

LETTER

Unexpected Management Choices When Accounting for Uncertainty in Ecosystem Service Tradeoff Analyses

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Bioeconomic model; decision making; ecosystem-based management; ecosystem services; multisector planning; second best theory; spatial planning; tradeoff analysis; uncertainty.

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Received

9 May 2016

Accepted

2 September 2016

Editor

Richard Zabel

doi: 10.1111/conl.12303

Abstract

Resource management and conservation increasingly focus on ecosystem service provisioning and potential tradeoffs among services under different management actions. Application of bioeconomic approaches to tradeoffs assessment is touted as a way to find win-win outcomes or avoid unnecessary stakeholder conflict. Yet, nearly all assessments to date have ignored inherent uncertainties in the provision and valuation of services. We incorporate uncertainty into the ecosystem services analytical framework and show how such inclusion improves optimal decision making. In particular, we show: (1) “suboptimal” solutions can become optimal when uncertainties are accounted for; (2) uncertainty paradoxically makes stakeholders value conservation despite their lack of preference for it; and (3) substantial losses or missed gains in ecosystem service provisioning can be incurred when uncertainty is ignored. Our results highlight the urgency of accounting for uncertainties in ecosystem services in tradeoff assessments given the widespread use of this approach by government agencies and conservation organizations.

Introduction

Communities face increasingly complex and uncertain decisions about how to effectively and efficiently manage natural resources, in particular as new uses of lands and oceans move into places already allocated for other purposes. These changes increase the number and diversity of stakeholders who benefit from, and often whose welfare depends on, how resources are managed. As such, environmental management decisions frequently involve tradeoffs among different ways that people use and value ecosystems. To better articulate the nature of tradeoffs among services and identify solutions that may reduce conflict and promote win-win solutions, decision makers are increasingly turning to an ecosystem services (ESs) decision framework (Polasky *et al.* 2008; White *et al.* 2012;

Lester *et al.* 2013), which models the potential supply and value of services from ecosystems under different management schemes (Tallis *et al.* 2012). The economic theory underpinning this approach has been around for decades, but it has only recently begun to gain significant traction in the context of resource management and conservation.

Recent advances in modeling ES have extended the tradeoff framework to many classes of problems, including conservation planning (Klein *et al.* 2013), reserve network design (Costello *et al.* 2010; Halpern *et al.* 2011; Rassweiler *et al.* 2012), land use regulation (Polasky *et al.* 2008; Goldstein *et al.* 2012; Johnson *et al.* 2012), habitat conversion (Barbier *et al.* 2008; Zavaloni *et al.* 2014), and resource use permitting (Kim *et al.* 2012; White *et al.* 2012). One of the general

outcomes from these diverse studies is that modeling the full set of possible management actions commonly identifies solutions that provide more value to more stakeholder groups, thereby potentially reducing conflict, than would normally arise in less comprehensive policy discussions (Rassweiler *et al.* 2014). Although there is a great potential value in scientific forecasts of diverse options, one of the challenges is that such forecasts have inherent uncertainty. To date, the theory and application of bioeconomic assessments of these tradeoffs have largely ignored such uncertainty (Nicholson *et al.* 2009; Johnson *et al.* 2012; Grêt-Regamey *et al.* 2013).

