A Simple Framework for Science-based Marine Spatial Planning in Data-limited Situations: A Theoretical Case Study

Un Marco (De Trabajo) Simple para la Planificacion Espacial marine Basada en Ciencia: Es Estudio de Caso

Une Approche Simple pour la Planification Spatiale Marine: Un Exemple de Cas

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ABSTRACT

Marine spatial planning is an increasingly important strategy for achieving a variety of stakeholder objectives in ocean environments. While many papers have offered general guidelines for marine spatial planning, few have presented scientific analytical tools and approaches that could guide the development of an actual plan. Using the Caribbean island of Barbuda as a case study, we present a simple approach for evaluating alternate marine spatial plans, offering a user-friendly framework to facilitate stakeholder engagement and data driven decision making. We generate a variety of hypothetical marine spatial plans, where each plan is designed to meet a specific ecological, fishery, or tourism objective. We then develop a tradeoff analysis to evaluate and compare these alternate plans, including determining their fishery impact by employing a simple spatial population dynamics model of the Caribbean spiny lobster (*Panulirus argus*) fishery and calculating changes in yield over time. Ecological and tourism impacts are evaluated based on the total area and diversity of habitat that is protected, and protected habitat that is accessible to recreational divers/snorkelers, respectively. This study demonstrates how a marine spatial plan can be successfully designed and evaluated using the best available science to balance a diverse set of objectives, even in data-limited situations.

KEY WORDS: Marine spatial planning, tradeoff analysis, small-scale fisheries, Barbuda

INTRODUCTION

Effective and comprehensive marine resource management solutions are critical to preserving ecosystem services such as fisheries, ecotourism, and cultural values (Crowder and Norse 2008, Foley et al. 2010). Marine Spatial Planning (MSP) has been identified as a valuable framework that allows decision-makers to preserve ecosystem services while zoning for diverse human uses (Douvere 2008, Crowder and Norse 2008). MSP has been defined as “a practical way to create and establish a more rational organization of the use of marine space and the interactions between its uses, to balance demands for development with the need to protect marine ecosystems, and to achieve social and economic objectives in an open and planned way” (Ehler and Douvere 2009). Key features of an MSP process include defining goals and objectives, creating potential plans, evaluating these plan scenarios against planning objectives, and gathering input from relevant stakeholders.

MSP is challenging because ocean ecosystem services often exhibit complex interactions whereby the delivery of one service may compromise the delivery of another (Lester et al. 2013). For example, the impacts of fishing may have a negative effect on recreation and tourism. Thus, stakeholders that value different ecosystem services may have competing objectives for the outcomes of MSP. Within this context, managers are challenged to evaluate MSP options and make decisions that achieve multiple, often conflicting, objectives. In making these difficult decisions, scientific input is highly valued due to its objectivity, rigor, and the explicit characterization of uncertainty (Carr et al. 2010). Yet, scientific knowledge about marine systems is complex, which can make integration of science into policy problematic, especially when decision makers have a limited scientific background (Carr et al. 2010). Thus, many papers have attempted to make scientific knowledge more accessible for environmental management by distilling complex knowledge into scientifically based operational guidelines or “rules of thumb” for marine spatial planning (NRC 2001, Carr et al. 2010, Foley et al. 2010, Fernandes et al. 2012).

However, as shown by Rassweiler et al. (2014), existing scientific rules of thumb may do little to improve outcomes for MSP. Instead, a wide variety of analytical decision-support tools have been developed to help guide decisions toward effective management schemes (COS 2011). For example, the use of spatially explicit models can deliver better results than scientific rules of thumb, and allows for the transparent evaluation of ecosystem service tradeoffs (Rassweiler et al. 2014). A tradeoff analysis approach allows decision makers to identify situations where tradeoffs among ecosystem services are unavoidable, as well as situations where win-win outcomes are possible. It can also identify inefficient marine spatial plans, where the provision of at least one ecosystem service can be increased at no cost to another (Lester et al. 2013).

As an additional complication to scientifically informed MSP, the effects of a spatial plan on an ecosystem service may change over time. For example, in the case of a Caribbean lobster fishery, marine reserves may have a negative impact on the fishery in the short term, as fishing grounds are closed and landings decline. However, over time, well-placed no-take zones can benefit the fishery, rather than simply decreasing landings, as they protect the spawning lobster population and large adults move into fishable waters (Lipcius 2001, Ley-Cooper 2014). Therefore, it is important to include temporal dynamics in spatial models, so that tradeoffs can be accurately assessed over relevant timescales. However, few analytical

MSP methods currently incorporate temporal dynamics (Klein et al. 2009, Allnut et al. 2012). White et al. (2012) did develop a robust, quantitative framework for conducting dynamic ecosystem service tradeoff analysis to inform marine spatial planning and applied it to a case study in Massachusetts. This type of analysis allows users to quantitatively compare tradeoffs among multiple marine spatial plans over time, greatly improving the ability of stakeholders to reach desirable outcomes. However, this type of approach is extremely data-intensive. Here, we apply a similar approach, but have simplified the analysis to provide a dynamic analytical tool that can guide MSP in data-limited regions and help improve marine spatial planning outcomes (Costello et al. 2010).

