosystem service delivery and measurement, project costs, sizes, challenges and successes, planning, and design (Appendix A). It was designed for ease of interaction for respondents; all questions could be answered by selecting among options, and respondents could provide more detail on most questions if they desired. Many questions asked for average values across projects, so we could get at typical project outcomes rather than only those of the most successful projects. The survey was also designed to identify past and ongoing projects relevant to our literature review. Survey requests were sent out starting in May 2013, and final results were collected by early July 2013.

Results *- Survey*

Fifty-three restoration practitioners responded to our survey, with fifty-one respondents completing the survey by going the website, and two respondents sending their input by email. Forty-nine respondents finished the fifteen-question survey, but not all respondents answered every question; the number of responses will be reported with each question result. 54.5% of respondents had worked on greater than four restoration projects (n=44). As expected, the vast majority of projects (90.2%) were designed to include coral recovery benefits, followed by fish production and recreation (43.1% each)(n=51). Coral recovery and fish production were the most commonly measured benefits (84.0% and 42.0%, respectively, n=50). 19.6% of respondents reported designing projects to deliver coastal defense benefits (n=51), but only 10.0% of respondents reported measuring coastal defense benefits (n=50) (Figure 1).

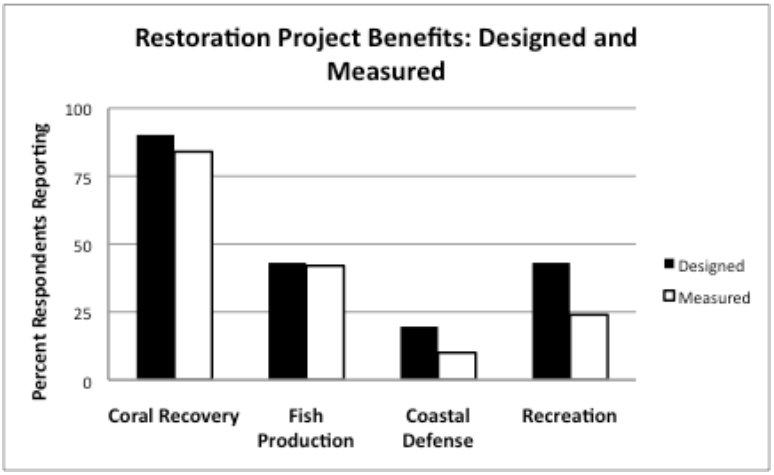

Figure 1: Ecosystem services and benefits included in project design, and measured benefits.

Costs of projects ranged widely, with 60% of projects costing US $100,000 or less. Labor was reported to be the greatest cost by the most respondents (15 respondents), with equipment, pre-construction costs, and materials reported as comprising the greatest parts of budgets by 9, 8, and 8 respondents, respectively. Maintenance and monitoring was reported to be the second greatest cost by 14 respondents (n= 42). Projects were most often monitored for 1-3 years (42.9% of respondents), although 23.8% of respondents reported monitoring projects for 4-5 years, and 9.7% reported monitoring for greater than 10 years (n=42).

Table 1: Costs of projects

Cost Range for largest project (USD)	Percent reported (n=47)
< 10,000	17.0
10,001-50,000	29.8
50,001-100,000	12.8
100,001-500,000	17.0
500,001-1,000,000	10.6
1,000,001-3,000,000	6.4
< 3,000,000	6.4

Table 2: Average amount of time projects were monitored

Length of monitoring	Percent reported (n=42)
< 1 year	16.7
1-3 years	42.9
4-5 years	23.8
6-10 years	7.1
> 10 years	9.5

Table 3: Average transplant survival across projects

Transplant survival (%)	Percent reported (n=32)
< 25	6.3
26-50	15.6
51-75	31.3

> 75	46.9

Table 4: Size of respondents' largest projects

Area of largest project (m^2)	Percent reported (n=35)
< 10	14.3
11-100	25.7
101-500	20.0
501-1,000	14.3
1,001-5,000	11.4
5,001-10,000	8.6
>10,000	5.7

Respondents who used transplanting had generally good results, with 46.9% reporting greater than 75% survival on average, and only 6.3% reporting less than 25% survival on average (n=32). Most respondents' largest projects were relatively small, with 74.3% covering less than 1,000 m^2; 5.7% reported restoring areas greater than 1 ha (n=35). This is significant because projects that deliver appreciable coastal protection benefits will tend to be larger, and success has not been widely achieved for larger-scale projects.

Coral gardening was the most highly ranked method, with 28 respondents reporting 'good' or 'excellent' results. Other concrete structures were reported to deliver 'good' or 'excellent' results by 17 respondents. The lowest ranked method was unconsolidated rubble, with 15 respondents reporting 'fair' or 'unacceptable' results (n=47).

Table 5: Project challenges

Greatest challenges to project success	Percent reported (n=41)
Bleaching	34.1
Water quality	31.7
Overfishing	24.4
Sedimentation	41.5
Storm damage	29.3
Predation	19.5
Disease	17.1
Lack of funding for maintenance	63.4
Damage from anchors, recreational activities	24.4
Lack of effective management/political problems	43.9
Vandalism	12.2

The greatest challenge to project success was lack of funding for maintenance, reported by 63.4% of respondents; 43.9% reported lack of effective management or political problems as a challenge. Sedimentation, bleaching, and water quality were also major challenges, reported by 41.5% and 34.1%, and 31.7% of respondents, respectively (n=41).

Results - *Literature Review*

We present costs of reef restoration projects, and active restoration techniques that provide coastal defense. Case studies, including project costs and outcomes, are included for each of the approaches. Methods for valuation of coastal defense benefits are also detailed. There is an overall lack of information available for complete project costs, but

this is changing as more attention is given to this problem. Improved reporting of costs, coupled with accumulation of long-term data from existing restoration projects, will make robust comparisons among methods more accessible.

Costs associated with reef restoration

Costs of reef restoration projects can be broken down into capital and operational costs. Capital costs can be further divided into pre-construction costs, which include feasibility studies, site surveys, objective setting, planning and design, and construction costs, which include substrate preparation, labor, materials, stock, and transport. Operational costs are comprised of management, maintenance, and monitoring costs, which include materials, equipment, wages, expenses, and administration costs (Spurgeon and Lindahl 2000). The extent of construction required is the most critical factor affecting cost, including substrate preparation, installation of artificial reefs, or both. The amount of labor required is also an important component of overall project costs (Spurgeon 2001).

A good understanding of costs associated with restoration efforts is essential for planning effective restoration projects, but itemized costs of conducting restoration projects are rarely available. Overall costs of projects are sometimes reported in the literature, but detailed costings are not usually available, and there are no standard accounting procedures for documenting and scaling costs (Spurgeon and Lindahl 2000, Edwards 2010). An emphasis on cost-effectiveness makes it difficult for planning costs of projects based on those of similar projects, as sources of volunteer labor and borrowed gear are not equally available for all projects. Edwards et al. (2010) developed a costing framework for transplanting projects to allow for comparisons of cost-effectiveness among restoration methods. It identifies costs for each of six project stages: (1) collection of source material; (2) set-up of nursery/culture facilities; (3) establishing coral material in nursery; (4) maintenance of cultured corals; (5) transfer of corals from nursery to restoration site; and (6) maintenance and monitoring of transplants at the site. They have provided an Excel-based tool for assessing costs (see Appendix B for a fuller description). This framework could be expanded to include costs associated with project planning, substrate preparation, and installation of artificial reef structures to plan for and compare costs of projects designed to deliver coastal defense.

Low-cost transplantation ranges from US $2,000 – 13,000/ha; costs quickly rise with density of outplanting, and can be closer to $40,000/ha. Major physical or structural restoration project costs can be quite different from coral gardening or transplantation projects, and can be on order of US $100,000 – 1,000,000s/ha (Edwards & Gomez 2007). In Table 6, we present the costs and benefits of restoration projects using various methods designed to stabilize substrate and provide coastal defense benefits. Most reported project costs were greater than US $100,000, but the Rock and Rubble Piles projects in Indonesia and the Philippines are notable examples of low-cost projects. However, as with our survey results, few of the projects included in the table measured coastal defense benefits, with the exception of the BioRock® "Necklace" and two of the Reef Ball projects. Only the Reef Ball project at the Marriott Hotel, Grand Cayman (Case study 2) included measurements of changes in beach width and morphology as well as changes in wave height and transmission in the lee of the structure.

Method	Who	Primary Goals	Location	Date	Costs	Benefits	Monitoring	Effectiveness	Considerations/Constraints
BioRock, "The Necklace"	Global Coral Reef Alliance; unnamed hotel	coastal defense	Ihuru, N Male Atoll, Maldives	1996	1/10 of cost of concrete wall with same dimensions (~$50,000 based on $10,000,000/km)	beach grew by 15m in 3 years	ground images	good	power supply issues
BioRock Anti-Wave (BAW), 12 units	Global Coral Reef Alliance; unnamed hotel	coastal defense	Gili Trawangan, Lombok, Indonesia	Dec 2008	?	beach grew, but then eroded from storm	ground images	initially good, but failed	power supply issues
Reef Ball breakwater with transplants	Stanford Development Company, ReefBall Foundation	improve beach, recreation	Maiden Island, Antigua	Nov 2003 - Mar 2004	~$1,770,000 + volunteer contributions	good fragment survival; fish spawning	coral survival	good as of 2007	extremely labor intensive
Reef Ball breakwater	Gran Dominicus Resort	improve beach, recreation	Gran Dominicus, Dominican Republic	completed July 1998; modified through 2001	?	beach growth (but erosion from hurricane necessitated intervention); natural coral recruitment	beach width, sand volume, coral spp present	good	has needed modification
Reef Ball breakwater	Marriott, Red Sail Sports, Cayaman Dept. of Environment, Reef Ball Foundation, local community	erosion mitigation	Mariott Hotel, Grand Cayman Island	11/02 initial; additions 11/05, beach filling 12/05; coral transplants 2006	?	beach accretion, but not under storm surge conditions	beach profile surveys, beach width, berm/seawall elevation, wave height/transmission, ground/aerial photographic images	good as of 2008	diminished protection under storm conditions
SHED concrete blocks	UK Department for International Development	restore from coral mining - stabilize rubble surface	Maldives	1990-1991	£367,500 (US $572,500) (for whole project, not just SHED); 1m^3 SHED blocks cost US$320/m^3 (US$1.6 million/ha) does not include transportation, monitoring	good coral recruitment	3.5 years; coral recruitment, growth, survival; monitored after 1998 bleaching	good	expensive
Armorflex mattresses	UK Department for International Development	restore from coral mining - stabilize rubble surface	Maldives	1990-1991	£367,500 (US $572,500) (for whole project, not just mattresses); Armourflex mattresses cost US$100/m^2 (US$1 million/ha); does not include transportation, monitoring	moderate coral recruitment	3.5 years; coral recruitment, growth, survival	good/fair	expensive
Rock Piles - unconsolidated	The Nature Conservancy, Putri Naga Komodo, and Taman Nasional Komodo	restore from blast fishing	Komodo NP, Indonesia	large scale Mar-Sep 2002	total budget $40,000; TNC covered scientist salaries	coral recruitment, reef building; low-cost	coral recruitment, cover, fish populations	good/fair, but uncertain which configuration is best	need access to rock quarries
Rubble Piles - consolidated	various grants	restore from blast fishing	Calagcalag MPA, Philippines	Jun-Oct 2003	$10,560 or $3300 for 0.24 ha ($44,000/ha or $13,750/ha) depending on design; for materials and labor only	low-cost; no maintenance; intact through severe storm	coral recruitment/survival, fish biomass/community composition	good on small scale	small-scale pilot
BioRock, 100 4m dia domes	Bunaken NP	restore from blast fishing, other anthropogenic degradation	Bunaken NP, Indonesia	?	$823, 770 over 10 years	$4,962,300 over 10 years (tourism only)	fish and coral abundance	?	questionable data for costs
EcoReef, 650 modules	Bunaken NP	restore from blast fishing, other anthropogenic degradation	Bunaken NP, Indonesia	completed Jan 2004	$502,350 over 10 years	$4,466,070 over 10 years (tourism only)	fish and coral abundance	?	questionable data for costs
Rock Piles	Bunaken NP	restore from blast fishing, other anthropogenic degradation	Bunaken NP, Indonesia	?	$287,600 over 10 years	$4,466,070 over 10 years (tourism only)	fish and coral abundance	?	questionable data for costs

Coral Reef Restoration Approaches

Active restoration methods can be divided into biological restoration, most often involving transplanting of coral fragments or colonies, and physical restoration, including reef repair and artificial reef installation. Many projects are dual restoration projects, that is, they involve both physical and biological restoration. Both biological and physical

efforts may be important for restoring vertical relief, complex structure, and sediment supply necessary to provide coastal defense.

Biological restoration

Transplanting coral fragments or colonies onto a reef is a common restoration technique that has been used worldwide. Corals are transplanted from donor sites or sites at risk from construction or effluent, from 'corals of opportunity' that fragmented naturally or were broken, or coral fragments from in situ or ex situ nurseries. Transplanting can increase coral cover and biodiversity and improve reef structure and aesthetic value immediately. It also increases the growth potential of a reef, and its capacity to provide sand to the adjacent shoreline. Concerns with transplanting include decreased genetic diversity, impacts to donor colonies, and decreased survival and fitness of transplants, although much progress has been made in addressing these problems (e.g., tracking genotypes to avoid inbreeding or outbreeding depression; see Baums 2008, Johnson et al. 2011).