Uncertainty arises from inevitable limits to scientific knowledge about the natural world and the effects of humans' actions on it, often making outcomes hard to predict in isolation, and even more so in combination. Uncertainty also stems from unavailability of high-quality datasets required to parameterize and inform predictive models. Moreover, how individuals and organizations respond to information adds an additional layer of uncertainty. Although previous studies have considered the impact of uncertainty on environmental management and the provision of ES and conservation of biodiversity (Doyen & Béné 2003; Lande *et al.* 2003; Grafton & Kompas 2005; Regan *et al.* 2005b; Halpern *et al.* 2006; McCarthy & Possingham 2007; Johnson *et al.* 2012), it is rare for analyses of ES tradeoffs to incorporate uncertainties, either in the natural or the human systems that determine service provision and valuation. Ignoring uncertainty creates a number of potential challenges, such as misinformed losses and gains from implementing a policy, miscalculation about actual risks, and misunderstanding about the implications of stakeholder preferences. Furthermore, second best theory (Lipsey & Lancaster 1956) predicts that the optimal strategy in the first-best world (with perfect information) would no longer be the optimal in a second-best world (with uncertainties or imperfect information), while the suboptimal strategy in the first-best world would perform better than the optimal strategy in a second-best world. Here, we address several unresolved questions about how uncertainty affects decision making around ES provision and value, namely: (1) what are the potential losses or gains in total service provision when uncertainty is ignored?, (2) does uncertainty and associated risk tolerance of stakeholders affect the choice of optimal management solutions?, and (3) will stakeholder preferences for different services change given uncertainty?

Methods

We use two case studies to illustrate the incorporation of uncertainty into the ES tradeoff framework and the

effect of uncertainty on optimal decision making. Our case studies examine (1) tradeoffs between converting mangroves to shrimp aquaculture versus their preservation as nursery habitat for fisheries (see Supplementary Information) and (2) tradeoffs between conservation (fish biomass) and yield in a stochastic fish population in a region deciding whether to create a marine protected area (MPA). In both cases, we explore a range of possible management actions, focusing on how uncertainty in underlying models affects optimal actions.

Tradeoff between mangroves' nursery function and aquaculture production

We model the nursery function (N) of a mangrove patch as a function of the distance from the watercourse (x , in unit of 100 m) using an exponential decay model,

$$N(x) = e^{-k_N x}. \quad (1)$$

k_N dictates the rate of decay in the nursery function with distance. We assume that the nursery function can have two states, $k_N = \{0.1, 0.15\}$. On average, mangroves' nursery functioning is highest and least variable near the watercourse.

Clean water is a limiting factor for aquaculture productivity. As an aquaculture pond's distance from the watercourse increases, so does the chance of reduced productivity and level of variability in productivity (Binh *et al.* 1997). We assume the productivity loss function for aquaculture to have a similar form as that of the nursery function ($A(x) = e^{-k_A x}$), with aquaculture production having two possible states, $k_A = \{0, 0.3\}$. $k_A = 0$ implies that all patches have the same aquaculture suitability of 1, while $k_A > 0$ implies a declining aquaculture productivity with distance from the watercourse.

We assume that a favorable state ($k_N = 0.1$ and $k_A = 0$) occurs with probability $p = 0.5$ and an unfavorable state ($k_N = 0.15$ and $k_A = 0.3$) occurs with probability $p = 0.5$. On average, an approximately 10% reduction in aquaculture productivity is expected at $x = 2$ and an approximately 40% reduction can occur at $x = 10$.

The area adjacent to the watercourse is suitable for both aquaculture and nurseries. However, the conversion of mangrove patches near the watercourse reduces the nursery functioning of mangroves. Furthermore, aquaculture structures can interfere with the spatial connectivity of the nursery function in a nonlinear manner, i.e., establishing aquaculture structures adjacent to the watercourse reduces the accessibility of the interior mangroves, thus reducing the contribution of interior mangroves to the total nursery functioning of the system (see Supplementary Information).

We consider four heuristic scenarios/strategies for establishing aquaculture ponds: (1) single-sector decisions where aquaculture stakeholders optimize production by deploying ponds close to the watercourse (▲); (2) single-sector decisions where conservation/wild fisheries stakeholders optimize nursery function by directing aquaculture establishment away from the watercourse (■); (3) random deployments of aquaculture ponds (●); and (4) a block strategy where mangroves are grouped together from the smallest connected mangroves to the largest connected mangroves, and then groups of mangroves are converted into aquaculture ponds, from smallest to largest (◆).

Tradeoff between fish yield and fish biomass in a stochastic fish population

We consider a Ricker (1954) difference equation model to describe the dynamics of a hypothetical fish population. The total population biomass at time $t+1$ (i.e., B_{t+1}) is a function of the population biomass at the previous time step, i.e., $B_{t+1} = f(B_t)$ where

$$f(B_t) = B_t e^{r\left(1 - \frac{B_t}{K}\right)}. \quad (2)$$

The parameters r and K represent the population's growth rate and carrying capacity, respectively.

