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Forest conservation delivers highly variable coral reef conservation outcomes

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Abstract. Coral reefs are threatened by human activities on both the land (e.g., deforestation) and the sea (e.g., overfishing). Most conservation planning for coral reefs focuses on removing threats in the sea, neglecting management actions on the land. A more integrated approach to coral reef conservation, inclusive of land–sea connections, requires an understanding of how and where terrestrial conservation actions influence reefs. We address this by developing a land–sea planning approach to inform fine-scale spatial management decisions and test it in Fiji. Our aim is to determine where the protection of forest can deliver the greatest return on investment for coral reef ecosystems. To assess the benefits of conservation to coral reefs, we estimate their relative condition as influenced by watershed-based pollution and fishing. We calculate the cost-effectiveness of protecting forest and find that investments deliver rapidly diminishing returns for improvements to relative reef condition. For example, protecting 2% of forest in one area is almost 500 times more beneficial than protecting 2% in another area, making prioritization essential. For the scenarios evaluated, relative coral reef condition could be improved by 8–58% if all remnant forest in Fiji were protected rather than deforested. Finally, we determine the priority of each coral reef for implementing a marine protected area when all remnant forest is protected for conservation. The general results will support decisions made by the Fiji Protected Area Committee as they establish a national protected area network that aims to protect 20% of the land and 30% of the inshore waters by 2020. Although challenges remain, we can inform conservation decisions around the globe by tackling the complex issues relevant to integrated land–sea planning.

Key words: conservation planning; coral reef; Fiji; fishing; forest; integrated land–sea planning; protected area; spatial conservation prioritization; watershed pollution.

Introduction

Marine protected areas (MPAs) are the cornerstone of most marine conservation strategies as they are effective at reducing one of the most prevalent threats to marine ecosystems, overfishing (Halpern 2003, Lester and Halpern 2008). MPAs may not be able to adequately protect marine ecosystems in places where land-based activities (e.g., forestry) negatively impact marine ecosystems (Boersma and Parrish 1999, Klein et al. 2010*a*). As a result, scientists have argued for strategies that consider connections between the land and sea for protecting marine ecosystems, such as marine ecosystem-based management and integrated coastal zone management (Dubinsky and Stambler 1996, Cicin-Sain and Belfiore 2005, McLeod and Leslie 2009).

There are a variety of land–sea connections important to marine resource management, including (1) land–sea processes (e.g., oceanic foraging by seabirds nesting in coastal forests); (2) cross-system threats (e.g., pollution and sedimentation from watersheds); and (3) socioeconomic interactions (e.g., the impact of land-based threats on marine-based tourism) (Beger et al. 2010, Alvarez-Romero et al. 2011). The utility and necessity of incorporating land–sea connections into systematic conservation planning is well established (Dutton et al. 1994, Stoms et al. 2005, Gordon 2007, Olsson et al. 2008). Yet, planning for the land and sea is typically conducted separately, and we often act as though the ecological and socioeconomic systems are unconnected (Beck 2003, Stoms et al. 2005). Although conceptual frameworks for pursuing land–sea planning have been proposed (see Stoms et al. 2005, Beger et al. 2010,

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Alvarez-Romero et al. 2011), there are few practical examples of how to implement these ideas into marine conservation planning.

Decisions about the location of protected areas are often supported by spatial conservation prioritization analyses. With few exceptions (see Tallis et al. 2008, Hazlitt et al. 2010), approaches for identifying priorities inclusive of land–sea connections use large analysis units (e.g., ecoregions) and/or are across multiple countries (Halpern et al. 2009, Jenkins et al. 2010, Klein et al. 2010*a*). These "large-scale" approaches are informative for some types of conservation decisions (Mills et al. 2010), but are of limited utility when applied to protected area design, which would require modification and higher resolution data. Others have developed innovative approaches to the design of protected areas that are inclusive of land–sea connections and found that priorities for conservation change when land–sea connections are incorporated (Tallis et al. 2008, Hazlitt et al. 2010). Although each approach has advanced integrated land–sea planning, many aspects remain unresolved and are critical in informing how we set priorities for actions (Beger et al. 2010, Alvarez-Romero et al. 2011).

It is well established that land-based human activities can impact marine ecosystems (Fabricius 2005, Croke and Hairsine 2006, Walling 2006, Diaz and Rosenberg 2008, Halpern et al. 2008). Negative impacts can result from intensive land-use changes, whereas positive impacts can result from conservation actions that preserve or restore linked terrestrial systems. Understanding how and where terrestrial conservation influences marine ecosystems is an important yet unresolved aspect in land–sea planning. Here, we address this gap and develop a new integrated land–sea planning approach that can be used to inform fine-scale spatial management decisions, with a focus on forests and coral reefs. Our aim is to determine where the protection of forest can deliver the greatest return on investment for coral reef ecosystems. To address this aim, we demonstrate our approach in Fiji and show how it can be used to answer relevant questions to land–sea planning: (1) How and where does forest conservation reduce impact of nutrient and sediment runoff on coral reefs? and (2) If forests are protected for conservation, where are the priorities for coral reef conservation?