This study uses the island of Barbuda as a case study to demonstrate a framework for conducting a simple but dynamic tradeoff analysis that may be implemented for the spatial planning of marine reserves. In 2012, the Barbuda Council, the local governing body on the island of Barbuda, began a marine spatial planning process with support from the Waitt Institute. Data on habitat were collected and scientific assessments were conducted on reef species and fisheries. Using the scientific data as guidance, the community was engaged in an MSP process of designing proposals that addressed their concerns and priorities. In August 2014, the Barbuda Council enacted a set of new coastal regulations that used scientific data to address community objectives (Johnson 2014). The coastal regulations include zones designated for moorings and protection from fishing, as well as new fishing regulations to improve fishery sustainability.

The tradeoff analysis we present evaluates the impact of several hypothetical MSP proposals on defined tourism, ecological, and fishery objectives. In our tradeoff framework we use the benthic habitat data and designated mooring zones created by the above initiative. In addition, we simulate Caribbean spiny lobster (*Panulirus argus)* spatial population dynamics to evaluate the tradeoffs of hypothetical, single objective marine reserve proposals. We examine tradeoffs and how they may change with reserve design over time given specific ecological, tourism, and fishery objectives. The methods we demonstrate here allow users to take advantage of the important features of more complex MSP models, namely temporal dynamics and tradeoff analyses, but in a manner that is more suited for data-limited contexts.

METHODS

Study Area

Barbuda is a small island located in the Leeward Islands in the Eastern Caribbean, making up part of the country Antigua-Barbuda. Barbuda has a land area of 160.56 km^2 and a population of approximately 1,600 people (CIA 2013). Antigua and Barbuda State has an Exclusive Economic Zone of approximately 110,225 km^2 of maritime claim, and 3,500 km^2 of this area is composed of a large shallow shelf known as the Barbuda Bank. The shelf around Antigua and Barbuda has approximately 180 km^2 of reef area, of which 62% is considered to be under high or very high threat, with the largest threat coming from fishing pressure (Burke and Maidens 2004). Tourism accounts for over half of the country's National GDP (Burke and Maidens 2004), with many of the island's tourists coming specifically for diving and other marine-related activities in Antigua; however, there are no dive shops in Barbuda (Phillpott 2004).

Barbudan fisheries target coral reef and demersal species on the wide island shelf (FAO 2007). The Caribbean spiny lobster (*Panulirus argus*) is the main target and source of income for commercial fishermen in Barbuda (Horsford and Archibald 2006). Approximately 54 vessels with a total of 118 fishermen currently participate in Barbuda's lobster fishery and its estimated annual value ranges from US$370,000 to $782,000 (Horsford and Archibald 2006, CRFM 2011). An assessment of spiny lobster in Barbuda completed in 2006 indicated the stock was being harvested sustainably (Horsford and Archilbald 2006). However, in 2012, Chinese buyers that export lobster increased the price paid per pound and some local fishermen expressed concern that the lobster stock was being depleted as a result (Butler 2013).

Data

We generated a rasterized grid with a 250m x 250m cell resolution for Barbuda and the surrounding ocean. The grid included 13,860 cells and covered 866.25 km^2. Benthic habitat data around the island were generated using IKONOS-2 satellite imagery collected by DigitalGlobe in 2012 with a 16 m^2 resolution from depths from approximately 0 to 30 meters and obtained from SeaSketch (www.seasketch.org). For the purposes of marine spatial planning, all benthic data were classified into one of the following categories at a 250 m^2 resolution: sand, seagrass, patch reef, continuous reef, and/or hard bottom (Figure 1). The Barbuda Council has designated mooring zones for visiting boats around the island of Barbuda and we obtained approximate coordinates for these mooring zones (Figure 1). No sufficient fishery data were available for use in our analysis.

MSP Value

We assigned an ecological, fishery, and tourism value to each 250 m^2 cell. The ecological value of a cell was determined by the number of habitats present, with the presence of seagrass and coral reef habitat given twice the weight of sand and hardbottom habitat (Figure 2a). We chose this metric because preserving habitat heterogeneity and diversity is key for a healthy ecosystem (Foley et al. 2010) and because in the Caribbean, coral reef and seagrass habitat in particular play a crucial role in marine ecological functioning (Harborne et al. 2006). The tourism value of a cell was determined by proximity to mooring zones. Cells inside a mooring zone were given the highest tourism value because these areas could be used by dive boats. Cells within 500 m of a mooring zone were given a value equal to half of the highest value, and cells between 500 - 1000 m of a mooring zone were given a value equal to 25% of the highest tourism score. A tourism score of zero was given to all other cells (Figure 2c). The fishery value for a cell was determined by presence of adult lobster biomass, which was assumed to occur in cells containing hardbottom or coral reef habitat (Figure 2b).