Given any positive finite values of K and r , Equation (2) is always positive. Additionally, f is positive for any positive finite stochastic values of K and r . Therefore, Equation (2) is an ideal model for investigating the effect of harvesting on the population given various magnitudes of population stochasticity. The nonzero equilibrium point of Equation (2) is globally/asymptotically stable for $r < 2$. We use $r = 0.5$ in our model.

We assume a constant-yield fishery where a fixed target yield (Y) is set. Given harvesting, the population model (Equation (2)) becomes

$$f(B_t) = B_t e^{r\left(1 - \frac{B_t}{K}\right)} - Y. \quad (3)$$

The maximum sustainable yield (MSY) occurs when $f'(B^*) = 1$, where $B^* = B_{MSY}$ is the steady-state population biomass at MSY. Hence,

$$\left(1 - \frac{r B_{MSY}}{K}\right) e^{r\left(1 - \frac{B_{MSY}}{K}\right)} = 1. \quad (4)$$

There is no analytic solution to Equation (4). Given $r = 0.5$ and $K = 1,000$, B_{MSY} and Y_{MSY} were derived numerically: $B_{MSY} = 467.7971$ and $Y_{MSY} = 142.6161$.

Since the point at MSY is unstable (Brauer & Sánchez 1975; Beddington & May 1977; Roughgarden & Smith 1996; Murray 2007), given population variability, it is necessary to engineer the system to achieve an optimal sustainable yield. We limit the engineering to MPA es-

ablishment. We assume that an MPA reduces the fishing ground and thus the yield. We simply assume that yield at MSY is reduced in proportion to the size of the MPA, i.e., $Y = (1 - \text{MPA.size}) * Y_{MSY}$, where the MPA size is from 0 (no MPA) to 1 (no fishing allowed).

We introduced a uniformly distributed noise $\nu(a,b)$ with zero mean and a symmetric limit of a and b to the carrying capacity of the population, i.e., $f(B_t) = B_t e^{r\left(1 - \frac{B_t}{K \pm \nu(a,b)}\right)} - Y$. We ran variability from 0 to $\pm 0.99K$ in steps of $0.01K$. For each instance of variability, we derive the size of the MPA needed to achieve an optimal sustainable yield. We ran the population for 1,000,000 time steps at each instance of variability.

ES tradeoff, utility, and indifference curve

Production theory, which deals with the production of goods and services given resource input, has been applied successfully to ES tradeoff assessments (e.g., White *et al.* 2012; Lester *et al.* 2013). ES can exhibit tradeoffs or synergistic relationships whose nature can be identified easily by plotting the production of services under different policy scenarios on a Cartesian coordinate system whose axes represent the assessed services. An indifference curve is a convenient graphical representation of stakeholders' preferences within the service production space. A curve contains the possible combinations of ES values (ES bundles) where stakeholders are indifferent about any of the combinations. Multiple, nonintersecting indifference curves represent the sets of indifferent ES bundles; an arrow is usually used to indicate the direction of the most preferred sets of ES bundles. The selected ES bundle will depend on both the production of ES and the stakeholder's preference.

The above framework assesses expected services produced by diverse policy options, and then uses these outcomes to make several broad management recommendations. First, outer-bound solutions are the best-case options (points a–f in Figure 1A); the choice between these options depends on stakeholders' relative preferences for different services, represented by indifference curves (Figure 1A). Second, interior points are suboptimal (e.g., option "g") and thus poor decisions, as one can improve the outcome of at least one service at no cost to other services. Policy option "c" has the highest utility in our example; it is positioned at the third indifference curve and the rest are positioned below it (Figure 1A). Third, the shape of the curve (concave in Figure 1A) is used to define the nature of the tradeoff between services (in this case, weak to moderate). Yet, all of these management guidelines ignore the potential influence of uncertainty in modeling these management outcomes (Figure 1B).

