Setting Conservation Priorities

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A generic framework for setting conservation priorities based on the principles of classic decision theory is provided. This framework encapsulates the key elements of any problem, including the objective, the constraints, and knowledge of the system. Within the context of this framework the broad array of approaches for setting conservation priorities are reviewed. While some approaches prioritize assets or locations for conservation investment, it is concluded here that prioritization is incomplete without consideration of the conservation actions required to conserve the assets at particular locations. The challenges associated with prioritizing investments through time in the face of threats (and also spatially and temporally heterogeneous costs) can be aided by proper problem definition. Using the authors' general framework for setting conservation priorities, multiple criteria can be rationally integrated and where, how, and when to invest conservation resources can be scheduled. Trade-offs are unavoidable in priority setting when there are multiple considerations, and budgets are almost always finite. The authors discuss how trade-offs, risks, uncertainty, feedbacks, and learning can be explicitly evaluated within their generic framework for setting conservation priorities. Finally, they suggest ways that current priority-setting approaches may be improved.

Key words: **conservation prioritization; systematic conservation planning; reserve design; decision theory; biodiversity; surrogates; costs; threats; vulnerability; likelihood of success; threatened species; hotspot; focal species; uncertainty; risk; feedbacks**

Introduction

Conservation activities must be prioritized so that scarce funds and resources are used efficiently and effectively to prevent long-term loss and degradation of biodiversity and ecological systems. A plethora of impacts are causing a mass extinction event (Lawton and May 1995), including habitat destruction and fragmentation, overexploitation of natural resources, invasive species, global climate change, and emerging diseases (Groombridge and Jenkins 2002). While we can attempt to increase the resources available for conservation, at present funding is insufficient in the context of current threats, and conservation competes with other societal priorities, such as food production, human habitation, and resource extraction (Abbitt *et al*. 2000; James *et al*. 2001; Balmford *et al*. 2004; Naidoo and Iwamura 2007).

Land-use allocation patterns around the world are not favorable for biodiversity conservation. Almost 500 million hectares of the Earth's surface is allocated to agriculture: arable and permanent cropland is estimated to cover approximately 1500 million hectares and permanent pasture approximately 3400 million hectares (FAO 2008). By comparison, terrestrial protected areas cover only 1840 million hectares (WDPA Consortium 2004). Global spending on conservation is also far lower than in other sectors. For example, in 2007 the total annual revenue of the world's largest non-government conservation organization, The Nature Conservancy (www.nature.org), was one tenth of the net profit of Wal-Mart. At a local level, expenditure on environmental

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The Year in Ecology and Conservation Biology, 2009: Ann. N.Y. Acad. Sci. 1162: 237–264 (2009).
doi: 10.1111/j.1749-6632.2009.04149.x

protection is often much lower than expenditure in other sectors. In Australia, for example, local governments receive over AUS$2.6 billion in revenue for environment protection activities, which equates to only 13% of their total revenue, and only a fraction of that is spent on biodiversity conservation (Australian Bureau of Statistics 2004).

People have long set conservation priorities, with subsistence communities declaring particular areas or species exempt from exploitation (Wright and Mattson 1996; Chandrashekara and Sankar 1998). Today the scientific literature documents a range of priority-setting approaches (Smith and Theberge 1987; Costello and Polasky 2004; Brooks *et al*. 2006; Sarkar *et al*. 2006). In its various forms, conservation priority setting seeks to identify where, how, on what, and/or when we should act first to conserve biodiversity efficiently, on the assumption we cannot do everything everywhere at once. Much of the conservation priority–setting literature concerns the identification of new protected areas or networks of protected areas, and this is often referred to as systematic conservation planning (Margules and Sarkar 2007). In this review, we explore a broad array of priority-setting approaches that concern not only the locations we wish to conserve, but also the biological assets we are concerned about, and the conservation actions available to increase their chance of persistence. There are also a wide variety of tools to assist the identification of conservation priorities, covering a broad spectrum from mathematical to intuitive.

The first aim of this chapter is to provide a generic framework for setting conservation priorities. It is based on the principles of classic decision theory (Clemen 1996; Elster 2003). While we describe the process of setting conservation priorities mathematically, the terminology we have employed is flexible and the framework reflects logical decision making. This framework encapsulates the key elements of any problem, such as the objective, the constraints, and our knowledge of the system. The system knowledge encompasses the key elements of our area of interest, which may be an entire biome, a country, or a parcel of land (herein termed locations). The key elements constitute the assets we wish to protect (a single species, a group of species, habitats, or an ecological service or function), the key threats to these assets, the actions likely to abate the threats, and the costs associated with their implementation.

The second aim of this chapter is to evaluate the benefits and risks of allocating funds with and without considering these elements (e.g., of allocating funds to particular assets without considering the cost of the action requiring implementation in order to conserve the asset in a particular location). On the basis of our review, we conclude that while the prioritization of assets or locations is commonplace, prioritization approaches that do not consider the conservation actions required to protect the assets in particular locations are incomplete.

The third aim of this chapter is to illustrate how multiple conservation objectives can be integrated. We discuss examples of trade-offs when there are disparate conservation objectives and how trade-offs can be explicitly evaluated within the generic framework for setting conservation priorities.

We acknowledge the challenges associated with the rational and efficient allocation of conservation funds, including lack of a clear conservation objective, a paucity of biological and economic data, uncertainty about the likely success of investments, and societal values or political processes that can influence investment decisions. The final aim of this chapter is to identify some opportunities for reducing these challenges when setting conservation priorities.

Defining a Conservation Prioritization Problem: What Are the Key Elements?

Conservation prioritization problems can be structured as classic optimization problems, with an objective function, mathematical descriptions of our knowledge of the system and the state variables, control variables, constraints, and system equations (Possingham *et al.* 2001; Wilson *et al.* 2007). In systematic conservation planning there have been two broad class of prioritization problem—the minimum-set and maximal-coverage problems—defined mathematically in Box 1.

In all problem formulations, the objective function reflects our conservation goal and has an explicit measure of performance. In a conservation context we may seek to maximize the number of species conserved to an adequate level, or to minimize the number of species that are expected to go extinct over the next 100 years. Often there is more than one objective. In almost all circumstances the strategy that optimizes one objective will not optimize other objectives, which means that compromises must be made. Objectives can be combined into a single objective using a weighted sum or some other function; however, the use of weights presents challenges that we discuss later.

