Title

Optimization of Economic Policies and Investment Projects using a Fuzzy Logic based Cost-effectiveness Model of Coral Reef Quality: Empirical Results for Montego Bay, Jamaica

Authors

Jack Ruitenbeek*

H.J. Ruitenbeek Resource Consulting Limited
RR#2 – Site 52 – C21, Gabriola Island, B.C., Canada V0R 1X0
Email: <hjruiten@web.net>

Mark Ridgley

TERRA Team, Hawaii, USA;
Department of Geography; University of Hawaii
2424 Maile Way, Honolulu, Hawaii 96822
Email: <ridgley@hawaii.edu>

Steve Dollar

School of Ocean and Earth Science & Technology
University of Hawaii, 1000 Pope Rd., Honolulu, Hawaii 96822
Email: <sdollar@iniki.soest.hawaii.edu>

Richard Huber

World Bank – Latin America and the Caribbean Region (LCSES)
1818 H Street N.W., Washington, D.C. 20433
Email: < rhuber1@worldbank.org>

Keywords ***Cost-effectiveness; fuzzy logic; optimization; reef management.***

*Corresponding Author — Jack Ruitenbeek, Tel +1 250 2478436, Fax +1 250 2478492

Original MS Submission Date — 07 December 1998

Revised MS Submission Date — 15 June 1999

Abstract

For effective mitigation of human impacts, quantitative models are required that facilitate a comprehensive analysis of the effects of human activity on reefs. Fuzzy logic procedures generate a complex dose-response surface that models the relationships among coral abundance and various inputs (e.g., physical damage, sedimentation, nutrient influx), within the context of the abiotic marine environment. This is linked to a nonlinear economic structure incorporating technical interventions (e.g., pollution treatment) and policy interventions (e.g., taxation) in eight economic sectors. Optimization provides insights into the most cost-effective means for protecting coral reefs under different reef quality targets.

The research demonstrates that: (i) it is feasible to use fuzzy logic to model complex interactions in coral reef ecosystems; and, (ii) conventional economic procedures for modeling cost-effectiveness can result in sub-optimal policy choices when applied to complex systems such as coral reefs. In Montego Bay, Jamaica, up to a 20% increase in coral abundance may be achievable through using appropriate policy measures having a present value cost of US$153 million over 25 years; a 10% increase is achievable at a cost of US$12 million.

Introduction

Some coral reef areas in the tropics are under particularly heavy pressure and are deteriorating; a recent World Bank report on coral reefs identified such ecosystems as the highest priority areas for conservation (Hatziolos et al. 1998). This paper provides a description of the methods and results of studies undertaken under a World Bank research project using a least-cost modeling framework for coral reef management and protection. The site investigated looks at the Montego Bay Marine Park and surrounding area with a view to identifying the least cost interventions for coral reef management.

The Montego Bay site was chosen for a number of reasons. Foremost, recent political commitment in the region has resulted in the establishment of the Montego Bay Marine Park as a protected area that will be managed to promote sustainable reef-based tourism while still accommodating a local fishery. Originally under public jurisdiction, a bold experiment was undertaken when the park was transferred to private management in 1996. A group of concerned citizens, which formed the Montego Bay Marine Park Trust in 1992, obtained responsibility from the Government of Jamaica to manage the park under the authority of the Natural Resources Conservation Authority. Moreover, impacts on the park are varied, ranging from over-fishing to pollution impacts from sedimentation, ocean dumping from cruise ships, and influx of nutrients through ground and surface water transport. From an ecological perspective, the area has been studied over a long period of time as there is continued interest in the precise extent and cause of reef degradation (O'Callaghan 1992, Hughes 1994, Sullivan and Chiappone 1994, Louis Berger 1995, Lapointe et al. 1997).

The area is economically important, as it supports a recently established free trade zone and the reefs themselves provide important economic services. Gustavson (1998) places tourism and recreation values at US$315 million and coastal protection at US$65 million; artisanal fisheries are valued at US$1.3 million. Spash et al. (1998) place the non-use benefits of the Montego Bay Marine Park area at almost US$20 million. Ruitenbeek and Cartier (1999) estimate that the area's biodiversity resources have an expected value of US$70 million to the pharmaceutical industry through marine bioprospecting, although none of this value is currently captured under existing institutional regimes. Such values are threatened by ongoing reef degradation.

A key conceptual problem facing policy makers is a lack of quantitative models and procedures designed to facilitate a comprehensive economic and ecological analysis, including identification, measurement and prediction, of the effects of economic activity on coastal marine ecosystems. In particular, the degradation of coral reefs has not been extensively analyzed in a framework amenable to economic policy analysis. This has made it difficult to develop a priority ranking of policy and investment interventions in terms of their cost-effectiveness; i.e., there are no means by which to formulate least-cost plans to control continued deterioration. While this paper focuses its empirical work on Montego Bay, the models developed here are generic in nature, are transferable to other sites, and are relevant to management problems associated with optimizing the benefits achievable from coral reefs and their contiguous coastal ecosystems.

A cost-effectiveness analysis framework is therefore developed; the focal point of the project is to render cost-effectiveness in terms of coral reef management and protection opportunities. The potential scope of the overall general model includes all conceivable economic activities, interventions, and environmental impacts in the coastal zone. The models developed to date, however, are somewhat more limited as they intend to explore selected methodological and practical issues in economic and ecological modeling of coral reefs.

Specifically, three research problems are addressed simultaneously within the current framework:

- <u>Normalization of Impacts</u>. First, we ask whether it is feasible to render the impacts of various economic activities in terms of a single biophysical parameter. Conventional ecological approaches to this problem such as those employed by Tomascik and Sander (1985, 1987) yield "dose-response" functions, but such functions are not typically capable of covering the full range of economic activities.
- <u>Separability of Benefits and Costs</u>. Second, we ask whether it is economically meaningful to separate economic benefits from costs in analyzing management choices. Conventional economic approaches to this problem rely

on integrating benefits and costs (Dixon 1993, Cesar 1996, Berg et al. 1998) or, when benefits are not quantifiable, on ranking choices within a cost-effectiveness framework (Eskeland 1992, Ruitenbeek 1992).

- Identification of Preferred Options. Third, we ask whether one can identify any clear preferred management options for a specific site: Montego Bay, Jamaica.

To place this research in perspective, it is useful to illustrate the management problem in terms of how it is often dealt with using conventional cost-effectiveness frameworks.

Conceptually, a conventional analysis framework would provide a ranking of the cost-effectiveness of various policy or project interventions. The outcome of any modeling effort would be a cost curve of the type shown in Figure 1. This figure shows costs and benefits on the vertical axis, and some indicator of coral reef quality on the horizontal axis. The step-wise cost curve represents a series of interventions, each of which results in a reduction of negative environmental impacts; these interventions will over time cause an increase in coral abundance. The first few interventions are relatively inexpensive, and may have no net costs associated with them if, for example, they concomitantly generate economic benefits not associated with coral reef improvement. Subsequent interventions become more expensive, on a cost per unit basis. Figure 1 also shows a declining benefit curve, which illustrates what is typically called a "damage function." The damage function shows the marginal benefit associated with the reduced environmental damage (e.g., increased fishery productivity, higher tourism potential, or reduced shoreline erosion). Under this conventional construct, an economic optimum occurs where the benefit and cost curves intersect. The framework is often regarded as useful even if benefits are uncertain or not known: in such a case it is often argued that the most cost-effective interventions should be undertaken first and that, from a management perspective, one need only systematically move up the cost curve.