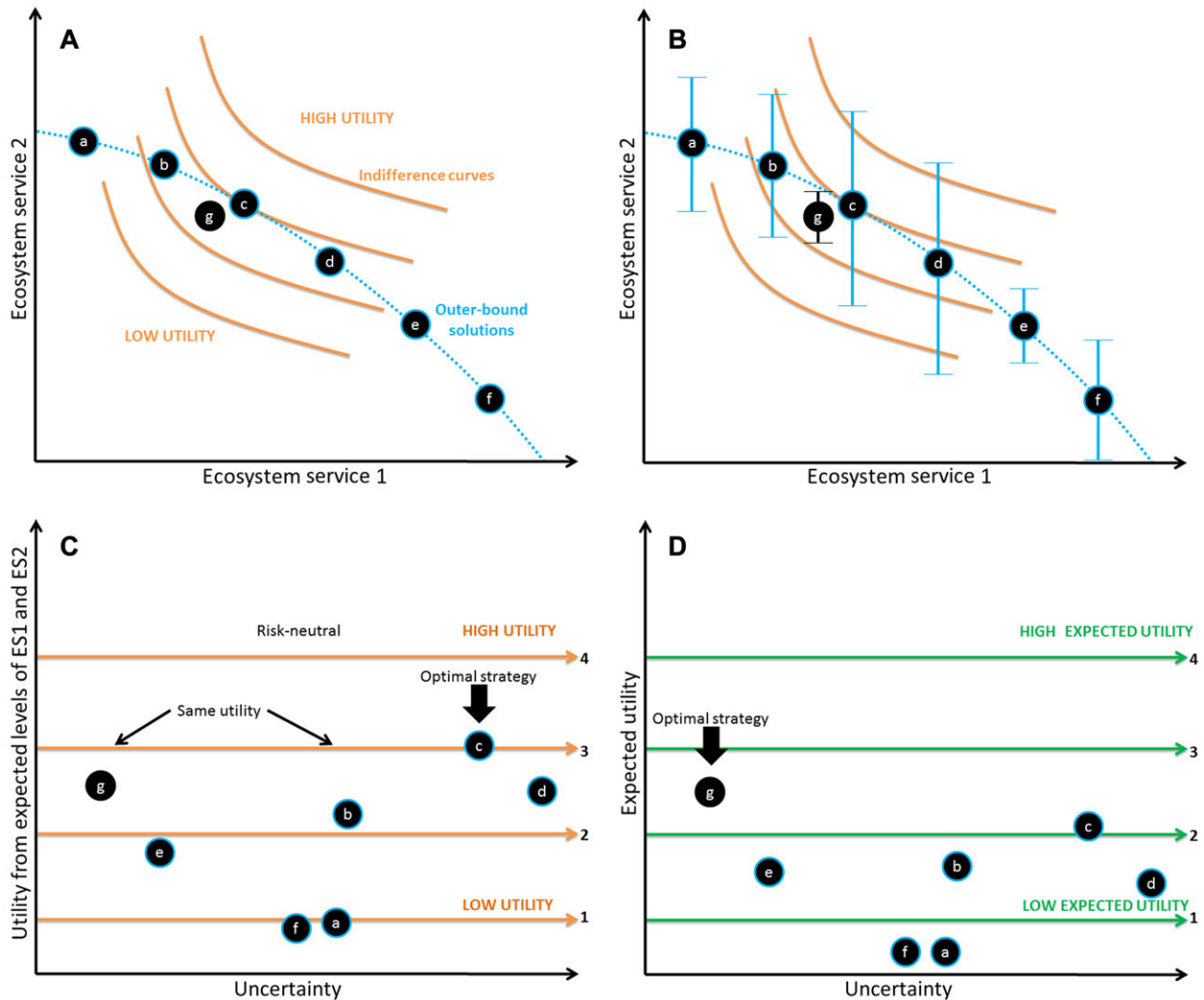


Figure 1 Ecosystem services (ES) tradeoff analysis with uncertainty. (A) Conventional representation of policy options in ES tradeoff analysis. (B) Policy outcomes with uncertainty. Uncertainty is only in ES 2. (C–D) Proposed graphical representation (utility v. uncertainty) for analyzing policy options with uncertainty under the ES tradeoff analysis framework. Points within a single line have the same utility. (C) A stakeholder that does not account or care for risk would choose policy option “c.” (D) Example where accounting for risk makes a risk-averse stakeholder chooses policy option “g” instead of “c.”