What we are required to know about the system (the system knowledge) will depend on the particular problem at hand—in a conservation prioritization context we would want to know what the assets are, what the threats are to these assets, what actions can be taken to abate these threats or otherwise improve the state of the assets, and the cost of carrying out those actions, which may vary over the area of interest and through time. We may also require additional knowledge, such as the uncertainty associated with key parameters and whether some outcomes are random or stochastic. The state variables represent the assets in the area of interest. The assets might be biological (such as habitat types, populations of threatened species, the distribution of invasive species, and ecosystem processes) or nonbiological (such as the locations of landholders that are amenable to habitat restoration or the distribution of water flows through a catchment).

The control variables reflect the things we could do. In the context of conservation prioritization we control how much money or resources we direct toward different conservation actions in any location and at a particular time. These control variables will directly or indirectly influence the states of our assets. The constraints limit the choice of control variables. In maximal-coverage prioritization problems (Box 1) constraints may include a budget, how many parcels of land can be restored each year due to operational and seasonal limitations, or may reflect the area of land that can be conserved annually. In minimum-coverage problems (Box 2) the constraints may represent the minimum amount of conservation that we aim to achieve for each asset. For example, we could have a target amount of each species requiring habitat protection or a target for the frequency of flooding required in a river basin (an ecosystem process asset). The system equations reflect our understanding of how our state variables (and also our objective function) change as a function of each other, system parameters (such as the chance an area is cleared for urbanization), and the control variables. The overall aim is to find a solution through manipulation of the control variables that has the highest possible value of the objective function subject to our constraints. While optimal solutions might be desired, multiple near-optimal solutions are often sought for the sake of flexibility and the ease of calculation and communication.

The conservation prioritization problem can be described mathematically. A general version of the conservation prioritization problem subject to a annual budget, b_t, is described below. Let x_{jkt} be the amount of money to be spent on action k in location j in year t. For example, one possible action might be to spend \$10,000 removing an invasive species from a single location in any one year. Each year the cost of all the actions across all the locations must be less than our annual budget, so we have the constraint:

$$\sum_{j=1}^{N}\sum_{k=1}^{P} x_{jkt} \leq b_t \quad \text{for every year } t. \qquad (1)$$

where N is the number of locations and P is the number of possible actions. Determining our budgetary constraints is generally straightforward, although obtaining data on the costs of different conservation actions in different locations can be challenging (Naidoo *et al.* 2006). Formally, we have the further constraint that x_{jkt} is greater than zero, since we cannot spend negative dollars on a conservation action.

Next we need a set of states in every location and time, y_{ijt}. For example, in each location and in each year we might need to know the size of a population of a threatened species, the percentage of plants being adequately pollinated, the condition of a particular habitat type, or the genetic diversity within a population. Our choice of state variables will depend on many issues, such as whether our state variables are assets (e.g., the population size of a threatened species) or whether the state variable influences our assets (e.g., the population size of an invasive species). Our objective will be to maximize some weighted combination of the state variables. As we describe later, choosing appropriate weightings can be difficult, but we assume for now that weights can be derived. The objective is to maximize over a time frame, T, of interest:

$$\sum_{i=1}^{M}\sum_{j=1}^{N}\sum_{t=1}^{T} w_{ijt} f(y\ldots) \qquad (2)$$

where w_{ijt} is a weighted value assigned to the amount of state variable (or asset) i, in place j, at time t. In principle the value we assign to any asset at any time and location can be a complex function of all the assets in the system $f(y\ldots)$. In this approach, money itself can be a state variable and we can replace the annual funding constraint by a dynamic accounting equation. Finally, we need a series of dynamic models that show how our actions and all the state variables interact to move, invariably in a stochastic way, the whole system from one state to another through time. Ultimately the whole problem is complex, which is why more specific and tractable versions of the general conservation prioritization problem exist (see Box 1).

Box 1. Mathematical Definition of Two Broad Class of Conservation Prioritization Problem: The Minimum-Set and the Maximal-Coverage Problems

Minimum-Set Problem

The objective of the minimum-set problem is to minimize the resources expended while meeting a given set of conservation targets (Pressey 2002). Each location j has a cost c_j and each asset i has a target r_i. The variable x_j equals 1 if location j is selected for investment, otherwise it equals 0. The contribution to the conservation of asset i by the selection of location j is contained in a matrix with elements a_{ij}. The objective is to minimize the cost:

$$\sum_{j\in P} c_j x_j$$

subject to the constraint that the targets are met:

$$\sum_{j\in P} a_{ij} x_j \geq r_i$$

for every asset i.

There are simple and complex versions of the minimum-set problem. All locations might be assumed to have equal cost or we can allow each location (or location–action combination) to reflect the actual monetary or social cost. Each asset may also be described in different currencies (e.g., number of individuals, extent of occurrence, probability of occurrence) and individual targets can be set for each asset. If more than one action is under consideration, we can set asset targets for each action and in each location, for example, we might aim to represent 20% of the range of a species in a strict protected area and 30% in an area managed for sustainable-resource extraction.

Maximal-Coverage Problem

The objective of the maximal-coverage problem is to maximize some measure of "benefit" (in a simple case, this might be the number of targets met for our assets), given a fixed budget or resources that can be expended (Church and ReVelle 1974). That is, the objective is to maximize

$$\sum_{i \in I} f\,(y_i(x.))$$

subject to

$$\sum_{j \in J} c_j\, x_j \leq b$$

where c_j and x_j are as previously defined, and y_i is the amount of feature i conserved in reserve system $x.$, and f is a function that turns that into a value. The maximum available expendable budget is b, which is in the same units as c_j. Like the minimum-set coverage problem, there are multiple variations to the maximal-coverage problem. The problem may be solved without applying targets and the budget may or may not be sufficient for meeting all targets. While budgets are typically finite, they are not necessarily fixed and the budget can be updated through time if more or fewer funds become available. In the simplest case, if the target allocated for asset i is achieved, y_i equals 1, and otherwise it equals 0. Alternatively, the benefit can be measured by a set of functions representing the incremental gains in the conservation of each asset per dollar invested. The functions relating benefit to the costs of acquiring these benefits can be linear, meaning that benefits are continuous, or curved to represent situations where benefits diminish or increase with each dollar invested. Assets can also be differentially weighted to emphasize investment in those that we value highly (e.g., assets that are locally rare or threatened) (Arponen *et al*. 2005, 2007).