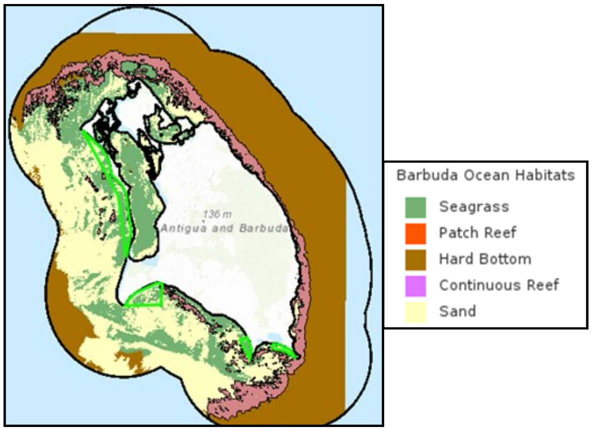

Figure 1. Benthic habitat data in Barbuda generated by satellite imagery. Approximate designated mooring zones are indicated in green. Figure provided by Sam Purkiss at Nova Scotia University.

We generated nine hypothetical marine reserve design proposals using Marxan, a free conservation planning software that uses simulated annealing to generate spatial reserve systems that achieve specified objectives (Figure 3) (Ball et al. 2009). Each proposal was designed to meet one objective while also fulfilling a percentage of protected waters requirement. The ecological objective was defined as maximizing the sum of the ecological value of cells present in a marine reserve network, the tourism objective was defined as maximizing the sum of the tourism value of cells present in a marine reserve network, and the fishery objective was to minimize the sum of fishery value of cells present in a marine reserve network. Three proposals were generated for each objective, containing 6%, 19%, and 31% of Barbuda's waters in marine reserves (Table 1).

Delay-Difference Model

While Marxan can be used for static design of MPA networks, it does not include spatial or temporal dynamics, such as using fish stock population dynamics to predict how these stocks will be impacted by the reserve systems in the future. Therefore, a Deriso-Schnute delay-difference model (Quinn and Deriso 1999) was used to simulate the spatial population dynamics of Caribbean spiny lobster (Quinn and Derios 1999, Sala et al. 2013). We generated a hypothetical initial lobster biomass to populate the model. Adult lobsters were assumed to occur in all cells that contained coral reef or hard bottom habitat (Peacock 2973, Saul, 2004). Lobster larvae were assumed to settle in cells containing seagrass habitat and migrate to adult habitat at the age of maturity (Peacock 2973, Saul 2004). An arbitrary carrying capacity (K) for adult lobster biomass was set for the waters around Barbuda and distributed equally among cells with adult habitat. A fishing harvest rate (f) of 0.5 per year was assumed for all cells containing adult lobsters. The model tracks lobster biomass in each area each year and accounts for growth of average individuals. *P. argus* life history parameters used in the model are presented in Table 2. Mature biomass of lobsters in patch i at the beginning of year $t+1$ is:

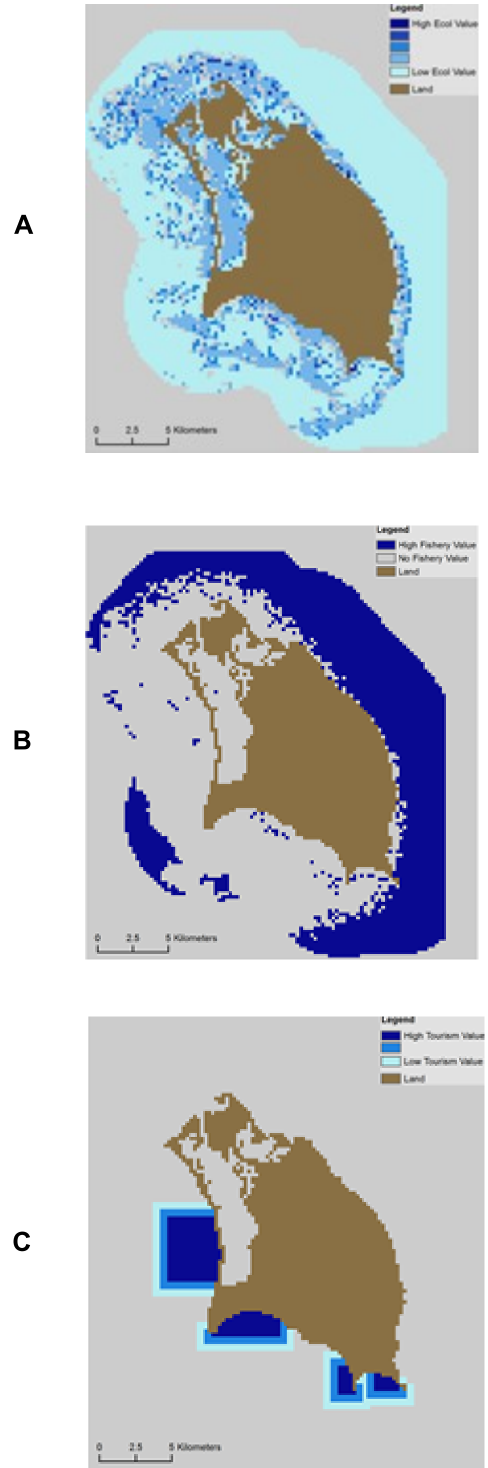

Figure 2. (a) Ecological value by area based on habitat diversity and presence of coral reef and seagrass habitat (b) Fishery value by area based on presence or absence of adult lobster habitat, and (c) Tourism value by area based on proximity to mooring zones.

Table 1. The nine marine spatial plan proposals that were generated using Marxan and evaluated in the trade off analysis.