This research, however, places in question this simplified conventional approach. The cost curve of the type contemplated in Figure 1 depends on the separability and independence of individual interventions. In complex systems, such independence rarely exists: cumulative or synergistic impacts of pollutants on reef health, for example, must be reflected in management decisions. Reliance on a conventional cost-effectiveness model can, in such cases, lead to incorrect decisions. We demonstrate this empirically through developing a generic complex systems model that does not rely on the separability assumptions inherent in the conventional model, and through applying this generic model to a practical case study site in Montego Bay, Jamaica. [1]

Methods and Assumptions

General Statement of Problem and Model Structure

The model developed in this research consists of two distinct sub-models: (i) a biophysical or ecological reef impact model relying on fuzzy logic; and, (ii) an economic model describing current and future economic activities, policy interventions and pollution loads in Montego Bay. The sub-models are linked and run side by side either in a simulation mode or an optimizing mode to predict future reef quality, economic activity levels, and economic policies.

The objective of the model is to achieve a target coral reef quality {Q} by identifying an optimal set of interventions {$\mathbf{S}^{o}$} such that the cost {C} of implementing this intervention set is minimized.

As noted above, conventional approaches to this type of problem have used a cost curve formulation, in which the cost of each potential intervention is analyzed along with its resultant impact on reef quality. A measure of cost-effectiveness (in terms of $ per % of coral cover improvement, for example) is then derived. An optimal set of interventions then involves selecting first those interventions with a low cost-effectiveness measure ($ per % improvement), and subsequently moving into higher measures. A supply curve is then derived similar to that shown in Figure 1.

But this conventional approach is flawed in many real life circumstances. The flaw relates to the nonlinear nature of the response function, and the effects of cumulative

---

[1] The focus of this work is on the cost function and is presented in full in Ruitenbeek et al. (1999). The full report provides a comprehensive explanation of methodological assumptions and selection of reef variables and fuzzy logic rules, as well as extensive sensitivity results relating to different assumptions of policy options, initial conditions and planning time horizons. In a complementary World Bank Research Committee project, a benefit function relating to coral reef benefits at Montego Bay is treated in Ruitenbeek and Cartier (1999). These studies, and related background documents, are available from the World Bank or from the following internet site: http://www.island.net/~hjr.

interventions. It is readily shown, for example, that such an approach can lead to non-optimal results if the response function is unresponsive to small changes in inputs (such as sediment or pollution loads) but very responsive to large changes in inputs. In such circumstances, the first intervention will inevitably have very low cost-effectiveness (as it will generate zero response) while subsequent interventions will have higher cost-effectiveness. The appropriate analytical framework is therefore not to look at the problem from the point of view of individual interventions, but from the point of view of a group of interventions having a cumulative effect.

Generalized Optimization Problem

The overall optimization problem involves selecting an optimal level of coral reef quality such that net benefits are maximized. To derive this result, we generate a cost function C(Q) and, in the generalized conceptual cost model, we consider the following:

Q = scalar indicator of coral reef quality (% coral abundance)

$\mathbf{F}$ = vector of biophysical factors that influence coral reef quality

F_j = level of factor j such that $j=1,2,3,\ldots,J$

$\mathbf{S}$ = vector of economic interventions

S_k = level of economic intervention type k such that $k=1,2,3,\ldots,K$

I_k = unit level of economic intervention type k

n_k = number of units of intervention type k such that $S_k=n_k*I_k$ and $n_k\geq 0$

$\mathbf{n} = \{n_1,n_2, \ldots, n_K\}$

C_k = cost of intervention S_k

r = discount rate

The following describes the full least cost optimization problem:

For a given target $\hat{Q}$, Minimize C by choosing **n** subject to:

$$C = C_1 + C_2 + \ldots + C_k \qquad \text{(Eq. 1)}$$
$$Q\{\mathbf{F}\} \geq \hat{Q}$$
$$F_j = F_j\{\mathbf{S}\}$$
$$C_k = C_k\{n_k, r\}$$

The cost function, which can subsequently be used in an overall benefit-cost optimization, is then simply $C=C\{Q\}$ for all technically viable levels of Q. Through simulation or iteration a cost curve envelope can be derived with each point on this curve representing a vector of interventions.

Biophysical Model Structure

The purpose of the biophysical model is to describe the relationship Q{**F**}in the above optimization problem. The general biophysical model is based on a generic coral reef system model (Figure 2). It relies extensively on fuzzy logic based systems in describing a complex dose-response function. The final system, and the selection of input and output variables, was derived based on expert knowledge and incorporates successive results of tests using prototype models.

In general, a reef-impact model should exhibit at least two key features. First, it should represent existing knowledge of reef ecology at a detail and within the bounds of accuracy sufficient for project evaluation. A particular requirement to achieve this aim is the model's ability to show the effects of nonlinear relationships among pollutants, coral reefs, and the reefs' larger marine environment. Second, the model should be operable and provide useful results with the information available at or for any location of potential application. This is a crucial requirement, since quantitative data on many oceanographic and biotic variables are frequently sparse, inaccurate, patchy, of short duration, or otherwise deficient for conventional analytical (i.e., exhibiting closed-form solutions) or numerical modeling. On the other hand, considerable qualitative data are available for almost all reefs of the world. Much of these data are in the form of expert knowledge or human judgment, derived either from formal education or from first-hand experience. In poor tropical countries, the latter may well be the dominant form of information available, in terms both of quality and abundance (Johannes 1981); in some locations it may be the only form available.

These two desiderata correspond to two defining characteristics of the model: (i) the recognition of the role played by the physicochemical environment in influencing the interaction between inputs (such as pollutants) and reef biota and other processes; and, (ii) the use of a fuzzy logic approach to represent cause-effect relations.

How material inputs affect a reef is a function not only of the magnitude and concentration of the inputs and the condition of the reef at the time, but also of such oceanographic variables as hydrodynamics (e.g., mixing and residence time). These determine the concentration and ultimate exposure of the input to the reef, and the supply of chemical reactants, upon which the uptake and utilization of nutrients by biota depends.

Data deficiencies, coupled with marked limitations on resources for reef research and management in the developing tropics, led to the adoption of a fuzzy logic (or fuzzy sets, fuzzy systems) approach. With the theory first introduced in the 1960s (Zadeh 1965), fuzzy logic has proven adept at describing and helping to manage a variety of complex nonlinear systems, initially those dealing primarily with electromechanical control of industrial and manufacturing processes (Kosko 1993, McNeill and Freiberger 1993), but more recently geophysical, ecological and economic systems (Ayyub and McCuen 1987, Kainuma et al. 1991, Bardossy and Duckstein 1995, Munda 1995, Meesters et al. 1998). Fuzzy methods possess a number of features making them particularly applicable to the prediction and management of these latter systems. First, they enable rigorous, quantitative system modeling even though the variables and their interrelationships are described initially (i.e., as inputs to the model) in qualitative terms. This is especially appropriate when human knowledge about the behavior of systems, such as reef ecosystems, is approximate and imprecise at best, rendering adequate parameterization all but impossible. The ability to accommodate qualitative data about reef systems means that more information about them, from more and different kinds of sources, is likely to be available. Since fuzzy logic allows systems to be described as sets of if-then, linguistically-specified rules relating inputs to outputs, it thus offers great potential to utilize human judgment and experiential knowledge, rather than being dependent upon mathematized theory or quantitative databases.

An Informal Introduction to Fuzzy Modeling

We provide here a brief, qualitative reprise of the bare essentials of fuzzy rule-based modeling. More detail is given in Ridgley et al. (1995) and Ridgley and Dollar (1996), as well as in standard references (Kosko 1992, Bardossy and Duckstein 1995, von Altrock 1995).