Assets

In systematic conservation planning, the ecological purpose of a reserve system is to sustain representative samples of the full range of biodiversity and ecosystem processes of the region in which it lies (Margules and Pressey 2000). Typically, conservation prioritization analyses focus on (1) protecting particular species (e.g., threatened, umbrella, or flagship species; see later in this chapter), (2) protecting areas of high species richness or areas with high endemism, and/or (3) protecting functioning ecosystems and their associated ecosystem processes. Knowledge of our assets is therefore a key component of a conservation prioritization problem—these are the species or other facets of biodiversity that we wish to conserve. Data are often incomplete, however, even for well-known taxa such as birds and mammals, and worse for lesser-known taxa like insects and fungi. Our knowledge of ecosystem processes is perhaps the least complete (Pressey *et al*. 2007). This paucity of data forces us to use biodiversity surrogates in conservation priority setting.

The surrogates (such as using birds as a surrogate for all vertebrates) are the state variables in the objective function. Proposed surrogates include well-known taxonomic groups, habitat types, and ecological classifications of land and water (Ferrier *et al*. 2000). The level of support in the literature for surrogates and their ability to represent other elements of biodiversity are variable (Reyers and van Jaarsveld 2000; Beger *et al*. 2003; Faith *et al*. 2004). Surrogates have been found to perform well in some cases (Howard *et al*. 1998; Oliver *et al*. 1998) and poorly in others (van Jaarsveld *et al*. 1998). Rodrigues and Brooks (2007) found cross-taxon surrogates to be more effective than surrogates based on environmental data, particularly environmental surrogates that are based only on abiotic data.

Many conservation plans involve setting targets—amounts of a biodiversity asset at which we feel the asset is conserved. The benefits gained for each asset through investment in their conservation can be quantified with or without use of targets. Targets can be set in a multitude of ways and can, for example, reflect the percentage of historical extent of a vegetation type or a minimum viable population size for a species. The development of targets can be informed by ecological theory and empirical knowledge in order to accommodate the requirements of species for their persistence, for example, by accounting for information on life history characteristics, habitat connectivity requirements, and threatening processes (Burgman *et al*. 2001). Targets that are generalized for sociopolitical purposes (such as, protect 10% of every habitat type) have limited scientific basis and should not be interpreted as the minimum amount of habitat requiring protection in order to ensure species persistence (Soulé and Sanjayan 1998; Noss 2004). The use of targets is implicitly based on the assumption that there is a critical level of protection required and until that level is achieved there are no benefits, or that the benefits accrue linearly (Arponen *et al*. 2005; Carwardine *et al*. In Press). The relationship between the investment and the benefit derived is of course more complicated and can take a variety of shapes (see later). In general the per-unit benefit of increasing the level of a protection of an asset will decline, that is, benefit functions should have slopes that decrease, and hence show diminishing returns (Davis *et al*. 2006; Wilson *et al*. 2006).

Threats

We need to understand the threats to biodiversity in order to evaluate the risk of losing species or habitats if no conservation action is taken; that is, the vulnerability of different locations and/or assets (Wilson *et al*. 2005). It is an inefficient use of funds to invest in locations where the risk of exposure to threats is low or where there are no biodiversity assets impacted by a particular threat. Ignoring threats in conservation prioritization is only justifiable when there is enough money to abate all threats at once, which is rarely the case. Incremental conservation investment is much more common and during a protracted process of protection, biodiversity (or the surrogates for biodiversity) might be lost or degraded.

Threats are highly variable spatially and there are numerous ways to evaluate the vulnerability of different locations and the assets they contain (Wilson *et al*. 2005). Furthermore, there are numerous ways for how information on vulnerability might be used to identify priority assets or locations for conservation investment (Redford *et al*. 2003; Brooks *et al*. 2006). In the general conservation prioritization problem outlined earlier, threats can be included as another state variable that, in general, has a negative impact on state variables that we value. Under such circumstances the state variable might be the size of an invasive-species population or the rate of sea-level rise due to global climate change. At a local scale, the former threat can be mitigated but the latter cannot without unrealistic expenditure. Other threats that may be prohibitive to mitigate include diseases for which there is no known cure, a volcanic eruption, or an invasive species for which an effective control mechanism has not yet been identified. For other threats there may, in theory, be an action that stops the threat, but the scale and cost of implementation might be prohibitive (e.g., a global reduction of fossil-fuel use to reduce sea-level rise).

Costs

For all known actions there is an associated cost and impact on assets of concern. The cost of each action might reflect the cost of land purchase (Ando *et al*. 1998; Polasky *et al*. 2001), the cost of management (e.g., the cost of controlling an invasive species) (Balmford *et al*. 2003; Wilson *et al*. 2007), opportunity costs (i.e., the profits that are forgone when a conservation action is undertaken) (Naidoo and Iwamura 2007; Carwardine *et al*. 2008a), stewardship costs (i.e., the compensation that a landowner is willing to accept to manage their land for conservation) (Carwardine *et al*. 2008b), social costs (e.g., the number of people that will be disadvantaged by the creation of a protected area (Luck *et al*. 2004), transaction costs (i.e., the costs associated with negotiating an economic exchange or establishing a conservation program) (Naidoo *et al*. 2006), and the costs of information acquisition and planning (e.g., the costs associated with undertaking surveys or conducting prioritization analyses) (Balmford and Gaston 1999; Gardner *et al*. 2008; Grantham *et al*. 2008).