Materials and Methods

Study region

Initiatives for conserving Fiji's marine and terrestrial ecosystems have been supported by communities, nongovernment organizations, and government. At the national scale, there have been multiple efforts to identify priorities for conservation (WWF 2004, Fiji Department of Environment 2007) and the government has committed to protecting 30% of its inshore waters and 20% of its land by 2020 (Jupiter et al. 2011). In protecting its marine resources, the benefits of employing an ecosystem-based management approach have been acknowledged, supporting the inclusion of integrated land–sea planning and socioeconomic factors into the decision making process (Clarke and Jupiter 2010). A national Protected Area Committee (of which S. Jupiter is a member) was formed through the Department of Environment in 2008 to develop policies and priorities and support the establishment of an adequate and representative protected area system (Jupiter et al. 2011). To support these processes, we demonstrate our land–sea planning approach on Fiji's three largest islands: Viti Levu, Vanua Levu, and Taveuni. We use the best available data to represent its coastal catchments, forested areas, coral reefs, land-based runoff, fishing pressure, and opportunity costs of implementing terrestrial and marine protected areas, described below. Data were summarized to 1-km^2 planning units, each of which could be selected for protection and evaluated in terms of its contribution to our planning objective.

Land–sea prioritization approach

The first step in formulating any conservation problem is to define a quantifiable objective (Possingham et al. 2001). Here, our objective was to maximize coral reef condition through investment in terrestrial protected areas across Fiji's coastal watersheds. We created a simple model to estimate coral reef condition as influenced by watershed-based pollution and fishing impacts, as no other suitable models were available to address our objective. These stressors were chosen as they are the only stressors for which we have consistent data across the whole study region that can be mitigated through implementation of a protected area. We acknowledge that a more comprehensive range of stressors influence the condition of coral reefs (Halpern et al. 2008), and given available data, any model of coral reef condition or ecosystem state (Tallis et al. 2011) could be used to implement our prioritization approach.

We modeled the condition, C, of each 1-km^2 coral reef, i ($i = 1, \ldots, 7759$), as a function of watershed-based pollution and fishing pressure:

$$C_i = (e^{-\alpha p_i})[(e^{-\beta f_i})(1 - \delta) + \delta]$$

where p_i and f_i are variables that quantify the amount of watershed-based pollution and fishing pressure at each reef, assuming no conservation strategies are implemented. The remaining parameters are constants, where α indicates the rate of coral reef degradation with increasing watershed-based pollution (see Plate 1), β indicates the rate of coral reef degradation toward a condition of δ with increasing fishing pressure, and $0 \leq \delta \leq 1$ is the expected condition of a heavily overfished coral reef with no watershed-based pollution. We populated the model variables (p_i, f_i) using existing spatial data, whereas the constants (α, β, δ) were derived from the literature, where possible, and varied to determine the sensitivity of the prioritization outcome to their value.

Watershed-based pollution

We model the condition of coral reefs relative to each other. Thus the values obtained from our model do not represent the actual condition of each coral reef; they can only be used to compare the condition between reefs. Our model of relative coral reef condition relies upon information about the amount (p_i) and impact (α) of watershed-based pollution on reefs. We determine the amount of watershed-based pollution, p_i, reaching reef i by adding the pollutants coming from all watersheds with runoff reaching reef i:

$$p_i = \sum_{l=1}^{M} V_{li} F_l$$

where V_{li} is the amount of pollution from watershed l ($l = 1, \ldots, M$) reaching reef i assuming all terrestrial vegetation has been cleared, and F_l is the proportion of watershed l that is not forested or protected. To estimate V_{li}, we use a method developed by Reefs at Risk Revisited that represents a proxy for sediment, nutrient, and pollutant delivery to coral reefs given limited data (Burke et al. 2011). Their method relies upon information about land cover type, slope, soil characteristics, precipitation, dams, and mangroves to predict the amount of pollution produced by each watershed, but does not consider additional types of land-based pollution resulting from livestock, urban or industrial sources. While we recognize that the concentrations of suspended sediment, nutrient species and associated pollutants delivered by streams to the nearshore is affected differentially by land use, human density and various downstream physical and biological processes (McKergow et al. 2005*a*, *b*), the Reefs at Risk model represents the best available information as we have no local data with which to parameterize new hydrological models. We improved upon their model by using higher resolution land cover and mangrove distribution data and ran it for 391 coastal watersheds. The pollution from each watershed was then distributed to each coral reef through a distance-based plume model, developed in Halpern et al. (2008) and used by Burke et al. (2011). Within each watershed, we assumed that the source of pollution is evenly distributed. To evaluate the impact of protecting existing forest on the amount of pollutants reaching each reef, V_{li}, we use a control variable that allows us to protect each 1-km^2 unit of land, j ($j = 1, \ldots N$), which in turn influences the amount of forest in catchment F_l:

$$F_l = \left[\frac{\sum_{j=1}^{N_l} (1 - w_j x_j)}{N_l} \right]$$

where w_j is a state variable that equals 1 if j is initially forested and x_j is a control variable that equals 1 if j is protected, otherwise both variables equal 0. Only sites that are initially forested and subsequently protected contribute to forest cover. We arbitrarily considered j to be forested if at least 90% of it was densely vegetated. We defined vegetation using a land cover map that we developed based on a mosaic of satellite imagery acquired from 2000 to 2002, which are the most recent freely available images with minimum cloud cover. The satellite imagery was captured at a resolution of 30 m with Landsat 5 Thematic Mapper. We produced the land cover map using a supervised maximum-likelihood classification, which assumes that the statistics for each class in each band are normally distributed and calculates the probability that a given pixel belongs to a specific class.

Finally, we explored a range of estimates ($0.01 \leq \alpha \leq 1.49$) for the rate of coral reef degradation with increasing watershed-based pollution. These estimates represent the gradient of change across various coral reef attributes known to be sensitive to land-based pollution (e.g., coral cover, octocoral cover, octocoral richness, and macroalgae cover) (Fabricius and De'ath 2004, Fabricius et al. 2005).