Some insights about the role of uncertainty in ES tradeoff evaluation can be gained from plotting the policy options into the utility versus uncertainty axes (Figure 1B–D). For example, a stakeholder who does not account for or care about uncertainty might choose policy “c” as expected (Figure 1C), i.e., the expected utility is constant at any level of uncertainty. A risk-averse stakeholder would instead try to minimize the chance of an undesirable outcome, or avoid any risk at all, and would account for uncertainty (e.g., Mangel 2000) leading to the choice of policy “g” (Figure 1D). In other words, a risk-averse stakeholder would rationally choose policy option “g,” an interior solution, over policy “c.”

We formalize the above idea by using the concept of utility and expected utility (Von Neumann & Morgenstern 1944; Varian 2009). Utility is a numerical description of a stakeholder’s preferences. For the aquaculture-nursery function tradeoff, we use a common utility function (U) of the form

$$U = \left(\sum_i A_i \right)^\alpha + \left(\sum_i N_i \right)^\alpha, \quad (5)$$

where $N_i = N(x_i)$ and $A_i = A(x_i)$ for patch i distance x_i . The summation is evaluated throughout the 1,125 mangrove grid cells: areas that can be conserved or converted into aquaculture ponds. The parameter α describes

Table 1 Expected utility of two bundles of services produced by two different policies for different levels of risk-aversion (α) (see Figure 2 for the bundles' positions)

α	0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.8	0.9	1.0
ES bundle 1	3.57	6.56	11.94	21.77	39.78	72.85	133.65	245.60	452.04	833.18
ES bundle 2	3.63	6.70	11.99	21.82	39.71	72.31	131.74	240.16	438.02	799.29

Note: $\alpha = 1$ is a risk-neutral stakeholder. The lower the α , the more risk-averse the stakeholder is. Shaded boxes indicate optimal solution for different α .

