Ecological Applications, 24(6), 2014, pp. 1357–1373
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Informed actions: where to cost effectively manage multiple threats to species to maximize return on investment

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Abstract. Conservation practitioners, faced with managing multiple threats to biodiversity and limited funding, must prioritize investment in different management actions. From an economic perspective, it is routine practice to invest where the highest rate of return is expected. This return-on-investment (ROI) thinking can also benefit species conservation, and researchers are developing sophisticated approaches to support decision-making for cost-effective conservation. However, applied use of these approaches is limited. Managers may be wary of "black-box" algorithms or complex methods that are difficult to explain to funding agencies. As an alternative, we demonstrate the use of a basic ROI analysis for determining where to invest in cost-effective management to address threats to species. This method can be applied using basic geographic information system and spreadsheet calculations. We illustrate the approach in a management action prioritization for a biodiverse region of eastern Australia. We use ROI to prioritize management actions for two threats to a suite of threatened species: habitat degradation by cattle grazing, and predation by invasive red foxes (*Vulpes vulpes*). We show how decisions based on cost-effective threat management depend upon how expected benefits to species are defined and how benefits and costs co-vary. By considering a combination of species richness, restricted habitats, species vulnerability, and costs of management actions, small investments can result in greater expected benefit compared with management decisions that consider only species richness. Furthermore, a landscape management strategy that implements multiple actions is more efficient than managing only for one threat, or more traditional approaches that don't consider ROI. Our approach provides transparent and logical decision support for prioritizing different actions intended to abate threats associated with multiple species; it is of use when managers need a justifiable and repeatable approach to investment.

Key words: Australia; biodiversity; cost effectiveness; decision support; multiple species; multiple threats; natural resource management; regional-scale management; return on investment; spatial prioritization; species richness; threatened species.

Introduction

Species are imperiled by numerous threats (Wilson 1992); abating these threats requires decisive management action. As a conservation practitioner, it is difficult to decide which management action to enact first and where to enact it (Marris 2007), especially given constraints of time (Pimm et al. 1995) and money (James et al. 1999). One choice is to act in the sites that may maximize the number of species protected against impending threats to provide the greatest benefit (Brooks et al. 2006, Evans et al. 2011*b*). However, this approach ignores the cost of management, and it might lead to managing expensive sites and missing out on cheaper sites that might still benefit many species (Balmford et al. 2000). By incorporating economic costs into conservation decision-making (Naidoo et al. 2006), more species or locations may benefit from management actions (Possingham et al. 2001, Joseph et al. 2009, Carwardine et al. 2012). Although conservation practitioners are already likely to tacitly consider costs in their decisions, it is difficult to mentally consider the trade-offs in such complex decisions. On the other hand, if a manager has the explicit goal of maximizing species benefits from management actions for abating threats at least cost, the dilemma of where to act first can be resolved by placing the problem in a conservation decision theory framework (Possingham 2001*a*). By using conservation return on investment (ROI) (Murdoch et al. 2007) to reveal which actions are most cost effective, a manager can make an informed decision to transparently prioritize the necessary actions expected to provide the greatest benefit to species (Bottrill et al. 2008).

Manuscript received 21 April 2013; revised 18 November 2013; accepted 10 December 2013; final version received 7 January 2014. Corresponding Editor: E. Nelson.
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The difficulty of deciding how and where to manage multiple threats has led to a number of alternative approaches being developed. Much of the traditional systematic conservation planning literature on threat management addresses only one action at a time for multiple species, such as targeting the threat of habitat loss with reserves (Margules and Pressey 2000). In contrast, economic approaches to spatial prioritization of conservation (Busch and Cullen 2009, Walsh et al. 2012) generally focus on single-species management with more than one action. Conservation effort allocation needs to be strategic to most efficiently mitigate the multiple threats acting in the landscape. Efficiency in this context is related to the spatial distribution of threats, how difficult the threats are to abate, and how many species might benefit from threat mitigation (Klein et al. 2010, Evans et al. 2011*b*). Yet doing everything, everywhere, isn't an option. Simultaneously considering the costs and benefits of multiple threat management actions leads to an understanding of where to expect the greatest return to species (Wilson et al. 2007, 2011*a*). Nevertheless, on-the-ground, fine-scale spatial prioritization of multiple actions is still not routine (van Teeffelen and Moilanen 2008).

Cost-effectiveness analysis (CEA) is a form of ROI that is commonly used to evaluate human health care options relative to costs (Weinstein and Stason 1977). CEA is also applicable to managers making decisions for investing in single or multiple conservation actions under resource constraints (Hughey et al. 2003, Murdoch et al. 2007). In both cases, benefits are measured in nonmonetary units. ROI provides guidance to managers and practitioners by clarifying where the highest rate of conservation return, i.e., the greatest benefit, is expected from an investment in one or more conservation actions. In the ROI framework, managers explicitly specify what outcome they want to achieve (i.e., an objective), define the benefits they expect from conservation action, and estimate the costs it would take to achieve them (Possingham et al. 2001, Mace et al. 2006).

Research surrounding the use of ROI to explore trade-offs in prioritizing conservation efforts is progressing rapidly. Conservation ROI is becoming increasingly complex, by integrating important system considerations. ROI informs efficient land acquisition, while achieving species complementarity (Ando et al. 1998, Balmford et al. 2000, Underwood et al. 2008). Incorporating additional considerations of land conversion and on-going habitat loss threats (Wilson et al. 2006, Withey et al. 2012), off-reserve lands (Polasky et al. 2005), and alternative land uses (Wilson et al. 2010) refines the analysis, as does considering additional actions such as policy incentives (Nelson et al. 2008, Lewis et al. 2011), habitat restoration (Goldstein et al. 2008, Wilson et al. 2011*b*), and abatement of threats such as invasive species (Wilson et al. 2007). Moreover, multi-objective ROI analyses reveal the contributions of desirable ecosystem services (Daily 1997) such as carbon sequestration and freshwater filtration (Kovacs et al. 2013, Kramer et al. 2013) in addition to the benefits of habitat conservation. These conservation ROI analyses provide informative outcomes in directing management by incorporating increasingly complex information, algorithms, and important dependencies.

However, although the conservation ROI research provides compelling evidence that the approach directs where best to efficiently invest in conservation action, there is less evidence that the ROI approach is being used extensively by conservation practitioners (Murdoch et al. 2007, Boyd et al. 2012). While ROI is also routinely used as a business performance indicator to provide strategic direction (e.g., Venkatraman and Ramanujam 1986), its surprising lack of use to rank and prioritize on-ground conservation action indicates a knowing–doing gap, whereby research findings are not widely accepted and applied by practitioners (Knight et al. 2008). A key limitation to the effectiveness of a ranking system such as ROI is the gap between its availability as a decision support tool, and its implementation by decision-makers (Cullen and White 2013). Developing more sophisticated ROI approaches is undeniably valuable, but the perceived complexity of the analytics could make the solutions less approachable to practitioners. As an alternative, our research seeks to make ROI accessible as a decision support tool to a manager who makes day-to-day decisions about implementing conservation action at a regional or local scale.

Here we demonstrate an ROI approach as decision support for a conservation practitioner/manager with the conservation objective of recovering threatened species through threat mitigation actions. Time and resource constraints also apply to managers making the effort to gain additional expertise in prioritization techniques, and those responsible for biodiversity management need quick, transparent, and easy-to-understand decision support tools that are useful at a fine scale. The calculation logic should be straightforward, so that managers can satisfy stakeholders (donors, funding agencies, and the general public) that important decisions are being made in a defensible manner. Using systematic planning software to prioritize spatial conservation actions at least cost or without a budget constraint (e.g., Marxan and Zonation; Ball et al. 2009, Moilanen et al. 2009) is a widely respected approach. On the other hand, practitioners are still sometimes wary of what are perceived to be complex "black-box" algorithms (e.g., simulated annealing in Marxan), although their use is grounded in quantitative decision theory that provides a solid basis for decision support applications (Possingham 2001*b*). Similarly, increasingly complex ROI approaches can be difficult to perform and interpret by practitioners.

Our research provides an alternative both to decision support solutions that are perceived to be complex, as well as the default position of not using decision support. The approach is a simplified, but rigorous and systematic, way to use ROI to prioritize conservation actions that considers multiple species and multiple threats. We promote a basic ROI based on costs and benefits because it is a straightforward concept to understand, relate to, and communicate to stakeholders and donors. The approach requires only working knowledge of commonly used technology (a geographic information system and spreadsheets), yet provides informative guidance for decision-making by allowing decision-makers to explore and easily visualize how uncertainty and variation in costs or benefits might drive different decisions. Due to the ease with which we can visualize what drives investment decisions, our approach makes the strength of ROI accessible to practitioners who are hesitant to implement complex approaches, yet need a transparent way to account for the way they invest limited funding.

Research aim

We describe an ROI decision-making framework to guide natural resource managers in achieving efficient allocation of conservation resources to address multiple threats to multiple threatened species. We also show the difference in decisions made with ROI, and without ROI. We use a case study of a bio-diverse natural resource management region in Australia that is typical of any administratively defined location, worldwide, where a manager must make decisions about where to focus action to abate threats to species. Our research informs a conservation practitioner on using a straightforward ROI analysis to evaluate the cost effectiveness of management action to benefit species at least cost. We model ROI using the relationship between the level of investment in a given management action, and the expected conservation outcome, i.e., threat abatement to species affected. We compare our results to those of alternative approaches to decision making. Specifically, we explore three questions to guide management action strategies across space:

1) How much better is an ROI approach when compared to more traditional conservation planning approaches of allocating resources or arbitrary action?
2) How do alternative ways of defining the expected benefits of conservation management change the spatial priorities and level of investment required to manage single or multiple threats in an ROI approach?
3) If managers can choose single or multiple management actions across the landscape depending on where actions are most cost effective, how can ROI inform whether one or more action(s) are allocated to an area for a given budget?

We illustrate how to implement our ROI approach using a case study of a regional natural resource management area where multiple species would benefit from actions to abate multiple threats to their persistence. Our goal is to show that an uncomplicated ROI analysis can provide useful decision support for conservation managers.

Methods

Case study

To illustrate our framework of using ROI to make investment decisions to abate multiple threats, we use an example from the Burnett-Mary Natural Resource Management (NRM) Region, which is located in southeast Queensland, Australia. Administrative boundaries of Australia's 56 NRM regions are ecologically defined bioregions or watershed catchments. Encompassing the catchments of the Burnett and Mary Rivers, the case study region covers 55 000 km^2. The region contains diverse ecosystems and has a history of land use decisions that threaten its biodiversity; for example, the now presumed extinct Paradise Parrot (*Psephotus pulcherrimus*) was last sighted in the study area in the 1920s (Olsen 2007). The regional managers are under pressure to successfully cope with a range of threats to priority threatened species (DERM 2010*a*), yet there are limited financial and technical resources for strengthening management practice (Robins and Dovers 2007), such as prioritizing management action. We divided the region into a grid of 25-ha potential management sites for a fine-grained analysis of species' habitat and management requirements. We analyzed 129 894 sites, those with remaining or regrowth native vegetation (Queensland Herbarium 2010*a*, *b*), that are presumed to provide habitat for species in the region.