Transplanting is labor intensive and expensive, especially as density of outplanting increases, and has produced varying results (Edwards and Gomez 2007, Rinkevich 2008). Because transplanting is often one of the most expensive stages of a project, the development of low-cost methods has been an important target of research. These methods seem to work well in lower energy environments, but are generally not sufficient for higher energy sites (e.g. Bowden-Kerby 2001, Lindahl 1998). Edwards (2010) provide guidelines and design specifications for various types of in situ nurseries, including smaller models of floating nurseries that could be easily implemented on a local scale. Precht (2006), Edwards and Gomez (2007), Johnson (2011), and especially Edwards (2010) provide thorough discussions of techniques and considerations for biological restoration.

Artificial reefs

Artificial reefs have been used for decades as fish aggregating devices, as they add complex habitats that attract fish. They can provide coastal defense benefits immediately, as well as stable substrate for coral settlement or transplanting. They have been increasingly used to relieve pressure from recreational and fishing activities on natural reefs (Nichols 2013, Polak and Shashar 2012, 2013). A wide variety of materials have been used for artificial reefs; the most common are concrete and natural rock (Baine 2001). To be effective for coastal defense, artificial reefs should provide vertical relief and topographic complexity; for facilitation of coral recruitment, they should contain as many vertical (and as few horizontal) surfaces as possible. Artificial reef modules for coral reef restoration, such as Reef Balls, BioRock®, and EcoReefs, have been developed and marketed, but so far these methods have not been rigorously tested (Kaufman 2006). Costs (when available) and benefits for marketed and non-marketed artificial reef approaches and case studies are presented in the following sections.

Reef Balls

Reef Balls are prefabricated concrete artificial reef modules that come in many sizes and designs for different site and project needs. They are the most commonly used artificial

reef modules; over half a million reef balls have been deployed in 4000 projects in 66 countries; a list of the countries is provided in Appendix C (reefball.org). The Reef Ball Development Group was founded in 1993; the company licenses contractors to manufacture the units. The Reef Ball Foundation (RBF), a non-profit organization, has developed specific protocols for deploying, anchoring and transplanting corals to Reef Ball products, and is often directly involved with projects utilizing Reef Balls through its Coral Propagation and Rescue Teams. Spurgeon (2001) cited Reef Balls as a good low-cost approach with benefit enhancement, while Edwards and Gomez (2007) asserted that the vast deployment of Reef Balls provides a small amount of complex substrate at the cost of millions of dollars.

Reef Balls are made of concrete mixed with microsilica to match the pH of seawater, with textured surfaces to promote settlement. The modules have holes designed to form feeding currents for sessile invertebrates and to maximize fish habitat and are often designed with attachment points for coral transplants. The holes also help dissipate wave energy. Reef Balls can be used as semicircular breakwaters, as examined in Armono and Hall (2002). The operating depth of Reef Ball breakwaters is limited by module height, so they can only be deployed in water < 2 m, unless they are placed on a rubble mound or concrete base. Reef Balls have been marketed specifically as breakwaters through a subsidiary, Reef Beach, and 12 of these projects have been completed, 10 in tropical waters (reefbeach.com). Five of these projects were deployed on sandy and hard substrate in front of resort beaches, designed for sand accumulation, and three projects included coral transplantation. One Reef Ball breakwater, constructed from 2002 to 2003 at Hotel Mayan Palace in Qunitana Roo, Mexico, was destroyed by storms. Reef Beach attributed this outcome to substandard concrete mixes and faulty anchoring methods, and has subsequently not allowed clients to build their own breakwaters. Information regarding these breakwater projects is limited, but we have compiled the available information on these projects in Appendix C. In the case studies, we have included those projects that were located in coral reef environments, and for which we could find a reasonable amount of information.

Reef Balls come in various sizes for different types of uses. The standard Reef Ball (1.83 m wide, 1.16 m tall) costs ~$350/module; the largest module, the Goliath Reef Ball (1.83 m wide, 1.52 m tall) costs ~$450/module (reefball.org). Reef Ball molds can be purchased for US ~$1,000; materials cost ~US $40/$m^2$ of coverage (Fox et al. 2005). Use of Reef Ball molds must be authorized by RBF. It is difficult to assess the total costs of Reef Ball projects, as volunteer labor and borrowed equipment typically subsidize any large projects. Coral Team members are paying volunteers, which can save US ~$75,000 in labor costs for a typical Reef Ball breakwater deployed near a resort. Each volunteer pays RBF $1,000-1,500, and resorts typically supply rooms, SCUBA gear, and boat support. Reef Balls are usually deployed via barges (although lift bags can be used), which can add considerably to project costs. A memo quoting a cost of $1,500,000 for one project noted that the figure does not include costs of transplantation, although transplanting was part of the project plan. This makes it difficult to compare the costs of Reef Balls to other technologies.

BioRock®

BioRock® is a patented method that uses electrolytic deposition of calcium carbonate to build artificial reef structures. The technology was originally developed for use in construction, as 'Seacrete' or "Seament.' It uses low voltage, low amperage direct current through a cathode and an anode to stimulate calcium carbonate formation around stainless steel structures. The developers reportedly noticed increased growth of calcifying marine organisms during early tests of the technology. They attributed this to alkaline conditions at the cathode site causing an increase in carbonate ion concentrations in the adjacent water, raising the carbonate saturation state and providing more favorable thermodynamic conditions for calcification. Thus, less metabolic energy is expended in calcification, freeing up energy for growth, reproduction, and resilience to stress (Goreau et al. 2004). BioRock® has been credited with causing increases in coral growth rates by 3-5x, 2-20x faster healing from breakage, 16-50x higher survival of bleaching, and 100-1000x higher settlement and reproduction (Goreau 2010), but these claims have been considered somewhat dubious by the academic community, and a recent survey of restoration practitioners in the Caribbean found BioRock to be the least effective method included (Young et al. 2012). Peer-reviewed studies using the BioRock method have been generally inconclusive regarding its benefits.

The technology requires a reliable power source, which has proven to be a challenge to many BioRock projects. Schumacher and Schillak (1994) found that corals cannot settle on the structure when electrochemical deposition of calcium carbonate is occurring. The current must be alternated between 'dead' periods when corals settle and grow to a critical size on the structure, and shorter 'live' periods when interstitial spaces and be filled in with electrochemically deposited limestone. Coral fragments, however, can be transplanted directly to live BioRock structures. A study conducted in the Red Sea found that four species of coral nubbins (*Acropora variabilis, Stylophora pistillata, Pocillopora damicornis, Pavona darians*) transplanted to a live BioRock structure cemented within 2-8 weeks. Survival varied among species, and ranged from 36% (*P. damicornis*) to 72% (*A. variabilis*) for the site that was monitored longest. Although the authors note that their survival rates compared favorably with previously published nubbin survival rates, an unpowered control was not included in the experimental design, so increased survival of nubbins on powered BioRock structures could not be concluded (van Treeck and Schumacher 1997).

Sabater and Yap (2002, 2004) compared the growth and survival of *Porites cylindrica* nubbins transplanted to powered BioRock structures in the Philippines to nubbins transplanted to unpowered identical structures and to naturally occurring colonies. Their results indicated that electrolysis can increase growth and survival of coral transplants, but these effects decrease over time, making it difficult to draw conclusions about the overall long-term effects of the technology.

Borell et al. (2010) compared responses of *Acropora yongei* and *A. pulchra* to an electric field on a cathode, an electric field insulated from the cathode, and controls over four months, and found that mineral accretion had either no effect or negative effects on coral transplants. Calcium carbonate precipitated on the cathode cemented both species to the

cathode within three weeks. Both species grew significantly more in the electric field without the cathode than on the cathode or in control treatments, but *A. pulchra* grew significantly more and had greater zooxanthellae densities on the cathode than in control treatments. *A. yongei* grew the least in cathode treatments. Growth was linked more closely to zooxanthellae density and photosynthetic performance than treatment. The authors suggest that the enhanced survival in the presence of mineral accretion observed by Sabater and Yap (2002, 2004) was due to the better attachment of treated nubbins compared to untreated nubbins, which were attached by placing them in mesh within the arrays. Borell et al. (2010) attached fragments in the non-cathode electric field and control treatments more securely, so the added stability imparted by the mineral accretion did not increase survival as in Sabater and Yap's (2002, 2004) study. They could not draw any conclusions about the direct or indirect effects of the treatments on the very different growth and survival observed within and between the closely related species.

Benedetti (2005) tested larval settlement of the octocoral *Corallium rubrum* in the Mediterranean on powered and unpowered electrolytically deposited carbonate and limestone marble plates. Using very low current densities (~1 $\mu A/cm^2$), they found that settlement was five times lower on the powered plates than on unpowered electrolytically deposited aragonite and marble plates. They used stainless steel plates with Zn anodes, and did find them to be cheap enough to recommend using electrolysis to manufacture settlement plates.

BioRock® structures are self-repairing if properly supplied with electricity, and the material that is deposited is very strong, with a load bearing strength of 80 MPa, or about three times as strong as Portland cement. Because BioRock® structures are made of rebar and metal mesh, they can be built in any shape and in large sizes very cheaply, so they can be designed as breakwaters. Goreau et al. (2012) described a BioRock® Antiwave structure, designed with a sloping shape to minimize drag and prevent scour, and enhance sediment deposition. It cements itself to the bedrock, and thus is very stable. Units can be fully or partially packed with rocks to increase wave reflection, and are most cost effectively deployed as staggered units. One such project is included in the case studies, as well as another BioRock® project designed to promote beach growth.

EcoReefs

EcoReefs are ceramic snowflake-shaped modules that are designed to maximize vertical surface area for coral recruitment and provide fish habitat. The snowflake shape is intended to create a hydrodynamic current field that facilitates coral settlement and feeding (Moore and Erdmann 2002). The modules are lightweight and easily installed, and cost US $30-70 per module (Apostolakos et al. 2007), or US ~$70/$m^2$ (Fox et al. 2005). We were unable to locate any restoration projects using EcoReefs that were relevant to coastal defense, although the modules are being used in nursery culture.

Concrete structures

Concrete is the most commonly used material for artificial reefs, and it has been used in many forms other than the ubiquitous Reef Ball. Because it is widely available and relatively cheap, concrete is used in both large and small-scale projects, making it

difficult to estimate the extent to which it has been used for reef restoration. It is used as both as substrate and to attach large coral colonies to reefs or artificial structures, as it is more sturdy and cheaper than epoxy. Cinder blocks have been widely used in both small scale projects and fixed nurseries. Similar materials, such as steel slag (Mohammed et al. 2012) and pulverized fuel ash-concrete from coal production (Lam 2003), have also been used with success for coral transplantation and settlement.

Concrete mattresses and pillows have been used extensively at ship grounding sites with mixed results; they have caused problems with shifting during storms and crushing substrate (Symons et al. 2006), but have been used with relative success for stabilizing the substrate and attracting some natural coral recruitment. Large (~1m^3) blocks provide more vertical surfaces to attract coral settlement, and provide immediate vertical relief to attenuate waves, but they are very costly. Concrete has also been molded into variously configured artificial reef modules, often in community-based projects.

Rock and rubble piles
The marketed artificial reef technologies include designs to maximize coral and fish habitat, and to provide natural reef-like structure. However, the superiority of these modules to "an artful pile of boulders" has not been rigorously demonstrated (Kaufman 2006). Indeed, rock and rubble piles can be effective and low-cost in locations where rock or rubble is plentiful, such as reefs that have been degraded by coral mining, blast fishing, or storm events. We include one rock and one rubble pile project among the case studies presented below.

Case Studies
Case study 1: Reef Ball Breakwater - Gran Dominicus Resort, Dominican Republic
The first coral reef Reef Ball breakwater to be implemented was at Gran Dominicus Resort, Grand Cayman. It was completed in July of 1998, and modified through 2001. The breakwater was deployed ~53 m offshore, and was comprised of ~450 Reef Ball (1.2 m high, 2 m diameter at base) and Ultra Ball (1.4 m high, 2 m diameter base) modules. It was composed of three sections, each three rows wide, in a depth of 1.6-2.0 m, with the crests of the modules 0.2-0.8 m below MWL. The tidal range is 0.4 m at the site. The shoreline is oriented east-west with wind and waves come predominantly from the southeast, transporting sediment from east to west. A control profile was established east of the breakwater site, along three profiles along the site, so that effects of the breakwater could be established.

The substrate at the site was limestone with a light coating of sand, so the central cavities of the modules were filled rocks to provide weight, and fiberglass rebar was driven or drilled into the bottom to provide sliding stability. The modules remained stable through Hurricanes Georges and Mitchell in September and October 1998, but the storm surge and wave heights were greater than could be attenuated by the breakwater, and significant erosion occurred. After the storms, the beach was nourished and small rocks were added to both sides of the breakwater. Rocks were also lined up to connect the eastern end of the breakwater to the shore. This caused too much sand to build up, so some sand was

removed, as well as the rocks shoreward of the breakwater that had become scattered about the swimming area.

The control profile showed no change in beach width between February 1999 and April 2001; the eastern, western, and central profiles at Gran Dominicus showed +13 m, +10 m, and 0 m change in beach width, respectively. Sand volume changes were +44.25m^3/linear m, +25.65m^3/m, and +2.0 m^3/m, at the eastern, western, and central profiles. From April 2001 to June 2002, the control profile showed no change, and the eastern and western profiles showed 0 m and +1 m changes in beach width. The central profile showed a +1-2 m change in beach width from November 2001 to July 2002.