Fishing pressure

Our model of relative coral reef condition relies upon information about fishing pressure, f_i, and the impact of fishing on coral reefs (β and δ). We estimated initial fishing pressure on a coral reef prior to implementation of a protected area, Z_{i0}, and incorporated a control variable into f_i that allows us to protect each 1-km^2 unit of coral reef, such that

$$f_i = Z_{i0}(1 - y_i)$$

where y_i is a control variable that equals 1 if i is protected, otherwise is 0. We considered protected to mean "no-take" but acknowledge that there are very few no take zones in Fiji (Mills et al. 2011). Fine-scale spatially explicit data representing fishing pressure prior to protection, Z_{i0}, are rarely available as they are costly to collect and most places have implemented some form of fisheries management (Scholz et al. 2004, Klein et al. 2008). Fishing information in Fiji only exists for small regions (Adams et al. 2011) or is from global data sets that are too coarse for this analysis (Halpern et al. 2008, Burke et al. 2011). However, because fishing pressure on coral reefs in Fiji is roughly correlated with coastal population (Teh et al. 2009), we estimated the relative initial fishing pressure on each coral reef using a model based on coastal population that assumed fishing is allowed on all coral reefs. We defined Z_{i0} as the number of people within a 35-km buffer of a coral reef, i, using high resolution population data (United States Department of Energy 2008). The 35-km buffer was chosen for two reasons: (1) it captures population data in Fiji's coastal districts, where fishers are most likely to live; and

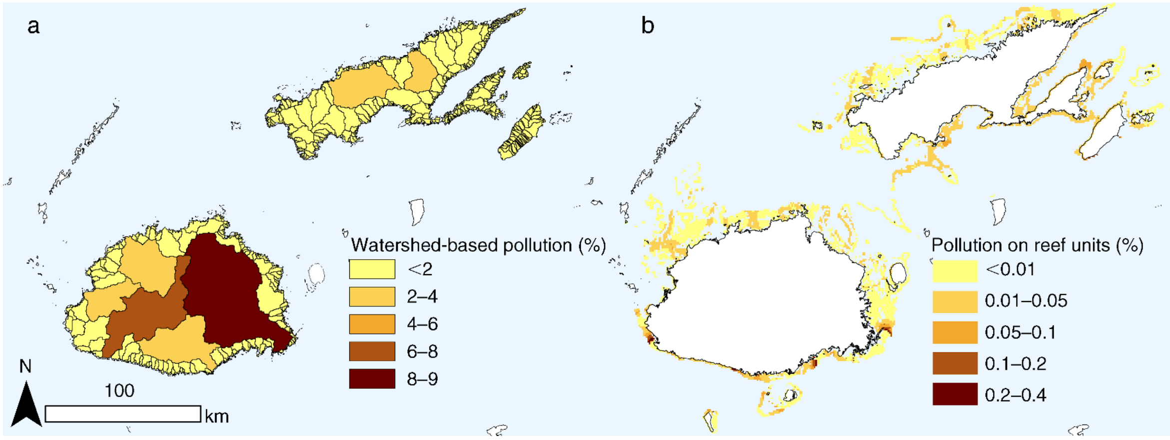

Fig. 1. Modeled amount of watershed-based pollution (a) in each watershed and (b) on each reef pixel, assuming all forest has been cleared. Amounts are expressed as the percentage of total pollution in the entire study region, which is the three largest islands of Fiji: Viti Levu, Vanua Levu, and Taveuni.

(2) it provides a nonzero value for all coral reefs, which is representative of the reefs in the study region as most are fished.

In coral reef ecology, there is a great deal of uncertainty about the impact of fishing on coral reef health. The uncertainty may arise due to differing definitions of reef condition, conflicting species- or tropic-level responses (Mumby 2006, Mumby and Harborne 2010), lag times in observing effects (Selig and Bruno 2010, Graham et al. 2011), and changes that may only occur after specific thresholds are surpassed (Dulvy et al. 2004). Given this uncertainty, we used a range of hypothetical values to estimate β and δ. We varied β relative to α as they both are measures of decline, where β ranged from 0.5α to 2α. Given the constraint that $0 \leq \delta \leq 1$, where higher values represent coral reefs in better condition, we vary this parameter between 0.1 and 0.5 to represent a range of possible values estimating the condition of an overfished coral reef with no watershed-based pollution.

Algorithm

The value of implementing a terrestrial or marine protected area in any 1-km^2 unit of forest or reef, respectively, was calculated as the increase of relative coral reef condition across all reefs. Thus, the cost-effectiveness of implementing a terrestrial or marine protected area was defined as the improvement in relative condition across all reefs, divided by the cost of implementing the action at the forest or reef pixel, respectively.

Opportunity costs

We estimated the opportunity costs associated with implementing forest protected areas in Fiji using premium payments to landholders and rent data from three existing projects across an area of 117.4 km^2: (1) Cakaudrove logging concession; (2) Naboro water catchment project; and (3) Namau water catchment project, information that was obtained from the Fiji Native Land Trust Board. We determined the present value of annual rents over 99 years using a discount rate of 5%. The annual lost-opportunity cost of each program to the landowners was \$738/km^2, \$2231/km^2, and \$1619/km^2, respectively (values are Fijian dollars, FJD). Just as the cost of these and most conservation projects differ across a region (Carwardine et al. 2008), we expect that the cost of protecting forest across Fiji are spatially heterogeneous. However, we were unable to determine what drives differences in forest conservation costs across Fiji and so we used the average opportunity cost from the three projects (\$1520/km^2).

Similar challenges were faced when trying to estimate the opportunity cost of coral reef protected areas in Fiji. The most representative cost estimate was derived from Adams et al. (2011), who modeled the opportunity cost of marine protected areas in Fiji's Kubulau region. They found that spatial variations of opportunity cost were, in part, driven by reef type. We calculated the maximum potential opportunity cost on fringing and non-fringing reef to be FJD\$4762/km^2 and FJD\$1649/km^2, respectively, and used this information to predict the opportunity cost of protecting coral reefs across our study region. Although we are uncertain if the relative costs between land and sea conservation are representative of reality across the entire planning area, it is not relevant here as we do not conduct a trade-off analysis between land and sea conservation actions.