Framework for conservation return-on-investment analysis

We conducted ROI analysis to measure the increase in conservation outcome per unit cost of management actions taken to reduce threats in the study region. The ROI produces a measure of conservation cost efficiency (Murdoch et al. 2007). In our case, we define management outcomes, and therefore conservation efficiency, to be abating a given threat to the species that are affected by that threat. Critical steps to applying ROI to a conservation decision framework (Possingham et al. 2001) are: (1) identifying the problem, (2) defining realistic expected benefits, (3) integrating realistic costs, and (4) combining information on expected benefits and costs to solve the management funding allocation problem.

Step 1. Problem definition

Our problem lies in determining how and where to cost effectively mitigate threats to species in a given landscape. We aim to find the most cost effective management strategy to secure selected species (Appendix A) across a typical natural resource management region by addressing two expert-identified threats to those species (DERM 2010*a*): invasive species (fox) predation, and habitat degradation (from cattle grazing). The red fox is recognized as one of the world's worst invasive alien species (Lowe et al. 2001), and there is overwhelming evidence that invasive predators have negative impacts on a broad range of native vertebrates in many parts of the world, including Australia (Mack et al. 2000, Burbidge and Manly 2002, Blackburn et al. 2004, Saunders et al. 2010). Red foxes are listed as a Key Threatening Process in Australia by the Environment Protection and Biodiversity Conservation (EPBC) Act 1999, with management actions such as bait poisoning addressed in a national Threat Abatement Plan (Anonymous 1999). Unsustainable grazing is also known to negatively impact the species composition, function, and structure of ecosystems (Fleischner 1994). To mitigate this threat, introduced herbivores are removed or reduced, and can result in increased species richness and abundance of small mammals (Legge et al. 2011), although recovery is likely to be dependent upon time, ecosystem type, and extent of habitat alteration (Read and Cunningham 2010).

We set the goal for solving our problem as follows. For a given level of investment, maximize the net expected benefit of the actions we take to mitigate threats that imperil a set of target species. As a frame of reference, we also want to examine the full potential benefit of action if management funding were not limited.

Step 2. Define realistic estimates of expected benefits

Our 20 target species (Appendix A: Table A1) were previously selected by experts for management action on the criteria of probability and consequences of extinction, and potential for affecting successful recovery (DERM 2010*a*). Species include seven vascular plants, one fish, four reptiles, one amphibian, three mammals, and four birds that are threatened at the national, state, and/or regional level. We obtained species' spatial distribution data across the study area from one of three sources, based on the best data available for a species (Appendix A: Table A1): (1) species distribution models (SDMs) modelled (N. A. Auerbach, M. C. Evans, and H. P. Possingham, *unpublished data*), using Maxent software (Phillips et al. 2006, Dudík et al. 2010); (2) Species of National Environmental Significance (SNES) range maps (DEWHA 2008), or (3) Atlas of Living Australia (ALA) point locations (ALA 2012).

To model the distribution of fox and grazing threats across the landscape, we used indirect threat modeling (see the following paragraphs), i.e., the spatial distribution of a particular threat is represented by the collective distributions of species affected by that threat (Evans et al. 2011*a*). For the two threats we considered, 7 of the selected species are only vulnerable to predation by the red fox, 11 are only vulnerable to habitat degradation caused by cattle grazing, and 2 species are susceptible to both threats (DERM 2010*a*) (Appendix A: Table A1).

We need to define a measure for the expected benefits to species that captures the management goal (Guikema and Milke 1999). Here, we explore defining biodiversity benefits with the primary goal of threat management. For example, are managers most interested in prioritizing threat management in areas with many species (high species richness)? Or do managers also want to target species with rare or vulnerable habitats? Alternative choices will lead to different prioritizations of management actions (Nicholson and Possingham 2006). The traditional approach to systematic conservation planning for reserve selection often uses data on the distribution of species, i.e., species richness, to inform decisions. However, to link these distributions explicitly with actions for threat management, a manager might also want to account for restricted habitats and/or vulnerability to multiple threats, as areas with species more likely to respond to targeted threat management might be considered higher priority for management.

We assume

1) that the region is made up of m sites labelled $i = 1, \ldots, m$
2) that the region contains n threatened species labelled $j = 1, \ldots, n$, and
3) that there are p threats labelled $k = 1, \ldots, p$.

In the first instance, let the expected benefit of acting to abate threat k to species j at site i (b_{ijk}) be a value of one if a species is vulnerable to a threat k, and a value of zero if not. Our indirect threat mapping approach to species' threat vulnerability (Wilson et al. 2005) means that if a species is known to be susceptible to a threat, the species is assumed to be vulnerable across all of its mapped habitat. If all values are the same (in this example, a value of one), this vulnerability is portrayed as being spatially homogeneous. However, other values between zero and one could be used to represent spatial heterogeneity of a threat, variability in the vulnerability of a species to a threat, or uncertainty in the benefit of managing for a threat. For simplicity, to demonstrate our approach, we make the assumption that a threat to a species is abated if managed (similar to Murdoch et al. (2007)), and the species will receive full benefit. This allows additive calculation of benefits to multiple species. Here, we compare three different ways to define expected benefits that depend on how much detail about the species and their threat is included, using data of increasing levels of complexity. The first defines benefit at a site to be the number of species (species richness) that will be secured by a particular threat abatement action (Metric 1), with the benefit B_{ik} of abating threat k at site i for n species formally described as follows:

$$B_{ik} = \sum_{j=1}^{n} b_{ijk}\, a_{ij} \tag{1}$$

where a_{ij} represents the presence of a species j in site i, $a_{ij} \in \{0, 1\}$ with a value of one indicating species presence; and $b_{ijk} \in \{0, 1\}$, with a value of one indicating that species j is vulnerable to the threat and would benefit from management.

The second metric considers habitat rarity in addition to species richness (Metric 2), with the benefit B_{ik} of abating threat k at site i for n species calculated as

$$B_{ik} = \sum_{j=1}^{n} b_{ijk} \frac{a_{ij}}{A_j} \tag{2}$$

where the total number of occupied sites of each species j is given by

$$A_j = \sum_{i=1}^{m} a_{ij}. \tag{3}$$

This means that the benefit of acting at a site for a species is proportional to the fraction of the range of a species that the site encompasses. If it is the entire range of a species, the benefit is large, whereas if it is a small part of the range of a species, the benefit is relatively small.

The third metric adjusts Metric 2 for species richness and habitat rarity by accounting for threat vulnerability (Metric 3), as we may want to manage in less risky areas (Game et al. 2008). Assume that there are multiple threats K at site i and a species receives only partial benefit from a management action that addresses only one threat. Assume that if we abate a threat k at a site, the management action secures the species in that site only proportionally to the number of threats to that species, and the benefit is proportional to the fraction of the range of the species. Here, the benefit B_{ik} of abating threat k at site i for n species is

$$B_{ik} = \sum_{j=1}^{n} \frac{b_{ijk}}{K_j} \times \frac{a_{ij}}{A_j} \tag{4}$$

where K_j is the total number of threats to species j, and b_{ijk}/K_j weights the benefit by the proportion of threats acting on the species across its range in the study region. For acting on one threat for a species vulnerable to only one threat, the value of b_{ijk}/K_j is one, whereas acting on one threat for a species vulnerable to two threats will result in a value of one-half = 0.5. For our two-action case study, $K_j \in \{1, 2\}$.

We applied each metric to estimate the cumulative benefit B_{ik} of abating threat k for all species j that occur at site i to examine benefit metrics of increasing complexity. In each metric we assume that each management action is equally effective at abating a specific threat.

Step 3. Integrate realistic cost estimates

Threat abatement comes at a cost, and management and opportunity costs are two specific types of conservation expenditures (Naidoo et al. 2006). Heterogeneity in costs across a landscape has been found to substantially affect results of systematic conservation planning (Ando et al. 1998, Polasky et al. 2001, Naidoo et al. 2006), and there is an increasingly large body of literature that describes how to calculate costs of management over space (e.g., Balmford et al. 2003, Naidoo et al. 2006, Polasky et al. 2008, Moilanen et al. 2011*b*, Carwardine et al. 2012, McCarthy et al. 2012). Although costs of conservation actions are not certain, it is usually better to accept estimation errors of even 50% rather than not consider cost at all (Murdoch et al. 2007). We calculated the cost of two possible management actions across the landscape over a 20-year period: removing grazing (an opportunity cost of foregone agricultural profits, e.g., Naidoo and Adamowicz [2006]), and controlling an invasive species (an on-ground management cost, e.g., Wilson et al. [2007]).

We estimated the opportunity cost to cease grazing through a stewardship agreement, based upon foregone agricultural profits (Marinoni et al. 2012), similar to other conservation planning studies (e.g., Naidoo and Iwamura 2007, Carwardine et al. 2008), although landowner bargaining power creates some uncertainty in this cost assumption (Lennox and Armsworth 2013, Lennox et al. 2013). We assumed landowners would accept a financial stewardship payment to cease grazing on land populated with threatened species, to mitigate income volatility (Mouysset et al. 2013). The dominant land use ($\sim$65%) in the study region is grazing of natural vegetation (DERM 1999). To calculate a stewardship payment, we used a snapshot of agricultural profitability for the year 2005/2006 in Australia (Marinoni et al. 2012), which was based upon costs, revenues, yields, and commodity (including livestock production) areas that were derived from land use, detailed census information, water resource use, and production costs. Where the overall cost of production outweighed the returns and resulted in negative profits, we set the yearly profit value to a minimum of \$10/25 ha to simulate providing landholders with a token payment. (All dollar values are in Australian dollars, AUD.) The stewardship payment was calculated to be equal to the net present value (NPV) of the opportunity cost of foregone agriculture profits over 20 years at a discount rate of 2% (Appendix B).

We estimated the cost of fox control using bait poisoning with sodium fluoroacetate (1080), a substance that naturally occurs in some native Australian vegetation and is used to control nonnative predators (Saunders and McLeod 2007). We calculated the cost of reducing fox impact on native fauna with four 14-day baiting campaigns per year modeled as a roadside/grid baiting strategy after Carter et al. (2011). Costs may be higher if longer campaigns are needed for better outcomes. We accounted for labor, transportation, and bait price in the calculation of expenses for this strategy, and calculated the costs to be equal to the NPV of fox-baiting over a period of 20 years at a discount rate of 2% (Appendix C).