By 2004, many of the Reef Balls had been colonized by massive corals. That year, two *A. palmata* fragments were transplanted to two Reef Balls, and the rest of the modules were moved further offshore because the swimming area had become too small. The *Acropora* modules were not moved because there was concern the corals would not survive. It was suggested by the advising engineer that two more rows of Reef Balls could be added to enhance wave attenuation, but it is unclear if this modification was implemented. A 2007 monitoring effort recorded naturally recruited *Porites astreoides, Millepora complenata, Meandrina meandrites,* encrusting *Diploria cervosa*, and *Siderastra sideria.* The *A. palmata* fragments planted in 2004 had grown from 0.001 m^2 to 0.2 m^2. A report derived from the monitoring recommended planting of more corals, but not on the older reef balls because the fouling community had developed enough to outcompete coral recruits (reefball.org, artificialreefs.org).

Case study 2: Reef Ball Breakwater - Marriott Hotel, Seven Mile Beach, Grand Cayman
The Marriott Hotel, on the southern end of Seven Mile Beach, Grand Cayman's largest tourist attraction, commissioned a Reef Ball breakwater to combat severe erosion of the hotel's beach. The north-south oriented beach typically experiences erosion during the wet season, May to November, and recovers from December to April. Beach width can vary by as much as 15 m during an average year due to these fluctuations (Arnouil 2008).

The breakwater was constructed in November 2002. It was deployed 52 m offshore and consisted of five rows of 200 Ultra Ball modules, weighing 1360 kg each, forming a structure 7.6-9.1 m wide, 1.1-1.4 m high, in ~1.5 m water depth. Modules were secured by fiberglass rebar that was driven into the hard bottom and through the modules to prevent sliding. The breakwater had a gap in front of the widest and shallowest part of the natural reef, seaward of the breakwater. An additional 32 modules were added in November 2005. After the additions were completed in 2005, the beach was nourished with ~5000 m^3 of sand placed along ~300 m shoreline (Arnouil 2008). Corals were planted to the Reef Balls on 2006. Over 400 fragments were propagated and secured to the modules with epoxy, and over 100 colonies that had been damaged by Hurricane Ivan in 2004 were stabilized underneath the Reef Balls and secured with hydraulic cement. The coral phase took three weeks, including one week of work on the site. Six more Reef Balls were added to mark locations and for more restoration and monitoring efforts, and a red mangrove restoration utilizing specialized "mangrove root mimic" Reef Balls was planned (reefball.org), but information regarding these plans is not available.

A monitoring program was implemented to gauge the success of the breakwater, which included beach profile surveys, beach width measurements, and ground and aerial photographic images (Arnouil 2008). The beach was 0 m wide in November 2002, and it grew by 15 m in the three months after the breakwater was put in. Beach width reached 25 m four years after in February 2007. Along all profiles, the beach was eroded at ~3.2 m/yr before the breakwater was installed, and it has accreted ~3.4 m/yr in the years after installation through June 2008. Berm height and width increased through 2007 (0.6-0.9 m and 6.0-9.1 m, respectively), and the elevation at the seawall increased (0.3-0.6 m)
The breakwater reduced wave heights by ~60% and percent wave energy transmitted decreased with increase wave height in conditions with no storm surge. In storm surge conditions, with increased water depth and wave height, percent wave energy transmitted increases, so the breakwater may not prevent erosion in storm conditions with continuous winds. Seven Mile Beach experienced some erosion during Hurricane Ivan in 2004, but this was mitigated because the beach did not receive continuous onshore winds (Arnouil 2008).

Case study 3: Reef Ball Breakwater - Maiden Island, Antigua
The Maiden Island Total Reef Restoration Project was a dual restoration project undertaken by RBF and Stanford Development Co. Ltd.; it stands as the flagship Reef Ball Breakwater project, and largest Reef Ball project to date, with ~3500 Reef Balls deployed. Maiden Island lies ~1 km from the northeastern coast of Antigua; its windward reef was badly damaged by Hurricane Luis in 1995. The island has a tidal range of 0.4 m, and experiences trade winds most of the year that create waves 0.6-1.2 m in height.

The project began in October 2003 and was completed in April 2004. First, Stanford Development removed 50 tons of beach-incompatible dredged sand that was placed on the island by the US Navy from dredging a channel between Antigua and Maiden Island, and replaced it with natural sand. RBF contractors manufactured the Reef Balls on site. 2000 units were completed in three weeks at the site, which had 100 workers building units 24 hours a day, seven days a week. The total cost was US $1,770,000; labor costs were mitigated through the use of 274 volunteer person-days. The project was comprehensive, and had seven major goals, the first of which was to improve the recreational value of the beach by attenuating wave energy.

Pre-construction surveys were used to guide the design of the fringing breakwater. SCUBA divers marked contour lines with rebar and neon cord every 30 ft, and the force necessary to drive the rebar into the substrate was measured to estimate bottom firmness. Surveyors used a total station survey methods to get coordinates and elevations for the rebar stakes to track the ‘biological tide line’ for healthy reef growth. A jet probe and rebar/hammer method to was used measure the depth of sand over hard substrate at survey points, so that the capacity of the sediment to support the modules and anticipated degree of settling could be calculated, and type of anchoring determined. Barges crews worked from dawn to dusk daily to deploy the modules. Because the bottom composition varied considerably across the site, four different attachment methods were employed. Anchoring cones were used in areas with ‘normal’ bottoms; spikes were used on ‘hard

packed' bottoms; battered pilings were used on soft bottoms; and modules were driven in with rebar for hard bottoms. For added stability, the modules on the inside and outside of the five rows had solid bottoms and were 25-50% heavier than modules in the inner three rows, which had hollow bottoms.

The breakwater on the windward side of Maiden Island was comprised of 1200 Reef Balls. Modules of eight different sizes were used to maintain the top of the breakwater at the 'biological tide line.' Over 100 different designs were used, including the "Layer Cake" model designed to provide ledge and complex void spaces to create habitat for a diverse array of reef inhabitants. Modules were also designed with special bottoms for seagrass and conch habitat and many were filled 1/3 with rubble to stabilize the substrate and provide fish habitat.

Over 5000 nursery corals were transplanted to the breakwater. An additional 17.5 tons of corals at risk from dredging operations was rescued, propagated, and transplanted to the breakwater, including four species that were absent from Maiden Island after Hurricane Luis. Coral Rescue teams used a rapid propagation technique developed by RBF. Coral colonies were collected and brought to a'propagation table' with an upper working surface and a lower surface underwater where fragments were kept pre- and post-processing. A quick-setting cement was mixed and poured into cups, and corals were fragmented and placed in the cement. Mounted corals were returned to seawater after 2 minutes, and could be outplanted 20 minutes later. A six-member Coral Team can make 1,000 plugs in one day. Reef Balls are designed with holes for the cement plugs, speeding the outplanting process. A relatively unskilled diver can transplant 100 colonies in an hour, compared to the ~10 corals that can be planted in an hour by a skilled diver with traditional methods. Of the > 5000 fragments planted, 200 had been lost as of 2007. These were all elkhorn and staghorn corals; other species had survival rates > 95%.

In addition to planting corals, volunteers introduced 500 *Echinometra lucunder* to the site to provide grazing pressure. A special fish 'corridor' was incorporated into the breakwater design to facilitate juvenile fish migration to deeper waters, which ends at a 'pinnacle' designed to promote fish spawning. A Gray snapper spawning was observed at the pinnacle. Guided snorkeling and diving trails were also created. The project team worked with researchers at Nova Southeastern to test specially designed invertebrate enhancement plates; genetic banks for transplanted elkhorn and staghorn corals were established. On the leeward side of the island, an additional 2000 Reef Balls were deployed to create another fringing breakwater. They were specially designed as mangrove planters, and 4200 red mangroves from Nova's mangrove nursery were planted. The mangroves provide additional habitat, erosion control, and water quality enhancement (reefball.org).

No monitoring data is available past the 2007 survival data, and detailed data is not available, so it is unclear how the transplants have fared in the nearly ten years that have passed since the project was finished. No data is available regarding the mode and magnitude of shoreline response. The owner of Maiden Island, R. Allen Stanford, was accused of orchestrating an $8 billion Ponzi scheme in 2008; the Antiguan government

seized 254 acres of Stanford's land in February 2009, including the 26 acre Maiden Island, where Stanford had pledged to spend EC $135 million (US ~$50,000,000) on the restoration effort and construction of his home. We contacted the Antiguan Department of Fisheries, which had expressed interest in protecting the project site, to find out whether monitoring or any other work has been done there, but we have not received a reply. RBF has revoked Stanford Corporation's authorization to use the 50 molds they purchased due to Stanford Corporation's financial delinquency.

Case study 4: BioRock® Antiwave project - Gili Trawangan, Lombok, Indonesia
Twelve BioRock® Antiwave units were deployed in Gili Trawangan, Lombok, Idonesia, in December 2008. Sand accumulated shoreward of the structures within 2 weeks, and the structure was quickly colonized with mussels and oysters. Sand and reef rubble built up on both sides of the breakwater within months. Google Earth images show beach growth within 8 months of deployment; the beach grew by 1 m in 1 year and did not erode in the monsoon season, while adjacent beaches eroded. The structure apparently did not interfere with longshore sediment transport, although data to support this was not presented. Reportedly, the hotels that commissioned the project did not expand it as planned, and power was not maintained, leading to rusting of the structures. A storm eroded the shoreline in 2012 (Goreau et al. 2012).

It is not clear why the Antiwave units would have stopped functioning, given that electrolytically deposited carbonate has shown to be an effective cement which seems, by other accounts, to be stable after it is not powered. It is unclear when the hotels stopped powering the structures, but it would seem that they would have retained breakwater function, at least for a few years. Perhaps more likely, the design was not sufficient to defend the beach from a storm of great magnitude. Furthermore, 1 m of beach growth in one year is small compared to seasonal changes in beach width that are widely reported.

Case study 5: BioRock® project - Ihuru, N. Male Atoll, "The Necklace"
A coral restoration project designed to "build a beach" in Ihuru, N. Male Atoll, Maldives, seems to have had better results. The 45 m long, 4-8 m wide, 1.5 m high welded steel structure, termed "The Necklace," was constructed in 1997. Coral colonies, mostly massive, that were "loose" on nearby reefs were transplanted to the structure. The structure was intended to be the first of a series of structures that would encircle the island, but later structures were not added. Reportedly, most corals on the structure survived the 1998 bleaching event, showing 16-50 times greater survival than corals on adjacent reefs, although survival data for the structure and for other reefs is not available. The beach grew by 15 m in 3 years, and did not erode during the 2004 tsunami. The reef and beach are apparently healthy as of April 2012 based on photos showing massive corals growing on the structure (Goreau et al. 2012).

Case study 6: Concrete artificial reef modules - Palau Weh, Aceh, Indonesia
A community-based project on Palau Weh was implemented after recovery funds were allocated from the 2004 tsunami that struck the Indian Ocean. The concrete was three parts aggregate (mostly from building rubble) mixed with cement, and it was poured into molds containing coral rubble and plastic refuse to reduce weight and iron rebar to

strengthen the modules. The modules dried on the beach for 1-4 weeks, and divers deployed the ~1.25 m^2 modules at 2-9 m on sand and rubble substrate. They were conditioned for 4-6 weeks (indicated by oyster colonization), and were then scrubbed with wire brushes to provide space for coral recruitment. ~25 cm long fragments of the staghorn corals *Acropora formosa* and *A. subglabra* were collected from healthy reefs and attached to PVC pipes extending up from the modules using cable ties. Transplanting was done in June 2006, August 2007, and December 2008 – January 2009. 260 modules for a total of ~250 m^2 of habitat was created.

The areas around fragment bases were scrubbed one month after transplanting to reduce competition with other benthic organisms, and corals were periodically maintained and were righted, reattached, or replaced as necessary. In November 2009, five replicate artificial reef units from the one, two, and three year old deployments were randomly selected for monitoring. Coral cover increased with module age, and ranged from ~24% for one year old modules to ~64% for three year old modules. Coral recruitment on the one and three year old modules was nearly twice that on the natural reef; recruitment was dominated by Acroporidae (45% of total recruits), Faviidae (23%), and Pocilloporidae (20%). Fish species richness increased with age of artificial reef modules; pelagic predators were present on some three year old modules.

Each module cost US $12.75 for materials, and required 40 person-hours to complete, for a cost of US $21.25 per module. Transportation costs were US $45 per module. The project directly employed three people for three months. The project was funded by at least US $8,500 in grants. It provided economic benefits to the community; the site was used extensively by both international and domestic tourists for diving, snorkeling, and fishing, and there was anecdotal evidence that the project increased tourism to the area. The project was strongly supported by the local community. Unfortunately, a May 2010 bleaching event resulted in nearly 100% mortality for transplanted corals; staghorn corals on nearby reefs fared the same as transplants. The project was, however, deemed a success in raising community awareness and support, and providing economic benefits to the community (Fadli et al. 2012).