Proposal Number	Proposal Objective	Total Area in Marine Reserve (km^2)	Percent (%) of Study Area Ocean in Marine Reserve	Percent (%) of Ecological Value in Marine Reserve	Percent (%) of Fishing Value in Marine Reserve	Percent (%) of Tourism Value in Marine Reserve
1	10% of total ecological score in Marine Reserves	39	6	10	4	19
2	33% of total ecological score in Marine Reserves	137	19	33	15	53
3	50% of total ecological score in Marine Reserves	222	31	50	41	42
4	Same total area is included in Marine Reserves as Proposal #1, but areas of highest fishing value are avoided in Marine Reserves	39	6	6	0	0
5	Same total area is included in Marine Reserves as Proposal #2, but areas of highest fishing value are avoided in Marine Reserves	137	19	16	1	23
6	Same total area is included in Marine Reserves as Proposal #3, but areas of highest fishing value are avoided in Marine Reserves	222	31	22	1	39
7	Same total area is included in Marine Reserves as Proposal #1, but mooring and anchoring areas are favored	39	6	10	1	87
8	Same total area is included in Marine Reserves as Proposal #2, but mooring and anchoring areas are favored	137	19	34	13	100
9	Same total area is included in Marine Reserves as Proposal #3, but mooring and anchoring areas are favored	222	31	48	30	100

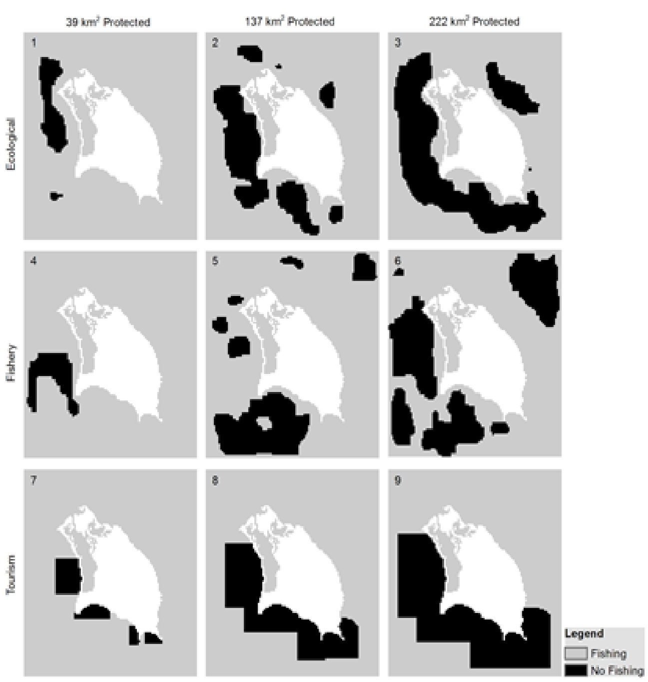

Figure 3. Marine reserve design for each of the 9 marine spatial plan proposals that were generated in Marxan and evaluated in the tradeoff analysis.

$$B_{i,t+1}=S_{i,t} * M_{i,t} * \rho + * s_{i,} t * M_{i,t} - s_{i,t} * S_{i,t}-1 * Mi_{,t}-1 * \rho - S_{i,t} * w_{k-1} * R_{i,t} * \rho + w_k * R_{i,t+1}$$

Where B_t is the biomass in patch i after adult movement in year t, $S_{i,t}$ is annual survival of lobsters year k and older, k is the age lobsters reach reproductive maturity and recruit to the fishery, ρ is the Brody growth coefficient, w_{k-1} is the average mass one year prior to recruitment, w_k is the average mass at recruitment, and $R_{i,t}$ is recruitment in patch i in year t.

Total annual survival S_t is calculated as:

$$S_{i,t} = s(1 - f_{i,t})$$

Where, s the natural survival and f is the annual harvest rate in patch i at time t

Given an equilibrium biomass for each patch $B_{i,0}$, the model solves for $R_{i,0}$:

$$B_{i,0} = \frac{R_{i,0} * w_k - S * w_{k-1}}{1 - S + \rho S^2}$$

The number of eggs (E) produced in each patch (i) at time (t) was calculated as:

$$E_{i,t} = \left(\frac{B_{i,t}}{2}\right) * g$$

Where ($B_{i,t-2}/2$) represents a 1:1 male to female ratio and g is a parameter representing fecundity at weight.

A closed system is assumed for all movement and dispersal. The number of larvae (D) reaching each patch (i) in the study area at time t is calculated as:

$$D_{i,t} = \sum_{j=1}^{13,860} E_{i,t} * p_{i,j}$$

Where $p_{i,j}$ represents the proportion of larvae moving from one patch to another. Larvae are dispersed in a Gaussian fashion, with the proportion of larvae dispersing from one patch to another decreasing with distance between patches:

$$p_{i,j} = exp(\frac{d_{i,j}^2}{2{\sigma_L}^2})$$

Where $d_{i,j}$ is the distances between two patches (i and j) and σ_L describes the larval dispersal range for the species.