The cost of abating threat k at site i is C_{ik}.

Step 4. Solve the problem

Solving the problem of how to maximize the expected benefits of managing target species under the constraint of a given budget is a type of "knapsack problem," which is a mathematical formulation in combinatorial optimization that maximizes an objective function subject to a single resource constraint (Pisinger and Toth 1998). In other words, here we seek to optimize the allocation of resources to threat management under financial limitations, as in other research addressing the prioritization of management actions given a restricted conservation budget (e.g., Joseph et al. 2009).

Comparing expected benefits in an ROI approach.—To examine how different ways of calculating benefits affect decision-making, we first calculated the cost effectiveness ce_{ik} of abating threat k, for each site, i, represented by the benefit of acting to mitigate the threat k for all species, divided by the cost of doing this

$$ce_{ik} = B_{ik}/C_{ik}. \tag{5}$$

We ranked all sites in order of ce_{ik} for each action and each of the three benefit metrics. ROI analysis requires an evaluation of how much benefit we would expect for a given investment, which we defined as the cumulative benefit ($\sum_{i=1}^{m} B_{ik}$) for each management strategy (and each benefit metric) after sites were ranked by ce_{ik}. In addition to managing only for foxes and only for grazing, we included a third management strategy of the combined management actions. We combined the information on each single-threat abatement action (here two actions, grazing and fox control) into a single list that was ranked in order of ce_{ik}. If a site had high cost effectiveness for managing one threat, but lower cost effectiveness for managing the second threat, it might be selected only for managing the first threat under a small budget, but under a larger budget both threats could be managed.

We then found the benefit of selecting sets of sites for given budgets. Mathematically, this is a "knapsack" problem

$$\max \sum_{k=1}^{p} \sum_{i=1}^{m} x_{ik} B_{ik}$$

subject to

$$\sum_{k=1}^{p} \sum_{i=1}^{m} x_{ik} C_{ik} \leq \text{Budget} \tag{6}$$

where x_{ik} represents the act of selecting and managing site i for threat k and is therefore a value of one if selected or zero if not. This algorithm finds the maximum cumulative benefit of selecting the action by site combinations (out of a total of $p \times m$ choices ranked by cost effectiveness), whose cumulative costs are less than or equal to a given budget. By removing the budget constraint, this equation will calculate the total ROI (i.e., the cumulative expected benefit) for a given level of investment (calculated as cumulative costs).

To understand the ROI as sites are added to the management site selection based on their original ce_{ik} rank, we need to plot the cumulative expected benefit against the cumulative cost. We did this for each of the three management strategies, and for each of the three benefit metrics. To determine whether ROI is better than an arbitrary approach that does not consider cost effectiveness, we calculated and plotted cumulative ROI curves for randomly selected management sites. For a more realistic comparison to potential selections by a manager optimizing for either benefits or cost, we calculated maximize-benefit and minimize-cost ROI curves (with no other constraints). In these cases, sites were ranked by species richness with disregard of cost for the former, and by cost with disregard of species richness for the latter. We also conducted sensitivity analyses to clarify (1) whether costs or expected benefits drive the differences in cost effectiveness between sites, by reanalyzing the ROI using constant costs or constant benefits, and (2) whether results are robust to variation in cost estimates that might be driven by the uncertainties outlined above. To visualize the changes in benefits with cumulative investment, we fitted the ROI curves determined by cost-effective selection of management sites using a cubic smoothing spline in R, and fitted the ROI curves determined by random selection of management sites with linear regression. To compare the distribution of priority sites under different benefit metrics, we mapped the variation in expected benefits and cost effectiveness for managing each threat across the landscape. We also compared the cost effectiveness and percentage of total possible expected species benefit with different budgets relative to an unrestricted budget, for each of the three management strategies (foxes, grazing, and combined).

Spatial priorities for single or multiple actions using ROI analysis.—The steps above demonstrate how to find the return on investment for a given management strategy. Our final aim was to solve the problem of choosing where to implement one or more management actions across the landscape for a given budget, dependent upon the expected benefits of the benefit metric incorporating the most information (Metric 3). We selected the best management strategy, i.e., the one that resulted in the highest and quickest benefits for a given budget based on the results of the ROI approach (see *Comparing expected benefits in an ROI approach*). We illustrated ranked cost efficient sites for this "best strategy" under a total budget of \$10 million by mapping spatial locations according to whether one or more actions was selected on the basis of cost effectiveness by the knapsack approach.

For the described analysis, we processed the spatial data using a geographic information system (GIS) (ArcGIS v10) and exported the spatially indexed data (habitat benefit and management cost maps) from the

GIS into text files. We then imported the data to spreadsheets (Microsoft Excel) for calculating the new ROI attributes. We calculated species' benefits using the three different metrics, and cost-effectiveness values. Next we sorted data by cost-effectiveness values to determine ROI rankings for each site, and then we cumulatively summed benefits and costs for discrete budgets. In other words, the solution is a greedy-heuristic based on the rank order of the benefit–cost ratio. We joined the calculated attributes back to the spatial data in the GIS for map display, based upon the unique spatial index number.

Results

The expected benefit of threat management is high with low initial investment when management sites are selected based on their cost-effectiveness rank. In comparison, benefit is linearly related (linear regression models in all cases $R^2 > 0.99$; $P < 0.001$; df $\geq 22\,517$; Appendix D) to investment with indiscriminate selection of management sites (Fig. 1). Most of the expected benefit to threatened species is achieved when <\$10 million is spent in this region over 20 years, if measured using Metric 2 or 3 (~67%; Fig. 1). This is illustrated by the steepness of the ROI curves when cumulative management cost is <\$10 million, with diminishing returns as more money is invested in managing each threat. When the management goal is focused on species richness only (Metric 1), rather than benefits that include addressing habitat restriction (Metric 2), and vulnerability (Metric 3), the slope of the ROI curve is flatter (Fig. 1a). Even so, expected benefits to species are more than four times as high if investing \$10 million in cost-effective, species-rich sites using a combined threat management strategy (Metric 1; expected benefit = 19.2%), than if the same amount of money is randomly invested (expected benefit = 4.5%; Fig. 1a). However, by further specifying a management goal that prioritizes restricted habitats with less vulnerability in bio-diverse areas (Metric 3), the contrast is even greater: expected species benefit is 16 times higher when \$10 million is spent on management sites that are prioritized using cost effectiveness (Metric 3; expected benefit = 66.4%) as compared to random selection (cumulative benefit = 4.2%; Fig. 1c).

Returns from investing in more realistic traditional conservation planning scenarios (maximize benefit or minimize cost) are expected to be higher than if resources are arbitrarily allocated, but lower than if investing in cost-effective actions prioritized using Metric 3 (Fig. 2). Particularly in the maximize benefit case for grazing management action (Fig. 2b), very expensive sites appear to drive costs up while returning little overall benefit.

When we reanalyzed the ROI curves keeping either benefits or costs constant, the comparisons between curves varied depending on the metric used to calculate benefit (Appendix E). When benefits are kept constant

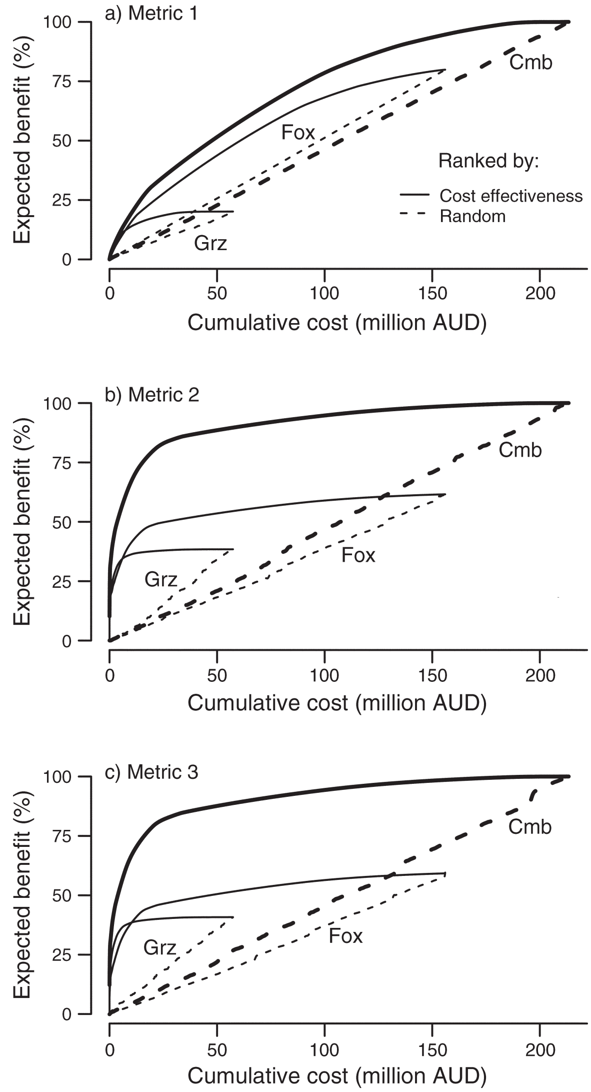

Fig. 1. Total return-on-investment (ROI) curves show greater expected benefit for the same level of investment (x-axis scales show cumulative costs in million AUD [Australian dollars]) when management sites are chosen by ranked cost effectiveness as compared to arbitrary/random selection (see Appendix D for linear regression statistics). Alternate action strategies are fox-baiting (Fox), grazing control (Grz), or combined fox and grazing control (Cmb). ROI curves also show that choosing management sites based on different management goals also has contrasting results, as seen when prioritizing action for areas with (a) high species richness (Metric 1); (b) and in addition, restricted habitats (Metric 2); (c) and in addition, less threat vulnerability (Metric 3). Of 129 894 (500 × 500 m) potential management sites, 22 519 of the sites with grazing vulnerability and 86 790 of the sites with fox vulnerability were matched with associated management costs. Out of these sites (109 309 possible action × site combinations), 16 727 of the sites have vulnerability to both threats.

and costs are spatially variable, cost effectiveness is necessarily driven by costs. However, we found that with constant benefits, the ROI curve is more similar to the curve that uses the variable costs and expected benefit