Case study 7: Reef blocks, Concrete mattresses, and chain-link fencing - Maldives
Clark and Edwards (1994,1995,1999, Clark 2000) examined the effectiveness of different techniques to stabilize a reef flat in the Maldives that had been severely degraded by coral mining. In late 1990-early 1991, they deployed four sets of artificial reef structures, totalling 360 tons of concrete. The four treatments were: 1 m^3 Shepherd Hill Energy Dissipators (SHED) modified with PVC and concrete infill to reduce open space; two sets of Armorflex-220 flexible mattresses anchored with paving slabs, one with coral colonies transplanted from a healthy reef, and chain-link fencing anchored with paving slabs. Each treatment had three 5 x 10 m replicate sites, and treatments were compared with three 50 m^2 degraded sites and three 50 m^2 sites on nearby unmined reef as controls. Study sites were 05 ±1.8 m below lowest astronomical tide; the tidal range at the site is ~1 m. Plots were monitored for fish population structure and biomass, coral recruitment, growth, survival, and loss of transplants.

The first corals to recruit to the structures were *Pocillopora* and *Acropora* spp., followed by massive *Porites* sp. Over the 3.5 year study period, the SHED structures had 12.6 recruits/m^2, and Armorflex mattresses, both with and without transplants, had an average of 4 recruits/m^2, with 18 recruits/m^2 on vertical edges. Within 3.5 years, each SHED structure had ~500 colonies each, with some as large as 30 cm in diameter. Outer and inner vertical surfaces attracted the most recruits to SHED structures. The coral community was dominated by acroporids and pociloporids on both the SHED and Armorflex structures, at nearly equal abundances, with a few massive colonies.

530 colonies, collected by denuding three 50m^2 patches of healthy reef, were transplanted to the three of the Armorflex structures and secured with cement. Of the transplanted colonies, 19-37% were swept off the mattresses by wave action, almost all within the first 7 months, while another 18-29% became loosened. Because of the high rates of loss, economic cost of transplanting, and lack of increase in recruitment to transplanted structures, the authors concluded that transplanting was not an effective technique for the area.

Fish were immediately attracted to the artificial reefs. In one month there were five times as many fish species on the structures compared to baseline surveys, and twice as many species on the Armorflex mattresses. Within a year, the artificial reefs had similar or greater species richness and densities of reef fish than unmined reef flats, but the community composition differed between natural and artificial reefs. After 2.5-3.5 years, the SHED structures supported the greatest fish biomass (~4.5 tons/ha); Armorflex mattresses supported 1.5-2.2 tons/ha. The pristine reef supported ~1.2 tons/ha, and the degraded reef had 0.3 tons/ha. The large difference between SHED structures was due to the large fish using the blocks for shelter.

A survey done 4-6 weeks of the bleaching event that began in April 1994 found all corals on the artificial reef structures to have bleached. A June 1998 inspection showed most massive corals recovered, and nearly all branching corals dead and covered with algae. Many more corals had recruited to both the SHED and Armorflex structures since the last monitoring in 1994, with massive corals colonizing the internal surfaces of SHED areas and the interstitial areas on the mattresses. Corals surviving the bleaching event were mostly *Porites* and *Pavona* spp. Researchers visited the structures again in February 1999, and noted recruitment of *Pocillopora damicornis.*

Another quantitative survey was done on the SHED structures in March 1999. A total of 205 recruits to were found, with densities ranging from 0.6-2.8 recruits/m^2, with highest settlement on outer vertical surfaces. Branching corals, as before, were the dominant recruits, comprising ~67% of new recruits, with massive corals representing ~33%, a much greater proportion than the 1991 recruitment, when massive corals were 2% of recruits. This seems to reflect the disproportionate survival of massive corals following the bleaching event.

Overall, the effectiveness of the artificial reef structures was proportional to their cost and structural complexity. SHED structures were the most effective at £210/m^2 (US $320),

followed by Armorflex mattresses at £66/m^2 (US $100) for plain treatments and £97/m^2 (US $147) for mattresses with transplants. Chain-link fencing was both the least effective and expensive at £26/m^2 (US $40); though even the chain-link fencing started to show signs of cementation and consolidation after 3.5 years. These costs included deployment costs but exclude shipping costs. Armorflex mattresses could be as effective as SHED units in the long term if sedimentation is not a problem. A large part of the costs of Armorflex mattresses and chain-link fencing was the concrete slabs; if they could be attached more cheaply their cost-effectiveness would be greatly increased. The project was funded by the UK Department for International Development; the total cost was £367,550 (US $557,685) (http://r4d.dfid.gov.uk/), although this figure may not represent total project costs (and only covers through 1994).

Case study 8: Rock Piles - Komodo National Park, Indonesia

Fox et al. (2005) tested four different configurations of rock piles to consolidate rubble at four sites on a reef that had been heavily degraded by blast fishing. Rock piles were chosen after they fared best in smaller pilot studies using wide mesh fishing net to consolidate rubble, which became covered by rubble, cement slabs pinned to rubble, whichwere overturned, and rock piles atop rubble, which became filled in by rubble, but could be modified. The four rock pile designs tested on larger scales (1000 m^2) were: complete rock coverage (~75 cm high); rock piles (1-2 m^3, spaced every 2-3 m); spur and groove morphology parallel to the current; and spur and groove perpendicular to the current (~75 cm high, 2 m wide, spaced every 2-3 m). The spur and groove designs were based on naturally occurring reefs. All designs were constructed from 140 m^3 of limestone rock from March to September of 2002. The limestone and sandstone rocks were sourced from a local quarry and transported by cargo boat, from which they were dropped over the side and arranged by SCUBA divers at depths of 5-10 m.

All four designs were successful in facilitating coral recruitment and growth over the study period, but it was unclear which design was best. The total coverage design kept rubble out, but covered the smallest area. Evenly spaced piles covered the greatest area, but rubble was entrained into the area. The spur and groove morphology parallel to the current could be particularly effective in promoting coral accumulation on the spur while allowing rubble to be flushed out along the spurs at sites where rubble movement is unidirectional. The spur and groove morphology perpendicular to the current might enhance coral settlement by generating turbulent flow.

After one year, mean recruitment to the rock piles was 7.3 recruits/m^2, with the lowest recruitment on evenly spaced piles (3.5 recruits/m^2) and highest on the complete coverage design (14.2 recruits/m^2). The high recruitment indicated that larval supply at the sites is sufficient so that transplantation is not necessary. After six years, coral cover was highest in spur and groove parallel designs, but not significantly so. Coral cover was variable among sites and designs, but was lowest at the low current site, and negligible in control rubble plots. Success was greatest at moderate current sites: rubble tended to fill in all designs at high current sites, while sedimentation was a problem at low current sites.

This study was included as a case study in Edwards (2010), and the authors have provided a detailed costing of the project (Haisfield and Fox). The project budget was US $40,000, and was funded by the David and Lucille Packard Foundation and The Nature Conservancy. The budget for physical rehabilitation was US ~$30,000. This included cargo boat rental for 76 trips (US $17,000) with 8-12 person crews to load, transport and unload rocks, 1 boat driver, and 1 volunteer, coordinated by a team of 2 divers (1 scientist, 1 park ranger), to finalize the rock configurations underwater. A speedboat transported divers at a fuel cost of US $3,380. The 2275 m^3 of rock was transported in 910 truckloads, totaling US $7,078. Park ranger stipends were US $2,500, and external consultants cost US $10,000. Scientists' salaries were paid by The Nature Conservancy. Each monitoring trip required 8-10 boat days and two scientists trained in coral identification. Pile sizes were measured by park rangers. Overall estimated cost per m^2 of each design was: US ~$17 for complete coverage; US ~$5 for spur and groove rows; US ~$3 for piles; US ~$5 on average.

Rock piles are a low cost way to stabilize substrate in rubble fields, and costs could potentially be reduced by half for larger spatial applications. Transplantation is not necessary at sites where larval supply is good, and implementation of stable structures greatly enhances coral recruitment within a year. To be cost effective, however, there must be a local source of rocks. Rock piles produced the best results at sites with moderate currents.

Case study 9: Rubble Piles - Calagcalag Marine Protected Area, Philippines
Raymundo et al. (2007) constructed rubble piles secured with plastic mesh in a pilot study on a reef degraded from blast fishing in a Marine Protected Area in the Philippines. Little recovery had occurred on blasted reefs since the MPA's establishment in 1988. Much of the reef within the MPA was healthy, but the rubble field was located where the greatest coral cover and fish diversity would be found. Researchers established the rubble stabilization treatments adjacent to a healthy reef, used as a reference ecosystem, and part of the rubble field was used as a control. Rubble was stabilized in five 17.5 m^2 plots, three deployed during coral spawning season (June 2003), and two deployed before the storm season (October 2003). The rubble surfaces of the plots were stabilized by overlaying the rubble with 2 cm-wide mesh anchored with rebar stakes, and with existing coral heads that were accommodated by cutting holes in the mesh. Rock piles made of reef rock and cement were constructed on land and placed on top of the stabilized rubble. The 1 m high piles were hollow and pyramid-shaped, and were spaced 1 pile/0.5 m^2. They were designed to give added weight and stability to the substrate, and to provide habitat availability and surface area for coral settlement. The three reef zones were monitored for fish diversity and biomass and coral recruitment and every 3-4 months for 4 years. During the second year, funds were obtained to get a patrol boat to curb the continued poaching within the MPA. The third year, a community officer was hired to increase management capacity and build ties with the Fisher's association, the mayor, and the local Department of Environment and Natural Resources. This allowed funding to be allocated by both offices for maintenance, monitoring, and enforcement, leading to decreases in poaching. Herbivorous fishes doubled in biomass by 2006 after

rehabilitation and enforcement; planktivores and piscivores also increased in biomass, especially on rehabilitation plots.

Coral recruitment increased on rehabilitation plots over time on plots deployed during spawning season, from an average of 0.5 individuals/m^2 in September 2003 to 4.5 individuals/m^2 in March 2005. Recruits to the mesh had a mean survival rate of 63.4 ± 32% at 10 months. Most of the mortality was caused by shifting rubble during an October 2004 storm. The recruiting community was composed of Faviidae, Poritidae, and Acroporidae, and was similar to that of the surrounding reef. Recruitment and survival on the adjacent rubble fields was negligible. Recruits to the mesh grew both upward and downward, with extensive basal growth cementing the rubble together. Plots stayed stable over the three storm seasons, and all rock piles remained upright. Mortality from the October 2004 storm was greater on sloping plots; because the storm represented the greatest magnitude experienced in the area, the authors conclude that the design was stable, especially given the rapid cementing of plots by new recruits. It was difficult to separate the effects of the rehabilitation from those of enforcement – the rapid increase in fish biomass in all three treatments was taken as evidence of the important role that enforcement played. Fish biomass increased much more on healthy reefs and rehabilitation plots than on rubble fields. Thus, increased enforcement of fishing restrictions in unrehabilitated rubble fields might be a less efficient use of resources than in healthy or rehabilitated areas. The cost of labor and materials was US \$75/17.5 m^2 plot. The plots did not require maintenance, and the fishers association provided additional volunteer labor. Expanding the design to the entire 0.24-ha area of the rubble field would cost US \$3300. This is one of the most cost-effective methods for substrate stabilization and artificial habitat creation, especially among those addressing rubble stabilization.

Coastal defense benefits and valuation

Reef-derived benefits are often valued using a total economic value (TEV) approach, which seeks to identify and value each benefit provided by the reef in a given location. Benefits are characterized broadly into direct use, indirect use (including coastal defense), and non-use values. Tourism, fisheries, and coastal defense are the greatest contributors to TEV, but this report will focus on coastal defense benefits. The various approaches for ecosystem service valuation are outlined in Spurgeon (1992, 2001), Cesar (2000), and Burke et al. (2005, 2008). This report will focus on methods for valuing coastal defense provisions.

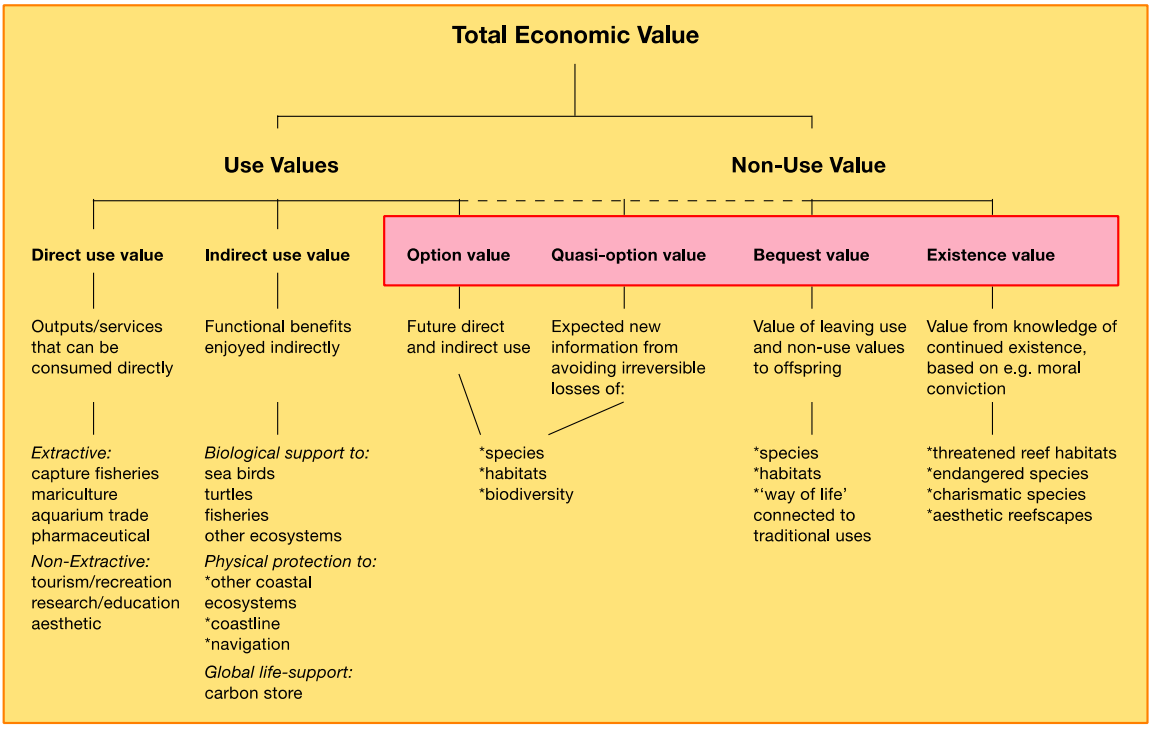

Figure 2: Services comprising reefs' Total Economic Value (from Cesar 2000 (Barton 1994)).