The number of papers presenting economically grounded conservation priority setting approaches has grown (Ando *et al*. 1998; Balmford *et al*. 2000; Polasky *et al*. 2001; Stewart *et al*. 2003; Moore *et al*. 2004; Stewart and Possingham 2005; Wilson *et al*. 2006; Wilson *et al*. 2007; Bode *et al*. 2008; Carwardine *et al*. 2008a,b; Klein *et al*. 2008), as conservation scientists recognize the benefits of explicitly considering the spatially variable costs of conservation. Accounting for the costs of conservation activities is essential for post hoc evaluations of whether certain actions were a cost-effective use of resources (Ferraro and Pattanayak 2006; Halpern *et al*. 2006). There is also increasing evidence of the importance of explicitly incorporating economic data in determining priorities (Bode *et al*. 2008). Studies have shown that conservation plans can be up to 10 times more efficient (Polasky *et al*. 2001; Naidoo *et al*. 2006), meaning that we can make better use of the limited funds available for conservation. Conservation costs typically vary by two to four orders of magnitude more than biodiversity data, and conservation outcomes have been found to be more sensitive to cost data than biodiversity data (Bode *et al*. 2008). Such issues are of primary importance in a dynamic and uncertain world, and are particularly relevant in the context of fluctuating property markets where costs can vary stochastically and unpredictably. This also has important implications for the types of data that are prioritized for collection—data on costs has been identified as an immediate priority (Naidoo *et al*. 2006; Bode *et al*. 2008).

To undertake full-cost accounting we must consider immediate and long-term costs. While the cost of land purchase for a new protected area is incurred immediately, the cost of managing the protected area will continue indefinitely. The simplest approach is to endow management costs in perpetuity and use discounting rates for costs that are incurred over time to reflect the fact that money spent now is worth more than money spent in the future. There are also interdependencies and interactions between the costs of different conservation actions that should be accounted for. Take for example two conservation actions: predator control and fire management. Predator-control management may cost \$$x$/ha, and fire management may cost \$$y$/ha; however, carrying out these actions together may equate to less than \$$x + y$/ha, as transportation and labor costs can be shared. Or to fit with the general version of the conservation prioritization problem [Eqs. (1) and (2)], if we spend \$$y$/ha on fire management and \$$x$/ha on predator control, the benefits are likely to be greater than undertaking them separately.

Overcoming the tendency for costs to be seen as secondary to biological data in conservation priority setting represents a major hurdle for conservation science (Odling-Smee 2005). This hurdle exists because, first, obtaining accurate cost data at an appropriate resolution is challenging and a frequently neglected activity (Naidoo *et al*. 2006). Second, biologically focused conservationists can be hesitant to allow nonbiological factors to influence the allocation of scarce resources, for fear that poorer conservation outcomes would result from "giving up" on species or ecosystems that are expensive to save (Pimm 2000; Bottrill *et al*. 2008). Finally, the general lack of problem definition can be a contributing factor: when locations are prioritized without an action in mind it is impossible to account for the cost of the action.

Most current conservation priority setting does not follow the basic structure outlined in this section. A general tendency is for conservation scientists and practitioners to seek to prioritize locations or assets, without considering the overall objective and the actions required in different locations to conserve the assets. In short, it is common to solve only part of the conservation prioritization problem, because the problem itself is not properly defined.

Prioritizing Assets: How Much Money Should We Invest in the Conservation of Pandas vs. the Conservation of Snails, and Is This the Right Question to Ask?

Much conservation effort is focused on particular species. These species might be keystone, umbrella, flagship, indicator, or focal species (Table 1). Species can make a logical target for conservation investment due to their public appeal (and raising public awareness can increase the pool of funds available for conservation), the availability of reasonable-quality data, and the lowered monitoring and evaluation costs resulting from a focus on fewer assets (Mace *et al*. 2007). It has been acknowledged that setting priorities based on a subset of species can be prone to bias, and the choice of species is problematic because many of the assumed ecological relationships are often untested (Simberloff 1998; Lindenmayer *et al*. 2002) (Table 1). Single-species management also ignores the suite of ecological processes that maintain and sustain such species, and the management of one species may conflict with the management of another (Simberloff 1998).

Many countries, states, and agencies prioritize funds for species recovery by focusing on threat status and prioritizing investment in those species at most risk of extinction. The threat status might be determined by the International Union for Conservation of Nature and Natural Resources (IUCN) red-listing criteria, which use quantitative rules to assign a risk of extinction (IUCN 2003). Such an approach is supported in many countries by legislated requirements for the protection of endangered (and especially charismatic) species and has a strong moral and ethical basis. It also has important implications for the availability and use of funds for biodiversity conservation. In Australia, for example, at a national level, critically endangered and endangered species are given preference for conservation spending (Possingham *et al*. 2002). Although not explicitly stated, this approach assumes that a focus on threatened species will result in the fewest extinctions (Possingham *et al*. 2002; Bottrill *et al*. 2008).

Using a threatened species as a surrogate for other species or habitats may be inappropriate if the presence of a threatened species does not indicate habitat of good condition or that is of high value for other species (Rubinoff 2001; Possingham *et al*. 2002). A sole focus on threatened species may also result in the inefficient use of funds, and an inefficient investment of resources can entail substantial opportunity costs. Imagine, for example, if securing the most endangered species will cost millions of dollars with limited collateral benefits, while several less endangered taxa could be secured by a single, comparatively cheaper conservation action.

An example of a conservation program focused on a single threatened species is that of the California condor, *Gymnogyps californianus*. Over 20 years, US$35 million was invested in the California condor recovery and release program, increasing the number of individuals in the wild from zero to 68 by 2002 (Alagona 2004) to more than 150 in 2008 (Walters *et al*. 2008). Currently US$5 million is spent annually on the project (Walters *et al*. 2008). Most funding for this species' conservation was from California or Oregon-based donors, particularly those focused on "birds of prey" (e.g., Peregrine Fund), along with the United States Fish and Wildlife Service. Given the largely non-fungible nature of the funds available for its conservation, it is likely that this condor program is a cost-effective use of conservation funds. It is also likely that the species would have gone extinct without a dedicated conservation program. However, without evidence to the contrary, we assume that the relative cost-effectiveness of this investment and the associated opportunity costs in terms of species that might have otherwise benefited were not assessed a priori (Walters *et al*. 2008). While the conservation achievements of the California condor project should not be undervalued, it is possible that the dollars allocated to the condor could have ensured the persistence of many more species, particularly if a long-term and global view is taken.