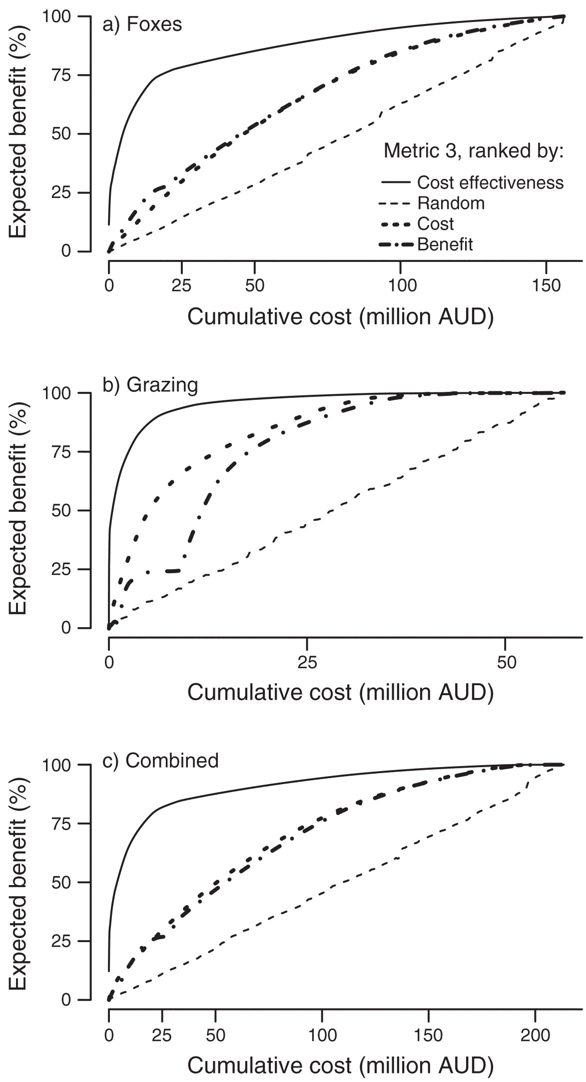

FIG. 2. Total ROI curves show the relationship between investment (cumulative cost: million AUD) and the percentage of the total benefit expected to be returned from the alternative threat action-abating strategies of (a) fox-baiting; (b) grazing control; (c) combined fox and grazing control, in areas with high species richness, restricted habitats, and less threat vulnerability (Metric 3). As compared to choosing sites based on ranked cost effectiveness, choosing management sites either by cost (ranked low to high with disregard to benefit, i.e., minimize cost) or benefit (species-richness ranked high to low with disregard to cost, i.e., maximize benefit) can result in lesser conservation outcomes for the same level of investment. All are superior to random (arbitrary) action.

values for Metric 1, whereas it is more similar to the random allocation of investment curves for Metrics 2 and 3. In contrast, when costs are held constant and cost effectiveness is driven by benefits, the results are reversed and the ROI curve is more similar to the random allocation of investment curve for Metric 1, whereas the curves for Metrics 2 and 3 are closer to the curves that use the variable costs and expected benefit values. The maximize-benefit and minimize-cost scenarios appear to closely mirror the constant-benefit approach of each metric. Varied management cost produces similarly shaped ROI curves for implementing cost-effective threat management action (Appendix G: Fig. G1).

The spatial distribution of expected benefits differs from that of cost effectiveness (ce_{ik}) values for both fox threat action (Fig. 3a) and grazing management (Fig. 3b). The most cost-effective areas generated using benefit metrics that take into account species richness and restricted habitats (Metric 2) plus areas of less threat (Metric 3) vary from those selected using Metric 1 (species richness only), particularly under low budgets (Fig. 3, cost-effectiveness columns). Differences between the spatial distribution of priority areas chosen using Metric 2 and Metric 3 to determine cost effectiveness are less prominent, but are still present. Areas with high expected benefit of grazing management (Fig. 3a) and fox management (Fig. 3b) also vary markedly across the landscape (Fig. 3, benefit columns).

Cost effectiveness of threat management action is highest with initial investments, and decreases as more money is invested, with diminishing returns. Using Metric 3, the most cost-effective strategy with the highest expected species benefit is to simultaneously implement grazing management along with fox-baiting across the landscape (Fig. 4). For a budget of \$10 million, it is almost twice as cost effective to manage for both threats ($\sum ce_{i(\text{both threats})} = 7.85$), than to solely manage for foxes ($\sum ce_{i(\text{fox})} = 4.52$) or grazing ($\sum ce_{i(\text{grazing})} = 4.55$; Fig. 4a). Furthermore, the first \$1 million spent on management is the most cost-effective allocation of money to abate threats to species, with rapidly diminishing returns on additional money invested; spending \$1 million on a combined management strategy is greater than five times more cost effective than spending \$10 million ($\sum ce_{i(\text{both threats})} = 42.65$ vs. 7.85; Fig. 4a. (Appendix F: Tables F1 and F2 detail ce_{ik} values under discrete budgets for each metric in Appendix F). Likewise, spending only \$1 million delivers more than a third (36%) of the total possible expected benefit to species, i.e., if there was unlimited money to spend on a combined management strategy (Fig. 4b). In addition, >66% of the possible expected species benefit is achieved with the first \$10 million budgeted for management; an investment of \$50 million would increase expected species benefit only by 21% over the initial \$10 million investment (Fig. 4b).

Cost effectiveness guides which management action to implement with additional investment for the greatest expected benefit to threatened species (Fig. 5a). For each increment in investment in the best strategy, we plotted what percentage of that budget was allocated to either action. If budgeting \$10 million, approximately half of the investment is in fox management, after initially greater investment in grazing management (Fig. 5b). Mapping the spatial distribution of cost-effective sites for management action indicates high priority areas for

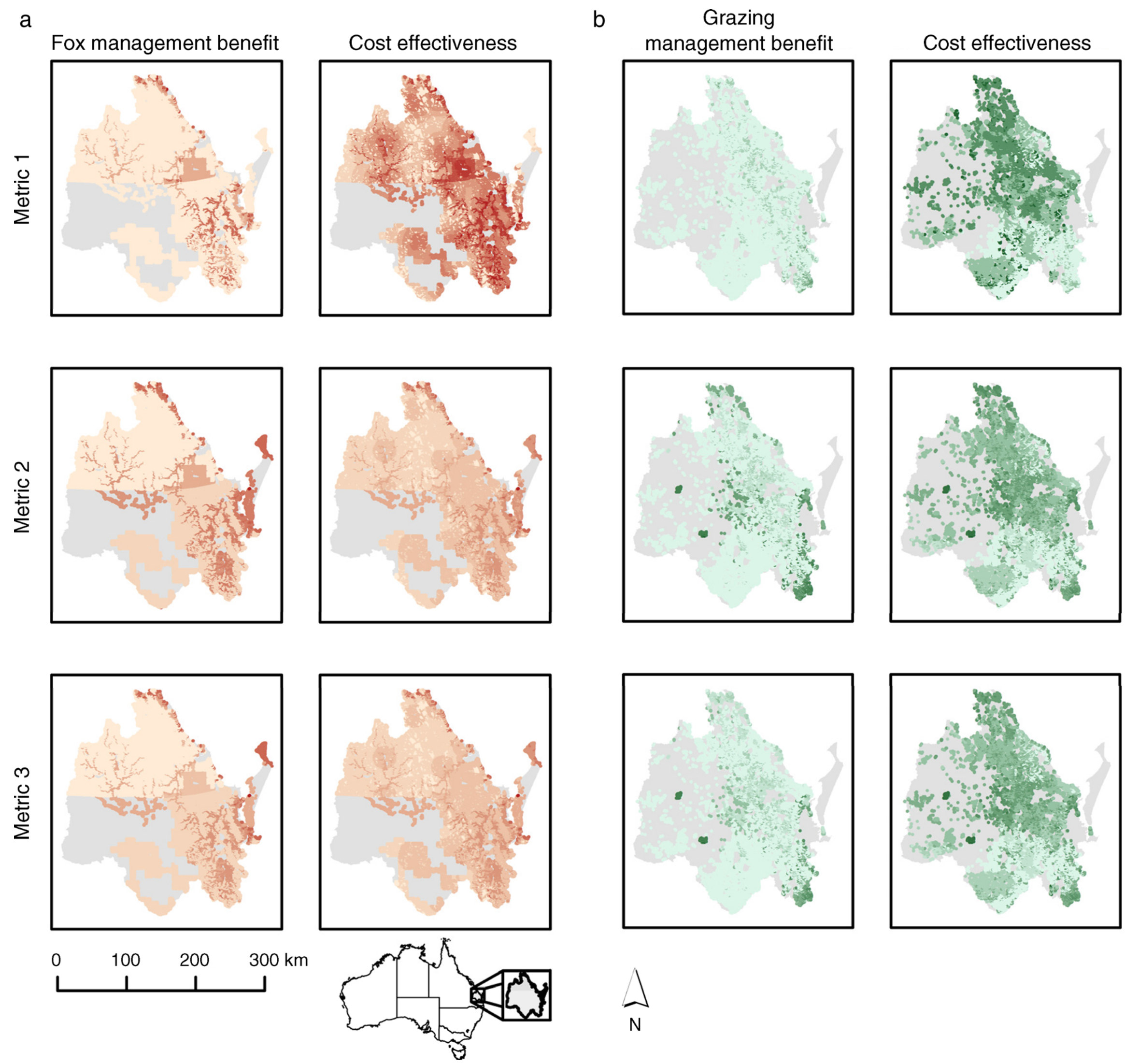

FIG. 3. The spatial distribution of the expected benefits of management for species who are vulnerable to (a) fox predation or (b) grazing threats are not necessarily the same as cost-effective (ce_{ik}) locations of managing those threats. This result indicates that priorities differ depending upon data incorporated into the decision-making process; benefit metrics reflect different emphasis in achieving the management goal of threat abatement. Shading (light to dark) indicates low to high benefit and cost-effectiveness values for fox and grazing management. Expected benefits were estimated using three metrics: Metric 1 addresses species richness, Metric 2 also includes restricted habitats, and Metric 3 also accounts for multi-threat vulnerability. For display, data were normalized by dividing every benefit or cost-effectiveness value by the maximum possible summed benefit or cost-effectiveness value, respectively, for each threat and each metric. Maps are shaded using "geometric intervals" based on classes delineated by natural data groupings, a balance between highlighting middle and extreme values.

managing the threats of habitat degradation, invasive species predation, or both (Fig. 6).