Fuzzy rule-based models relate a set of inputs to a set of outputs. The inputs in this case refer to nutrient and sediment influx, physical oceanographic characteristics (“mitigators”), and biotic state variables; outputs also refer to biotic state variables, although not necessarily the same as the biotic inputs. Once inputs and outputs have been identified, the first step is to define the range of possible values (measurements) for each one and to divide that range into a set of overlapping intervals. Each interval defines a fuzzy set, referring to a relative magnitude of that input (e.g., high, medium, low); fuzzy sets are thus sometimes referred to simply as "adjectives." Such intervals are based on expert judgment.

Fuzzy sets are so-named because of the ambiguity associated with the membership of certain values in those sets. Such ambiguity is characteristic of the linguistic terms we use to label the sets (again, e.g., high, medium, low). A particular quantitative value (e.g., 25%) could be associated with more than one fuzzy set (e.g., both low and medium). How plausible it is that the value in question belongs to a particular fuzzy set is termed its degree of membership, represented by a number between 0 and 1.0, inclusive. Most quantitative values are associated with more than one fuzzy set, usually to different degrees. A value's membership in a given fuzzy set is determined by its membership function. Membership functions are usually represented as geometric figures – triangles or trapezoids are the two most common ones – whose "tops" correspond to the full membership of 1.0, bases (the intervals defining the fuzzy sets) to a value of 0.0, and sides to intermediate values. Thus, we can conceive of each membership function as having a certain area associated with it.

Given a set of inputs and outputs, their fuzzy sets, and corresponding membership functions, the next step is to specify input-output rules in terms of the fuzzy sets. The set of such rules, called the knowledge domain, defines a mathematical relation and constitutes a fuzzy system, also called a fuzzy associated memory (Kosko 1993.) For example, a (hypothetical) 2-input, 1-output rule could be: "If nitrogen influx is HIGH and residence time is LOW, then coral abundance is HIGH." Each input in a rule is called an antecedent, each output a consequent.

With the knowledge base established, three steps can transform a given set of quantitative inputs, with their corresponding membership degrees, into quantitative outputs. First, a process of scaling determines the degree to which each rule applies, called the rule's activation level. If a rule has a single antecedent, the activation level is the value's membership in that fuzzy set. If the rule has two or more antecedents with different membership degrees, fuzzy logic operators are used to determine the most appropriate activation level. The activation level is then used to scale the output fuzzy set by reducing its area and shape accordingly. Second, in a combination step, all scaled consequents from active rules (i.e., whose activation levels are positive) are combined via superposition – that is, superimposing the scaled fuzzy outputs on top of each other. Third, a defuzzification step transforms the fuzzy composite consequent to a single quantitative ("crisp") output value, either that corresponding to the centroid of the consequent set, or that having the maximum degree of membership.

As fuzzy logic explicitly incorporates uncertainty within its rule system, the use of error analyses – such as the confidence intervals or goodness-of-fit measures commonly found in statistical analyses – is problematic. Most fuzzy logic analyses ignore the error issue entirely, while others argue that it is not meaningful (Zadeh 1965) or that it is more meaningful to present detailed sensitivity results (Munda 1995). Bardossy and Duckstein (1995) outline potential methods for confidence interval analysis, but acknowledge that these are not comparable to statistical techniques and are not readily interpreted in policy making or in verifying the reasonableness of models. For these reasons, few studies attempt to present confidence intervals as measures of robustness. In our specific modeling problem, the use of error analysis is further complicated by the use of complementary non-linear constrained optimization techniques that could introduce unknown biases; there is, as yet, no formal treatment of this particular issue in the literature. Under the circumstances, this paper does not present any explicit error analyses, although the full study (Ruitenbeek et al. 1999) contains numerous sensitivity analyses that generally support the methodological findings presented here.

Figure 2 depicts the variables and structure of the fuzzy model. Variables, variable names, and fuzzy set ranges are defined in Table 1. The organization into levels slows the proliferation of rules with the addition of variables: with three fuzzy sets per input, and a deterministic water quality transform function, not more than 747 rules would ever be needed to saturate the knowledge base. Modeled as a single level system, over 177,000 rules would have been required. The system of fuzzy logic rules in effect represents a multi-dimensional dose-response function.

<u>Effective Nutrient Concentrations</u>. Nutrients (primarily nitrogen and phosphorus) are among the most important potential anthropogenic impacts to coral reefs. While nutrients may or may not directly impact coral growth and physiology, depending on the concentration, the major effect of increased nutrients on corals is likely a decrease in their competitive advantage over benthic algae, which can exhibit increased growth rates with increased nutrient concentrations. However, we recognize that the "effective nutrient concentration" that can affect algal abundance is not necessarily the same as nutrient loading, when loading is distinguished between N loading and P loading. The reasoning for this differentiation is based on the unifying concept in biological oceanography that plants (whether phytoplankton or benthic plants) have a definite atomic ratio of C:N:P. In phytoplankton the ratio is commonly expressed as the "Redfield ratio" with a numerical value of 106:16:1. In benthic marine plants, the ratio is variable, but has an estimated median value of 550:30:1 (Atkinson and Smith 1983). A corollary to this standard compositional ratio of marine plants is the observation that the net uptake and release of nutrients through biochemical processes also tend toward the same ratio. Thus, the nutrient in shorter supply to make up the appropriate tissue ratio will generally be the limiting nutrient to plant growth. As a result, if only one nutrient (N or P) is elevated while the other remains at low concentrations, the effect in terms of plant growth is likely to be substantially less than if both nutrients increase correspondingly. With this concept of uptake ratios as a basis, the input of "effective nutrient loading" is determined by the ratio of N loading to P loading. The rule base states that when loading of N and P is unequal, the effective loading remains equivalent to the nutrient in shortest supply. There is, however, a caveat to this rule making. Coral reefs are capable of fixing atmospheric nitrogen to form organic nitrogen. There is no equivalent biochemical process for phosphorus. Thus, if the ratio N:P of the water flowing over a reef is low relative to the uptake ratio of plants on the reef, the capability exists for nitrogen fixation to raise the potential uptake of phosphorus. On the other hand, if phosphorus is the nutrient in low relative concentration, there is no potential to increase uptake potential through atmospheric supply. As a result, we consider phosphorus the limiting nutrient in our rule base, and the input variable of "effective nutrient concentration" is equivalent to the "effective phosphorus concentration."

Water-Quality Transform Function. This model converts sediment and effective nutrient loadings at specified locations into effective nutrient concentration, depth of sediment deposition, and concentration of suspended sediment over the reef. The model uses a simple fuzzy rule-based water quality transform that approximates a conventional (non-fuzzy) water-quality model described in Rijsberman and Westmacott (1996).

Algae-Nutrient-Grazing Subsystem. The reasoning behind this subsystem is, simply, that the primary effect of elevated nutrient levels on coral is the enhanced growth of algae which, ceteris paribus, may compete with coral for hard substratum or perhaps even smother existing live coral. However, grazing by fish, sea urchins, and other fauna will help check the proliferation of algae. Thus a quite parsimonious function for determining algae levels is derived from nine rules describing the interplay between the effective nutrient concentration and grazing pressure.

Sediment Deposition versus Suspended Sediments. Distinction is made between the input variables sediment deposition and suspended sediments because these factors can be considered to affect coral community structure differently. While suspended sediment is often considered a detriment to coral growth and reproduction, it has been documented that many reef areas contain high percentage cover of coral in areas where suspended sediments is normally considered high. Species composition in such areas may be substantially different than in areas with low suspended sediment primarily as a result of the physiological capability of some species to efficiently eject sediment from living polyps. As a result, reef composition may vary dramatically between areas of differing levels of sediment suspension, but one reef assemblage cannot necessarily be considered inferior to the other. Coral cover then, in contrast to coral species mix, may not vary significantly with suspended sediment.