Setting priorities primarily based on how threatened a species is reflects poor problem formulation. An assessment of threat status reveals the urgency of conservation intervention—it does not provide information

TABLE 1. Common Types of Species Used in Setting Conservation Priorities, with a Summary of the Strengths and Weaknesses of Each Approach

Species type	Justification	Key references	An example	Strengths	Weaknesses
Keystone	Changes in the population of the keystone species will have a large impact on other species	(Paine 1969; Power *et al*. 1996)	Cassowary (*Casuarius casuarius*)	Ecological meaning	The identity of keystone species may not be known with certainty and perceived relationships are largely untested
Umbrella	Conservation of the umbrella species will ensure the persistence of other species due to their habitat requirements	(Shrader-Frechette and McCoy 1993; Roberge and Angelstam 2004)	Jaguar (*Tapirus terrestris*)	Ecological meaning.	Perceived relationships are largely untested. The requirements of one species may not reflect that of others
Flagship	Charismatic species that will garner public attention and financial support	(Shrader-Frechette and McCoy 1993)	Giant panda (*Ailuropoda melanoleuca*)	Improved public support for conservation	May not reflect local priorities or values and unlikely to reflect the conservation urgency of other species
Indicator	Changes in the population of these species reflects environmental change	(Landres *et al*. 1988; Noss 1990)	Freshwater mussels	Early warning of environmental change	Unlikely to reveal casual impacts and account for complex ecological interactions
Focal species	Species with high sensitivity to the dominant threats and whose distribution reflects the spatial, compositional, and functional attributes that should be present in the landscape	(Lambeck 1997; Lindenmayer *et al*. 2002)	Gray wolf (*Canis lupis*)	Strategic approach that reflects the conservation context	Choice of species may be biased and data dependent

about the immediate and ongoing cost of conserving the species, the likelihood the conservation effort will work, and the associated benefits to other assets. If, on the other hand, we identify and prioritize actions to abate the key threats to the assets that we value, then we are immediately drawn to a discussion of what those actions protect, what the costs and likelihood of success of these actions are, and whether the actions are socially and politically feasible. Joseph *et al*. (In Press-b) provide a rational approach for prioritizing investments in species conservation (Box 2). With some added complexity, this approach could be used to evaluate the overall benefit of undertaking each action, in terms of the provision of collateral and incidental benefits.

Box 2. Prioritizing Projects for Threatened Species Conservation

Joseph *et al*. (In Press-b) provide a decision theoretic approach for choosing between projects (i.e., bundles of actions required to secure a threatened species) that aim to conserve species in a region. The approach ranks each project (i) by its cost-efficiency (E_i), calculated as the product of the biodiversity benefit (B_i), species value (W_i) and the probability of success (S_i) divided by the cost (C_i):

$$E_i = \frac{B_i \times W_i \times S_i}{C_i}$$

Given a fixed budget for management of threatened species, the top-ranking projects are selected until the budget is expended. Thus, the approach is similar to a maximal-coverage problem (without complementarity) (Box 1); where the set of projects selected maximizes the total number of species secured within a total budgetary constraint (K).

$$\sum_i C_i \leq K$$

This prioritization approach may be applied at any scale. Species values may be economic, social, biological, or political (Faith 1994; Vane-Wright *et al*. 1994; Rodríguez *et al*. 2004; Isaac *et al*. 2007; Marsh *et al*. 2007), or species may be considered equal. The algorithm provides a near perfect greedy solution to the Knapsack problem (Martello and Toth 1990; Weitzman 1998; Hartmann and Steel 2006).

Joseph *et al*. (In Press-a) apply their "Project Prioritization Protocol" (PPP) to explore resource allocation for managing the threatened species of New Zealand. This represents the first real application of cost-effective resource allocation for the management of multiple threatened species.

While the approach is quantitative, rational, and repeatable, it can explicitly explore the subjective elements to conservation priority setting where threatened species are concerned. In the case of prioritizing resources for New Zealand's threatened species, the authors investigated the consequences of valuing species. Species were weighted by their taxonomic distinctiveness and compared with a scenario where species were valued equally. A trade-off exists between funding the management of a greater number of the most cost-efficient and least risky projects and funding fewer projects to manage the species of higher value. Their application of the PPP to 32 of New Zealand's threatened species resulted in far better outcomes (i.e., a greater number of species expected to be secure) than by the commonly used approach of ranking projects by the threat status of species that the projects aim to protect.

Prioritizing Locations: How Do We Identify a Biodiversity Hotspot or a New Protected Area, and Is This the Right Question to Ask?

A number of approaches are available for prioritizing locations to conserve biodiversity (Table 2). These approaches have also leveraged a substantial amount of money for conservation, with biodiversity hotspots alone mobilizing at least US$750 million of funding (Brooks *et al*. 2006). The approaches reviewed in Table 2 are not exhaustive, although they are representative of the range of approaches currently in use.

Locations are typically prioritized using some measure of biodiversity importance (e.g.,

TABLE 2. Different Prioritization Approaches Used to Identify Priority Locations*

Prioritisation approach	Conservation planning principle	Criteria	Key references	How information on vulnerability is used	Characteristics of typical priority locations
Biodiversity hotspots	Irreplaceability and vulnerability	Thresholds for endemism (≥1,500 endemic plant species) and habitat loss (≥70% of original habitat extent has been lost)	(Conservation International 1990; Mittermeier *et al*. 1998, 2004; Myers *et al*. 2000)	(+)	Fragmented
The global 200 ecoregions	Representation	Scoring approach based on species richness, endemism, taxonomic uniqueness, unusual ecological or evolutionary phenomena, and global rarity	(Olson and Dinerstein 1998; Dinerstein *et al*. 2000; Olson and Dinerstein 2002)	(+/−)	Either fragmented or intact
Endemic bird areas	Rarity	Thresholds based on population size: the distributions of two or more restricted-range bird species must overlap	(Stattersfield *et al*. 1998)	NA	Either fragmented or intact
Crisis ecoregions	Vulnerability	Thresholds based on area of habitat converted and area of habitat protected	(Hoekstra *et al*. 2005)	(+)	Fragmented
High biodiversity wilderness areas	Adequacy	Thresholds based on habitat intactness and extent, and human population density	(Mittermeier *et al*. 1998, 2003)	(−)	Intact
Last wild places	Adequacy	Threshold of 10% of each biome least affected by human development	(Sanderson *et al*. 2002)	(−)	Intact
Important bird areas	Rarity	Thresholds based on the population size of globally threatened, restricted-range, biome-restricted migratory, and/or congregatory bird species	(Evans 1994)	(+)	Either fragmented or intact
Alliance for zero extinction sites	Vulnerability	Thresholds based on the occurrence of highly threatened species	(Ricketts *et al*. 2005)	(+)	Either fragmented or intact
Key biodiversity areas	Rarity	Thresholds based on the population size of globally threatened and restricted-range species and congregatory and biome-restricted species	(Eken *et al*. 2004)	(+)	Either fragmented or intact

*Each prioritization approach is summarized in terms of the conservation planning principle employed, the specific criteria used, and how the information on vulnerability is used [(+) vulnerable areas are prioritized, or otherwise (−)]. The characteristics of typical priority locations are also summarized.

irreplaceability, rarity, adequacy) and/or a measure of threat or vulnerability, although different priority locations for conservation investment are typically identified (Redford *et al.* 2003; Brooks *et al.* 2006). Where differences occur they are likely due to the application of different data, analysis at different spatial scales or resolutions, or due to differing institutional values, constraints, and associated objectives.