DISCUSSION

The importance of considering conservation cost effectiveness for species protection through habitat acquisition is well understood (e.g., Ando et al. 1998). This concept is routinely used in systematic conservation planning (Margules and Pressey 2000) where algorithms are used to identify areas at a fine scale that efficiently meet feature targets at minimal cost (Ball et al. 2009), for producing priority rankings for biodiversity and alternative land uses (Moilanen et al. 2011*a*), and more recently for multifaceted conservation ROI analyses that consider additional dependencies (e.g., Kovacs et al. 2013, Kramer et al. 2013). However, with added algorithmic sophistication comes additional computational requirements and difficulty in translating results for stakeholders. Simplicity can be achieved through reducing the complexity or amount of data required to inform decision analysis, or through simplifying the decision-making approach itself. In this study we

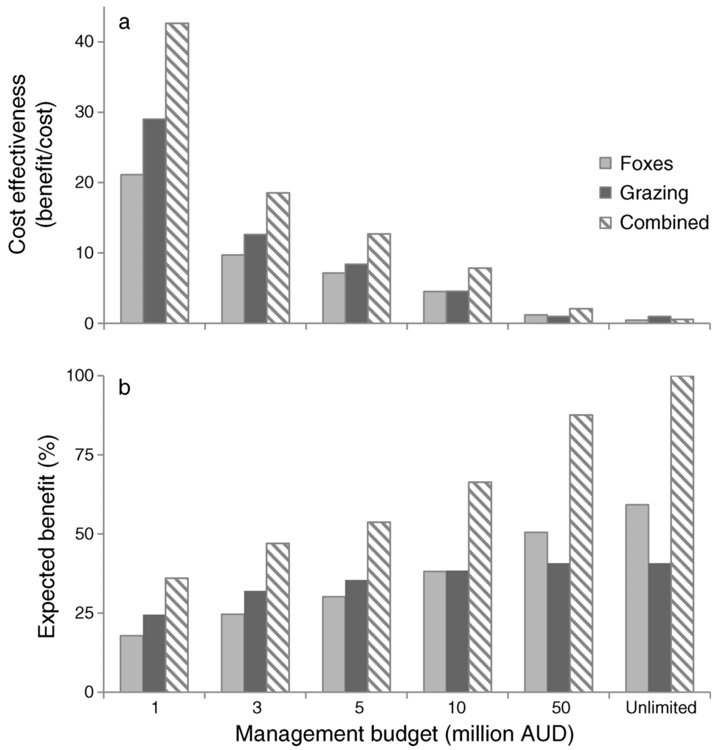

FIG. 4. A greater percentage of species are expected to benefit from simultaneously managing threats with a combined management action strategy, but cost effectiveness is greatest with initial investment and returns diminish rapidly. Constrained management budgets for fox, grazing, and combined management (in millions of Australian dollars) are plotted vs. (a) cost effectiveness, and (b) expected species benefit when management site selection across the region is by ranked cost effectiveness. Here, the goal is to benefit species by prioritizing threat abatement action for areas with high species richness, restricted habitats, and less threat vulnerability (Metric 3).

address both, by using a case study to explore the results of using different types of data to inform cost-effective prioritization of management actions, and secondly, by demonstrating a simple cost-effectiveness prioritization approach that uses a basic ROI as an alternative to more complex analyses. We focus on a fine-scale prioritization of multiple management actions to benefit multiple threatened species in an administratively managed natural resource region. Our approach improves on traditional hot spot allocations of funding based only on species and threats that do not account for costs and actions (Brooks et al. 2006) by demonstrating a technique for selecting management actions that optimizes ROI. The practical relevance of the technique, in addition to its transparency and ease of use, encourages uptake and utilization for management decision-making (Venkatesh and Davis 2000). Our approach allows managers to determine priority locations where one or more actions would be most cost effective to mitigate threats (Fig. 6).

Informed investment when budgets are low can yield high expected benefits to species

Small expenditures on management action result in high expected benefits to species when management sites are prioritized by cost effectiveness (Figs. 1, 2, and 4), compared with allocating funding based on species richness alone, or to minimize spending (Fig. 2). Other research has also found ROI to yield greater conservation outcomes per dollar than either of these approaches (Ferraro 2003, Naidoo et al. 2006, Murdoch et al. 2010).

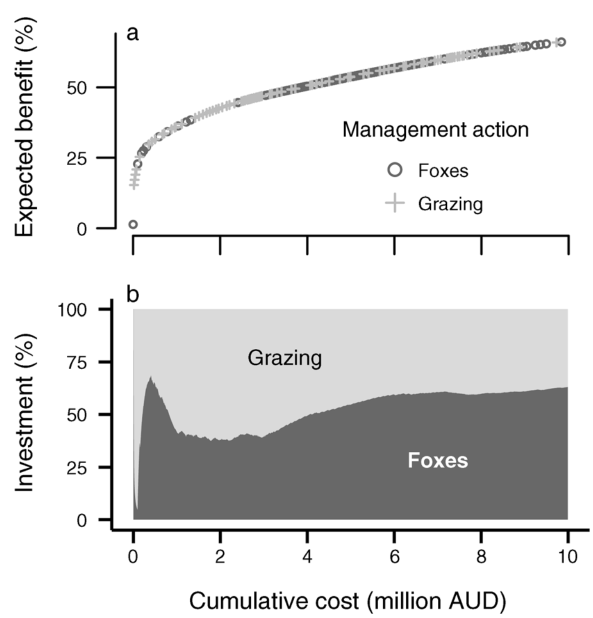

FIG. 5. (a) Cost-effectiveness guides which threat abatement action to implement with additional investment (cumulative cost in millions of Australian dollars) in management for the greatest expected benefit to species under the constraint of a $10 million budget. Only every 50th management site is plotted for easier visualization. (b) Investment in management costs is approximately half and half for fox and grazing action. Areas of high species richness, restricted habitats, and less threat are prioritized (Benefit Metric 3).

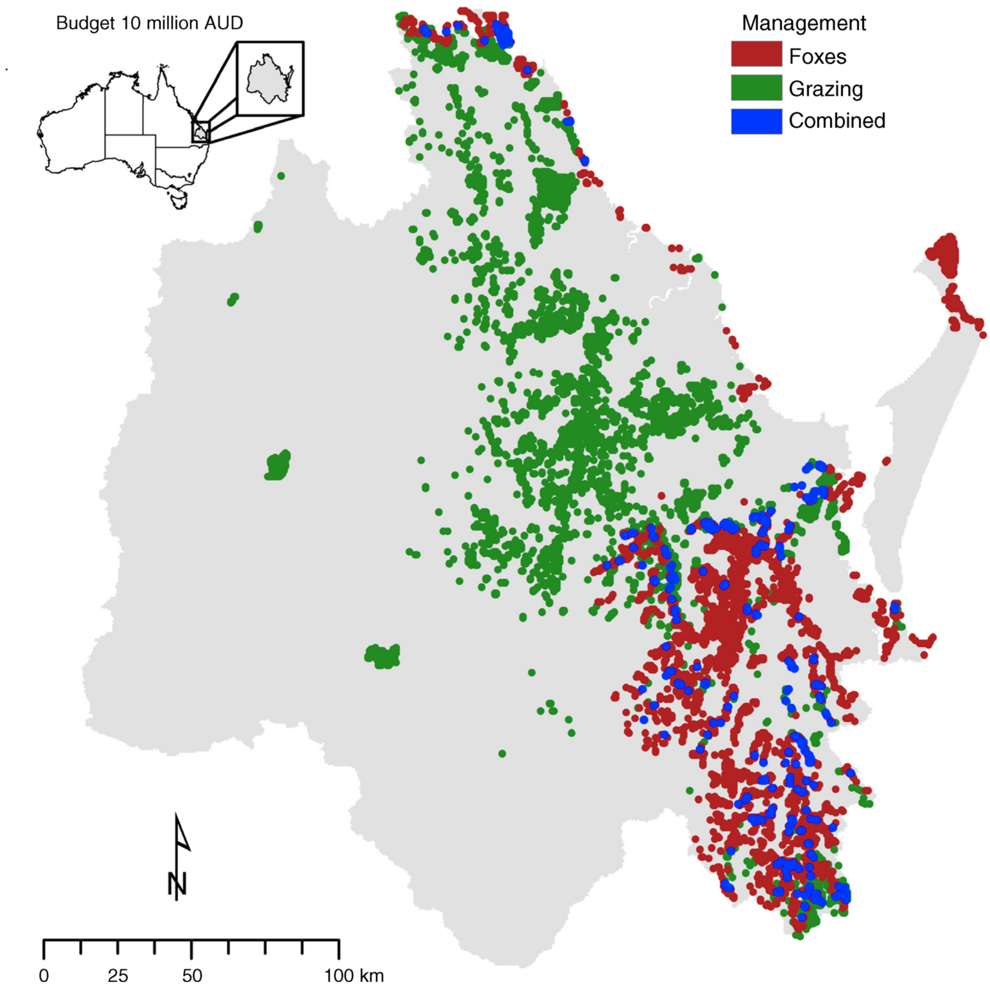

FIG. 6. Mapping the most highly ranked cost-effective sites for acting to abate threats to species shows where to efficiently spend a management budget of $10 million in the Burnett-Mary Natural Resource Management Region of Queensland, Australia. Here, the goal is to benefit species by prioritizing threat abatement action for areas with high species richness, restricted habitats, and less threat vulnerability (Metric 3). Symbols are not to scale.

We also compared our ROI results to a baseline of uninformed action, and found that allocating funding in an arbitrary manner (Figs. 1 and 2) yields low returns. It is important to highlight that ROI for threat management is high initially, but returns rapidly diminish. This means that investors receive the highest relative returns at low levels of investment, if money has been spent strategically on the most cost-effective locations and actions first (Figs. 1, 2, and 4). This finding supports arguments against delayed action (Grantham et al. 2008, 2009) and highlights the benefits of acting immediately even when limited funding is available to invest in management.

ROI results are sensitive to the way in which expected benefits are defined and how benefits and costs co-vary

Conservation ROI can be used to focus management decisions, when the management goal is clearly defined by the benefit metric. Our ROI results were sensitive to the metric used to measure benefit to species (Figs. 1 and 3), which indicates that explicitly stated management goals (and benefits) can effectively direct where the greatest benefit to species is expected to be obtained and at what cost. Differences in the returns on conservation investment and the spatial distribution of priority sites for management actions depended on the level of complexity and type of data used to parameterize benefit metrics (Figs. 1 and 3). Our benefit measures reflected priorities for management action based upon criteria of species richness, restricted habitat area, and number of threats to species (vulnerability). By comparing different benefit metrics we demonstrate that decision-making with multiple criteria relevant to the decision-making context narrows the array of choices about where to manage effectively (Fig. 3; Metric 1 vs. Metric 2 or Metric 3). Managers need to be aware that allocating funding based on species richness alone (a common goal of systematic conservation planning), rather than vulnerability to threats or other relevant ecological information such as rarity of habitat, might prioritize inefficient areas. If appropriate ecological data are available and can be applied as we have demonstrated, an ROI approach can yield enormous gains in conservation efficiency (Murdoch et al. 2010). Most of the priority species in this analysis are not known to be impacted by both threats, explaining the small difference in priority locations when the benefit measure accounts for multiple threats in addition to species richness and restricted habitat (Fig. 3; Metric 3 vs. Metric 2). Further discrimination would become more apparent if additional threats (e.g., we did not model the cost of the threat of fire frequency or invasive plants) or species (e.g., we did not include invertebrates in our analysis due to lack of data) were included.