On the other hand, sediment deposition appears to be universally more detrimental to living coral reef structure. Coral planulae cannot settle in areas where soft sediments continually cover the bottom, and may not survive in areas where sediment deposition is episodic but a regular occurrence. In areas of highly variable water motion, sediment deposition may occur occasionally during periods of high input and low water motion, with subsequent clearing of the deposited material when water motion increases. While adult colonies of some species may tolerate coverage by settled sediments for short periods of time (hours to days), coverage for longer periods is lethal to virtually all species. As a result, in our model, sediment deposition has a considerably stronger adverse effect upon reefs than suspended sediment. It is also important to understand that while these two input variables can co-vary (e.g., high sediment deposition in areas of high sediment suspension), it is not unusual to find reef areas where the input variables are very dissimilar, generally as a function of water motion. For example, in areas with normal high water motion from wave forces, suspended sediments can be high with virtually no deposition. On the other hand, in areas with low water motion, and limited flushing as a result of physiographic structure, sediment input may be low, resulting in relatively low suspended sediment. However, because there are insufficient physical forces to remove sediment, deposition is high. This is a typical situation in lagoonal areas which often have soft sediment bottoms with little coral development.

Fishing Pressure. While corals themselves are sometimes the target species (mainly for curio collectors), fishing pressure is generally considered to have an important indirect impact on coral reefs. Removing a large percentage of the grazers or piscivores on any reef may cause changes in the balance between corals and algae, which can result in phase shifts in reef structure. While fishing pressure is considered an important variable, it is inherently difficult to measure and quantify for input into the model. We have chosen to employ the units of measurement presented by McClanahan (1995) in his coral reef ecosystem-fishery model which is aimed at determining the impacts of fishing intensity and catch selection on reef structure and processes. Based on field data, McClanahan estimates that a man can catch 25 kg/ha/day of fish at maximum fish biomass. This clearly depends on the techniques used and should be seen as a relative measure. We use this number as a maximum value and scale downward to create membership classes. It should be acknowledged that this variable is likely to be the most difficult to quantify in any applied situation, but it nevertheless is a necessary input for an effective model.

Economic Model Structure

Accounting for intermediate variables in the fuzzy model, the reduced form of the output and inputs to the integrated complex system function are the following.

Parameters that are listed with a "*" are regarded as fixed for any given site and are not normally affected by the impacts arising from economic interventions.

$Q=Q\{F_1, \ldots, F_9\}$= Coral Abundance on Available Substrate
F_1 = Suspended Sediment
F_2 = Sediment Deposition
F_3 = Physical Damage
F_4 = Fishing Pressure
F_5 = Relief*
F_6 = Grazing Pressure*
F_7 = Initial Effective Nutrient Concentration*
F_8 = Nitrogen Loads
F_9 = Phosphorous Loads

Various computer modeling and simulation platforms were tested to find an efficient system that could handle the biophysical parameters as well as the economic optimization procedures. Final modeling was conducted using MATLAB™ 5.2 software relying on the specialized Fuzzy Logic Toolbox and the Optimization Toolbox (Mathworks 1998). In modeling the relationships, fuzzy rule-based systems were initially defined for each system and were subsequently modified to improve computational efficiency. The modifications included use of Sugeno transforms instead of Mamdami transforms and the specification of a fuzzy inference system for the water quality transform. All optimization routines relied on a sequential quadratic programming method, which is the most efficient algorithm for optimizing over nonlinear surfaces (Han 1977, Powell 1978, Gill et al. 1981, Floudas and Pardolos 1992); identification of global optima was assured through specification of different starting points to ensure convergence.

The economic model structure consists primarily of two components. One component involves the definition of a "unit intervention set," including the costs of each of the unit interventions. The second component incorporates an economic activity "baseline" that represents a base case level of activity and impact in the absence of any interventions. The baseline level of activity corresponds to $n_k=0$ for all k=1 to K. Cost information for the various interventions was based on location specific data for Montego Bay (GMBRC 1996). In general, the simplified form of the cost function takes the form:

$$C_k = 0 \text{ if } n_k=0 \tag{Eq. 2}$$

$$C_k = n_k C1_k + n_k C2_k/r \text{ if } n_k>0$$

where C1 is the capital cost of a unit intervention and C2 is the annual operating cost of a unit intervention of type k. Each of these at a "unit scale" will have some impact on economic activities and on the inputs to the biophysical model (i.e., on the vector **F**).

The economic baseline component essentially involves projecting all economic activities under the assumption of no interventions. A resultant baseline vector **F0** is generated, with a corresponding level of coral quality that can be calculated as Q0=Q{F0} through evaluation using the fuzzy model.

At this stage, the model can be used in two different modes: simulation or optimization. In simulation mode, the model determines the consequences of a given intervention set. An intervention set is defined by the vector **n**, and each n_k could take on a user-specified value from zero to some upper bound which is dictated by feasibility constraints (for example, it will not permit replanting more than 100% of the watershed). In optimization mode, the only input is the target reef quality ($\hat{Q}$) and the model will generate the least cost combination given constraints on each n_k. The output is a vector **n**.

Modeling Scenarios and Interventions

The model can forecast economic activity, pollution and impact loads, and resultant coral quality over a 55 year period, which reflects the local government's long-term planning horizon for the impacts of future capital projects. Results in this paper, however, focus on a 25 year horizon.

The underlying forecast of economic activity is divided into the following sectors:

- Municipal Sector (domestic). Migration into the area is regarded as a significant element in future economic development of the region, and demands on municipal waste treatment services will escalate. Wastes from the domestic sector thus are a potentially significant contributor to overall pollutant loading.
- Agribusiness Sector. This sector is selected because it is one of the major growth nodes in the area and has high pollution potential. Although agriculture itself is not an important contributor to regional product, value

added processing may become increasingly significant in the free trade zone and elsewhere.

- Light Manufacturing Sector. This sector is highlighted because of its high pollution potential for metals, sediments, nutrients and toxic compounds. Also, growth may be expected to increase given the desire for industrial expansion in and around the free trade zone.
- Heavy Manufacturing and Construction Sector. This sector also has high pollution potential, although its pollutants have traditionally been mainly sediment loads and solid wastes leading to potential physical damages on the reef.
- Hotel and Tourist Service Sector. This sector is an important current component of the local economy and will continue to be a major player in the future. As such, interventions relating to this sector are likely to have a significant impact on water demands and on overall pollution loads.
- Forestry and Agriculture Sectors. These sectors are included for completeness, and because of their high potential pollution loads. In the Montego Bay area, however, their relative contributions to economic output are small.
- Offshore Transport Sector. Offshore shipping contributes to recurrent oil spills in the area. It is expected that these recurrent impacts, as well as the risk of an oil spill, will escalate with increased processing in the free trade zone and elsewhere.

In any particular simulation, or optimization, the baseline forecast is chosen as a status quo case. This describes conditions in the absence of any active interventions. We use as a reference case a rapid growth scenario developed on the basis of consultations with and documents provided by the Greater Montego Bay Redevelopment Corporation (1996). The forecasts represent relatively rapid growth over a 20 year period, tapering off to lower levels over the remainder of the 55 year period. Specifically, population is expected to grow by about 2.5% annually for 20 years, and 1% annually in the longer term. Real economic output in the manufacturing and processing sectors is expected to range between 3% and 5% in the near to medium term, and 1% to 1.5% in the long term. Tourism and hotel industry growth is expected to average about 3% annually for 20 years, tapering off to 1% annually afterwards. Forestry and agriculture are expected to realize only modest growth in the near term (less than 1% annually) and no real growth over the long term as land is converted to municipal requirements.