One important distinguishing factor is whether the approach for prioritizing locations incorporates the principle of complementarity; that is, whether it allows for the selection of areas that complement one another in terms of the assets conserved (Justus and Sarkar 2002). Some approaches evaluate sets of locations on the basis of their collective representation of biodiversity assets such that the whole is not merely the sum of the parts. Hotspot approaches sidestep the issue of complementarity by focusing on endemic species (complementarity is not relevant when there is no overlap in assets between locations), and scoring approaches ignore complementarity meaning that they will be inefficient when protection of multiple assets is sought (Smith and Theberge 1987; Pressey and Nicholls 1989; Possingham *et al.* 2006).

Another important distinguishing factor is the way each approach integrates information on vulnerability (Table 2). At one end of the spectrum lies the identification of endemic bird areas (EBAs), which does not employ information on vulnerability, but focuses on rare or restricted-range species (although, see Caldecott *et al.* 1996). In contrast, some approaches place high importance on areas of both high vulnerability and high biodiversity value (e.g., biodiversity hotspots), and other approaches give priority to areas with low vulnerability and high biodiversity value (e.g., high biodiversity wilderness areas).

Whether vulnerable areas are given preference or avoided depends upon the objective of the priority-setting approach and whether the threat is perceived to be stoppable or unstoppable. For example, the aim of the vulnerability assessment employed in the biodiversity hotspots approach is to highlight the urgency for conservation action, as without rapid action the habitat loss to date in these hotspots threatens the remaining biodiversity (Mittermeier *et al.* 1998). In contrast, the global 200 ecoregions aim to achieve the greatest long-term representation of biodiversity, and hence typically prioritize locations that are not vulnerable (Dinerstein *et al.* 2000; Groves *et al.* 2002), although in some cases high-risk areas might be sought if they are important for representation. Generally, using this approach, each ecoregion is ranked in terms of its suitability for conservation, prioritizing areas with low human use, and conversion of natural land cover on the assumption that these areas will be cheaper to manage and that assets in these areas will likely have a higher probability of persistence (Abell *et al.* 1999; Ricketts *et al.* 1999; The Nature Conservancy 2000; Groves *et al.* 2002). A similar rationale is employed in identifying wilderness areas (Sanderson *et al.* 2002). Future threats are often also analyzed in the ecoregional planning approach to gauge the urgency of conservation actions, but typically after the candidate priority areas have been identified (Olson *et al.* 1998; Dinerstein *et al.* 2000).

While it is clear that there are differences between priority-setting approaches in terms of how information on vulnerability is used, there is no simple rule as to whether favoring or avoiding vulnerable areas provides the best outcomes for biodiversity conservation. There might be logical merit in avoiding vulnerable areas if the probability of success in an area under high threat is low if conservation investment in these areas will stimulate social or political conflict and accelerate further biodiversity loss. Nonetheless, when areas of high biodiversity value are threatened, then vulnerable areas must be allocated high priority in order to avoid biodiversity loss.

Most approaches do not provide a sequencing of locations for conservation investment and simply identify locations as either a priority or not. In these cases, additional decisions are needed to determine the order that each location must be invested in. Another important distinguishing factor between approaches is whether or how the costs of conservation are considered. No approach explicitly accounts for the costs of conservation in each location (Brooks *et al*. 2006), although conservation organizations are actively pursuing methods and data to do so (Murdoch *et al*. 2007; Wilson *et al*. 2007; Bode *et al*. 2008).

The conundrum of how to prioritize investments through time in the face of threats (and also spatially and temporally heterogeneous costs) can be solved by proper problem definition. Using the general framework for setting conservation priorities outlined earlier in the chapter (Eqs. 1 and 2), we can rationally integrate multiple criteria and devise schedules of where and when to invest conservation resources (Costello and Polasky 2004; Meir *et al*. 2004; Drechsler 2005; Wilson *et al*. 2006).

The majority of the prioritization approaches outlined in Table 2 comprise the analysis of scores or the use of thresholds. Scoring systems can be broadly defined as any approach for evaluating or ranking options (assets, locations, actions) based on summing the weighted scores for a number of criteria (e.g., global 200 ecoregions). Multiple Criteria Analysis is a formal scoring approach developed for operations research in the 1940s and has been applied in natural resource management and for the conservation of biodiversity (Fernandes *et al*. 1999; Hajkowicz *et al*. 2000; Villa *et al*. 2002; Mendoza and Martins 2006; Moffett and Sarkar 2006). The approach integrates disparate and competing objectives (see the section on tradeoffs later in this chapter) and captures the preferences of stakeholders by allowing them to control the relative weights given to each criterion. Criteria can be diverse and may include biodiversity importance, fishing value, and ecotourism potential.

There are, however, challenges associated with weighted scoring systems. For example, there can be subjectivity associated with how much weight to be given to different criteria. As a result, scoring systems are susceptible to manipulation by preferentially modifying weights according to user preferences for specific outcomes. Because each option is scored independently of other options, the average reigns—low values in a few criteria can overwhelm high values in a single criterion. Thus, weights are not necessarily reflected in the outcomes. There can also be challenges in combining scores for criteria that have different currencies. Combining the scores for "apples and oranges" to obtain a final overall score can obscure useful information held in the individual scores obtained for the separate criteria.

Thresholds, as opposed to targets, are measured at an individual site level. For example, a threshold may specify that priority locations must have at least three threatened species or be at least 50% intact. Threshold methods, such as biodiversity hotspots (Myers *et al*. 2000), have the advantage of being easy to communicate, but the simplicity comes at a price. If a location meets all but one criterion, then it would fail to be prioritized, and locations that are prioritized because they exceed a threshold by a small degree are considered of equal value to a location that exceeds a threshold by a large amount. Similar to weights in scoring methods, it is difficult to rationally determine the value of thresholds, even though they have such a dramatic effect on the prioritization outcome.