The number of recruits (R) settling in patch i at time t is calculated as:

$$R_{i,t} = \frac{\alpha_i S_{i,t}}{\beta_i + S_{i,t,}}$$

Where α_i and β_i are patch specific density dependent Beverton-Holt parameters calculated as:

$$\alpha_i = \frac{R_{i,0} * L_i}{5h-1}$$

$$\beta_i = D_{i,0} \left(\frac{1-h}{5h-1}\right)$$

Where h is population's steepness parameter and L_i is a vector describing the available seagrass habitat in each patch.

Once lobsters become mature, they are assumed to move out of seagrass habitat to areas of coral reef and hard bottom habitat in the same Gaussian fashion that lobsters are distributed.

$$p_{i,j} = exp(\frac{d^2}{2\sigma_J^2})$$

Where i represents seagrass habitat and j represented adult habitat and σ_J represents juvenile movement range.

Adult movement between patches of adult habitat is also determined using Gaussian movement:

$$p_{j,j} = exp(\frac{d^2}{2\sigma_A^2})$$

Where σ_A is the adult movement range**.**

Yield (Y) from each cell containing adult lobster habitat (i) was calculated for each time (t) as:

$$Y_{i,t} = B_{i,t} * f_{i,t}$$

Where, B is adult lobster biomass in a cell and f is the annual harvest rate of cell i in time t. Initial harvest rate and initial lobster biomass were equal in all cells with adult lobster habitat, so each cell had either a high fishery value or no fishery value.

The delay-difference model was run with no marine reserves and an annual harvest rate of 0.5 applied to all cells with adult lobster habitat until the stock reached equilibrium. The starting spawning stock biomass used for our model was set at 22% of the stock's virgin stock biomass ($0.22B_0$), which is the equilibrium biomass that is reached when an annual harvest rate of 0.5 is applied to virgin stock biomass. These results served as a baseline against which to compare after marine reserve implementation.

For each of the nine proposals, harvest rates were set to 0 for all cells within a marine reserve, and the removed fishing effort (harvest) was proportionately redistributed among cells that were not included in reserves and had adult lobsters present.

Tradeoff Analysis

The ecological score of a proposal was calculated as the sum of the ecological value of cells present in marine reserves relative to the sum of the ecological value of all cells in the study area:

$$\sum \frac{Ecological\ value\ of\ cells\ inside\ marine\ reserves}{Ecological\ value\ of\ all\ cells}$$

The tourism score of a proposal was calculated as the sum of the tourism value of cells present in marine reserves relative to the sum of the tourism value of all cells in the study area:

$$\sum \frac{Tourism\ value\ of\ cells\ inside\ marine\ reserves}{Tourism\ value\ of\ all\ cells}$$

The fishery scores for each proposal represent the cumulative annual total lobster yield after 10 years relative to the yield with no marine reserves after 10 years:

$$\sum \frac{Cumulative\ lobster\ fishery\ yields\ after\ 10\ years\ with\ mari}{Cumualtive\ lobster\ fishery\ yields\ after\ 10\ years\ with\ no\ mar}$$

Table 2. Caribbean spiny lobster (*Palunirus argus*) life history parameters used in the delay-difference model.

Parameter	Value	Reference
Natural Mortality (M)	0.36	Phillips and Kittaka 2000
Asymptotic Length (L_{inf})	186.030 mm CL	Leocadio 2008
Von Bertalanffy Growth Coefficient (k)	0.280	Leocadio 2008
Theoretical Age at which Size is 0 (t_0)	-0.115	Leocadio 2008
Age at Maturity	2.14 years	Leocadio 2008
Steepness (h)	0.970	SEDAR 2010
Weight-Length Parameter a	0.00184	Bertelsen 2001
Weight-Length Parameter b	2.82	Bertelsen 2001
Fecundity Schedule	Egg count = –231,212 + 91.88 CLmm^2	Bertelsen 2001
Longevity	12 years	CRFM 2008
Larval Movement Parameter (σ_L)	1500 km	Butler et al. 2011
Juvenile Movement Parameter (σ_J)	2000 m	Acosta 1999
Adult Movement Parameter (σ_A)	900 m	Bertelsen & Hornbeck 2010

We also examined yield after one year to determine changes in yield over time. Additionally, the change in adult lobster biomass in the year the stock reached equilibrium for each proposal relative to the adult lobster biomass in the same year with no marine reserves was examined.

We compared the ecological, tourism, and fishery scores for each proposal. The average and standard deviation of the three scores was calculated for each proposal to determine if one proposal performed higher across all objectives. The average scores and standard deviations were also calculated for proposals with 5.5%, 19%, and 30.9% coverage. The average scores and standard deviations were also calculated for proposals with ecological, fishery, and tourism objectives, which determined the placement of marine reserves.

RESULTS

The ecological, fishery, and tourism scores for each proposal are presented in Table 3. Biomass and yield reached equilibrium in the lobster model after ten years (Figure 5).

The highest ecological score (0.5) occurred in proposal 3, which was designed with the objective of maximizing ecological value and including 31% of Barbuda's waters in marine reserves (Table 3). The highest tourism score (1.0) occurred in proposals 8 and 9, which were designed to maximize the tourism score and include 19 and 31% of Barbuda's waters in marine reserves, respectively (Table 3). Eight of the nine proposals had a fishery score that was negative because yields were always lower with marine reserves compared to yields without marine reserves (Table 3). Proposal 4, designed to exclude fishing areas and include 6% of Barbuda's waters in marine reserves, had no impact on yield and received a fishery score of 0 (Table 3). In general, ecological and tourism scores were positively correlated across proposals, and negatively correlated with fishing scores (Figure 4).