The model incorporates eight active intervention types for Montego Bay. The interventions, and their approximate costs, are (all figures in US$):

1. Sediment Trap. This involves placement of a sediment trap close to the Montego River outlet before it empties into Montego Bay. The trap is a physical barrier that slows the water flow and prevents most of the sediments from entering Montego Bay; it also removes some solid litter that might cause physical damage to the reefs. It does not reduce nutrient loads to any significant degree. Effective operation of the trap requires regular (weekly) maintenance and removal of sediments, for disposal in clean fill sites. The capital cost of such a trap is estimated to be about $6 million, with annual operational costs of about $330,000. Smaller traps, at lower cost and efficiency, could be installed at various upstream locations.
2. Planting of Trees in Upper Watershed. This scenario reflects reforestation of the most degraded watershed areas around Montego Bay. This involves planting about 150,000 acres of trees, at a one time capital cost of almost $28 million (based on average reforestation costs for Jamaica). This intervention would lead to a substantial (almost 100%) reduction of sediment and nitrogen loads from this area.
3. Aeration of Waste. This involves installation of a common waste treatment aeration system in the Montego Bay free trade zone, capable of treating 416 tons per day of waste. It would result in a substantial end-of-pipe reduction in sediment and nutrients from the light industry in this zone. Costs of such a facility are estimated to approach $1 million, requiring also an additional $1 million annually for operation.
4. Large Scale Centralized Treatment Facility. This scenario involves installation of a common waste treatment facility capable of processing about one-quarter of the sewage and waste in the Montego Bay area. Installation of such a facility would reduce nutrient and sediment loads

associated with domestic, commercial and hotel waste streams; some modest decrease in physical impacts on the reef would also be evident. In theory, up to four of these might be built over the long term in Montego Bay; construction of additional units is, however, constrained by difficulties associated with connecting all areas, and with overcoming the common use of disposal wells. In the optimization modeling, therefore, the model limits this to only one such facility being constructed at a capital cost of about $50 million and annual operational costs of about $5 million. Smaller scaled down versions of this could also be constructed.

5. Agricultural Extension. This intervention reflects the establishment of technology transfer programs along the lines of internationally accepted waste reduction programs. Such programs are aimed at reducing pollutant loads (primarily from nutrients) through providing relatively low-cost (often self-financing) technologies to the agricultural and agroprocessing sectors. The intervention covers up to 10% of such enterprises in the area, and will cost $1.2 million to implement with an annual cost of about $120,000.

6. Outfall and Pump. This is a stand-alone intervention that would involve a sewage outfall and pump station to take the sediment beyond the reef edge (approximately 5 kilometers). The unit would cost about $1.8 million, along with $72,000 annually, and would mainly reduce sediment loads and physical impacts of wastes on the reef. Smaller versions at lower cost and efficiency are available.

7. Household Solid Waste Collection. This scheme involves establishing a small-scale waste collection system to connect about 30,000 people in squatter settlements or low-income areas to common waste handling facilities. Although the capital costs for this type of an arrangement are low ($72,000) the operating costs are relatively high ($36,000 annually). The effect this has on pollution loads will be to reduce sediment and nutrient loads from the household sector.

8. <u>Hotel Tax</u>. This intervention simulates the impact of a 25% land tax on the existing hotel/service sector, and is meant to illustrate the impacts of a policy intervention as opposed to some of the investment interventions considered elsewhere. While this tax is not directly attacking any specific pollutant, the increase in hotel operation costs is expected to dampen investment and decrease pollution loads. The administrative costs of such an intervention are estimated to be about $60,000 annually.

Results

While the model provides a dynamic forecasting environment, it was found that decision-makers find it most useful if reef quality can be expressed in terms of a single index relating to a single future reference year (Werners 1998). In all modeling summaries and optimizations, therefore, a "25 year equilibrium" level of coral abundance was selected as a benchmark. Precise interpretation of this figure is somewhat complex, but it essentially describes the long-term level of coral abundance on available substrate arising from the next 25 years of activities and interventions. It therefore consolidates initial conditions (taken as 1998) with future economic development activities (and their associated negative impacts) and any mitigative interventions (and their positive impacts).

The basic technical sensitivity of the reef impact model, calibrated for Montego Bay conditions, is shown in Table 2. Under static conditions of no growth and no mitigative interventions, with all stresses essentially remaining at current levels, it can be expected that a long term equilibrium level of 43% coral abundance would be achieved. Table 2 also shows that the greatest deterioration would arise from changes in pollution loading (N, P and sediments) while reef quality is less responsive to changes in fishing pressure.

The economic impacts of single technical interventions are shown in Table 3. The results also show that, in the "high growth" reference forecast, a long term equilibrium level of about 29% coral abundance would be expected. This decline, relative to the "no growth" case of 43% coral abundance, is attributable entirely to the increased impacts from economic activity in the absence of mitigating interventions. The results also indicate the potential impact of single interventions. No single intervention is capable of totally compensating for the negative impacts on coral abundance, although, if all interventions were executed, a level of about 49% coral abundance could be achieved. This, in fact, represents a 20.23% improvement of what would otherwise happen, and it would result in a present value cost in excess of US$150 million.

The results in Table 3 show the impact of single interventions relative to a “do nothing” scenario. Because of the nonlinearity of the coral reef response, it is not possible simply to add up these interventions to arrive at a cumulative impact. The model, in optimization mode, permits setting of a target level of coral abundance (or change in coral abundance over a reference case); results for such optimizations are summarized in Table 4. For any given target level, the optimization provides the least cost combination of interventions, permitting variable intensities from zero to unity. A zero indicates that the intervention is not undertaken, while any positive value shows partial or full implementation of a given intervention.

Discussion and Conclusion

Modeling results provide important insights into methodological issues as well as practical policy issues. A major methodological success of the exercise is that it was found to be feasible to model a large variety of economic and ecological parameters in a predictive system that permits comparison of policies. The fuzzy logic procedures, coupled with economic optimization tools, can take advantage of relatively sparse information sets.

The nonlinearity of underlying complex systems also places in question many conventional methods of cost-effectiveness analysis that assume separability of benefits and costs, and separability of the impacts of individual interventions. Inspection of the results illustrates a number of these points.

First, the nonlinearity of the coral quality response surfaces to individual interventions is shown in Table 3. Both the reforestation alternative and the waste aeration alternative achieve precisely the same level of coral abundance, because of a localized “plateau” in the coral quality response surface. Such localized plateaus in the ecological model are relatively common and are surpassed only through more investment through additional interventions; the first intervention in such cases will always have a high cost (in terms of $ per % improvement) compared to subsequent investments which move conditions beyond such a plateau.

Second, the fallacy of separating benefits from costs, and of using a continuous ranking of individual interventions, is shown in the optimization results in Table 4. In a conventional separable model with monotonically increasing marginal costs (such as that in Figure 1), an intervention that was undertaken at a low target level of coral improvement would also always be undertaken at a high target level of coral improvement. But this is clearly not the case here. Reforestation, for example, is part of the optimal intervention set at coral quality improvement targets of 14% and 20%, but it is not part of the intervention set at intermediate targets of 15% or 16%. Similarly, the intensity of the agricultural extension and hotel tax interventions do not increase monotonically. This is reflected also in the marginal cost (MC) curve inherent in Table 4; while generally it is increasing there are some localized decreases. The most significant implication this has for policy makers is that one can not simply pursue low cost interventions in the absence of some coral quality target, which will in turn be related to the economic benefits.

The fallacy of the conventional ranking procedures is also shown by inspection of the average costs of individual interventions (Table 3). Such average costs are often used as a means for ranking alternatives, and are usually calculated based on "initial" conditions. Reliance on such an indicator would lead one to conclude, for example, that reforestation was more economical than a hotel tax; but the optimization results show that at higher coral quality targets (between 15% and 18% improvement), a hotel tax is the most economical option. Again, some knowledge of the economic benefits is necessary before a "target" can be achieved in association with the available cost intervention.