Values for coastal defense are most often calculated as damages avoided from flooding and erosion due to the presence of an intact reef (Spurgeon 2001). Techniques for valuing the coastal defense function include percentage dependence, whereby the total value of economic activity, property, and infrastructure protected by the reef is calculated and multiplied by its degree of dependence on the reef. The change in productivity technique can be used to assess the differences in coastal defense value between healthy and degraded reefs. The replacement cost may also be used, whereby the value of coastal defense is assumed to be equal to the amount spent on alternative coastal defense provisions such as seawalls or concrete tetrapods to replace the protection formerly provided by the reef (Spurgeon 1992).

The percentage dependence technique is the most commonly used technique, and property values are most the commonly used proxy for the economic value of the reef. Coastal defense values are much higher in developed areas with tourism than for rural or agricultural areas, and these approaches rarely take into account damage to infrastructure such as schools and hospitals and other damages, so they are likely to be underestimates. The proportion of dependence on reef protection is determined by land use, sensitivity of the shoreline to erosion and flooding, the frequency and magnitude of storms, distance of the reef from shore, underlying coastal processes, and the wave absorbing capacity of the reef.

In practice, calculating accurate values for coastal defense that take into account many of the determinant factors is difficult and costly to do, so benefit transfers are often used to estimate coastal defense values based on values calculated for similar areas (Cesar 2000). Van der Ploeg et al. (2010) created the The Economics of Ecosystems and Biodiversity

(TEEB) Valuation Database, a searchable database of ecosystem service values for ecosystems worldwide, which facilitates well-informed calculation of benefit transfers (the database can be downloaded at http://www.fsd.nl/esp/80763/5/0/50).

The World Resources Institute (WRI) has developed a GIS-based model to calculate coastal defense values using a modified avoided damages approach which incorporates many physical factors that contribute to percent dependence, detailed in Appendix D.

Valuation studies

Coral reefs worldwide have been valued at US $29.8 billion yearly; coastal defense benefits are estimated to comprise $9 billion per year, second only to tourism at $9.6 billion, and much greater than fisheries at $5.7 billion per year. Worldwide, the coastal defense function of reefs averages US $31,721/km^2 (Cesar et al. 2003). Some local and regional valuation studies are discussed below; Table 1 in Appendix D includes values from these studies and others, calculated over a range of geography and methodology.

Cesar (1996) calculated ecosystem values for Indonesian reefs, combining categorizations of land use and tourism potential. They identified coastal defense values of US $820/km, $50,000/km, and $1,000,000/km for sparsely populated, moderately populated, and highly developed areas, respectively.

Spurgeon and Roxburgh (2005) calculated the value of American Samoan reefs and mangrove stands, accounting for spatial variation in coastal defense values. Coastal defense values ranged from zero for reefs far from villages to US $0.89/m^2 for reefs near valuable land prone to erosion, the total coastal defense value of American Samoan reefs was $447,000/yr. The analysis did not include sand supply, which adds to the coastal defense function; sand and rubble have been harvested for values of US $90-450/yd^3, yielding short-term economic benefits, while reducing coastal defense benefits.

In Tobago, coral reefs contributed to defense of 50% of the shoreline. Coastal defense provisions provided by reefs were calculated at $18-33 million for 2007, and $450-825 million over 25 years. Because the adjacent land is low-lying, one reef (Buccoo Reef) provided ~30% of the coastal defense value, US $5-10 million for 2007, and $140-250 million over 25 years. Thus, managers should prioritize protection of this reef (Burke et al. 2009).

Valuation studies for reefs worldwide have proliferated as restoration and conservation efforts have expanded, and their methods have become increasingly refined. However, it is not possible to simply add together the economic values of all relevant reef-derived benefits and proclaim the sum to be the reef's value. Some activities are incompatible; coral mining and blast fishing yield economic benefits, yet detract from the values of others, e.g. coastal defense, tourism. Tourism-related infrastructure greatly increases the value of land and thus the value of coastal defense provisions, but coastal development, much of it related to tourism overuse, is a major source of degradation for many reefs (Cesar et al. 2003). In turn, a loss of reef structure may directly harm the tourism industry. For example, beach erosion decreases the value of hotel real estate – for each

lost meter of beach in front of a resort in the Dominican Republic, the per-person hotel rate declines by $1.50 per night. At current rates of erosion, the tourism industry in the Dominican Republic may lose $52-100 million over the next decade (Vermeij 2013).

At best, valuation methodologies can accurately deliver values of reef-derived benefits, but they cannot resolve the issue of incompatibility. Decision-making tools have been adapted for reef restoration and coastal management, allowing managers to evaluate costs and benefits of proposed management schemes. These tools include Benefit-Cost analysis, Multi-Criteria Analysis, and Habitat Equivalency Analysis; these methods, along with a valuation and option appraisal case study, are discussed in depth in Appendix E.

Discussion

Active coral reef restoration is a rapidly growing field; thousands of projects have been undertaken worldwide. Coastal defense is one of the most economically valuable and socially vital ecosystem services provided by reefs, so efforts to restore services lost to reef degradation should include explicit planning for and measurement of coastal defense benefits. However, the survey results indicate that projects rarely incorporate coastal defense into project design (19.6% of respondents reported planning), and even more rarely are coastal defense benefits measured (10% of respondents reported measuring). The literature review results indicate the same trend, with some exceptions, including some of the case studies presented here, most notably the Reef Ball breakwater projects. This makes it extremely difficult to assess the extent to which existing projects might be providing coastal defense. Coral and fish measurements are prioritized more highly, and were measured more often.

The costs of reef restoration projects can be extremely high, with major physical restoration projects costing US millions of dollars, and smaller projects costing in the tens to hundreds of thousands. Lack of funding for maintenance was a reported to be the greatest challenge to success by 63.4% of survey respondents. Similarly, 58% of Caribbean restoration practitioners reported lack of funding to be a significant obstacle in another recent study (Young et al. 2012). Maintenance is important to project success, particularly with those that include transplanting. The lack of funding for maintenance might be exacerbated by the lack of maintenance cost reporting in the literature (Spurgeon and Lindahl 2000), making it difficult to plan for and secure funding for maintenance. Increased reporting of detailed costings for maintenance, as well as increased stakeholder involvement and support, might help alleviate problems with project continuity and maintenance.

43.9% of survey respondents reported lack of effective management as the greatest challenge, which is likely tied to a lack of funding for management and enforcement of regulations. Lack of funding also affects how long projects are monitored. 60% of projects are monitored for three years or less; this makes it difficult to assess project outcomes over longer time periods. It seems that there is a dearth of funding for implementing, maintaining, and monitoring large-scale reef restoration projects that can deliver coastal defense benefits and other valuable services.

Spergel and Moye (2004) detailed 30 mechanisms to generate funding for marine conservation, including grants and donations, development rights, revenues from tourism, fishing, and mining, and for-profit investments. The private sector has become increasingly important and active in natural resource management, largely through public/private partnerships, as management models have shifted from purely biologically-driven to incorporating more social and economic factors (Spurgeon 2004). An example of this type of partnership is the Caribbean Challenge Initiative, which includes 10 Caribbean governments and many corporate and NGO partners, with the goal of conserving coastal habitats and implementing a sustainable finance model to fund marine conservation efforts. The vast majority of this funding is for MPAs, not for active restoration. However, this could be seen as a first step toward securing funding for active restoration after MPAs are established. At the very least, the more fishing and other uses are managed in coastal areas, the more amenable the habitats might become to active restoration efforts.

In a survey of Caribbean reef restoration practitioners, Young et al. (2012) reported that the highest ranked benefit in the Caribbean was public awareness and education, followed by fisheries habitat and increasing coral population. Public awareness was not included as an option in our survey, but it emerged as a major benefit in email communications with restoration practitioners. This was partially related to the general lack of funding: restoration efforts are often financed by resorts for tourism benefits, so exposure and outreach to guests naturally follows. Restoration efforts are also seen as a way of highlighting conservation within communities and getting people involved.

Coral gardening and concrete structures were the highest ranked methods in our study. Coral gardening was also the highest ranked method among Caribbean restoration practitioners, reported by Young et al. (2012). That study found that stabilization and artificial reefs were also ranked highly, and that the cheapest, lowest-technology methods were more effective overall.

Performance of artificial reef modules

Although half a million Reef Balls have been deployed worldwide, no rigorous scientific studies comparing their performance to other substrates have been performed. However, as Kaufman (2006) points out, a vast long-term experiment has begun, and we will see how they work over the long-term in various settings. They provide vertical relief and are easily adapted as breakwaters; their porosity contributes both to their function as breakwaters (Dhinakaran et al. 2012) and to their suitability as fish habitat. They are suitable substrate for coral settlement, and they can be used to stabilize rubble and rocks within the cavities for added stability to the substrate and to the Reef Ball modules. Performance concerns with Reef Balls include scour and settlement into sandy bottoms. It is difficult to directly compare costs of Reef Balls to other methods, and to other types of concrete structures. Reef Beach has reported 10 submerged breakwater projects in tropical waters, three of which involved transplanting corals. Some of these structures have needed extensive modification after storms or excessive beach infill, so long-term maintenance costs could be high. Respondents to our survey ranked concrete structures

(i.e. concrete other than Reef Balls) as more effective than Reef Balls. Studies comparing the effectiveness of Reef Balls and other concrete artificial reef modules would be useful, as concrete materials are often cheaply and widely available; other concrete modules might prove to be a better option than Reef Balls when costs and benefits are weighed.

Industrially produced concrete modules such as SHED structures have been shown to be effective for both coral recruitment and fish habitat (Case study 7, Clark and Edwards 1999), but these are expensive options. Community-based projects have implemented locally-made concrete modules cheaply using available materials with good initial results (Case study 6). Because concrete has performed acceptably as a settling surface, the general effectiveness of concrete structures will be largely determined by how well they are designed for a given site. 3D printing technology has recently been applied to artificial reef structures that have been deployed in Bahrain, using a special concrete (www.sustainableoceans.com.au). Costs would presumably be very high for such modules compared to other options, but pricing information is not yet available.

The inconclusivity of studies examining the effects of BioRock technology is cautionary. Goreau (2012) attributes the mixed results to researchers not using the technology properly, particularly by using excessive voltage and current densities. Goreau and Trench (2012) contains studies that have found large increases in coral growth and survival, accompanied by copious photographic examples of BioRock's successes. Projects published in Goreau and Trench (2012) used voltages of 3.8-17 V; it is not clear what amperages were used.

It is clear that species-specific effects exist, even among closely related species (Borell et al. 2010, using 12 V, ~2.8 ±0.1 A/m^2). The live current has been shown to inhibit coral settlement, even at very low current densities (~1 $\mu A/cm^2$, Benedetti et al. 2005). More study is needed for BioRock® to become accepted more widely, though some studies have acknowledged its potential (Moberg and Rönnbäck 2003, Rinkevich 2008). The electrolytically deposited mineral accretions are very strong, however, and seem to be appropriate for coral settlement when they are not powered. Materials are cheap and readily available, so using BioRock technology to build artificial reef breakwater structures might be a cost-effective approach, even if the technology does not enhance coral success. If, however, BioRock® is definitively shown to increase coral success, especially survival during bleaching events, it will certainly become an important tool in reef restoration.

There is a paucity of data on the performance of EcoReefs. They are expensive, and we got feedback from survey respondents that the modules are fragile, so they are probably best suited to low energy environments, where they might be less cost-effective than other methods.

Rock and rubble piles performed well as artificial reefs in the case studies presented, providing complex habitat and stability to rubble fields for relatively low costs. Rock piles may be stacked without consolidation and will remain stable in moderate energy environments, but there must be a local source of rocks for this method to be cost

effective. Rubble pieces can be treated similarly, but they must be stabilized by cementing or otherwise consolidated. Simply stabilizing the substrate in any effective way in rubble fields has been shown to facilitate coral settlement and, at least initial, cementation.

Apostolakos et al. (2007) used an MCA to evaluate the suitability of EcoReefs, BioRock®, Rock Piles, and simple transplantation. The study was more an exercise than a conclusive study, as data collection was not rigorous. It was, however, the only objective study that compared marketed restoration technologies with more simple approaches. The objectives considered were Ecological Sustainability, Economic Benefits, and Social Acceptability. Rock Piles and EcoReefs ranked first; Rock Piles due to its high score for Social Acceptability, and EcoReefs for its Ecological Sustainability (assigned due to modules' large vertical surface area). BioRock® was favored by tourists, but it had many drawbacks so was not recommended. Simple transplantation also had too many drawbacks to be recommended. Although no firm conclusions can be drawn from this study, it remains of interest because it allows for both quantitative and qualitative information to contribute to a quantitative ranking of restoration options.