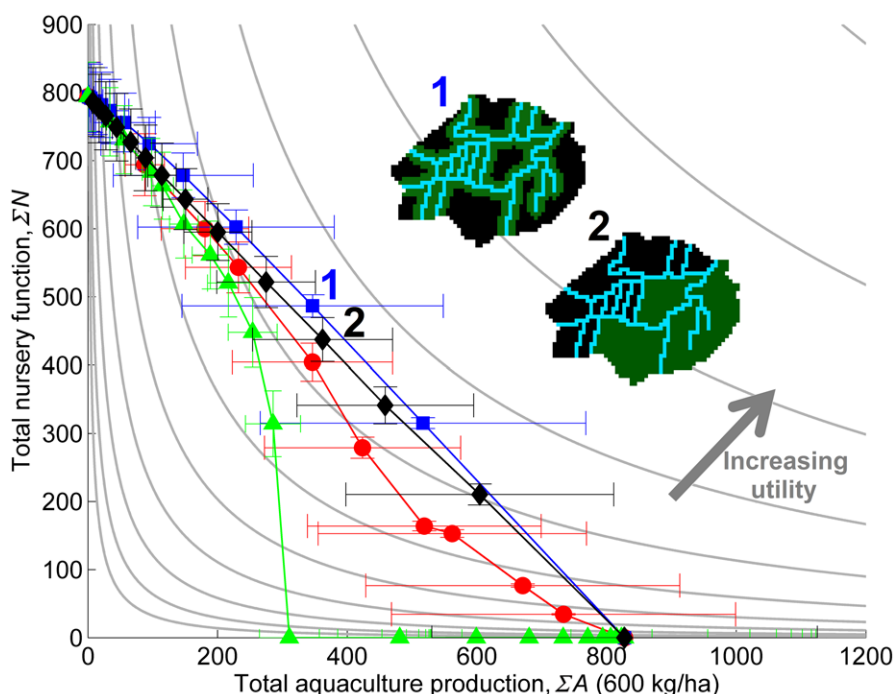


Figure 2 Tradeoff curves between the aquaculture production and nursery function of a mangrove forest in Viet Nam for variable heuristic decisions: (■) preferentially establishing ponds away from the mangrove edge, (●) random location of ponds, (▲) preferentially establishing ponds near the mangrove edge, and (◆) deploying ponds by blocks. Inset: Policy design for options “1” and “2” with black as ponds, green as mangroves, and blue as watercourse. $\alpha = 0.1$ is used for the indifference curves.

the risk preference of a stakeholder. $\alpha = 1$ is a risk-neutral stakeholder because U increases linearly with $\sum A$ and $\sum N$. A $0 < \alpha < 1$ describes a risk-averse stakeholder, i.e., the marginal gain in U decreases at increasing values of $\sum A$ and $\sum N$ (U is concave).

For any given configuration of aquaculture development, two states can occur (favorable state $(\sum_i A_i^+, \sum_i N_i^+)$ and unfavorable state $(\sum_i A_i^-, \sum_i N_i^-)$). These give rise to favorable utility U_+ with probability p_+ and an unfavorable utility U_- with probability $(1 - p_+)$. The expected utility of the ES bundle is

$$EU = p_+U_+ + (1 - p_+)U_- \tag{6}$$

Note that for a risk-averse stakeholder, EU is less than the utility of the mean production values, i.e., $EU < U'$, where $U' = \left(\frac{\sum_i A_i^+ + \sum_i A_i^-}{2}\right)^\alpha + \left(\frac{\sum_i N_i^+ + \sum_i N_i^-}{2}\right)^\alpha$; the opposite holds for a risk-loving stakeholder.

Results

Across each potential management strategy for mangrove conversion to shrimp ponds, incorporation of uncertainty in the underlying biophysical model determining service production leads to differences in predicted service provision of up to 60% (Figure 2). The worst-case scenario (▲) results when stakeholders neglect the interactions of the two ESs, acting in favor of a single sector (in this case, aquaculture). Optimizing aquaculture production requires ponds to be built along habitat edges. When all edge patches are converted into ponds, the nursery functioning of all interior mangroves vanishes. Uncertainty accumulates as total aquaculture production increases; uncertainty in aquaculture is highest when the total aquaculture production is highest. However, the uncertainty in the nursery function is highest instead when aquaculture ponds are preferentially placed along habitat