Apart from the above methodological issues, the model results do provide some practical insights to policy design decisions in Montego Bay. First, the results illustrate that some interventions are common to all "optimal policy sets" for intermediate levels of coral improvement. Specifically, household solid waste collection, installation of an outfall, and use of a sediment trap on the Montego River are relatively cost-effective interventions; use of these three interventions would impose present value costs of about

US$12 million and achieve a coral improvement in excess of 10%. By contrast, achieving the maximum potential improvement of 20% would entail present value costs of US$153 million.

In conclusion, we note that – as with all such modeling exercises – any such prescriptions should be complemented by good judgment on the part of policy makers. Manipulation of the models can provide insights into the generally desirability and impacts of various interventions, but such models never tell the whole story. In Montego Bay, for example, the water quality model remains a weak link; it still treats pollutant transport and mixing with a broad brush that neglects seasonal variations and potential localized impacts on, say, important diving sites. Such considerations are beyond the capacity of this analysis framework, although they may be of key importance to a dive industry that generates considerable local benefits through tourism.

Also, the current models do not adequately capture many dynamic elements of coral reef responses to human, and other, stresses. Limitations in coral reef science and data availability make it extraordinarily difficult to reconcile or benchmark models such as this (which predict long-term equilibrium conditions) against real field data (which measure current reef conditions, often under disequilibrium conditions). Consequently, this again calls for prudence in using and interpreting the results of such models. In our view, the model is most useful for providing guidance in the <u>changes</u> in reef quality induced by <u>localized human impacts</u>; the model is less robust in its predictive ability for <u>absolute</u> levels of reef quality in an environment characterized by both human-induced local stresses and other external stresses. Nonetheless, the messages of the model results are clear: pay greater attention to ecosystem responses; pay less attention to conventional constructs of cost effectiveness that assume linear behavior. Complex systems such as coral reefs are not likely to lend themselves to simple management solutions; modeling tools must strive to capture some of this complexity.

Acknowledgements

This work was funded under the World Bank Research Committee RPO#680-08. Development of earlier prototype fuzzy logic models benefited from modeling work by Mike Nullet, and from reviews by Subodh Mathur, Frank Rijsberman, and Susie Westmacott. Ridgley and Dollar (1996) developed the rules and selected the final variables used in the fuzzy model; the process was informed by an expert workshop convened in Honolulu on 19 June 1996, aimed at selecting and discussing possible variables to be included in the model. This workshop was attended by the following: Dr Marlin Atkinson (Hawaii Institute of Marine Biology), Dr Evelyn Cox (HIMB), Dr Richard Grigg (Department of Oceanography, University of Hawaii), Dr Donald Kinsey (Australian Institute of Marine Biology, ret.), Dr Stephen V. Smith (Oceanography, UH), and Dr Gordon Tribble (US Geological Survey). The rule base was corroborated through independent peer review by Dr James Maragos and Prof Robert Buddemeier. The authors gratefully acknowledge the assistance of Kirk Taylor, Jill Williams and Malden Miller of the Montego Bay Marine Park Authority in information collection and review of results through a series of workshops and seminars in Montego Bay over the period 1994 to 1999. Finally, the authors are grateful to Jean-Marc Bergevin, Roger Bradbury and one anonymous reviewer for their thoughtful comments on an earlier version of this article.

References

Atkinson MJ, Smith SV (1983) C:N:P ratios of benthic marine plants. Limnol Oceanogr 28(3): 568-574.

Ayyub B, McCuen R (1987) Quality and uncertainty assessment of wildlife habitat with fuzzy sets. Journal of Water Resources Planning and Management 113(1): 95-109.

Bardossy A, Duckstein L (1995) Fuzzy rule-based modeling with applications to geophysical, biological, and engineering systems. CRC Press, Boca Raton FL.

Berg H, Öhman MC, Troëng S, Línden O (1998) Environmental economics of coral reef destruction in Sri Lanka. Ambio 27(8):627-634.

Cesar H (1996) Economic analysis of Indonesian coral reefs. World Bank Environment Department, Washington.

Dixon JA (1993) Economic benefits of marine protected areas. Oceanus, fall: 35-40.

Eskeland G (1992) Demand management in environmental protection: fuel taxes and air pollution in Mexico City. World Bank, Country Economics Department, Washington.

Floudas C, Pardalos P (eds) (1992) Recent advances in global optimization. Princeton University Press, Princeton.

Gill PE, Murray W, Wright MH (1981) Practical optimization. Academic Press, London.

Greater Montego Bay Redevelopment Corporation (1995) Greater Montego Bay Area Plan 2014. Greater Montego Bay Redevelopment Corporation, Montego Bay.

Gustavson K (1998) Values associated with the local use of the Montego Bay Marine Park. Study prepared for the World Bank. World Bank, Washington.

Han SP (1977) A globally convergent method for nonlinear programming. Journal of Optimization Theory and Applications 22: 297.

Hatziolos ME, Hooten AJ, Fodor M (eds) (1998) Coral reefs: challenges and opportunities for sustainable management. World Bank, Washington.

Hughes TP (1994) Catastrophes, phase-shifts, and large-scale degradation of a Caribbean coral reef. Science 265: 1547-1551.

Johannes R (1981) Words of the lagoon. University of California Press.

Kainuma M, Nakamori Y, Morita T (1991) Development of an intelligent decision support system for environmental modeling and planning. Research Report 128. National Institute for Environmental Studies, Japan.

Kosko B (1992) Neural networks and fuzzy systems: a dynamical systems approach to machine intelligence. Prentice-Hall, Englewood Cliffs.

Kosko B (1993) Fuzzy thinking. Hyperion, New York.

Lapointe BE, Littler MM, Littler DS (1997) Macroalgal overgrowth of fringing coral reefs at Discovery Bay, Jamaica: bottom-up versus top-down control. Proc 8th Int Coral Reef Symp 1: 927-932.

Louis Berger International (1995) Montego Bay environmental monitoring programme: Northern Jamaica Development Project, Second Interim Report. Louis Berger International Inc., Montego Bay.

Mathworks (1998) MATLAB™: The language of technical computing, version 5.2. Mathworks, Natick MA.

McClanahan TR (1995) A coral reef ecosystem-fisheries model: impacts of fishing intensity and catch selection on reef structure and processes. Ecological Modeling 80: 1-19.

McNeill D, Freiberger P (1993) Fuzzy logic. Simon & Schuster, New York.

Munda G (1995) Multicriterion evaluation in a fuzzy environment: theory and applications in ecological economics. Physica-Verlag, Heidelberg.

O'Callaghan Consulting Services Limited (1992) Environment study report. Greater Montego Bay Redevelopment Corporation, Montego Bay.

Powell MJD (1978) The convergence of variable metric methods for nonlinearly constrained optimization calculations. In: Mangasarian OL, Meyer RR, Robinson SM (eds) Nonlinear programming 3. Academic Press, London.

Ridgley M, Dollar S, Bailey K (1995) Overview of the development of a prototype model to estimate future pollution-induced coral reef impacts. World Bank, Latin America and Caribbean Environment and Urban Development Division (LA3EU), Washington.

Ridgley M, Dollar S (1996) A coral-reef impact-assessment model using fuzzy rule-based logic, Phase II Interim Report. World Bank, Latin America and Caribbean Environment and Urban Development Division (LA3EU), Washington.

Rijsberman F, Westmacott S (1996) Cost-effectiveness analysis of coral reef management and protection: a case study for Curacao. World Bank, Latin America and Caribbean Environment and Urban Development Division (LA3EU), Washington.

Ruitenbeek HJ (1992) The rainforest supply price: a tool for evaluating rainforest conservation expenditures. Ecological Economics 6: 57-78.

Ruitenbeek HJ, Cartier CM (1999) Issues in applied coral reef biodiversity valuation: results for Montego Bay, Jamaica. World Bank Research Committee Final Report, World Bank, Washington.