Results

Runoff modeling

The results of the watershed-based pollution modeling are presented in Fig. 1, showing the percentage of the total pollution produced by each watershed and the distribution of pollution to each reef unit, *i*. Large watersheds generally have more pollution than smaller

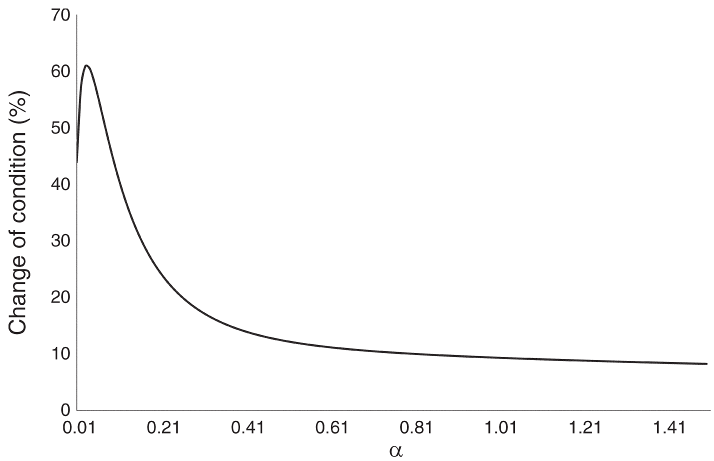

FIG. 2. Change in coral reef condition when α (rate of coral reef degradation with increasing watershed-based pollution) was varied between 0.01 and 1.49 for two conservation scenarios: (1) protect all forested areas and (2) clear all forested areas. The values of β (rate of coral reef degradation toward a condition of δ with increasing fishing pressure) and δ (expected condition of a heavily overfished coral reef with no watershed-based pollution) were set to 0.

ones because we assumed that all terrestrial vegetation was cleared. In most cases, coral reefs with the greatest amount of pollution were influenced by several nearby watersheds.

Prioritization

Prioritization results differed depending on which combination of constants, α (rate of coral reef degradation with increasing watershed-based pollution), β (rate of coral reef degradation toward a condition of δ with increasing fishing pressure), and δ (expected condition of a heavily overfished coral reef with no watershed-based pollution), were employed in the model. First, we explored the sensitivity of the approach to α by comparing the percent difference in total relative coral reef condition for two conservation scenarios: (1) protect all forested areas ($x_j = 1 \ \forall \ w_j = 1$) and (2) clear all forested areas ($x_j = 0 \ \forall \ w_j = 1$) (Fig. 2). The biggest difference (58%) in condition between the two scenarios was when $\alpha = 0.03$ and the smallest difference (8%) was when $\alpha = 1.49$. We would expect the greatest difference between the scenarios at an intermediate value of alpha, which happens to be 0.03. This is because when alpha is zero then forests don't affect reefs, hence there is no difference between the scenarios. Whereas when alpha is very large then reef health is very low whatever you do, hence the difference between the scenarios is also small. We then tested the sensitivity of total coral reef condition to parameters β and δ, using $\alpha = 0.03$ and $\alpha = 1.49$ (Fig. 3). Relative condition decreased as β increased, a trend more pronounced when $\alpha = 0.03$ (Fig. 3a). When $\alpha = 1.49$ (Fig. 3b), condition was less sensitive to changes in β. Regardless of α and β, condition increased as δ increased. We present the remainder of results spatially and produced them using: $\alpha = 0.03$ to highlight where priorities will be most different; $\beta = 0.03$, representing a situation where the impacts of fishing are equivalent to the impacts of watershed-based

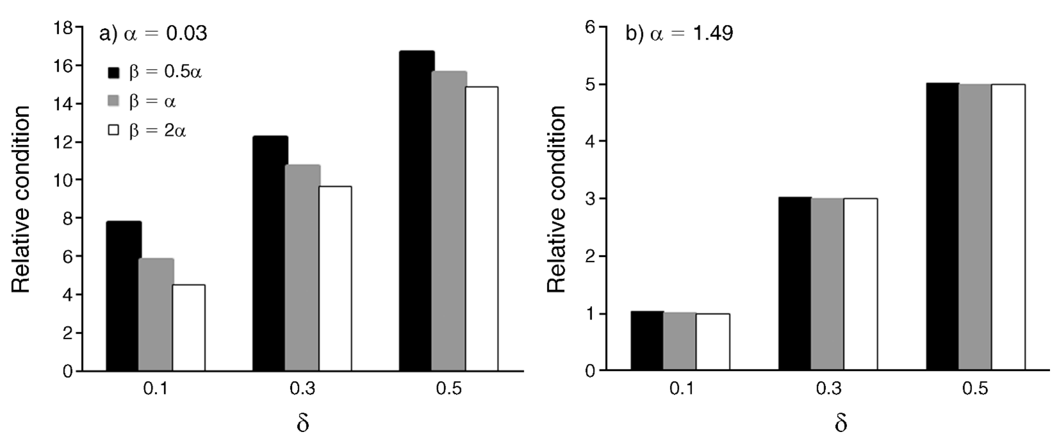

FIG. 3. Sensitivity of total coral reef condition to parameters to α, β, and δ. The condition value has been normalized to be relative to the scenario with a minimum condition for the constants α (the rate of coral reef degradation with increasing watershed-based pollution), β (the rate of coral reef degradation toward a condition of δ with increasing fishing pressure), and δ (expected condition of a heavily overfished coral reef with no watershed-based pollution), where $\alpha = 1.49$, $\beta = 2.98$, and $\delta = 0.1$.