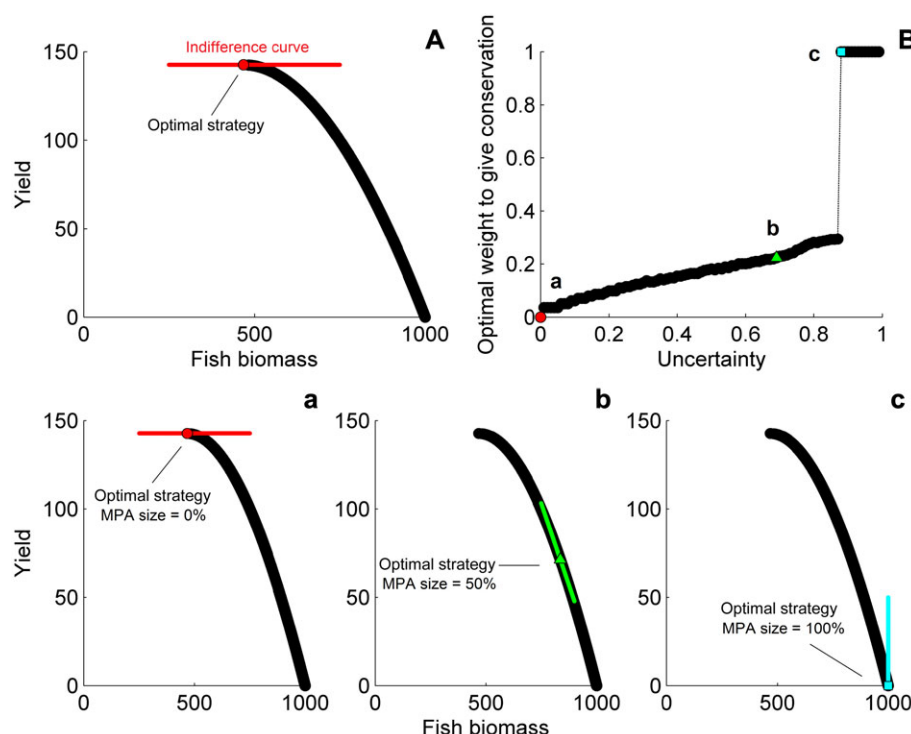


Figure 3 Tradeoff between conservation (fish biomass) and yield. (A) Fish biomass versus yield for variable size of marine protected area (MPA). The highest yield corresponds to Y_{MSY} (with no MPA). MPA has the effect of reducing yield. Closing the entire fishery resulted to zero yield and biomass at carrying capacity ($K = 1,000$). Stakeholders are assumed to have zero preference for conservation (horizontal indifference curve). (B) Effect of uncertainty on stakeholder's preference. With increasing uncertainty, stakeholders' preference appeared to shift toward conservation even though they have zero preference for it, i.e., (a) at zero uncertainty, the optimal strategy is to fish at MSY; (b) with uncertainty, the optimal strategy is to establish MPAs or lower fishing effort; and (c) at sufficiently high uncertainty, the optimal strategy is to not fish the stock at all. Uncertainty pertains to the level of variability in K , which ranges from $a, b = 0$ (uncertainty = 0) to $\pm 0.99K$ (uncertainty = 0.99) (see Methods).

edges, because ponds block interior mangroves and this limits water flow into the interior mangroves (Figure 2).

Based on tradeoff assessments without uncertainty, the optimal policy solution lies along the outer-bound (frontier) solutions (■, marked by "1," Figure 2). However, accounting for uncertainty makes a risk-averse stakeholder chooses an interior solution (marked by "2," see Table 1), because it has higher expected utility than any of the outer-bound solutions. The two solutions vary considerably in outcome. One involves utilizing the entire mangrove forest by converting mangrove patches that are away from the watercourse into aquaculture ponds, while the other involves converting mangroves to ponds by blocks, hence retaining other blocks of mangroves in their pristine state (Figure 2, inset). The optimal solution changes from an interior to an outer-bound as the stakeholder becomes less risk-averse, i.e., $\alpha \geq 0.5$ (Table 1).

In the second case, the stakeholder is assumed to care only about fish yield with no interest in conservation (i.e., horizontal indifference curve, weight toward conserva-

tion = 0, Figure 3A). Optimizing yield implies that the stakeholder would fish the stock at MSY and would not consider establishing an MPA. However, the fish population is unstable at MSY, and population variability may push the population to rapid declines. When considering uncertainty in model evaluation, the stakeholder should prefer to establish an MPA, or to lower fishing effort, as fishing at MSY with high variability can ultimately lead to population collapse. Optimizing catch without accounting for uncertainties would result to losses or missed gains for both catch and biomass, and these losses depend on the level of population stochasticity (Figure 4). Higher uncertainty implies a less optimal sustainable catch limit and higher fish biomass.

Indeed, the outcome of uncertainty is similar to an outcome where the stakeholder has more preference for conservation (question 3, see Figure 3). Incorporating uncertainty makes stakeholder behave "as if" they had strong biological or conservation preferences, even when they have no preference for conservation at all.