Ruitenbeek HJ, Ridgley M, Dollar S, Huber R (1999) Optimization of economic policies and investment projects using a fuzzy logic based cost-effectiveness model of coral reef quality: empirical results for Montego Bay, Jamaica – A least cost model for coral reef management and protection. World Bank Research Committee Project RPO#680-08 Final Report, World Bank, Washington.

Spash CL, van der Werff JD, Westmacott S, Ruitenbeek HJ (1998) Lexicographic preferences and the contingent valuation of coral reef biodiversity in Curaçao and Jamaica. Study prepared for the World Bank. World Bank, Washington.

Sullivan KM, Chiappone M (1994) A rapid ecological assessment of the Montego Bay Marine Park, Jamaica. The Nature Conservancy, Montego Bay.

Tomascik T, Sander F (1985) Effects of eutrophication on reef building corals: I. Growth rate of the reef building coral Montastrea annularis. Marine Biology 87: 143-155.

Tomascik T, Sander F (1987) Effects of eutrophication on reef building corals: III. Reproduction of the reef building coral Porites porites. Marine Biology 94: 77-94.

von Altrock C (1995) Fuzzy logic and neurofuzzy applications explained. Prentice-Hall, Englewood Cliffs.

Werners S (1998) Development of the user-friendly computer model COCOMO for coastal conservation in Montego Bay. Status report to World Bank, Latin America and the Caribbean Region (LCSES). World Bank, Washington. Mimeo.

Zadeh L (1965) Fuzzy sets. Information and Control 8(3)(June): 338-353.

Table 1. Input and output variables and their associated fuzzy sets. Typical values are shown for each fuzzy set, with corresponding ranges in square brackets. A "*" after a variable signifies an output variable. Where no values are shown for a specific fuzzy set, that set is not used; all input variables are defined by three fuzzy sets while outputs are defined by four or five fuzzy sets.

Variable	Low	Mlow	Med	Mhi	High
N loads (mmol m^{-2} day^{-2})	2 [0-6]		15 [5-50]		80 [40-200]
P loads (mmol m^{-2} day^{-2})	0.5 [0-1]		4 [0.8-6]		7 [5-10]
Effective Nutrient Load*	0.25 [0-0.5]	0.75 [0.4-1]	1.5 [0.8-3]	4 [2.5-6]	8 [5-10]
Effective Nutrient Concentration (µM)	0.02 [0-0.05]		0.1 [0.04-0.15]		0.3 [0.14-0.5]
Grazing Pressure (kg ha^{-1} day^{-1})	10 [0-30]		40 [25-100]		110 [80-150]
Algae* (%)	5 [0-20]		25 [15-50]	40 [25-60]	60 [40-100]
Sediment Loads (g m^{-2} day^{-1})	50 [0-100]		150 [80-500]		600 [450-800]
Suspended Sediment (g m^{-3})	0.6 [0-2.5]		2.5 [1.5-5]		5 [4-10]
Sediment Deposition (g m^{-2} day^{-1})	2 [0-10]		20 [8-50]		60 [45-80]
Physical Damage (index)	0.5 [0-1]		1 [0.5-2.5]		3 [2-4]
Algae (%)	5 [0-20]		25 [15-50]		60 [40-100]
Fishing Pressure (kg ha^{-1} day^{-1})	2 [0-5]		6 [4-15]		20 [12-25]
Relief (rugosity index)	1.2 [1-1.5]		2 [1.25-3]		4 [2.5-5]
Live Coral* (% on available substrate)	8 [0-15]	18 [10-25]	35 [20-50]	50 [40-65]	70 [60-100]

Table 2. Changes in Montego Bay coral reef quality arising from changes in key inputs. Coral abundance levels show long-term equilibrium arising from changes in physical impacts of human-induced activities on the reef ecosystem.

Scenario	Coral	Δ Coral
Base Case Conditions - No Economic Growth	42.73%	
Doubling of:		
Pollution Loads (N, P & Sediment)	21.83%	– 20.90%
Physical Damage	25.49%	– 17.24%
Fishing Pressure	39.80%	– 2.93%
All Inputs	6.82%	– 35.91%
Halving of:		
Pollution Loads (N, P & Sediment)	56.38%	+ 13.65%
Physical Damage	51.33%	+ 8.66%
Fishing Pressure	44.00%	+ 1.27%
All Inputs	76.18%	+ 33.45%

Table 3. Changes in Montego Bay coral reef quality arising from single interventions. Coral abundance levels show "25 year equilibrium," and resultant total cost (TC in millions $) and average costs (AC in millions of $ per additional % of coral abundance).

Intervention	Coral	Δ Coral	TC	AC
0. Base Case Conditions - High Economic Growth	28.94%	0.00%		
k1. Sediment Trap	32.13%	3.20%	9.30	2.91
k2. Planting of Trees in Upper Watershed	30.57%	1.63%	27.90	17.12
k3. Aeration of Waste	30.57%	1.63%	11.84	7.25
k4. Large Scale Centralized Treatment Facility	34.18%	5.24%	98.40	18.78
k5. Agricultural Extension	29.00%	0.07%	2.40	36.81
k6. Outfall and Pump	34.33%	5.39%	2.52	0.47
k7. Household Solid Waste Collection	30.73%	1.80%	0.43	0.24
k8. Hotel Tax	28.97%	0.03%	0.60	17.30
k1-8. All of the Above	49.17%	20.23%	153.40	7.58

Table 4. Optimization results for Montego Bay, showing levels of individual interventions required to achieve target coral reef quality, and resultant total cost (TC in millions $) and marginal costs (MC in millions of $ per additional % of coral abundance). Interventions are as follows: k1 = Sediment Trap; k2 = Planting of Trees in Upper Watershed; k3 = Aeration of Waste; k4 = Large Scale Centralized Treatment Facility; k5 = Agricultural Extension; k6 = Outfall and Pump; k7 = Household Solid Waste Collection; k8 = Hotel Tax.

ΔCoral (%)	k1	k2	k3	k4	k5	k6	k7	k8	TC	MC
0.25	0	0	0	0	0	0	0.13	0	0.06	0.24
0.50	0	0	0	0	0	0	0.26	0	0.11	0.20
0.75	0	0	0	0	0	0	0.39	0	0.17	0.24
1.00	0	0	0	0	0	0	0.58	0	0.25	0.32
1.25	0	0	0	0	0	0	0.71	0	0.31	0.24
1.50	0	0	0	0	0	0	0.85	0	0.37	0.24
1.75	0	0	0	0	0	0	0.98	0	0.42	0.20
2.00	0	0	0	0	0	0.04	1	0	0.53	0.44
2.25	0	0	0	0	0	0.08	1	0	0.64	0.44
2.50	0	0	0	0	0	0.13	1	0	0.76	0.48
2.75	0	0	0	0	0	0.18	1	0	0.87	0.44
3.00	0	0	0	0	0	0.22	1	0	0.99	0.48
3.25	0	0	0	0	0	0.27	1	0	1.10	0.44
3.50	0	0	0	0	0	0.31	1	0	1.22	0.48
3.75	0	0	0	0	0	0.36	1	0	1.33	0.44
4.00	0	0	0	0	0	0.40	1	0	1.45	0.48
4.25	0	0	0	0	0	0.45	1	0	1.56	0.44
4.50	0	0	0	0	0	0.49	1	0	1.68	0.48
4.75	0	0	0	0	0	0.54	1	0	1.79	0.44
5.00	0	0	0	0	0	0.58	1	0	1.90	0.44
5.25	0	0	0	0	0	0.63	1	0	2.02	0.48
5.50	0	0	0	0	0	0.67	1	0	2.13	0.44
5.75	0	0	0	0	0	0.72	1	0	2.24	0.44
6.00	0	0	0	0	0	0.76	1	0	2.34	0.40
6.25	0	0	0	0	0	0.80	1	0	2.45	0.44
6.50	0	0	0	0	0	0.84	1	0	2.56	0.44
6.75	0	0	0	0	0	0.89	1	0	2.67	0.44
7.00	0	0	0	0	0	0.93	1	0	2.78	0.44
7.25	0	0	0	0	0	0.97	1	0	2.88	0.40
7.50	0.03	0	0	0	0	1	1	0	3.19	1.24
7.75	0.10	0	0	0	0	1	1	0	3.85	2.64
8.00	0.17	0	0	0	0	1	1	0	4.52	2.68
8.25	0.24	0	0	0	0	1	1	0	5.18	2.64
8.50	0.31	0	0	0	0	1	1	0	5.83	2.60
8.75	0.38	0	0	0	0	1	1	0	6.49	2.64
9.00	0.45	0	0	0	0	1	1	0	7.15	2.64
9.25	0.52	0	0	0	0	1	1	0	7.80	2.60
9.50	0.59	0	0	0	0	1	1	0	8.45	2.60
9.75	0.66	0	0	0	0	1	1	0	9.10	2.60
10.00	0.73	0	0	0	0	1	1	0	9.75	2.60