ROI results are sensitive not only to the way in which expected benefits are defined but also to the range in values of management benefits and costs (and how these values co-vary). We demonstrated how to interpret whether the ROI is more sensitive to costs or benefits for a given scenario (Appendix E). For our case study, we found that costs drive differences in cost effectiveness when the range in expected benefits is small but the range in costs is large (e.g., Metric 1, Appendix E). ROI is often more dependent on cost differences rather than benefit differences (Ferraro 2003, Murdoch et al. 2007). If the range in expected benefits is larger than the range in costs (in our study Metrics 2 and 3 ranged in expected benefits by four orders of magnitude, whereas costs ranged by three orders of magnitude across the region), expected benefits are more likely to drive the early gains in efficiency (Appendix E). Compiling information on both the costs and the expected benefits is therefore crucial to determining where to achieve the greatest conservation outcomes using ROI. However, as a general rule of thumb, calculating cost effectiveness will be most useful where costs and expected benefits are heterogeneous across the landscape (Ando et al. 1998, Polasky et al. 2001, Naidoo et al. 2006). If cost is spatially homogeneous, it will not change the decision about where to implement a single management action (although it will inform the relative cost effectiveness of different management action considerations), and expected benefits alone might be used to inform decisions. This is rarely the case, although many previous conservation prioritizations have assumed constant costs or disregarded them completely (e.g., Brooks et al. 2006, Kremen et al. 2008).

We found that ROI results are robust to minor variations in management cost in that the shape of the curve remains similar with a reduction or increase in management costs, but benefits are accrued sooner if costs are lower than estimated or later if costs are higher (Appendix G: Fig. G1). However, in considering if management costs might change in the future (Arponen et al. 2010), our method also allows managers to visualize how considerably reducing the cost of an action (e.g., if landholder engagement reduces cost of stewardship agreements, or if innovation reduces cost of fox-baiting) might impact in which action they invest. High uncertainty in costs (i.e., up to 50%) can result in changes to the management actions being prioritized. For instance, we show that if grazing management costs are reduced by 50%, it is much more cost effective to invest in that action (Appendix G: Fig. G2). ROI also identifies high-priority sites for initial investment, which are similar despite cost uncertainty (Appendix G: Fig. G3). We have demonstrated a framework to account for variation in management costs, and to visualize how uncertainty in costs, or the potential for costs to change could alter (a) the selected actions and (b) where to act.

Multiple-action strategy for multiple species is better than single-action strategy

We found that a strategy that incorporates more than one action is more cost effective than single-action strategies, and benefits more species for a lower cost (Figs. 1, 2, and 4). This is because different actions provide benefits to different species by mitigating threats that act in different ways. This is an important finding given previous studies that have found no difference between overall outcomes of single- and multi-species management (Clark and Harvey 2002, Cullen et al. 2005). Moreover, while the predominant focus of conservation planning is on single-threat mitigation, such as reserve acquisition to stop habitat loss (e.g., Watson et al. 2011), it is clear that multiple management actions are needed to reduce species loss (Butchart et al. 2010). One explanation for previous lack of support for multi-species and multi-action threat management is the tendency to "lump" species together based on their habitat use in the same or similar region (Dobson et al. 1997), rather than grouping them according to vulnerability to specific threats. In this study, we explicitly grouped species based on their common threats as well as their co-location, and in doing so, focused management on eliminating or mitigating those threats as recommended for successful multi-species management (Clark and Harvey 2002).

By using a simple ROI approach, we were able to untangle the reasons behind one or more actions being selected at a given site (due to its cost effectiveness) to find a single solution (e.g., Fig. 6) to a decision problem where there are multiple threats and associated actions. In our case study, both reducing grazing pressure and controlling for foxes across the landscape was warranted, although both management actions are not always required or cost effective at the same location (Figs. 5 and 6). This makes our approach much easier to follow, and to translate the results of planning for multiple actions compared with more complex systematic conservation planning tools or ROI algorithms. The other approaches can produce multiple optimal solutions in which the trade-offs between the benefits and costs of doing different actions are obscured or at least potentially difficult to interpret by some.

Our approach assumes that there is no interaction between the threats being evaluated, and that their effects are additive. We locate cost-effective sites for managing both threats, although realistically, the impacts of threats may be synergistic (Brook et al. 2008) or antagonistic (Brown et al. 2013), and more threats exist in the study area than the two we examined. Dependencies or interactions can be added to ROI, but for simplicity of demonstrating our approach we did not do this. We have shown that managing more than one threat is more effective than managing threats independently. Accounting for dependencies would increase the efficiency, due to the possibility of cost-sharing across similarly managed species' projects (Briggs 2009, Joseph et al. 2009).

ROI informs where to act

Conservation ROI provides the means to visualize where particular management actions are most cost effective to implement given a budget (Fig. 6), aiding in stakeholder engagement. Alternative management actions affect species differently, and it is valuable to identify where a given management action can maximize conservation benefits (van Teeffelen and Moilanen 2008). We incorporate "indirect" threat maps in our analysis, assuming that different levels of species richness (derived from the summed habitat models of species that are known to be susceptible to a particular threat) can be used to represent areas of greater and lesser threat vulnerability (Wilson et al. 2005). We do this because natural resource managers are most concerned with acting on threats where species of interest occur or are likely to occur. An extensive literature discusses decisions about appropriate species for management focus, e.g., from focal (Lambeck 1997) to indicator (Tulloch et al. 2011) species; we chose previously identified priority threatened species (DERM 2010*a*). The choice of species for which to prioritize threat management is especially important when developing indirect threat maps for decision-making and conservation ROI, because the distribution of threats is dependent on the species and their associated vulnerabilities, as well the scale and level of detail in the predicted distributions (N. A. Auerbach, M. C. Evans, and H. P. Possingham, *unpublished data*).

Our ROI approach is particularly useful for ubiquitous threats, those that occur across most of the area of interest. It might be less useful for heterogeneous, localized, or patchy threats (e.g., a pocket of weed invasion, or disease), where we might need to know the precise location of the threat and not just the species vulnerable to it. However, certainty in threat data is difficult to come by, and our ROI approach provides first-pass guidance for decision-making in an uncertain world as to where management action could be focused.

We show that the sites that have the highest expected benefit of conducting threat management may not be the most cost effective (Fig. 3). Prioritizations that ignore costs can therefore lead to scarce resources being used inefficiently (Ando et al. 1998, Balmford et al. 2000), and opportunities to achieve conservation goals may be lost (Naidoo et al. 2006). Ultimately, however, assessing cost effectiveness may be just one part of the conservation decision-making process. Cost efficiency in conservation does not necessarily translate to effectiveness in management, i.e., work in practice (Arponen et al. 2010). This is because the least expensive solution may not be the best for long-term species persistence (Cabeza and Moilanen 2001), may be in marginal or infeasible locations (Gaston et al. 2001), or may depend on other socioeconomic concerns that were not considered (McBride et al. 2007, Knight et al. 2010). These additional considerations might enhance benefits (Di Minin et al. 2013), exacerbate threats (Faith and Walker 1996), or increase costs (Armsworth et al. 2006). Incorporating additional considerations such as these is possible in our ROI approach. For instance, feasibility or likelihood of successful management (or failure; a consideration that will decrease benefits in a heterogeneous way across the landscape) can be included in analyses by incorporating a probability of success into the benefit metric (Joseph et al. 2009, Tulloch et al. 2011, Tulloch et al. 2013). Alternatively, the importance of species for conservation might be considered by including a weighting factor in the benefit metric based on phylogenetic distinctiveness (e.g., Isaac et al. 2007, Joseph et al. 2009). For simplicity in presentation, we minimized consideration of these additional factors.

Conclusion

Our research demonstrates an ROI conservation decision-support approach that can be implemented to guide investment in cost-effective management action for abating threats to species. A cost-effectiveness analysis is straightforward: the calculation is based on the expected benefits to species for abating threats relative to the costs of action. Expected benefits are calculated using the spatial distribution of the species of interest, knowledge about which species are affected by a given threat, and additional management concerns, such as restricted habitats and vulnerability. Our approach integrates economics with conservation decision science to inform threat management for stakeholders interested in biodiversity conservation. Resource managers with limited capacity can apply the approach to any set of species and management actions to address abating threats to those species, employing commonly used GIS and spreadsheet software. Decisions about where to act efficiently are informed by cost effectively investing limited resources. Spatial conservation ROI analysis transparently and accountably locates the least expensive areas to manage specific threats for the greatest expected benefit to species. The ROI approach presented here provides decision support despite having only estimates of species benefits from, and management costs for, abating threats. Efficient use of limited resources to act on abating multiple threats to multiple species, informed by ROI, can be expected to deliver greater conservation outcomes to threatened species at less cost. A complex algorithm may not always prove a better solution to a conservation problem (Possingham et al. 2000), and a simple approach has more potential for uptake by conservation practitioners due to its ease of use and transparency.

Acknowledgments

This research was conducted with support from the Australian Government's National Environmental Research Program, the Australian Research Council Centre of Excellence for Environmental Decisions, and an Australia Postgraduate Award scholarship. The Queensland Government provided species presence data (WILDNET (DERM 2010*b*) and HERBRECS (DERM 2011) databases) used in species distribution modeling for this research. We appreciate the suggestions by the anonymous reviewers, which improved the research. We thank Rachel Lyons for her interest in, and discussions about, the case study research.

Literature Cited

ALA. 2012. Atlas of living Australia: sharing biodiversity knowledge. Global Biodiversity Information Facility Atlas of Living Australia. www.ala.org.au

Ando, A., J. Camm, S. Polasky, and A. Solow. 1998. Species distributions, land values, and efficient conservation. Science 279:2126–2128.

Anonymous. 1999. Threat abatement plan for predation by the Red Fox. Environment Australia, Canberra, Australia.

Armsworth, P. R., G. C. Daily, P. Kareiva, and J. N. Sanchirico. 2006. Land market feedbacks can undermine biodiversity conservation. Proceedings of the National Academy of Sciences USA 103:5403–5408.

Arponen, A., M. Cabeza, J. Eklund, H. Kujala, and J. Lehtomäki. 2010. Costs of integrating economics and conservation planning. Conservation Biology 24:1198–1204.

Ball, I. R., H. P. Possingham, and M. E. Watts. 2009. Marxan and relatives: software for spatial conservation prioritization. Pages 185–195 *in* A. Moilanen, K. A. Wilson, and H. P. Possingham, editors. Spatial conservation priorities action: quantitative methods and computational tools. Oxford University Press, Oxford, UK.

Balmford, A., K. J. Gaston, S. Blyth, A. James, and V. Kapos. 2003. Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs. Proceedings of the National Academy of Sciences USA 100:1046–1050.