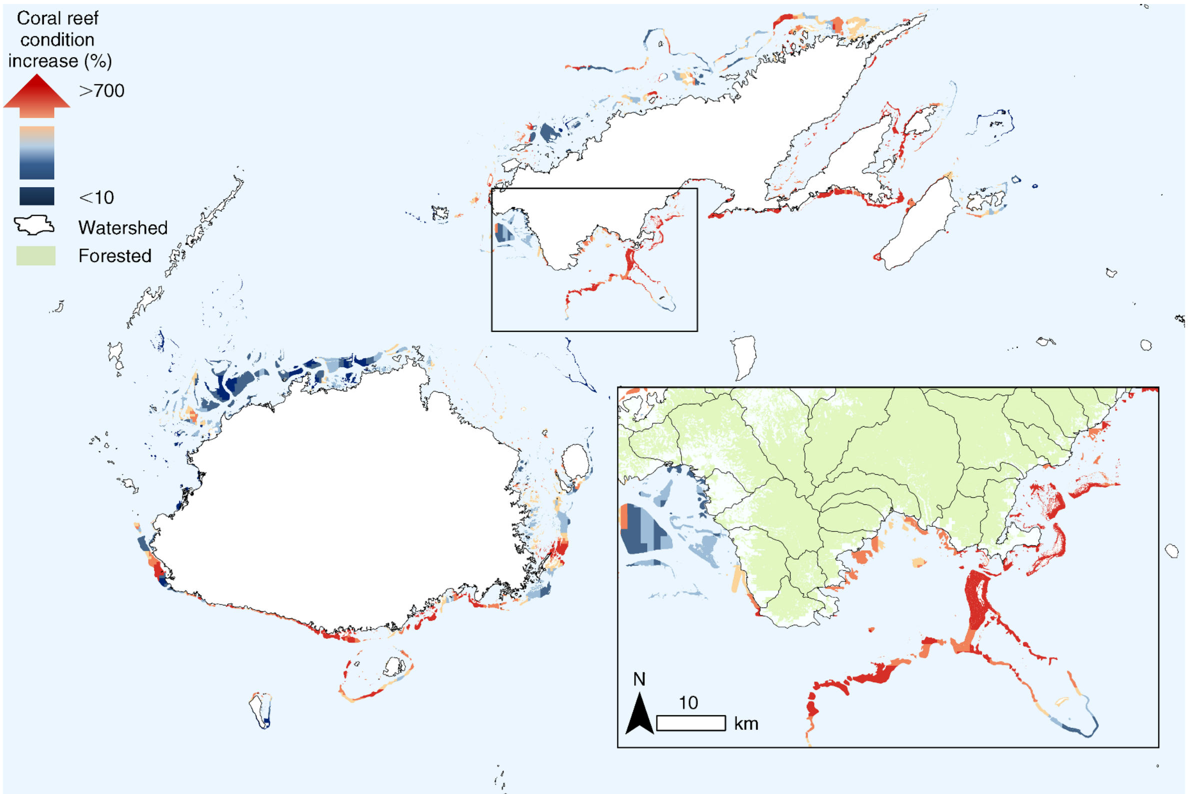

FIG. 4. Percentage increase in coral reef condition when all forests are protected rather than cleared. Results are reported for each 1-km^2 coral reef pixel. The inset shows results from southern Bua Province on the island of Vanua Levu in greater detail.

pollution (α); and $\delta = 0.1$, representing the worst case scenario for reef condition that we evaluated.

For the scenario where $\alpha = 0.03$, $\beta = 0.03$, and $\delta = 0.1$, total coral reef condition is 40.4% greater when all forests are protected rather than cleared. The relative difference in condition of individual coral reefs is shown in Fig. 4. Relative differences are generally most pronounced on reefs influenced by watersheds containing a large proportion of forest. Smaller differences are generally found either on reefs influenced by watersheds containing a smaller proportion of forest; and/or are far from land as they are affected by relatively less watershed-based pollution.

To evaluate where forest conservation most impacts relative reef condition, we calculated the cost-effectiveness of protecting each forested pixel (Fig. 5). The return on investment in protecting forests for coral reef condition diminishes with protection. For example, we found that protecting the top (i.e., most cost-effective) 2% of forest pixels delivers 490 times the benefit (i.e., improvement in relative reef condition if forest is protected) as protecting the bottom 2% of pixels (Fig. 5). The cost-effectiveness of protecting any forest pixel within a watershed is similar because: our model estimating watershed based pollution assumes each land pixel, *j*, was deforested and contributes equally to the production of pollution within a watershed; our estimation of terrestrial opportunity cost only varied with area of forest per pixel; and all forest pixels contained >90% forest.

Our final result shows the cost-effectiveness of protecting each coral reef pixel when all forest is protected for conservation (Fig. 6). Cost-effectiveness was calculated as the increase of relative coral reef condition across all reefs, divided by the cost of implementing a protected area at the reef pixel. This result highlights priorities for implementation of marine protected areas that aim to improve coral reef condition, where more cost-effective reefs are a higher priority for conservation.