Conservation prioritization, regardless of the approach employed (e.g., ecoregional planning, biodiversity hotspots, etc.), is scale-dependent. Given perfect knowledge, it would always be more efficient to prioritize a range of conservation actions at a fine resolution, even on a global scale. While comprehensive global data sets on biodiversity values, threats, and costs are improving (Hoekstra *et al*. 2005; Orme *et al*. 2005; Ceballos and Ehrlich 2006; Naidoo and Iwamura 2007; Naidoo *et al*. 2008; Underwood *et al*. 2008), data that are available at a global scale are still typically sparse and of varying quality. At local scales, we are more likely to be equipped with data to guide the choice of conservation actions. Rouget *et al*. (2003), for example, analyzed threats at the scale of the entire Cape Floristic Province and the scale of the composite Algulas Plain. The coarser scale analysis identified the broad priorities for conservation, whereas the finer scale assessment allowed for the analysis of more detailed and comprehensive data sets but did not provide an evaluation of the broader conservation context. Therefore the identification of broad priority locations, particularly in data-poor areas, may help refine the geographic extent within which specific actions must be identified, evaluated, prioritized, and applied to suit local conditions.

Prioritizing Actions: Should We Remove Weeds, Buy Land, Plant Trees, or Install Pollution Traps, and Is This the Right Question to Ask?

The creation of protected areas has traditionally been the primary action to achieve conservation outcomes, and is often used implicitly as a surrogate for the broader suite of conservation interventions available to protect biodiversity (Margules and Pressey 2000; Chape *et al*. 2005). Conservation practitioners of course routinely invest in a diverse array of activities such as fire management, invasive-species control, and revegetation, either in protected areas, or on government or privately owned land. In many places land acquisition is not feasible, and neither appropriate nor cost-effective (Pence *et al*. 2003; Stoneham *et al*. 2003; Possingham *et al*. 2006; Carwardine *et al*. 2008a). More recent approaches to conservation priority seek to prioritize between multiple actions (and locations) to achieve conservation objectives. Some approaches are also dynamic, capable of prioritizing actions temporally as well as spatially. In this section we discuss and illustrate by example some approaches to prioritizing multiple conservation actions.

Multiple Zoning of Land and Sea

Much traditional conservation land-use planning has focused on a binary decision-making framework, where the landscape or seascape is allocated to either protected or unprotected status. Realistically, there is a range of land (or sea) uses, which contribute differently to biodiversity conservation and have different costs of implementation (Box 3). We need a lot more information to zone an area for multiple uses. At a minimum, we need to know the contribution that putting each area into each zone will make to the conservation of every asset. This contribution could be asset-specific or constant across all assets. We also need to know the possible zone transitions and the cost of each transition. In more advanced scenarios, there may be desired spatial relationships between zones.

Box 3: Land-Use Zoning in Tropical Forest Regions

Tropical rain forest habitat is used for a diversity of land uses, ranging from protected areas to production forests. Each alternative land use contributes differently to the conservation (or destruction) of biodiversity. Some land uses provide habitat throughout all levels of forest strata, along with a diversity of food sources for fauna species occupying the forest. Other land uses are more restricted in their provision of habitat and food sources with the resultant floristic and faunal diversity reflecting these differences (Meijaard *et al*. 2006; Meijaard and Sheil 2008). Because of the relative sensitivity of species to different land uses, their contribution to the conservation of biodiversity varies (Dutton 2001; Jepson *et al*. 2001; Bengen and Dutton 2004; Curran *et al*. 2004; Meijaard *et al*. 2006; Nakagawa *et al*. 2006; Meijaard and Sheil 2007; Wells *et al*. 2007).

This situation presents a variety of challenges. The suite of possible land uses is complex and the control variable is not binary. Furthermore there may be constraints, upper and lower, on how much land that can put into different land uses, such as in protected areas or oil palm plantations. We can, however, examine different land allocation scenarios by accounting for these constraints, and the relative costs and benefits of different land uses. First, we need to know the relative sensitivity of our biodiversity assets to forest conversion, and then assign targets that accounts for this. We can then determine the contribution of each land-use zone to achieving the targets for each asset. With such information we can evaluate not only where to act but how to act in order to effectively and efficiently conserve biodiversity.

Conservation Action Planning

Conservation Action Planning (CAP) is one part of the *Conservation by Design* process developed by The Nature Conservancy (The Nature Conservancy 2000, 2003). It was developed as a stakeholder-driven planning approach for landscape-scale projects, but has also been used at regional and local scales. The approach is designed to be iterative and adaptive, recognizing that information is always incomplete, but should improve over the life of a project. The CAP approach is based on "The Five S's."

- *Systems*: The conservation assets in a landscape and the key ecological attributes that maintain their viability (generally, 6–8 assets are employed).
- *Stresses*: The types of destruction or degradation impacting each conservation asset.
- *Sources*: The agents generating the stresses.
- *Strategies*: The types of conservation actions that can be deployed to abate sources of stress.
- *Success*: Measures of biodiversity health and threat abatement.

The aim of the CAP approach is to identify actions that maintain healthy, viable occurrences of the conservation targets. By definition, healthy occurrences are not significantly stressed, and measures of success can include indicators of the effectiveness of conservation strategies and indicators of target viability.

The CAP process has the benefits of being stakeholder driven and iterative. The approach accounts for the key elements of a conservation prioritization problem and reflects a project-management tool that is a mixture of a focal species approach and a static multiple-action prioritization approach. The process encourages consideration and articulation of what conservation success in a landscape would look like and how progress toward this goal will be measured. A key limitation of the approach is that the consideration of the cost and likelihood of success of each action are not an integrated and explicit part of the planning process, and therefore have little influence on the initial identification and prioritization of conservation actions.