Site considerations to maximize benefits

It is generally agreed that active restoration should not be attempted unless local sources of degradation are removed, and that projects should be selectively done within MPAs or similarly regulated areas to improve chances of success. The average operational management cost of running an MPA is US $775/km^2/yr, which represents less than 0.2% of the value of 1 km^2 of reef in a year (Wells et al. 2006). In Case study 9, enforcement of no-take restrictions greatly improved the success of the restoration. This reinforces the importance of properly funding coral restoration by funding enforcement as well.

Coastal protection benefits associated with reef restoration can also be maximized in areas where seagrasses and mangroves occur or may be restored shoreward of the reef. Healthy reefs create sheltered conditions that are suitable for mangrove and seagrasses, which in turn filter out terrestrial sediments and nutrients in runoff. Mangrove restoration is less costly than coral reef restoration (Wells et al. 2006), so combining restoration of both habitats would create more benefits and could enhance the cost effectiveness of reef restoration projects.

Coastal defense benefits provided by artificial reefs can be enhanced by transplanting coral fragments or colonies to the structures, adding frictional drag and sediment sources for beaches. Even low technology methods can be costly, and are generally only suited for low to moderate energy environments, so transplantation is generally recommended only when larval supply is low and sources of degradation are mitigated. However, transplanting may be desired to increase outreach of a project, or when natural colonies are threatened by activities such as dredging. In some areas, such as Colombia's National Parks, artificial reefs are not permitted, so transplanting to the reef is the primary active restoration activity. In Rosario and San Bernardo National Parks, transplanting projects are designed to provide coastal defense, although these benefits are not measured

(Esteban Zarza, pers. comm.). Transplanting projects can also be designed to provide substrate stabilization by targeting coral species that form basal plates, such as *Porites rus* (Rojas et al. 2008). Sponge fragments show promise for stabilizing coral rubble; their use is non-consumptive and they can cement to rubble pieces within days (Biggs 2013).

Conclusions

Coastal defense functions of degraded reefs can be restored nearly instantly by the installation of artificial reefs when deemed appropriate, and on decadal timescales by stabilizing substrate in rubble fields where coral larval supply is plentiful. Maintenance of nurseries that rear thermally-tolerant and endangered corals to be transplanted to artificial reefs, or to reefs after disturbances, can promote reef accretion and shoreline stabilization by providing carbonate sources to reefs while maintaining critical genetic diversity. Artificial reefs should be designed to provide vertical relief and complex structure; wide widths attenuate more energy, other design components being equal. Rock and rubble piles and concrete structures are likely to provide the most cost-effective means of achieving these characteristics, depending on availability of materials and site requirements. Artificial reefs should be designed to maximize coastal defense in the absence of corals, but active reef growth is essential to keep pace with sea level rise and to provide sediment to beaches. Primary targets for coastal defense-oriented restoration projects should be low-lying areas that have lost reef structure that are not protected by traditional structures, as such areas are most vulnerable to flooding. Lack of funding is considered to be the biggest challenge to the success of projects by restoration practitioners.

Acknowledgements

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Appendix A – Survey

Reef Restoration projects

This survey gathers information for evaluating the relative success, costs, and benefits of different coral reef restoration approaches. We seek information on many types of benefits, but we are particularly interested in coastal defense benefits (e.g. erosion reduction, wave attenuation). Thanks in advance for your feedback.

If you would like a copy of the survey results, please enter your email address at the last question.

1. Which ecosystem services or benefits have your reef restoration projects been designed to deliver?

- Fish production
- Coastal defense
- Recreation (e.g. diving)
- Coral recovery
- None
- Other (please specify)

2. Which ecosystem services or benefits have you measured during or after restoration projects?

- Fish production
- Coastal defense
- Recreation (e.g., diving)
- Coral recovery
- None
- Other (please specify in box)

What metrics are used? (e.g., number of fish, beach widening)

3. In your experience, please indicate the quality of these methods for delivering restoration project success. If you have not used a method, please leave it blank.

	Excellent	Good	Fair	Unacceptable
BioRock				
EcoReef				

ReefBalls

Rubble unconsolidated

Rubble consolidated (e.g., gabion)

Other concrete structures

Geotextile tubes

Coral Gardening (e.g., transplants)

Coral larval seeding

Other (please specify)

Optional: Specify main reasons why methods were satisfactory or unsatisfactory.

4. What was the total cost of your largest restoration project?

< $10,000

$10,001-$50,000

$50,001-$100,000

$100,001-$500,000

$500,001-$1,000,000

$1,000,001-$3,000,000

>$3,000,000

Specify exact amount, or currency if non USD.

5. Please provide citations or URLs for publications, websites, fact sheets or other references on this project or on other projects designed to deliver ecosystem services (particularly coastal defense).

The following 10 questions are aimed at gathering more detailed information about project costs, monitoring,challenges, and success of the project. We would appreciate your feedback on these additional questions if you have time. Otherwise you can skip to the end - and only if you want- enter your email address to receive the results.

6. What are the top 3 ecosystem services (benefits) considered when designing your restoration projects? Please indicate the order of importance, 1 (top) to 3.

	1	2	3	Not Considered
Fish production				
Coastal defense				
Recreation (e.g., diving)				
Coral recovery				
Water quality enhancement				
Research/education				
None				

Other (please specify)

7. How long, on average, are projects monitored after restoration activities are completed?

- < 1 year
- 1-3 years
- 4-5 years
- 6-10 years
- > 10 years

8. What was the total area restored in your largest project?

- < 10 sq m
- 11-100 sq m
- 101-500 sq m
- 501-1,000 sq m
- 1,001-3000 sq m
- 3,001-10,000 sq m
- >10,000 sq m

Give exact dimensions, if known.

9. How many reef restoration projects have you worked on?

1

2-3

4-6

>6

10. If you have used transplanting in projects, what is the average survival of outplanted corals?

< 10%

11-25%

26-50%

51-75%

> 75%

How long after project implementation do you measure survival?

11. What have been the greatest project costs? Please indicate 1 (greatest) to 3 (least).

	1	2	3
Pre-construction costs (feasibility studies, site surveys, etc.)			
Materials			
Labor			
Equipment (e.g. boat time)			
Maintenance/Monitoring			

Other (please specify)

12. Which of the following site conditions do you most consider when planning a project? Please indicate the 3 most important conditions.

Wave energy/height

Coastal geomorphology/exposure level

- Angle of wave incidence
- Current strength/direction
- Storm frequency/seasonality
- Temperature range
- Water quality
- Existing coral genetic diversity
- Health of surrounding coral populations
- Presence/absence of predators
- Historical range of coral
- Existing management zones
- Distance of reef from shoreline
- Coastal vegetation
- Larval supply
- Longshore drift rates
- Development of shoreline property
- Human population adjacent to reef
- Depth
- pH
- Water exchange
- Other (please specify in box)

Specify top 2-3 most important conditions

13. Which of the following anticipated environmental changes have been considered when planning projects?

- Sea level rise
- Sea surface temperature increase
- Ocean acidification (pH decline)
- Storm intensity increase
- Human activity increase
- None

Other (please specify)

14. What are the greatest challenges to success of your projects? Please indicate the 3 biggest challenges.

- Bleaching
- Water Quality
- Overfishing
- Sedimentation
- Storm damage
- Predation
- Disease
- Lack of funding for maintenance
- Damage from anchors, recreational activities
- Lack of effective management/political problems
- Vandalism
- Other (please specify in box)

Indicate top 2-3 challenges.

15. What causes of degradation have affected project sites before restoration efforts?

- Physical damage from storms
- Physical damage from ship grounding
- Physical damage from recreational use (e.g. diving, anchors)
- Physical damage from fishing (e.g. blast fishing)
- Physical damage from coral mining
- Bleaching
- Poor water quality
- Sedimentation
- Biotic changes (e.g. Crown-of-thorns outbreak, disease)
- Overfishing
- Direct human contact

Other (please specify)

16. Thank you for your participation. If you would like a copy of the survey results, please enter your email address in the box below.

Appendix B: Costing framework for Restoration Projects

The costing framework developed by Edwards et al. (2010) is broken into six stages of a project, any of which can be omitted if irrelevant: (1) collection of source material; (2) set-up of nursery/culture facilities; (3) establishing coral material in nursery; (4) maintenance of cultured corals; (5) transfer of corals from nursery to restoration site; and (6) maintenance and monitoring of transplants at the site. Boat costs and fuel are likely to be significant if sites are offshore, and fuel will be a major operational cost if considerable distance must be covered; SCUBA will be required if sites are > 3m deep. At each stage, labor should be reported in person/hours so that costs can be scaled to local wages. Boat costs should be reported in number of boat days, and SCUBA costs should be reported in number of air tanks necessary to do a task. The amount of coral material should be reported for steps 1-5, so that costs may be scaled, and equipment required at each step should be itemized. Set-up and running costs should be differentiated when applicable, and lifetimes of facilities and equipment should be noted so the cost can be spread over the appropriate time period. To enable ease of comparison among methods, nursery costs should be reported as cost per coral surviving to transplantable size, and for outplanting, costs should ideally be reported as cost per coral colony surviving to maturity, but few projects have sufficient monitoring capacities for this to be feasible. Scientific monitoring is considered as a separate cost from maintenance and monitoring, as it does not directly contribute to project success. The authors have made an Excel spreadsheet available (www.gefcoral.org/Targetedresearch/Restoration/Informationresources/Costingrestoration; Figure 1B) to assist with planning projects. A "What if?" function is included so that changes to costs can be anticipated for different scenarios, e.g. varying survival in nursery or at transplant site. This framework could be expanded for coastal defense projects by explicitly including costs of labor and materials required for substrate preparation and artificial reef installation. Frameworks for pre-construction costs, which can be substantial, should also be included in costing schemes for coastal defense projects.

Asexual rearing to produce 10,000 coral fragments/year using *in-situ* modular tray nurseries

Local wage rates	Monthly	Monthly/160 Hourly
Skill level 1 (highest) salary - e.g. scientific adviser/expert	$900	$5.63
Skill level 2 (medium) salary - e.g. trained educated local	$560	$3.50
Skill level 3 (lowest) salary - e.g. trained manual labour	$210	$1.31

Local rates (air fills and boat hire)			
air per tank	$2	boat per day	$30

Task	Equipment/ consumables *USD*	Time input by personnel *Person-hours*	*Rate*	*USD*	*# air-tanks*	*USD*	*Boat time days*	*USD*	Total cost/yr *USD*	% total cost/yr
1. Collection of source material for 10,000 fragments	$30	20	$3.48	$70	20	$40	3	$90	**$230**	1%
2. Setting up 10,000 fragment tray nursery	$1,650	378	$2.73	$1,031	54	$108	6	$180	**$2,049**	7%
3. Establishing 10,000 fragments in nursery-culture	$142	1764	$2.41	$4,245	504	$1,008	63	$1,890	**$7,285**	26%
4. Maintenance of material in *in-situ* culture (1 year)	$109	720	$1.31	$945	240	$480	60	$1,800	**$3,334**	12%
5. Transfer to and attachment of 10,000 juveniles at restoration site	$815	3200	$2.41	$7,700	1200	$2,400	100	$3,000	**$13,915**	50%
6. Monitoring and maintenance at restoration site (1 year)	$0	192	$1.31	$252	96	$192	12	$360	**$804**	3%
Annualised total (nursery construction cost split over 3 years)	**$1,826**	**6274**		**$14,242**	**2114**	**$4,228**	**244**	**$7,320**	**$27,616**	100%
Capital equipment - costs split over 3 years	**$1,380**									
Diving equipment - costs split over 3 years	**$3,200**									

Diving gear	*Per set*	*Rate*	*Persons*	*Total*	
	Daily hire	$20	2 or 4	$16,520	
	Purchase	$800	4	**$3,200**	Cheaper to buy 4 sets

Boat days with 2 divers	75
Boat days with 4 divers	169

What if? Scenarios							
1.	% survival in nursery	***100%***	***95%***	***90%***	***85%***	***80%***	***75%***
a)	Cost per 1-year fragment in nursery	$1.40	$1.47	$1.55	$1.64	$1.75	$1.86
b)	Cost per 1-year transplant at restoration site	$2.79	$2.93	$3.10	$3.28	$3.48	$3.72
2.	% survival at restoration site (over one year)	***100%***	***95%***	***90%***	***85%***	***80%***	***75%***
	Cost per 2-year transplanted colony at restoration site	$2.87	$3.02	$3.19	$3.37	$3.59	$3.82

Figure 1B: Costing scheme for coral gardening project, including "What if?" scenarios (www.gefcoral.org/Targetedresearch/Restoration/Informationresources/Costingrestoration, Example 7.1a)