DISCUSSION

Despite the rapid development of spatial conservation prioritization research (Moilanen et al. 2009), few approaches have been developed to inform integrated land–sea conservation decisions (Tallis et al. 2008, Halpern et al. 2009, Hazlitt et al. 2010, Klein et al. 2010*a*). Existing approaches, including ours, typically focus on one aspect of land–sea planning, whether a land–sea process (Hazlitt et al. 2010) or cross-system threat (Tallis et al. 2008, Halpern et al. 2009, Klein et al. 2010*a*). Yet, in any planning region, there are many important land–sea processes, cross-system threats, and socioeconomic interactions operating at a range of

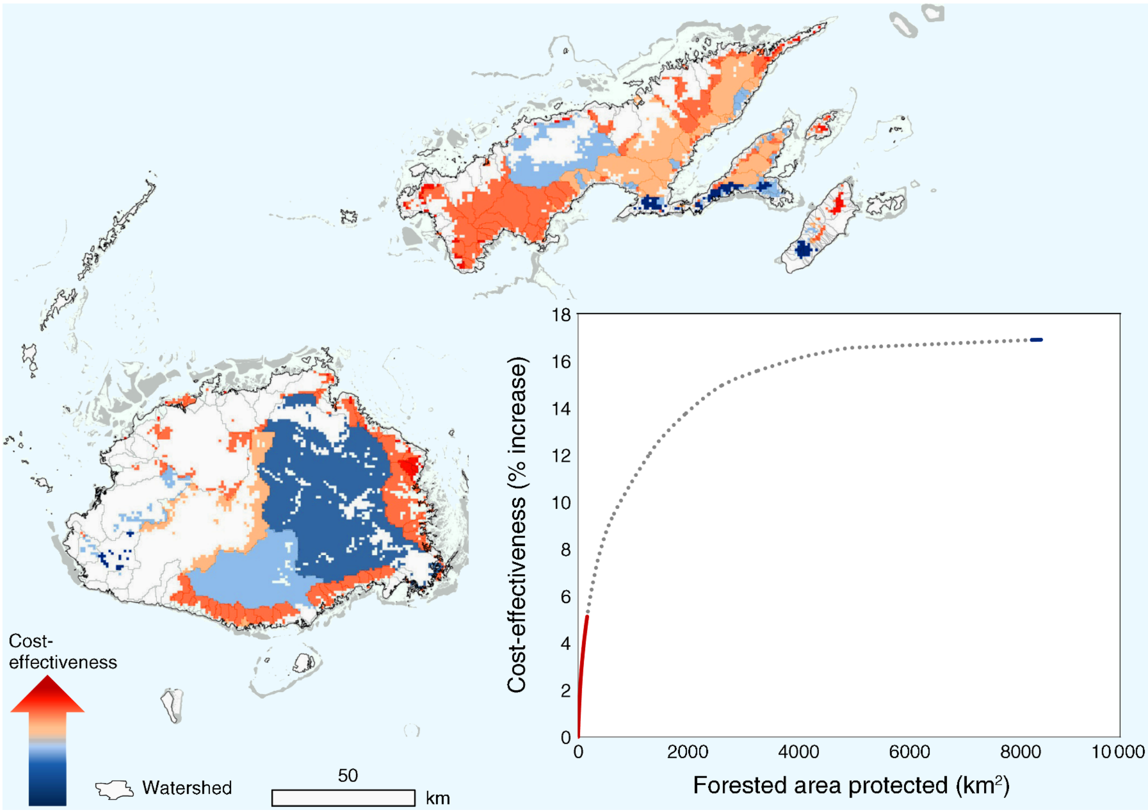

FIG. 5. Cost-effectiveness of protecting forests and cumulative benefits of protecting each forest pixel, highlighting the most (red, top 2%) and least (dark blue, bottom 2%) cost-effective areas for conservation. The cost effectiveness of implementing a terrestrial protected area was defined as the improvement in relative condition across all reefs, divided by the cost of implementing the action at the forest pixel. The colors of the lines on the graph correspond to the colors on the map: the red line is the top 2%, and the dark blue line (which is the end of the line on the right) is the bottom 2% of cost-effective areas. The dotted gray line represents the rest of the data.

spatial and temporal scales. The paucity of work in integrated land–sea planning is, in part, due to the difficulties of understanding and compiling data to represent any one of these connections (Alvarez-Romero et al. 2011). Although an ideal approach could accommodate a more comprehensive suite of land–sea connections, researchers must first overcome challenges associated with understanding and representing individual connections.

The aim of our study was to develop a fine-scale approach that could inform how and where the protection of land can deliver the greatest return on investment for marine conservation, and not to develop an index for coral reef condition. To solve the objective of maximizing coral reef condition, we had to estimate how threats to coral reefs influenced coral reef condition, but any estimation of condition could be used if available. We focused on two threats, watershed-based pollution and overfishing, and two conservation actions, protection of forests and coral reefs, and we evaluated the potential relative change to reef condition from removing those threats. We recognize that our model did not consider the range of interacting factors affecting condition, such as global stresses induced by climate changes (Anthony et al. 2011) (e.g., ocean acidification and sea surface temperature changes), local stress from explosions of coral predators (e.g., crown of thorns), coastal development, and natural stress from tropical cyclone damage (Wilson et al. 2006). Nor did our model take into consideration the differential impacts of each threat on reef types. Coral reef condition is challenging to model, especially across a large area like Fiji, due to the complex nature of ecological processes operating across multiple scales on coral reefs (Connell et al. 1997, Hughes and Connell 1999, Done et al. 2010) and limited data available to represent these processes. Our estimations for the amount of watershed-based pollution and fishing pressure influencing coral reefs across Fiji did not consider all relevant biophysical drivers (e.g., currents driving distribution of pollution on coral reefs) or socioeconomic factors (e.g., distance of coral reefs to markets) (Cinner and McClanahan 2006), as data were unavailable across the entire study region. Other watershed impact data were considered, including satellite estimates of sediments and chlorophyll (Maina