With the three objectives weighted equally, proposals 8 and 9 had the highest average score (0.38) (Table 3). However, proposal 8 (mean = 0.38, sd = 0.60) had a lower standard deviation, and a lower negative impact on fisheries relative to proposal 9 (mean = 0.38, sd = 0.67) (Table 3). Therefore, across all objectives, proposal 8 performed better, resulting in a tourism score of 1.0, an ecological score of 0.34, and a fishery score of -0.19 (Figure 4).

We did not observe an increase in fishing yield after the implementation of marine reserves relative to if there were no reserves for any of the proposals. However, the relative fishing yield after implementation of a marine reserve did increase over time (Figure 5). Specifically, proposals 2, 3, 8 and 9, all designed with the objective of maximizing ecological or tourism scores and to include 19 or 31% of Barbuda's waters in marine reserves, showed a sharp decline in yield one year after marine reserve implementation (Figure 5). However, for these proposals, after the first year of marine reserve implementation, yields increased annually and reached equilibrium after ten years, but never recovered to pre-marine reserve levels.

The impact of marine reserve coverage and placement varied across objectives. Coverage area of the reserve had a larger impact on ecological scores than placement (Figure 6), while placement had a larger impact on tourism scores (Figure 7). Both coverage area and placement had an impact on the fishery scores (Figures 6 and 7).

DISCUSSION

The incorporation of tradeoff analysis into MSP allows for the transparent quantification and analysis of the impacts of alternate spatial plans on multiple sectors. Our analysis shows that even in the absence of plans designed to meet multiple objectives, explicitly evaluating tradeoffs can improve outcomes for multiple sectors. For example, proposal 9, which was designed to meet tourism objectives, covers the same area of ocean and has a similar ecological score as proposal 3, but has a tourism score that is twice as high. It also has a lower negative impact on fisheries (-0.33 for proposal 9 vs. -0.42 for proposal 3).

An average score can be used to find the best plan across multiple potentially competing objectives, and a weighted average scores can be used to reflect the values of the community in terms of prioritizing different objectives. In our example, if all objectives are weighted equally, proposals 8 and 9 have the highest average score (0.38). If the objectives are not weighted equally and, for example, fishery objectives were valued twice as much as ecological or tourism objectives, proposal 8 would have the highest score (0.24).

Previous studies have found that marine reserves may result in net benefits to fisheries over time (Roberts et al. 2001, Russ et al. 2004, Ley-Cooper 2014). However, for marine reserves to benefit fisheries, spillover of biomass or export of larvae from inside the reserve to the outside fishing grounds must occur (Hilborn et al. 2004, Russ et al. 2004, Goni 2009). In our example, biomass did increase over time after the implementation of marine reserves, however, fishery yield did not increase. The decline in fishery yields as a result of the marine reserve in this study may have been an artifact of our modeling approach, including how we modeled adult movement, larval dispersal, and fishery fleet dynamics, and/or limited spillover effects for lobster from the simulated reserves.

In our model, settlement of larvae to seagrass habitat was limited by density-dependence. An increase in spawning stock biomass inside an MPA will not increase recruitment success outside of the MPA when recruitment is density-dependent (Goni et al. 2009). Juvenile habitat has been found as a bottleneck for production in other lobster stocks (Acousta 2001, Parrish and Polvina 2004, Ship 2011), and in general, fishery benefits from larval export do not occur, except when a spawning stock has been exploited to a level that has affected recruitment success (Kelly 2010, Ship 2011). In our model, an increase in biomass and larval production did occur; however, because recruitment was density-dependent and limited by the availability of seagrass habitat, any increase in biomass that occurred outside of the reserve was the result of adult lobster movement. Incorporation of the impacts of marine reserve protection on juvenile habitat quantity and quality

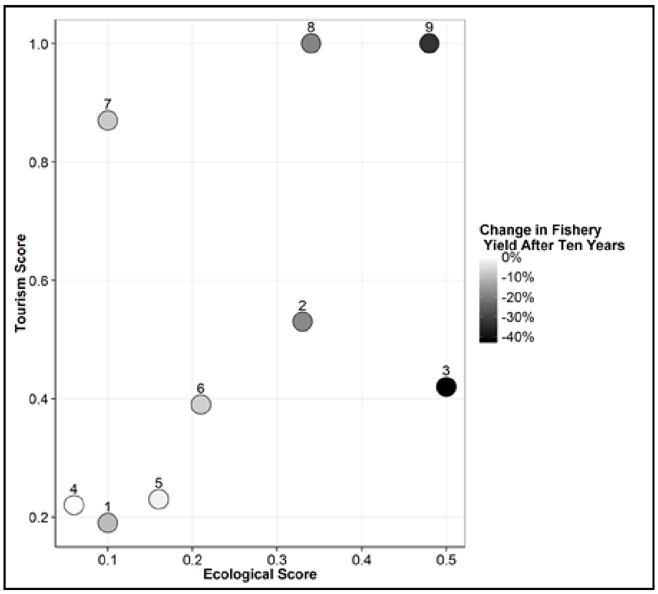

Figure 4. Model results for the tradeoff analysis. Each point represents a MSP proposal and its respective ecological, tourism, and fishery score.