Appendix C – Reef Ball projects

Table 1C: Reefball Breakwater Projects

Country	Location	What	Notes
Antigua	Maiden Island	breakwater with transplants, ~3500 Reef Balls on windward and leeward sides of island; completed 2004	successful as of 2007; Stanford Corp has 50 molds, but authorization has been pulled by RBF (case study 3)
Bahamas	Island School, Eleuthera	erosion control project; > 50 Reef Balls planted in groups; started in 2000	sponsored by Bahamas Reef Environment Educational Foundation
Cayman Islands	Marriott Hotel, Seven Mile Beach, Grand Cayman	breakwater, 5 rows, 200 modules; completed 2002-2005, corals transplanted 2006	beach has accreted (case study 2)
Dominican Republic	Gran Dominicus Resort	breakwater with transplants, ~450 Reef Balls, 3 sections of 3 rows; completed fall 1998, modified through 2001	beach eroded from 1998 hurricanes, was nourished; natural coral recruitment; beach stable to 2007 (case study 1)
Dominican Republic	Iberostar Resort	photos show irregularly shaped 3-4 rows; completed 2000	beaches stable to 2007; adjacent to Gran Dominicus project
Dominican Republic	Canola Hilton	three 35m long breakwaters, 100s of Reef Balls; ~2004 (?)	
Maldives	Thulhagiri Resort, North Male	pilot project, ~10 units; ~2000 (?)	work done by Cosmo Enterprise Pte. Ltd.; Sinbad Pte. Ltd.
Mexico	Lighthouse at Yalkubl Merida	erosion control project, 90 Pallet Balls; appears to be 3 rows from photos	beach accumulation as of 2005; Enrique Alvarez Del Rio, coastal engineering consultant; Javier Dajer Miguel Comsolida, licensed contractor
Mexico	Hotel Mayan Palace, Quintana Roo	breakwater, 1560 Reef Balls; completed 2002-2003	not anchored properly, wrong concrete mix used; many units destroyed by hurricanes; Reef Beach no longer allows clients to build own breakwaters because of outcome
Mexico	Puerto Progreso	120 Reef Balls, completed 2002	working as of May 2005; stable beach
United States	Carteret Community college, Morehead City, North Carolina	sill for erosion control, estuarine habitat restoration; ~2006	not enough rows, tight enough spaciing as recommended by Reef Beach; was used along other methods (oyster shells, stone breakwaters, etc) along 1200 ft; unclear if successful
United States	Chesapeake Ranch Estate, Maryland	3 rows; completed ?	did not stop erosion

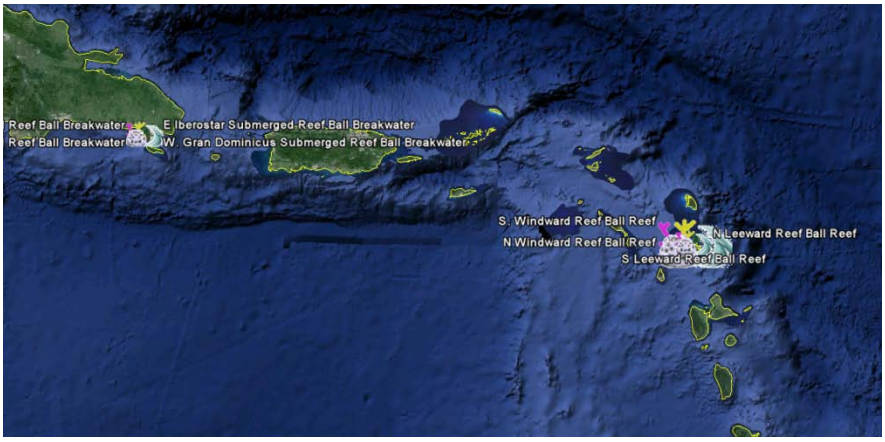

Figure 1C: Reef Beach Projects in the Caribbean; there is one further project in the Maldives.

Table 2C: Countries with Reef Ball projects

American Samoa	Hong Kong	Qatar
Antigua	Indonesia	Russia
Australia	India	Saudi Arabia
Bahamas	Iran	Scotalnd
Bahrain	Italy	Singapore
Barbados	Jamaica	Spain
Belize	Kuwait	Sri Lanka
Bonaire	La Reunion Island (France)	St. Estatia
Brazil	Malaysia	St. Lucia
Brunei-Muara	Maldives	St, Maarten
Canada	Mexico	St. Thomas
Cayman Islands	Nevis and St. Kitts	Switzerland
Columbia	Mustique	Taiwan
Costa Rica	Netherlands	Tahiti
Curacao	Maldives	Thailand
Dominica	Mexico	Trinidad and Tobago
Dominican Republic	Nevis and St. Kitts	Turkey
Ecuador	Mustique	Turks and Caicos
El Salvador	Netherlands	Ukraine
England	New Zealand	United Arab Emirates
Germany	Oman	United States
Guam	Palau	Venezuela
Guatemala	Philippines	
Haiti	Puerto Rico	

Appendix D – WRI/IMA Modified Avoided Damages Approach and Local/Regional Coastal Defense Values

First, the model identifies vulnerable land, defined as land within 1km of a coast that is at lower elevation than the projected combined storm surge and wave height of a 25 year storm. Next, protected land is identified, which includes land within 100m of fringing reef, land enclosed by a barrier reef, or land adjacent to windward shoreline that lies within the wind shadow of a reef or atoll in areas where barrier reefs occur far offshore. The relative stability (ability to resist erosion) is then ranked, using a coral reef index (incorporating reef type, continuity, and distance from shore) and up to nine physical protection factors, depending on availability and quality of data and geomorphological expertise. These factors include: coastal geomorphology, geology, presence of coastal protection (by either reefs or traditional structures), wave energy, frequency of hurricane events (by category), coastal elevation, coastal vegetation, and presence of damaging anthropogenic activities (e.g. sand mining). Each protection factor is rated 0-4 (see Table 1), with 4 indicating highest stability. For a given segment of coast, the coral reef index is added to the physical protection factors and an average is taken, yielding the relative total coastal protection (RTCP). The authors suggest five factors as being the minimum number incorporated into the model with some expectation of applicability (Burke et al. 2008).

Factor	***Level Of Coastal Protection***				
	Very High 4	**High 3**	**Medium 2**	**Low 1**	**None 0**
Coastal Geomorphology	Rocky, Cliffed Coastline	Soft (Limestone) Cliffs or Low Bluffs	Mangroves	Beaches	N/A
Coastal Geology	Igneous and/or Volcanic	Metamorphic	Sedimentary	Unconsolidated Sediments	N/A
Coastal Protection Structures	Protected by 2 prominent headlands and breakwater	Protected by 2 prominent headlands	Seawalls, Riprap or Breakwaters	Protected by one or two small headlands	No protection by headlands
Wave Energy (~ Max. Wave Height [cm])	< 20	20 – 40	40 – 60	>60	N/A
Coral Reef Index *(sum of 3 factors / 10 *4)* **Reef Type** **Reef Distribution** **Reef Distance (m)**	 Barrier Not applicable (N/A) < 250	 Patch N/A 250 - 500	 Fringe Continuous 500 - 1000	 Apron Discontinuous > 1000	 No reef present No reef present No reef present
Storm/Hurricane Events	Affected by 1-5 Tropical Storms every 10 years	Affected by at least 5 Trop. Storms every 10 years	Affected by at least a Category 1 every 25 years	Affected by at least a Category 3 every 25 years	N/A
Coastal Elevation (m)	> 12	5 – 12	1 – 5	0 – 1	< 0 (N/A) **
Coastal Slope (%)	6.2 – 9.7	2.6 – 6.2	1.1 – 2.6	0.4 – 1.1	N/A
Coastal Vegetation Index *(average of 2 factors)* **Type** **Distribution**	Mangroves > 75 % length of coastline	Coastal Woodlands 50% - 75 % of length	Thicket 25% - 50 % of length	Runners < 25% length of coastline	None No Vegetation
Coastal Anthropogenic Activities	No sand mining, coastal development, etc.	Misc. Other Activities	Either sand mining or coastal development	Sand mining and coastal development	N/A

Table 1D, from Burke et al. 2008

The relative stability imparted by the reef is then calculated by removal of the reef from the map and recalculation of the physical factor values. Percentage dependence, or relative reef contribution (RRC) is the percent decrease in total coastal protection factors (non-averaged) when the reef is removed. The RRC is calculated by dividing the reef index by the sum of the coastal protection factors, dividing this result by the RTPC, and raising the resulting value to the 0.5 power. This adjustment accounts for the disparity between the value given by simply averaging the reef index and protection factors, and

field observations of wave attenuation by reefs, as well as the variation in number of protection factors considered, given constraints on available data. (Burke et al 2008).

Finally, avoided damages are calculated by multiplying average property (land and built structure) values in areas classified as vulnerable and average property values in areas classified as protected by reef by the RRC and summing these values. Business revenues may also be incorporated, and a distinction between property values for land within 100m of the coast and those of land further inland yields more accurate results. The sum is multiplied by 4%, (to reflect the probability of a 25 year storm occurring in a given year), which gives the coastal defense value imparted by reefs on a yearly basis (Burke et al. 2008).

The WRI/IMA model is the most complete in accounting for many factors that affect coastal defense values, and it allows for modeling and planning for scenarios with varying wave heights and storm surge. It is somewhat amenable to benefit transfers, but it requires high-resolution data for coastal elevation, reef locations, and other coastal characteristics. It is also limited by availability of data on wave-induced damage caused by storms, thus calculated values should be seen as means within a confidence interval defined by the quality of available data. The method only provides valuation for a moment in time, and does not account for whether activities that affect provision of ecosystem services are sustainable; it also cannot be used to project declining coastal defense values with reef degradation, as quality data are necessary for each step toward degradation (Burke et al. 2008).

Table 2D: Local and regional coastal defense values

Location	Value	Method	Source
Indonesia - sparsely populated areas	US $820/km	avoided storm damages - agriculture	Cesar 1996
Indonesia - densely populated areas	US $50,000/km	avoided storm damages - housing, roads	Cesar 1996
Indonesia - locations where tourism is main activity	US $1,000,000/km	avoided storm damages - cost of maintaining sandy beaches	Cesar 1996
Indonesia/Philippines	US $5,500-110,000/yr	extrapolation from original calculation	Burke et al. 2002 (citing Cesar 1996)
West Lombok, Indonesia	US $905,000	cost of coastal protection structures to date	Riopelle 1995
Sampela, in Indonesia's Wakatobi National Park	£6898/yr; £58,730 present value (US $10,611/yr; $90,344 present value)	avoided damages - survey-based	Hargreaves-Allen 2004
Lami Town, Fiji	FJD 471/ha, 653,277/yr (US $248/ha, $343,624/yr)	avoided storm damages - building values - GIS	Rao et al. 2013
Navakavu, Fiji	US $826,140/yr; $9,086,882 PV over 99 years	benefit transfer	O'Garra 2012
New Caledonia	€115,000,000-219,400,000 total; €23.37-44.58/km^2/yr (US $152,513,000-290,968,280 total; US $31-59/km^2/yr)	modified avoided damages approach (WRI/IMA)	Pascal 2010
North Efate, Vanuatu	Int $38/ha, 29/capita (US $38/ha, 29/capita)	modified avoided damages approach (WRI/IMA)	Laurans et al. 2013 (Pascal 2011a)
Saipan, Northern Mariana Islands	US $8.04 million/year	avoided storm damage - GIS	van Beukering et al. 2006
Sri Lanka	US $160-$172,000/km^2/yr	avoided damages - erosion rates, land values	Berg et al. 1998
Sri Lanka	US $1,230,000-$4,100,000/km^2/yr	preventative expenditure (cost of replacement)	Berg et al. 1998
Guam	US $8.4 million/yr	avoided damages - building values - GIS	van Beukering et al. 2007
American Samoa	US $447,000/yr	preventative expenditure (savings from delaying installation/repair of coastal defense structures)	Spurgeon & Roxburgh 2005
Florida	US $170,000/km	preventative expenditure (cost of replacement)	Spurgeon 1999 (citing Posvord Duvivier 1996)
Caribbean - low development	US $2,000-20,000/km ($10-30 million total)	modified avoided damages approach (WRI/IMA)	Burke & Maidens 2004
Caribbean - medium development	US $30,000-60,000/km ($120-150 million total)	modified avoided damages approach (WRI/IMA)	Burke & Maidens 2004
Caribbean - high development	US $100,000-1,000,000/km ($620-2000 million total)	modified avoided damages approach (WRI/IMA)	Burke & Maidens 2004
St. Lucia	US $28-50 milion/yr; $700 million - 1.2 billion over 25 years	modified avoided damages approach (WRI/IMA)	Burke et al. 2008
Tobago	US $18-33 million/yr; $450-825 million over 25 years	modified avoided damages approach (WRI/IMA)	Burke et al. 2009
Belize	US $120-180 million/yr	modified avoided damages approach (WRI/IMA)	Cooper et al. 2009
Turks and Caicos	US $16.9 million/yr	coastal erosion, storm/hurricane damage	Carleton and Lawrence 2005
Montego Bay	US $107 million net present value	avoided damages - erosion rates, land values	Gustavson 1998
St. Martin Natural Reserve	€179,400-368,000 annual profits derived from MPA (US $237,000-488,042)	modified avoided damages approach (WRI/IMA)	Pascal 2011b

Appendix E – Option Appraisal Techniques

Benefit-Cost Analysis

Benefit-cost analysis (BCA) is used to assess the relative costs and benefits of a particular use of resources, so that various potential uses of resources can be compared. For coral reef managers, BCA can reveal the best option among various active and passive rehabilitation measures, as well as inaction. In BCA, all costs and benefits (TEV) associated with each option must be identified and assigned a monetary value. "Present-day" costs and benefits are calculated through discounting, which is similar to but opposite of adding interest, as benefits delivered in the future are less economically valuable than benefits currently being supplied. Present-day costs and benefits are then compared using the benefit-cost ratio:

$$BCR = \frac{\sum_{t=0}^{T} B_t / (1+r)^t}{\sum_{t=0}^{T} C_t / (1+r)^t}$$

where t = time over which costs and benefits are considered, and r = discount rate. An option whose BCR > 1 would be justified, and higher BCR values indicate increasingly efficient uses of resources (Spurgeon 2001).