Balmford, A., K. J. Gaston, A. S. L. Rodrigues, and A. James. 2000. Integrating costs of conservation into international priority setting. Conservation Biology 14:597–605.

Blackburn, T. M., P. Cassey, R. P. Duncan, K. L. Evans, and K. J. Gaston. 2004. Avian extinction and mammalian introductions on oceanic islands. Science 305:1955–1958.

Bottrill, M. C., et al. 2008. Is conservation triage just smart decision making? Trends in Ecology and Evolution 23:649–654.

Boyd, J., R. Epanchin-Niell, and J. Siikamäki. 2012. Conservation return on investment analysis, a review of results, methods, and new directions. Discussion paper. Resources for the Future, Washington, D.C., USA.

Briggs, S. V. 2009. Priorities and paradigms: directions in threatened species recovery. Conservation Letters 2:101–108.

Brook, B. W., N. S. Sodhi, and C. J. A. Bradshaw. 2008. Synergies among extinction drivers under global change. Trends in Ecology and Evolution 23:453–460.

Brooks, T. M., R. A. Mittermeier, G. A. B. da Fonseca, J. Gerlach, M. Hoffmann, J. F. Lamoreux, C. G. Mittermeier, J. D. Pilgrim, and A. S. L. Rodrigues. 2006. Global biodiversity conservation priorities. Science 58:58–61.

Brown, C. J., M. I. Saunders, H. P. Possingham, and A. J. Richardson. 2013. Managing for interactions between local and global stressors of ecosystems. PLoS ONE 8:e65765. http://dx.doi.org/65710.61371/journal.pone. 0065765

Burbidge, A. A., and B. F. J. Manly. 2002. Mammal extinctions on Australian islands: causes and conservation implications. Journal of Biogeography 29:465–473.

Busch, J., and R. Cullen. 2009. Effectiveness and cost-effectiveness of yellow-eyed penguin recovery. Ecological Economics 68:762–776.

Butchart, S. H. M., et al. 2010. Global biodiversity: indicators of recent declines. Science 328:1164–1168.

Cabeza, M., and A. Moilanen. 2001. Design of reserve networks and the persistence of biodiversity. Trends in Ecology and Evolution 16:242–248.

Carter, A., G. W. Luck, and S. P. McDonald. 2011. Fox-baiting in agricultural landscapes in south-eastern Australia: a case-study appraisal and suggestions for improvement. Ecological Management and Restoration 12:214–223.

Carwardine, J., T. O'Connor, S. Legge, B. Mackey, H. P. Possingham, and T. G. Martin. 2012. Prioritizing threat management for biodiversity conservation. Conservation Letters 5:196–204.

Carwardine, J., K. A. Wilson, G. Ceballos, P. R. Ehrlich, R. Naidoo, T. Iwamura, S. A. Hajkowicz, and H. P. Possingham. 2008. Cost-effective priorities for global mammal conservation. Proceedings of the National Academy of Sciences USA 105:11446–11450.

Clark, J. A., and E. Harvey. 2002. Assessing multi-species recovery plans under the Endangered Species Act. Ecological Applications 12:655–662.

Cullen, R., E. Moran, and K. F. D. Hughey. 2005. Measuring the success and cost effectiveness of New Zealand multiple-species projects to the conservation of threatened species. Ecological Economics 53:311–323.

Cullen, R., and P. C. White. 2013. Prioritising and evaluating biodiversity projects. Wildlife Research 40:91–93.

Daily, G., editor. 1997. Nature's services: societal dependence on natural ecosystems. Island Press, Washington, D.C., USA.

DERM. 1999. Land use mapping 1999 Queensland (ISO 19139). DERM (Department of Environment and Resource Management) QGIS (Queensland Government Information Service). http://dds.information.qld.gov.au/dds/

DERM. 2010*a*. Burnett Mary Natural Resource Management Region back on track actions for biodiversity. Queensland Government, Queensland, Australia. http://www.ehp.qld.gov.au/wildlife/pdf/burnett_mary_actions_for_biodiversity_complete.pdf

DERM. 2010*b*. WildNet (Database). Queensland Government Department of Environment and Resource Management, Queensland, Australia.

DERM. 2011. HERBRECS (Database). Queensland Government, Queensland Herbarium, Department of Environment and Resource Management, Queensland, Australia.

DEWHA. 2008. Species of national environmental significance (database). Department of the Environment, Water, Heritage, and the Arts. Australian Government, Canberra, Australia.

Di Minin, E., D. C. MacMillan, P. S. Goodman, B. Escott, R. Slotow, and A. Moilanen. 2013. Conservation businesses and conservation planning in a biological diversity hotspot. Conservation Biology 27:808–820.

Dobson, A. P., J. P. Rodriguez, W. M. Roberts, and D. S. Wilcove. 1997. Geographic distribution of endangered species in the United States. Science 275:550–553.

Dudík, M., S. J. Phillips, and R. E. Schapire. 2010. Maximum entropy modeling of species geographic distributions. Department of Computer Science, Princeton University, Princeton, New Jersey, USA.

Evans, M. C., R. A. Fuller, J. E. M. Watson, L. Venter, S. C. Bennett, P. R. Marsack, and H. R. Possingham. 2011*a*. The spatial distribution of threats to species in Australia. BioScience 61:281–289.

Evans, M. C., H. P. Possingham, and K. A. Wilson. 2011*b*. What to do in the face of multiple threats? Incorporating dependencies within a return on investment framework for conservation. Diversity and Distributions 17:437–450.

Faith, D. P., and P. Walker. 1996. Integrating conservation and development: effective trade-offs between biodiversity and cost in the selection of protected areas. Biodiversity and Conservation 5:431–446.

Ferraro, P. J. 2003. Assigning priority to environmental policy interventions in a heterogeneous world. Journal of Policy Analysis and Management 22:27–43.

Fleischner, T. L. 1994. Ecological costs of livestock grazing in western North America. Conservation Biology 8:620–644.

Game, E. T., E. McDonald-Madden, M. L. Puotinen, and H. P. Possingham. 2008. Should we protect the strong or the weak? Risk, resilience, and the selection of marine protected areas. Conservation Biology 22:1619–1629.

Gaston, K. J., A. S. L. Rodrigues, B. J. van Rensburg, P. Koleff, and S. L. Chown. 2001. Complementary representation and zones of ecological transition. Ecology Letters 4:4–9.

Goldstein, J. H., L. Pejchar, and G. C. Daily. 2008. Using return-on-investment to guide restoration: a case study from Hawaii. Conservation Letters 1:236–243.

Grantham, H. S., A. Moilanen, K. A. Wilson, R. L. Pressey, T. G. Rebelo, and H. P. Possingham. 2008. Diminishing return on investment for biodiversity data in conservation planning. Conservation Letters 1:190–198.

Grantham, H. S., K. A. Wilson, A. Moilanen, T. Rebelo, and H. P. Possingham. 2009. Delaying conservation actions for improved knowledge: how long should we wait? Ecology Letters 4:293–301.

Guikema, S., and M. Milke. 1999. Quantitative decision tools for conservation programme planning: practice, theory and potential. Environmental Conservation 26:179–189.

Hughey, K. F. D., R. Cullen, and E. Moran. 2003. Integrating economics into priority setting and evaluation in conservation management. Conservation Biology 17:93–103.

Isaac, N. J. B., S. T. Turvey, B. Collen, C. Waterman, and J. E. M. Baillie. 2007. Mammals on the EDGE: conservation priorities based on threat and phylogeny. PLoS ONE 2:e296.

James, A. N., K. J. Gaston, and A. Balmford. 1999. Balancing the Earth's accounts. Nature 401:323–324.

Joseph, L. N., R. F. Maloney, and H. P. Possingham. 2009. Optimal allocation of resources among threatened species: a project prioritization protocol. Conservation Biology 23:328–338.

Klein, C. J., et al. 2010. Prioritizing land and sea conservation investments to protect coral reefs. PLoS ONE 5(8):e12431. http://dx.doi.org/10.1371/journal.pone. 0012431

Knight, A. T., R. M. Cowling, M. Difford, and B. M. Campbell. 2010. Mapping human and social dimensions of conservation opportunity for the scheduling of conservation action on private land. Conservation Biology Series (Cambridge) 24:1348–1358.

Knight, A. T., R. M. Cowling, M. Rouget, A. Balmford, A. T. Lombard, and B. M. Campbell. 2008. Knowing but not doing: selecting priority conservation areas and the research-implementation gap. Conservation Biology 22:610–617.

Kovacs, K., S. Polasky, E. Nelson, B. L. Keeler, D. Pennington, A. J. Plantinga, and S. J. Taff. 2013. Evaluating the return in ecosystem services from investment in public land acquisitions. PLoS ONE 8:e62202.

Kramer, D. B., T. Zhang, K. S. Cheruvelil, A. Ligmann-Zielinska, and P. A. Soranno. 2013. A multi-objective, return on investment analysis for freshwater conservation planning. Ecosystems 16:823–837.

Kremen, C., et al. 2008. Aligning conservation priorities across taxa in Madagascar with high-resolution planning tools. Science 320:222–225.

Lambeck, R. J. 1997. Focal species: a multi-species umbrella for nature conservation. Conservation Biology 11:849–856.

Legge, S., M. S. Kennedy, R. Lloyd, S. A. Murphy, and A. Fisher. 2011. Rapid recovery of mammal fauna in the central Kimberley, northern Australia, following the removal of introduced herbivores. Austral Ecology 36:791–799.

Lennox, G. D., and P. R. Armsworth. 2013. The ability of landowners and their cooperatives to leverage payments greater than opportunity costs from conservation contracts. Conservation Biology 27:625–634.

Lennox, G. D., K. J. Gaston, S. Acs, M. Dallimer, N. Hanley, and P. R. Armsworth. 2013. Conservation when landowners have bargaining power: continuous conservation investments and cost uncertainty. Ecological Economics 93:69–78.

Lewis, D. J., A. J. Plantinga, E. Nelson, and S. Polasky. 2011. The efficiency of voluntary incentive policies for preventing biodiversity loss. Resource and Energy Economics 33:192–211.

Lowe, S., M. Browne, S. Boudjelas, and M. De Poorter. 2001. 100 of the world's worst invasive alien species: a selection from the global invasive species database. Species Survival Commission, World Conservation Union, Auckland, New Zealand.

Mace, G. M., H. P. Possingham, and N. Leader-Williams. 2006. Prioritizing choices in conservation. Pages 17–34 *in* D. MacDonald, editor. Key topics in conservation biology. Oxford University Press, Oxford, UK.