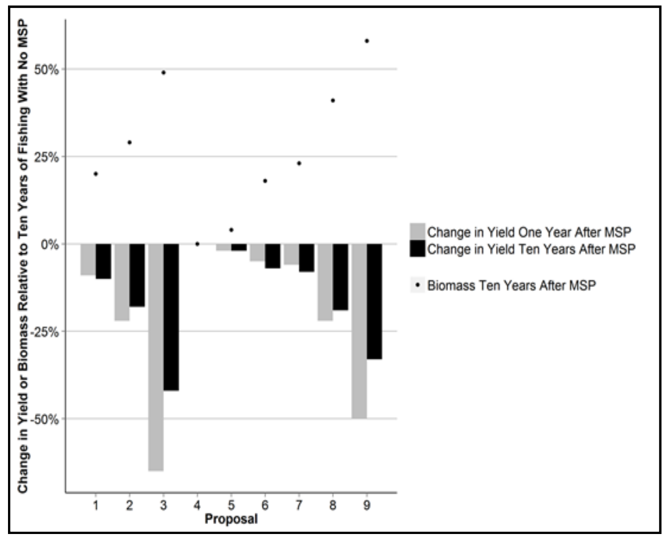

Figure 5. Change in biomass and yield relative to marine reserves. Represents change in annual future yields in year 1 and year 10 (not change in cumulative yields).
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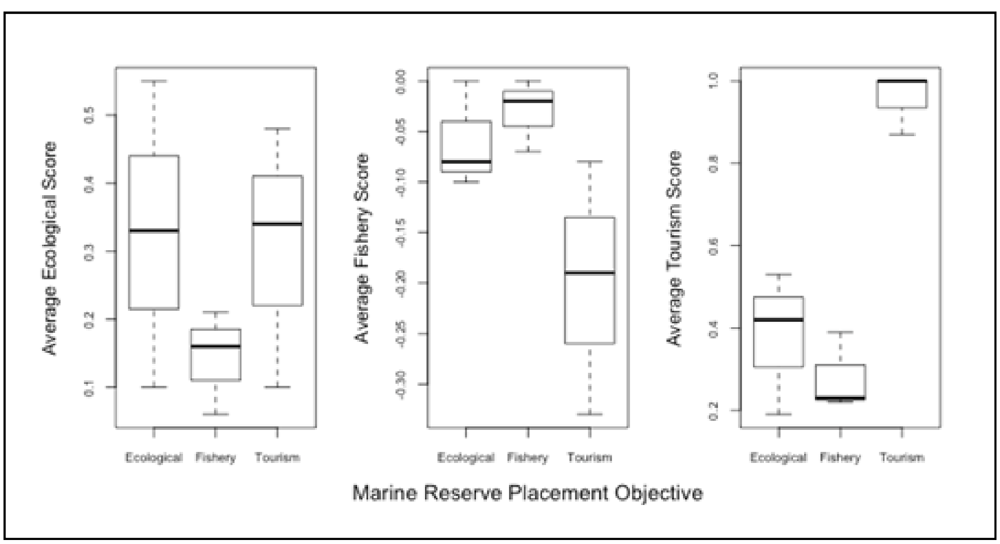

Figure 6. Average ecological, fishery, and tourism scores across 4%, 19%, and 31% marine reserve coverage.

could improve our model and would change fishery scores.

Determining a species' movement range and ability to move across a marine reserve boundary is critical in determining the impact of a reserve on surrounding fisheries (DeMartini 1993, Hobday 2010). In general, species with larger movement ranges relative to the MPA size will benefit less from the reserve (DeMartini 1993, McClanhan and Mangi 2000). Previous studies that showed marine reserves increased lobster fishery yields assumed lobsters had a larger movement range relative to reserve size (Ley-Cooper et al. 2014). In our study, lobster movement to outside of the reserve was minimal due to the assumed adult lobster movement parameter, and was not enough to counter the initial reduction in yield that occurred due to fishing ground closures. In New Zealand, marine reserves have generally had negative effects on lobster fisheries except in cases where reserve size relative to boundary length was small enough to allow for a larger portion of the lobsters to move outside of the reserve (Kelly 2003, Hobday 2010). The results of our model are sensitive to movement parameters, so it is important to include accurate estimates of adult and larval movement.

Furthermore, our model assumed fishing effort displaced by a marine reserve was evenly redistributed to all fishing areas outside of the reserve. In many cases, redistributed fishing effort will be concentrated near the boundary of reserves to best capture spillover (Ketllet et al. 2007, Goni et al. 2010). However, factors such as distance of marine reserve boundary from ports, fishery value, social customs, and expected benefit of the marine reserve are all potential factors that may play a role in determining how fishing effort is redistributed after implementation of a marine reserve (Stelzenmuller et al. 2008). If fishing effort in our study had been redistributed to capture spillover in biomass (due to adult movement), we may have observed a more consistent and larger increase in fishing value through the decade following marine reserve implementation.