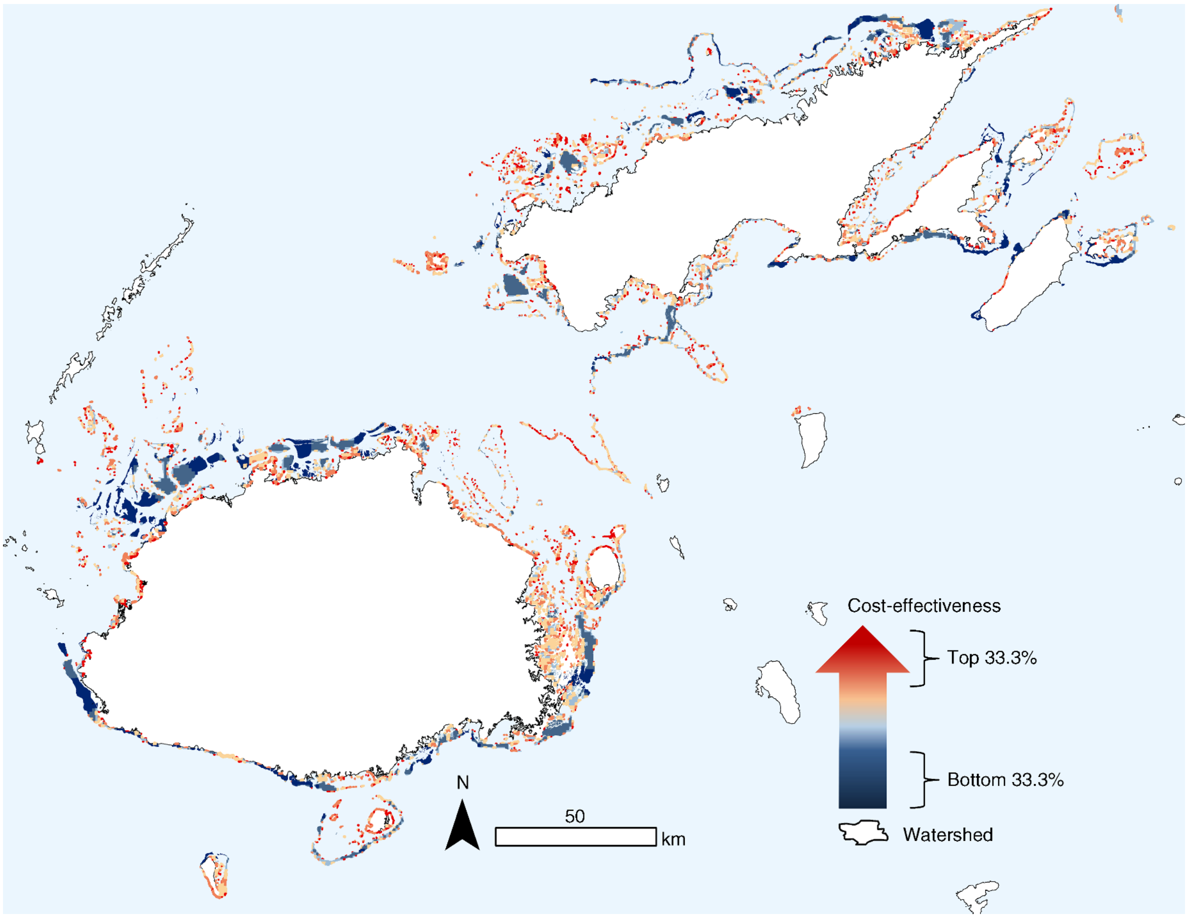

FIG. 6. Cost-effectiveness of protecting coral reefs, assuming all forest is protected for conservation. Cost-effectiveness of implementing a marine protected area was defined as the improvement in relative condition across all reefs, divided by the cost of implementing the action at the reef pixel.

et al. 2011), but upon review we found the data to be anomalous in several locations. In a more data-rich coral reef region, more sophisticated approaches could be used to estimate model watershed-based pollution, e.g., SedNet (Prosser et al. 2001, McKergow et al. 2005*a*) and fishing pressure, e.g., Open OceanMap (Ecotrust 2005). As knowledge about these processes increases and data to represent them can be produced, we will be able to improve models of coral reef condition, validate them, and use them to inform conservation decisions.

Despite these challenges, we developed a prioritization approach that demonstrates the benefits of forest conservation to coral reef ecosystems. We found that the conservation of Fiji's remaining forests could improve the relative condition of coral reefs in the study region by 8–58% (Fig. 2), depending on model constants (i.e., α, β, δ) used. If all Fiji's remaining forests were conserved or well managed, we show how our approach could be used to help prioritize areas for implementation of coral reef MPAs. For example, reefs influenced by heavily cleared watersheds are a low priority for implementing protected areas regardless of fishing pressure as stopping fishing will deliver little benefits. Reefs influenced by fishing and watersheds containing a large proportion of forest will benefit from the implementation of MPAs, though their exact placement would depend on socioeconomic factors (Carwardine et al. 2008, Klein et al. 2010*b*). Given that the protection of all of Fiji's remaining forests is unrealistic due to competing land-uses (e.g., agriculture, forestry, mining) and pressures (e.g., population growth), we identify where the priorities are for forest conservation in terms of how much they contribute to maximizing reef health per dollar spent. The return on investment in forest conservation diminishes as more forest is protected. This is because the protection of an equivalent amount of land in different watersheds will prevent different proportions of pollution from reaching the reef from that watershed. Typically, the most cost-effective forest is in watersheds that are heavily forested and influence a large area of coral reefs, thus contributing most to increasing reef health.