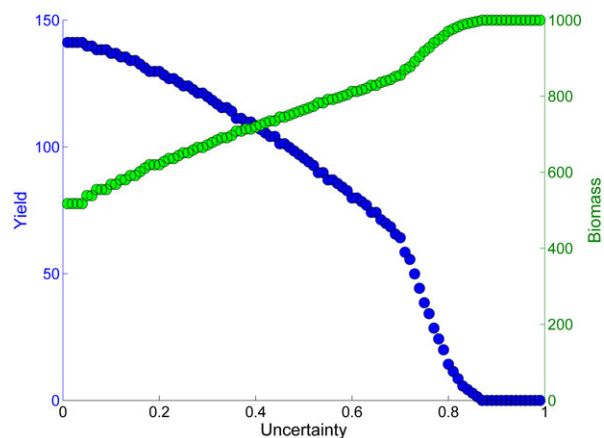


Figure 4 Losses and missed gains given various levels of uncertainties. Green indicates the mean lowest level of fish biomass that can be retained in the fishery, while blue shows the mean highest yield that can be obtained when uncertainties are accounted for.

Sufficiently large population variability results in the case where the stakeholder will support a strongly conservationist strategy and will fish the stock very little, as more intensive harvesting will only lead to population collapse (Figure 3B). Note that in general, MSY will not be the optimal harvest level when the fishery is concerned with profit (Gordon 1954; Scott 1955).

Discussion

Most ES tradeoff assessments to date have focused on outer-bound solutions as optimal, yet there is inherent uncertainty in the ability to achieve those solutions (Regan *et al.* 2005a; Kareiva *et al.* 2011; Rassweiler *et al.* 2014). We showed that interior solutions can achieve preferred outcomes, especially when accounting for stakeholder risk tolerance. This finding is consistent with the second best theory (Lipsey & Lancaster 1956), where the optimal solution in a world with perfect information becomes suboptimal in a world with uncertainties.

Incorporating uncertainty in tradeoff analysis also allows the visualization of hidden conservation bias in the indifference curve, as illustrated by the case where purely production-oriented stakeholders, regardless of their risk tolerance, would need to consider conservation more as uncertainty increases. Our result reinforces the precautionary approach to fisheries management of considering both fisheries yield and conservation by setting target production lower than MSY (Roughgarden & Smith 1996) or establishing MPAs to manage both fishing effort (Hastings & Botsford 1999) and uncertainty in fisheries (Hsieh *et al.* 2006).

Ignoring or underestimating uncertainty risks foregoing potential value that could be derived from ES (if the mean is underestimating value) or collapsing systems or setting expectations that are too high (if the mean is overestimated). In general, incorporating uncertainties into ES tradeoff analyses has three main advantages. First, risks and uncertainties in tradeoffs of policy options are explicitly illustrated and quantified; therefore, the costs and benefits of ignoring (or reducing) uncertainties are demonstrated. Second, uncertainties have the paradoxical effect of making stakeholders value conservation even though their preference is solely to maximize gain from an extractive ES. This hidden conservation bias in indifference curves cannot be visualized by traditional treatment of uncertainties. Such an effect may also arise with ESs that are tightly coupled and may be common in nature. And finally, the ES decision framework appeared to be a convenient and practical tool for communicating tradeoffs and risks.

Although it is widely known that uncertainty affects decision making, management within the ES framework has yet to embrace this reality. So far, we have explored a few types of uncertainties; there are broad classes of uncertainties in decision making focused on natural resource management that warrant attention (Regan *et al.* 2002). Factors including irrational behavior of stakeholders, discount rates, and market imperfection could influence the conservation outcome and interact with other ecosystem components. While this work also focuses on uncertainties in the context of provision and valuation of ES, another gap that merits attention is the flows and ultimate use of ES. Alternative methods based on Bayes' theorem, which quantify uncertainties in ES supply and demand given some level of available data, have been proposed (Villa *et al.* 2014). To address the complexities in the types and nature of these uncertainties, which themselves merit further evaluation, the framework presented here is general and can be used to account for other uncertainties in human systems, biophysical systems, and their interactions.

Acknowledgments

We thank K. Siegel for the language edit and the reviewers for the comments and suggestions that improved our work. The publication fee was supported by the UCSB Open Access Fund.

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