Table 4 (continued)

ΔCoral (%)	k1	k2	k3	k4	k5	k6	k7	k8	TC	MC
10.25	0.80	0	0	0	0	1	1	0	10.39	2.56
10.50	0.87	0	0	0	0	1	1	0	11.04	2.60
10.75	0.94	0	0	0	0	1	1	0	11.68	2.56
11.00	1	0	0.01	0	0	1	1	0	12.41	2.92
11.25	1	0	0.14	0	0	1	1	0	13.89	5.92
11.50	1	0	0.26	0	0	1	1	0	15.35	5.84
11.75	1	0	0.38	0	0	1	1	0	16.78	5.72
12.00	1	0	0.50	0	0	1	1	0	18.21	5.72
12.25	1	0	0.62	0	0	1	1	0	19.63	5.68
12.50	1	0	0.74	0	0	1	1	0	21.06	5.72
12.75	1	0	0.86	0	0	1	1	0	22.47	5.64
13.00	1	0	0.98	0	0	1	1	0	23.89	5.68
13.25	1	0.09	1	0	0	1	1	1	27.20	13.24
13.50	1	0.22	1	0	0	1	1	1	30.88	14.72
13.75	1	0.35	1	0	0	1	1	1	34.55	14.68
14.00	1	0.34	1	0.04	0	1	1	1	38.27	14.88
14.25	1	0.28	1	0.10	0	1	1	0.20	42.09	15.28
14.50	1	0	1	0.24	0	1	1	0.36	47.67	22.32
14.75	1	0.63	1	0.10	0	1	1	0.57	51.51	15.36
15.00	1	0	1	0.32	0	1	1	1	55.88	17.48
15.25	1	0	1	0.36	0	1	1	1	60.01	16.52
15.50	1	0	1	0.40	0	1	1	1	64.13	16.48
15.75	1	0	1	0.45	0	1	1	0.18	68.32	16.76
16.00	1	0	1	0.48	0	1	1	1	72.35	16.12
16.25	1	0	1	0.53	0	1	1	1	76.43	16.32
16.50	1	0	1	0.57	0	1	1	1	80.82	17.56
16.75	1	0	1	0.62	0	1	1	0.35	85.25	17.72
17.00	0.99	0	1	0.64	0.22	1	1	0.48	87.43	8.72
17.25	1	0.32	1	0.64	0	1	1	0.04	95.89	33.84
17.50	1	0	1	0.77	0	1	1	1	100.49	18.40
17.75	1	0	1	0.81	0	1	1	1	104.68	16.76
18.00	1	0	1	0.86	0	1	1	1	108.85	16.68
18.25	1	0	1	0.90	0	1	1	1	112.99	16.56
18.50	1	0	1	0.94	0	1	1	1	117.10	16.44
18.75	1	0	1	0.98	0	1	1	1	121.20	16.40
19.00	1	0.10	1	1	0	1	1	1	125.78	18.32
19.25	1	0.27	1	1	0	1	1	1	130.64	19.44
19.50	1	0.44	1	1	0	1	1	1	135.39	19.00
19.75	1	0.61	1	1	0	1	1	1	140.06	18.68
20.00	1	0.83	1	1	0	1	1	1	146.31	25.00
20.25	1	1	1	1	1	1	1	1	153.48	28.68

Figure 1. A conventional cost curve for cost-effectiveness analysis.

Cs= Measure of cost-effectiveness in reducing effect/impact indicator "E" by intervention 's'
E= Effect or impact indicator
s= policy intervention or investment
B*-B* = Marginal environmental benefits associated with reducing effect/impact indicator E

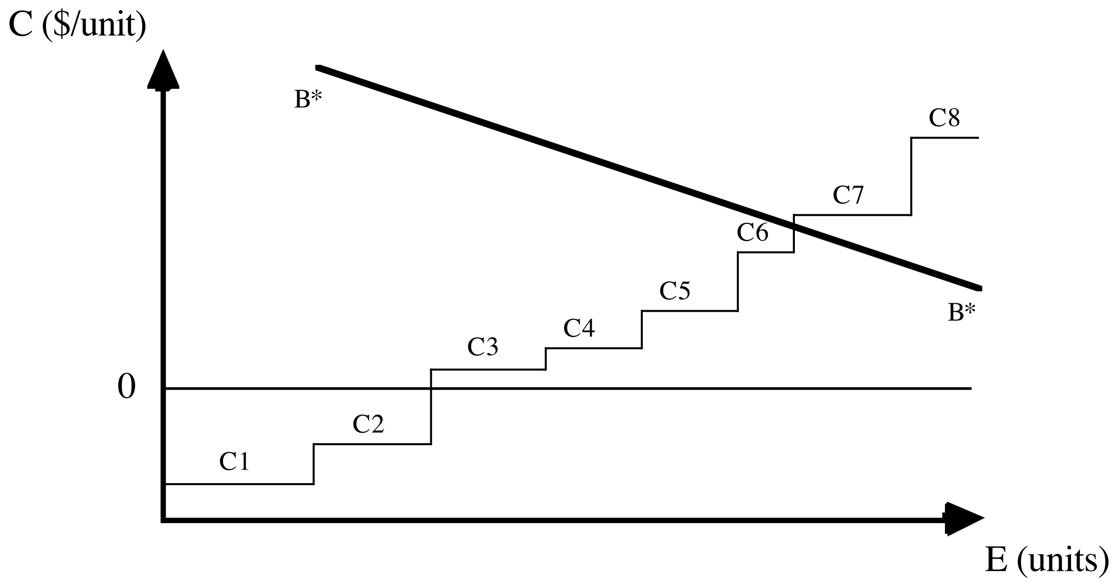

Figure 2. Coral reef impact model structure. The generic ecological sub-model consists of four stages: (i) N loads and P loads are converted to effective nutrient load in a fuzzy logic transform; (ii) sediments and nutrients are converted to nutrient concentrations, sediment deposition and suspended sediment at the coral reef site using a water quality transform function that can consist either of a deterministic linear transform, a deterministic non-linear transform, or a fuzzy logic based transform; (iii) nutrient concentration and grazing pressure are converted to algae cover in a fuzzy logic transform; and, (iv) six primary determining variables are converted into live coral cover using a fuzzy logic transform. Where a deterministic water quality transform is used, and where each input takes on three potential values (Low, Medium, High), the system requires a maximum rule base of 747 rules ($3^2 + 3^2 + 3^6$).