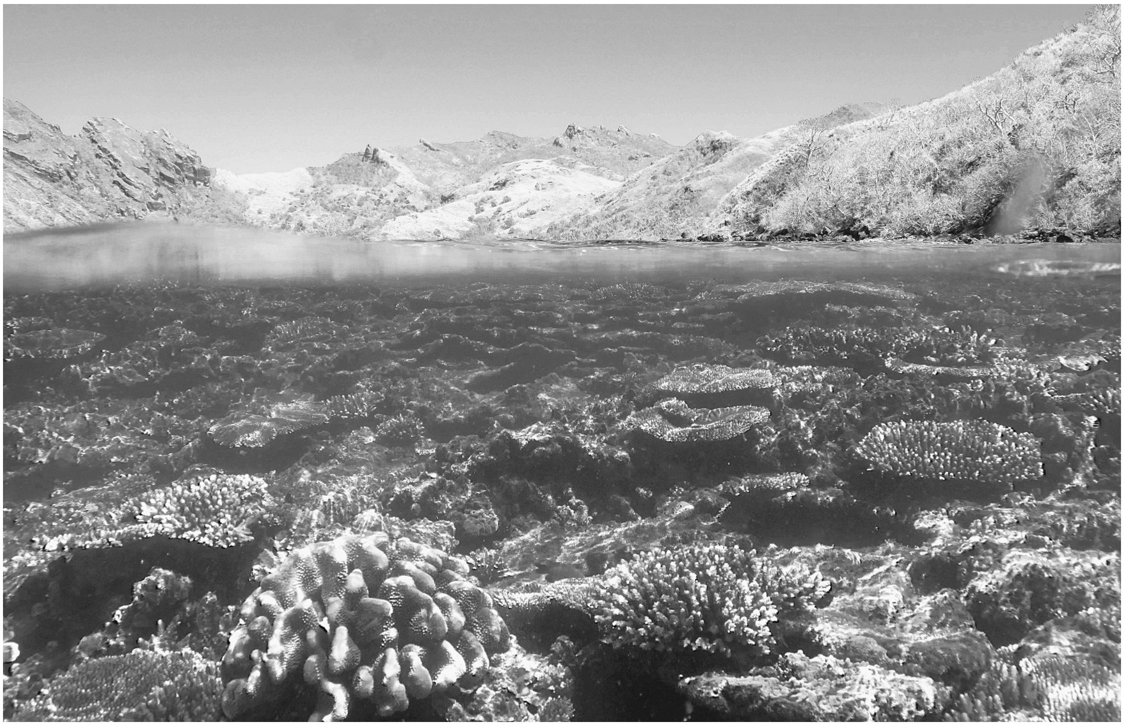

PLATE 1. Fringing reefs located next to heavily cleared islands of the Yasawa Group in Fiji. Reefs influenced by large, heavily cleared watersheds are a low priority for protection from fishing as they are likely to suffer degradation from chronic impacts of land-based runoff. Photo credit: S. D. Jupiter.

Our prioritization approach has made strides toward addressing the complexities of land–sea planning where marine conservation is the primary aim. In addition, this project highlights areas of land–sea planning that require further development and improved data. First, our model assumes that the protection of any unit of forest within a watershed has an equivalent contribution to coral reef condition; thus, we could not provide specific guidance on which areas within a watershed deliver the greatest return on investment. In reality, the amount of runoff, as well as suspended sediment and nutrient species, reaching waterways will vary spatially within a catchment based on specific land uses and management practices, soil erodibility, slope, proximity of clearing to waterways, and presence of riparian zones (McKergow et al. 2005*a*, *b*). Second, we did not consider how other land-based management actions could contribute to coral reef condition, such as restoration of areas where forest has been cleared and improved agricultural practices. Third, we did not consider the terrestrial benefits of protecting forest, including biodiversity and carbon sequestration (Venter et al. 2009), or the benefits to migrating aquatic species of protecting forest adjacent to coral reefs (Jenkins et al. 2010), which are key factors in deciding on the location of protected areas. Fourth, our model did not include any temporal benefits of management (Tallis et al. 2011) such as recovery of herbivorous fish populations, which could bring fished reefs back to an improved initial state. Finally, due to data availability, our conservation costs were crude as they were relatively homogenous across space, especially on the land, and did not consider the range of social and biophysical factors that drive spatial variations of conservation costs (Adams et al. 2010, 2011). For example, in Fiji it is impractical to farm and illegal to log on land with a slope $>30°$. Improving these estimations would impact the location of priorities on land and sea. Regardless, we include costs in our prioritization approach to emphasize the importance of including economic information when making any conservation decision.

The purpose of this paper was to develop a novel approach to identifying priorities for conservation to support the establishment of a national protected area system in Fiji, with a focus on prioritizing terrestrial areas to benefit marine conservation (Jupiter et al. 2011). Although the results will not be used to determine the exact location of protected areas in Fiji, they are instructive for the Fiji Protected Area Committee to provide "rule of thumb" guidelines (e.g., avoid coral reefs influenced by heavily cleared watersheds) on where resource allocation will be most cost-effective in Fiji to achieve the national targets for 20% and 30% protection of terrestrial and marine systems, respectively (Jupiter et al. 2011). Such rules of thumb approaches have been instructive for planning large-scale marine protected area networks in California (Carr et al. 2010). With improved biophysical and socioeconomic data, the results of our approach could be applied more spatially explicitly and be inclusive of considerations of other standard protected area network design principles (e.g., complementarity, representativeness, risk spreading, and adequacy; Margules and Pressey 2000), both in Fiji and across linked forest–coral reef systems around the globe. Such an analysis would also include the contribution of current protected areas and be able to prioritize for multiple conservation actions. Just as other areas of spatial conservation prioritization research have influenced conservation decisions (Fernandes et al. 2005, Green et al. 2009), we are hopeful that integrated land–sea planning will continue to develop and make a positive contribution toward protecting global important habitats and biodiversity.

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