BCA allows for quantitative comparisons among schemes – all options are evaluated with the same methodology, so if valuations are accurate, the choice with the highest BCR reflects the best use of resources. Because discounted values are used, a BCA tends to be biased against restoration, as benefits from restoring degraded habit develop over time (Spurgeon 2001). When considering whether or not to implement a restoration scheme, natural rates of recovery should be compared with projected recovery rates from project implementation. This requires knowledge of recruitment and growth rates for the site, as well as environmental parameters that influence recruitment and growth (Spurgeon and Lindahl 2000). This contributes to one of the drawbacks of BCA: as discussed in the previous section, accurately valuing all costs and benefits is a formidable and expensive undertaking. Benefit transfers can be used to estimate some values, although their use is sometimes controversial (Cesar 2000, Spurgeon 2004).

BCA does not deal well with non-marketed values. Non-use values such as cultural importance are substantial and can often exceed use values of coral reefs (Laurans et al. 2013, Spurgeon 2004). Non-use values are estimated using a Contingent Valuation Method, which uses surveys to indicate people's Willingness to Pay for the existence of a resource. These methods are susceptible to multiple biases (discussed in Cesar 2000), and there is concern that WTP conclusions are biased toward delivering benefits to the wealthy rather to than the poor (Spurgeon 2004); in many coral reef communities, especially in the Pacific, the cash economy is not well-established, and local users are not inclined to assign monetary values to reef benefits. There are quantitative methods of measuring cultural importance, such as the community's degree of familiarity with the reef, measured as the number of place names per km^2, or the number of fish species named. Qualitative measures such as the role of the reef in community identity, or in its

social and political relationships with other communities, can also help define the cultural importance of reefs (Laurans et al. 2013).

Least-cost Analysis and Cost-effectiveness analysis

BCA can be combined with other types of option appraisal analyses to assist in decision-making. If environmental objectives are agreed upon, a least-cost analysis (LCA) is a straightforward means of identifying the least expensive way to achieve certain goals. Cost-effectiveness analysis (CEA), which identifies the most cost-efficient means of achieving objectives, incorporates different degrees of improvement and their costs to help managers decide how much improvement is the most cost-effective (Spurgeon and Lindahl 2000). These techniques require knowledge of costs of the various schemes being weighed, but do not necessarily require valuation of all benefits, as they are applied when objectives have already been identified.

Multi-Criteria Analysis

Multi-Criteria analysis (MCA) is another type of decision-making tool, in which management objectives are defined, prioritized, and scored (Figure 3a, below). After objectives are weighted, management schemes can be evaluated under the criteria established by the MCA. MCA is well-suited for situations with many potentially conflicting stakeholder interests, and when there is limited data available. It is a time-intensive process, requiring extensive effort to gather and process stakeholder opinions on objectives, but is powerful in that it combines qualitative and quantitative analysis of options (Fernandes et al. 1999).

Fernandes et al. (1999) used MCA to identify management goals for Saba Marine Park, Netherlands Antilles, when concerns were raised that the park was being affected by tourism overuse. Interviews of the public and surveys of school children were used to gather stakeholder opinions about what the park's objectives should be. Multiple public meetings were held to structure the objectives, ensuring stakeholder involvement in

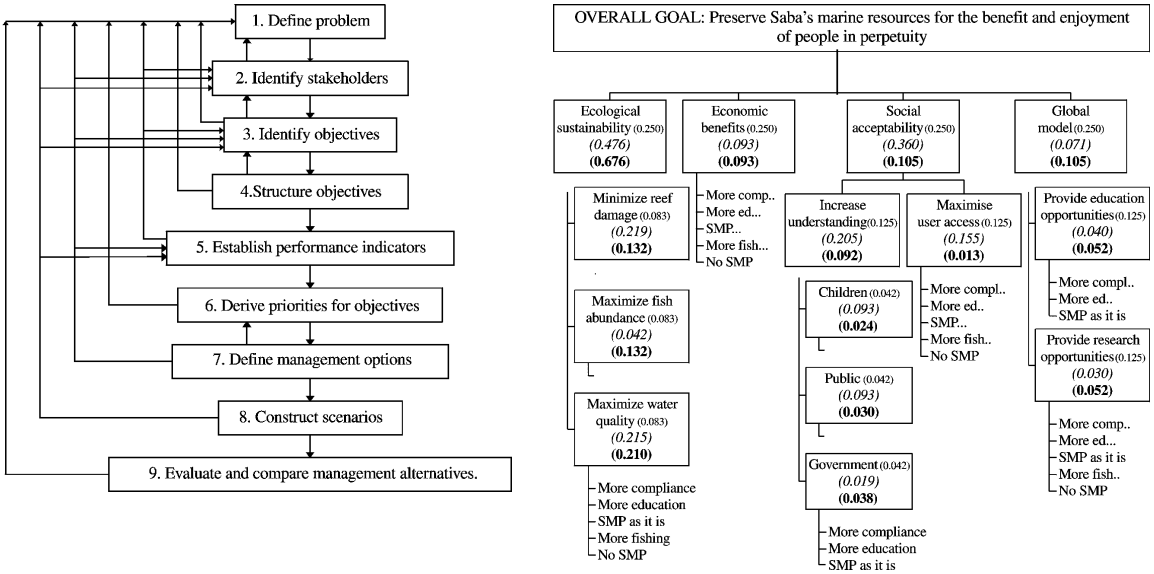

Figure 1E: Steps involved in MCA (a), from Fernandes et al. 1999; Saba Marine Park objectives (b), from Fernandes et al. 1999

the process. An overall goal of preserving and sustaining the marine resources of the park for the benefit of people was identified, with a hierarchy of supporting objectives (Figure 3b). The objectives were prioritized, and performance indicators for each objective were established (e.g., physical damage per number of coral colonies indicates the objective of minimizing reef damage). Management scenarios were constructed, and experts used an analytical process to identify to what degree the lowest-level objectives would be met under each scenario. The relative value of each scenario was determined using the weighting of objectives and expert analysis of lowest-level objectives. Final recommendations included retaining the park's "no fishing" zone; increasing enforcement of regulations and responses to community needs, including involvement in curriculum development at schools; and communicating positive messages about the marine park to the government. The authors note the strengths of MCA in facilitating a consensus among conflicting stakeholders, and the emergence of enforcement and education as highly valued by the community. It is an effective means of conveying the importance of Saba Marine Park to the community, and their goals for park management, to the government. The hierarchical process does have flaws that can alter results, so users of the technique must be well-educated on the process (Fernandes et al. 1999).

Habitat Equivalency Analysis

Habitat Equivalency Analysis (HEA) is a method that uses both biological and economic data to determine proper compensation for loss of coral due to damages, primarily due to ship groundings. HEA seeks to ensure that ecological services provided by restoration or replacement are equivalent to those services that were lost over the period between damage and return to baseline ecological functioning. It involves quantifying damage, determining the intrinsic value of the damaged reef, analyzing possible restoration schemes, estimating natural recovery rates, choosing compensatory actions, and scaling of the project over time until baseline ecological function is obtained. The most appropriate means of compensatory restoration may include outreach and education programs, better signage marking sites where grounding is a risk, establishment of nurseries or artificial reefs, monitoring studies to evaluate the success of past restoration projects, and a host of other measures (Precht and Robbart 2006). HEA has many limitations, and is discussed in detail in Milon and Dodge (2001). Because sites where coastal defense benefits are highly valued correspond with concentrated human activity, it seems unlikely that this method would be used to evaluate coastal defense-motivated projects, but lessons learned from HEA implementation would certainly be informative in planning projects.

Valuation and Option Appraisal case study, Lami Town, Fiji

As part of the UNEP Ecosystem-based Adaptation Flagship Program, an economic analysis of adaptation scenarios to prevent flooding in Lami Town, Fiji, was undertaken. The goal was to identify ways that Lami Town could use existing and potential ecological and engineering systems to adapt to climate change-associated risks over a 20 year period. At a participatory town council meeting, it was agreed that more frequent and intense storms posed the greatest threat, so storm protection was the focus of proposed action. Possible ecosystem-based and engineering actions were listed, as well as social

and policy options. Ecosystem-based options included revegetation, conservation, and preservation of habitats for coastal defense, while engineering options included building structures various such as sea walls and drainage ditches to decrease flooding. Social and policy options included such actions as rezoning, relocating people, and flood warning systems.

Costs associated with all potential actions were assessed, including installation, maintenance, labor, and opportunity costs for each option over 20 years, and an LCA was performed to identify the least-cost measures that would generate varying levels of storm protection. Then, the cost of taking no action was calculated by quantifying damages that would result from the flooding of Lami Town. This was based on lost income from business closure, losses to households, and health costs associated with flooding, and is described in detail below. The authors considered intangible effects of flooding, including increased stress and insecurity, but estimates were not made for these effects, as they are difficult to monetize. Tangible effects were divided into various direct and indirect costs (see Figure 2E, below); estimates of these values were based on extensive surveys following flooding in the nearby towns of Ba and Nadi.

Because a detailed flood map was not available, buildings within 30m of the coast were classified as vulnerable, and these were distinguished between industrial buildings or households. Informal settlements were also accounted for; the estimated 88 informal households identified in Lami Town were assigned the average value for formal houses. The authors note that although informal houses might be of less monetary value than formal houses, flooding damages are likely to represent a much greater percentage of household income for informal households than formal households, and most of the informal houses are in especially vulnerable areas on the coast or river. Household loss values included lost possessions, structural damage, lost wages, clean-up, medical costs, and evacuation costs, totaling FJD 4,153 (US $2,184) per household. Losses to business included structural damage, lost assets, clean-up, disrupted business, evacuation, relocation, extra salaries, and miscellaneous costs, totaling FJD 116,643 (US $61,354) per business. Health costs were estimated based on disease costs due to flooding in Rarawei, a low-lying Ba River town deemed to have similar flooding-related disease effects as Lami Town. Expected health costs after a flood were estimated to be FJD 46,396 (US $24,404) using the per-person estimate for Rarawei.

The total avoided damages due to flooding were FJD 31,157,057 (US $16,388,611) from lost business income, losses to households, and health care costs. Estimates of the costs of repairing government and private infrastructure, provision of flood relief supplies and services, and education losses were not made in the study, but these costs are likely to be high, and funds for government programs would likely be re-budgeted to cover costs of infrastructure and flood provisions. Thus, the avoided damages may be greater than what was calculated. However, the authors note that storms are likely to be more frequent due to climate change, and incremental damage from multiple storms might reduce the value of infrastructure and livelihoods, possibly making the avoided damages value an overestimate. Avoided damage estimates would be greatly improved by a detailed flood map, incorporating the factors used in the WRI/IMA approach to better identify areas that

are most vulnerable to flooding. Additionally, estimates of costs of repairing infrastructure and flooding provisions would make the total avoided damages calculation more robust.

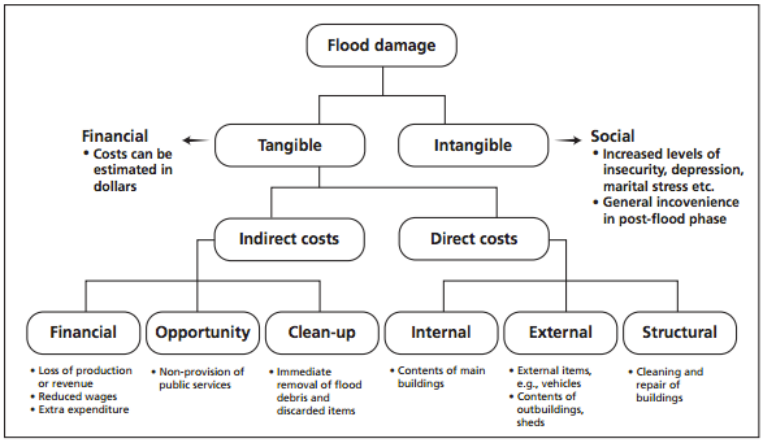

Figure 2E, Costs associated with flood damage, from Rao et al. (2013).

Once avoided damages had been calculated, a BCA was performed for the use of each adaptation option to reduce avoided damages by 10%, 20%, and 50%. Higher BCRs were calculated for the least expensive options of increased monitoring and enforcement of regulations, and reducing upland logging and coral extraction. The most expensive activities (building sea walls, reinforcing rivers, and increasing drainage) had the lowest BCRs. The least expensive options were not deemed to be effective enough by themselves, so four scenarios using ecosystem-based adaptation, engineering adaptation, and two combinations of these approaches were developed and assessed. Ecosystem service benefits provided by the coral reef and other habitats were calculated, using benefit transfers derived from valuation literature spatial analysis of the habitats. Coastal defense benefits of coral reefs were FJD 471/ha (US $248), or FJD 653,277 (US $343,624) per year; incorporating other reef-derived extractive benefits gave an overall value of FJD 658,491(US $346,366) per year. Finally, another BCA was performed incorporating avoided damage estimates, ecosystem service values and opportunity costs. In this analysis, ecosystem-based adaptation had the highest BCR, net present value, and annualized net present value, while engineering options had the lowest values. In each case, action was better than no action.

The analysis resulted in a suite of conclusions and recommendations incorporating all adaptation approaches, including protecting coral reefs and mangroves, strategic use of engineering options to protect built capital, and policy options to move toward an integrated management plan. The recommendations also included the development of a high-resolution elevation map for the area to enable better flood monitoring, and to examine assumptions used in the analysis regarding the effectiveness of the various options and improved costings to refine economic analyses (Rao et al. 2013).

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