Our marine spatial plan proposals only focused on the size and placement of marine reserves where fishing would be prohibited. We did not consider other fishery management strategies such as gear restrictions or seasonal closures. We assumed that tourism benefits and ecosystem benefits could occur simultaneously and did not account for the potential effects ecological benefits may have on fishing (e.g. effects of seagrass habitat protection on lobster stocks). If recreational diving and other tourism activity causes habitat degradation, separate areas should be designated for achieving diving (tourism) and marine conservation/ecological goals. In Barbuda, implementation of the new ocean zones is beginning, and space was allocated based on habitat data, information on the distribution of fishing effort, and community needs and priorities. The use of nets, a destructive fishing gear, has

Table 3. Proposal, score for each objective, and average score and standard deviation across all objectives.

Proposal	Ecological Score	Tourism Score	Fishery Score	Average Score	S.D.
1	0.10	0.19	-0.10	0.06	0.15
2	0.33	0.53	-0.18	0.23	0.37
3	0.50	0.42	-0.42	0.17	0.51
4	0.06	0.22	0.00	0.09	0.11
5	0.16	0.23	-0.02	0.12	0.13
6	0.21	0.39	-0.07	0.18	0.23
7	0.10	0.87	-0.08	0.30	0.50
8	0.34	1.00	-0.19	0.38	0.60
9	0.48	1.00	-0.33	0.38	0.67

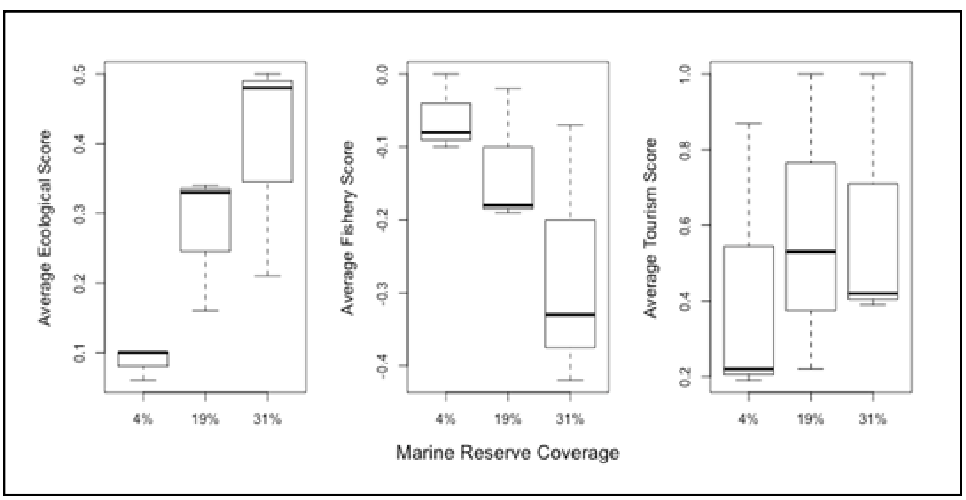

Figure 7. Average ecological, fishery, and tourism score across ecological, fishery, and tourism objective designs.

been banned in areas of high ecological value, mooring zones for boats have been designated, and marine reserves have been implemented in areas to maximize ecological value and minimize the cost to fisheries (Johnson 2014). Most importantly, a large lagoon area of seagrass and nursery habitat has been designated a no fishing zone for at least the next two years, which will likely benefit the lobster fishery over time.

Importantly, the proposals, objectives and value layers evaluated here are all hypothetical and are intended to demonstrate a useful approach/. In practice, marine reserve design should be based on clearly defined objectives specified by stakeholders (Pomeroy 2008, Ehler and Douvere 2009). Prior to conducting tradeoff analyses, stakeholder objectives should be clearly defined. The value of areas based on these objectives can be assigned using either quantitative or qualitative information, or a mixture of both. Fisheries impacted by the implementation of a marine reserve should be identified, and multiple fisheries may be aggregated or compared separately in the analysis, depending on the spatial distribution of the fisheries and the stakeholders' objectives.

This framework can be applied in data-limited situations, but still requires some inputs. The most critical components are information on the spatial distribution of ecosystem services and information on the population dynamics of key species. However, it is possible to use proxies for key inputs, such as habitat cover as a proxy for species distribution, and life history parameters needed to model population dynamics may be available from the literature or can be estimated from other data. In our analysis, the initial spatial distribution of lobster biomass was hypothetical. If this framework were being used to inform a real MSP process, spatially explicit abundance or catch data would be important to inform starting conditions for the model. Additionally, the framework can be adapted and built upon to improve accuracy in cases where more data are available. For example, more information about oceanographic currents and larval duration could improve the model's ability to capture larval dispersal patterns.

We have a presented a framework for evaluating marine reserve design proposals that can be easily adapted to other regions in the Caribbean that have more or less data available, and which can be applied to other fisheries. Although our proposals were hypothetical, we demonstrated that multiple objectives can be achieved with minimal tradeoffs through appropriate marine spatial planning. Our example illustrated that the impact of size and placement of a marine reserve may vary across objectives, and that life history of key species is important to consider when designing marine reserves. This framework presents a scientific tool that can be used to evaluate tradeoffs across a number of objectives to inform decision making during the MSP process.

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