Mack, R. N., D. Simberloff, W. M. Lonsdale, H. Evans, M. Clout, and F. A. Bazzaz. 2000. Biotic invasions: causes, epidemiology, global consequences, and control. Ecological Applications 10:689–710.

Margules, C. R., and R. L. Pressey. 2000. Systematic conservation planning. Nature 405:243–253.

Marinoni, O., J. Navarro-Garcia, S. Marvanek, D. Prestwidge, D. Clifford, and L. A. Laredo. 2012. Development of a system to produce maps of agricultural profit on a continental scale: an example for Australia. Agricultural Systems 105:33–45.

Marris, E. 2007. Conservation priorities: what to let go. Nature 450:152–155.

McBride, M. F., K. A. Wilson, M. Bode, and H. P. Possingham. 2007. Incorporating the effects of socioeconomic uncertainty into priority setting for conservation investment. Conservation Biology 21:1463–1474.

McCarthy, D. P., et al. 2012. Financial costs of meeting global biodiversity conservation targets: current spending and unmet needs. Science 338:946–949.

Moilanen, A., B. J. Anderson, F. Eigenbrod, A. Heinemeyer, D. B. Roy, S. Gillings, P. R. Armsworth, K. J. Gaston, and C. D. Thomas. 2011*a*. Balancing alternative land uses in conservation prioritization. Ecological Applications 21:1419–1426.

Moilanen, A., H. Kujala, and J. R. Leathwick. 2009. The zonation framework and software for conservation prioritization. Pages 196–210 *in* A. Moilanen, K. A. Wilson, and H. P. Possingham, editors. Spatial conservation prioritization: quantitative methods and computational tools. Oxford University Press, Oxford, UK.

Moilanen, A., J. R. Leathwick, and J. M. Quinn. 2011*b*. Spatial prioritization of conservation management. Conservation Letters 4:383–393.

Mouysset, L., L. Doyen, and J. Jiguet. 2013. How does economic risk aversion affect biodiversity? Ecological Applications 23:96–109.

Murdoch, W. W., S. Polasky, K. A. Wilson, H. P. Possingham, P. Kareiva, and R. Shaw. 2007. Maximizing return on investment in conservation. Biological Conservation 139:375–388.

Murdoch, W. W., J. Ranganathan, S. Polasky, and J. Regetz. 2010. Using return on investment to maximize conservation effectiveness in Argentine grasslands. Proceedings of the National Academy of Sciences USA 107:20855–20862.

Naidoo, R., and W. L. Adamowicz. 2006. Modeling opportunity costs of conservation in transitional landscapes. Conservation Biology 20:490–500.

Naidoo, R., A. Balmford, P. J. Ferraro, S. Polasky, T. H. Ricketts, and M. Rouget. 2006. Integrating economic costs into conservation planning. Trends in Ecology and Evolution 21:681–687.

Naidoo, R., and T. Iwamura. 2007. Global-scale mapping of economic benefits from agricultural lands: implications for conservation priorities. Biological Conservation 140:40–49.

Nelson, E., S. Polasky, D. J. Lewis, A. J. Plantinga, E. Lonsdorf, D. White, D. Bael, and J. J. Lawler. 2008. Efficiency of incentives to jointly increase carbon sequestration and species conservation on a landscape. Proceedings of the National Academy of Sciences USA 105:9471–9476.

Nicholson, E., and H. Possingham. 2006. Objectives for multiple-species conservation planning. Conservation Biology 20:871–881.

Olsen, P. 2007. Glimpses of paradise: the quest for the beautiful parakeet. National Library of Australia, Canberra, Australia.

Phillips, S. J., R. P. Anderson, and R. E. Schapire. 2006. Maximum entropy modeling of species geographic distributions. Ecological Modelling 190:231–259.

Pimm, S. L., G. J. Russell, J. L. Gittleman, and T. M. Brooks. 1995. The future of biodiversity. Science 269:347–350.

Pisinger, D., and P. Toth. 1998. Knapsack problems. Pages 299–428 *in* D.-Z. Du and P. M. Pardalos, editors. Handbook of combinatorial optimization. Kluwer Academic, Dordrecht, The Netherlands.

Polasky, S., J. D. Camm, and B. Garber-Yonts. 2001. Selecting biological reserves cost-effectively: An application to terrestrial vertebrate conservation in Oregon. Land Economics 77:68–78.

Polasky, S., et al. 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. Biological Conservation 141:1505–1524.

Polasky, S., E. Nelson, E. Lonsdorf, P. Fackler, and A. Starfield. 2005. Conserving species in a working landscape: land use with biological and economic objectives. Ecological Applications 15:1387–1401.

Possingham, H. P. 2001*a*. The business of biodiversity: applying decision theory principles to nature conservation. TELA Series No. 9. The Australian Conservation Foundation, Melbourne, Australia.

Possingham, H. P. 2001*b*. Models, problems and algorithms: perceptions about their application to conservation biology. Pages 1–6 *in* MODSIM2001: International Congress on Modelling and Simulation. Modelling and Simulation Society of Australia and New Zealand, Australian National University, Canberra, ACT, Australia.

Possingham, H. P., S. J. Andelman, B. R. Noon, S. Trombulak, and H. R. Pulliam. 2001. Making smart conservation decisions. Pages 225–244 *in* M. E. Soulé and G. H. Orians, editors. Conservation biology: research priorities for the next decade. Island Press, Washington, D.C., USA.

Possingham, H. P., I. R. Ball, and S. Andelman. 2000. Mathematical methods for identifying representative reserve networks. Pages 291–305 *in* S. Ferson and M. A. Burgman, editors. Quantitative methods for conservation biology. Spring-Verlag, New York, New York, USA.

Queensland Herbarium. 2010*a*. Biodiversity status of pre-clearing and 2006b regional ecosystems SE Qld (ISO 19139). Brisbane Botanic Gardens, Department of Environment and Resource Management QGIS (Queensland Government Information Service), Toowong, Australia. http://dds.information.qld.gov.au/dds/

Queensland Herbarium. 2010*b*. Vegetation management act high value regrowth vegetation version 2.0 (ISO 19139). Brisbane Botanic Gardens, Department of Environment and Resource Management QGIS (Queensland Government Information Service), Toowong, Australia. http://dds.information.qld.gov.au/dds/

Read, J. L., and R. Cunningham. 2010. Relative impacts of cattle grazing and feral animals on an Australian arid zone reptile and small mammal assemblage. Austral Ecology 35:314–324.

Robins, L., and S. Dovers. 2007. NRM regions in Australia: the 'Haves' and the 'Have Nots.' Geographical Research 45:273–290.

Saunders, G., and L. McLeod. 2007. Improving fox management strategies in Australia. Bureau of Rural Sciences, Canberra, Australia.

Saunders, G. R., M. N. Gentle, and C. R. Dickman. 2010. The impacts and management of foxes *Vulpes vulpes* in Australia. Mammal Review 40:181–211.

Tulloch, A., H. Possingham, and K. Wilson. 2011. Wise selection of an indicator for monitoring the success of management actions. Biological Conservation 144:141–154.

Tulloch, A. I. T., I. Chadès, and H. P. Possingham. 2013. Accounting for complementarity to maximize monitoring power for species management. Conservation Biology 27:988–999.

Underwood, E. C., M. R. Shaw, K. A. Wilson, P. Kareiva, K. R. Klausmeyer, M. F. McBride, M. Bode, S. A. Morrison, J. M. Hoekstra, and H. P. Possingham. 2008. Protecting biodiversity when money matters: maximizing return on investment. PLoS ONE 3(1):e1515. http://dx.doi.org/10.1371/journal.pone.0001515

van Teeffelen, A. J. A., and A. Moilanen. 2008. Where and how to manage: optimal selection of conservation actions for multiple species. Biodiversity Informatics 5:1–13.

Venkatesh, V., and F. D. Davis. 2000. A theoretical extension of the technology acceptance model: four longitudinal field studies. Management Science 46:186–204.

Venkatraman, N., and V. Ramanujam. 1986. Measurement of business performance in strategy research: a comparison of approaches. Academy of Management Review 11:801–814.

Walsh, J. C., K. A. Wilson, J. Benshemesh, and H. P. Possingham. 2012. Unexpected outcomes of invasive predator control: the importance of evaluating conservation management actions. Animal Conservation 15:319–328.

Watson, J. E. M., M. C. Evans, J. Carwardine, R. A. Fuller, L. N. Joseph, D. B. Segan, M. F. J. Taylor, R. J. Fensham, and H. P. Possingham. 2011. The capacity of Australia's protected-area system to represent threatened species. Conservation Biology 25:324–332.

Weinstein, M. C., and W. B. Stason. 1977. Foundations of cost-effectiveness analysis for health and medical practices. New England Journal of Medicine 296:716–721.

Wilson, E. O. 1992. The diversity of life. Harvard University Press, Cambridge, Massachusetts, USA.

Wilson, K. A., M. C. Evans, M. Di Marco, D. C. Green, L. Boitani, H. P. Possingham, F. Chiozza, and C. Rondinini. 2011*a*. Prioritizing conservation investments for mammal species globally. Philosophical Transactions of the Royal Society B 366:2670–2680.

Wilson, K. A., M. Lulow, J. Burger, Y.-C. Fang, C. Andersen, D. Olson, M. O'Connell, and M. F. McBride. 2011*b*. Optimal restoration: accounting for space, time and uncertainty. Journal of Applied Ecology 48:715–725.

Wilson, K. A., M. F. McBride, M. Bode, and H. P. Possingham. 2006. Prioritising global conservation efforts. Nature 440:337–340.

Wilson, K. A., et al. 2010. Conserving biodiversity in production landscapes. Ecological Applications 20:1721–1732.

Wilson, K. A., R. L. Pressey, A. Newton, M. Burgman, H. P. Possingham, and C. Weston. 2005. Measuring and incorporating vulnerability into conservation planning. Environmental Management 35:527–543.

Wilson, K. A., et al. 2007. Conserving biodiversity efficiently: what to do, where, and when. PLoS Biology 5:1850–1861.

Withey, J. C., et al. 2012. Maximising return on conservation investment in the conterminous USA. Ecology Letters 15:1249–1256.

Supplemental Material

Appendix A

The case study (*Ecological Archives* A024-081-A1).

Appendix B

Model of opportunity costs to cease grazing (*Ecological Archives* A024-081-A2).

Appendix C

Model of management costs to bait foxes (*Ecological Archives* A024-081-A3).

Appendix D

Linear regression statistics for random management ROI curves (*Ecological Archives* A024-081-A4).

Appendix E

What drives early gains in efficiency? (*Ecological Archives* A024-081-A5).

Appendix F

Cost-effectiveness of threat management under different budgets (*Ecological Archives* A024-081-A6).

Appendix G

Variation in management cost sensitivity analysis (*Ecological Archives* A024-081-A7).