Dynamic Multiple-Action Prioritization

In dynamic versions of the multiple-action conservation prioritization problem our management decision is how much of our budget to allocate to each conservation action at each time step. For each conservation action we need to know what it costs per unit area and its benefit to the relevant biodiversity assets (Table 3; Box 4). We also need to know the current investment in the conservation action and the remaining area that would benefit from further investment in the action. We can then generate dynamic investment schedules that reflect shifts in the allocation of funds as the return from investing in each conservation action diminishes. The investment schedule is determined by the interplay of three main factors: (1) the relationship between the additional area invested in each conservation action and benefit to biodiversity, (2) the cost of this investment, and (3) the existing level of investment. We can then seek to prioritize actions with the greatest benefit per dollar invested (i.e., to maximize gains), or if the rate of habitat loss is known, we can prioritize actions that will minimize the expected loss of biodiversity (i.e., to minimize losses) (Sarkar *et al*. 2006; Wilson *et al*. 2006; Murdoch *et al*. 2007).

TABLE 3. Examples of Three Conservation Actions Identified to Abate Key Threats to Biodiversity in Two Australian Ecoregions*

Conservation action and ecoregion	Invasive predator control		*Phytophthora cinnamomi* management		Revegetation	
	Eyre and York mallee	Mount Lofty woodlands	Eyre and York mallee	Mount Lofty woodlands	Eyre and York mallee	Mount Lofty woodlands
Number of assets protected	514	517	251	255	153	157
Cost of the action (US$/km^2)	301,154	301,005	514,592	514,443	7,091	6,942
Total area of the ecoregion (km^2)	60,896	23,786	60,896	23,786	60,896	23,786
Area already receiving action (km^2)	18,926	4,753	0	0	416	1,221
Area requiring action (km^2)	6,295	2,855	12,791	4,198	18,510	3,532

*The number of assets estimated to be protected by the action (i.e., the number of species sensitive to the threats that each action will abate) and the cost of investing in each action is provided, along with estimates of the area already receiving the action and the additional area requiring the action.

Multiple-action conservation prioritization has been shown to outperform more traditional approaches to conservation funding allocation that focus solely on protected-area establishment (Box 4). Although fully comprehensive studies are desired, multiple-action prioritization analyses suffer from the same limitations as other prioritization analyses discussed in this review. In particular, data and computational constraints often deem it prohibitive to fully account for complexities such as the temporal and spatial heterogeneity associated with information on costs, threats, and benefits, not to mention the stochastic nature of environmental and socioeconomic factors (Wilson *et al.* 2006, 2007).

Multiple-action prioritization problems can be potentially very complex. For example, there can be spatial dependencies and the impact (and cost) of actions will likely vary between assets. The change in value from investment in different actions depends on the relationship used to describe the objective function; common forms include threshold, linear, sigmoid, or concave (Fig. 1). Van Teeffelen and Moilanen (2008) evaluate the consequences of using different functional forms for the objective function. A linear relationship provides a continuous accumulation of benefit, whereas a threshold function reflects an assumption that the benefit peaks once the threshold is reached. A concave objective function reflects a situation of diminishing marginal gains. The sigmoid objective function reflects an assumption that low levels of investment will deliver low returns and that the returns increase rapidly after some amount of investment is reached and then tapers off.

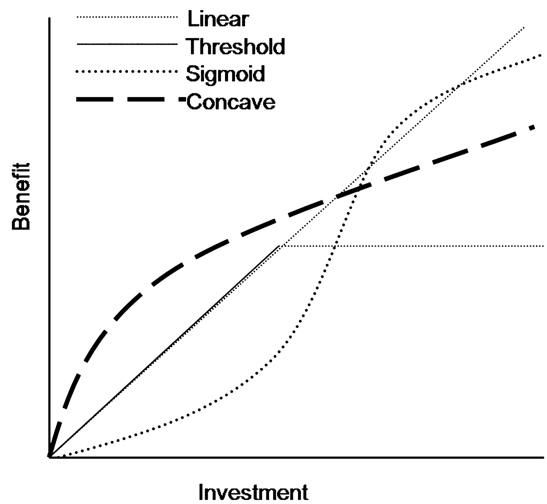

Figure 1. Four possible objective-function shapes: linear, threshold, sigmoid, and concave.

Box 4. Maximizing the Conservation of Biodiversity Through Investment in a Range of Conservation Actions Through Time: A Case Study from Mediterranean Ecoregions

Wilson *et al*. (2007) developed a framework for guiding the allocation of funds among alternative conservation actions that address specific threats and applied it to 17 of the world's 39 Mediterranean climate ecoregions. This subset of Mediterranean ecoregions covers parts of Australia (ten ecoregions), Chile (one ecoregion), South Africa (three ecoregions), and California and Baja California (three ecoregions). In total, they evaluated 51 ecoregion–conservation action combinations which were termed "ecoactions." In Australia, for example, ten Australian Mediterranean ecoregions were analyzed and three primary threats were identified: habitat fragmentation and salinization (with salinization the result of altered hydrological regimes); introduced predators (focusing on cats [*Felis catus*] and foxes [*Vulpes vulpes*]); and the soil-borne pseudofungus *Phytophthora cinnamomi*. The mitigating actions selected were (for each respective threat) strategic revegetation, invasive predator control and research, and a mix of precautionary and preventive measures, including the application of phosphite, quarantine, education, research, and communication. For each of these threats several pieces of information were obtained (Table 3).

Wilson *et al*. (2007) assumed that the marginal benefit decreases for each new unit area receiving investment, and therefore, the benefit diminishes with cumulative investment. This relationship was modeled using species-area curves, which were converted into species-investment curves. Only 24 ecoactions (of the 51 possible) received investment during the first five years. A mix of land protection and off-reserve management in South Africa was highly rated, along with invasive-plant control in Chile, California, and South Africa. These conservation actions yielded the greatest marginal return on investment over five years because the biodiversity benefits were high and the costs were comparatively low. However, this does not involve simply prioritizing the cheapest actions, or the most species-rich ecoregions; rather, this information is integrated to find the most cost-effective combinations.

The ecoaction-specific framework was then compared to a model of conservation that focuses only on land acquisition. Over five years almost four times as many species can be protected using the ecoaction approach (2780 vs. 703 species). These results suggest that investing in a sequence of conservation actions targeted toward specific threats, such as invasive-species control and fire management, will deliver greater biodiversity returns than by relying solely on acquiring land for protected areas.

We can also incorporate complementarity within this dynamic multiple-action prioritization framework. Complementarity is important if we aim to minimize the cost of the conservation action under consideration, ensure that all biodiversity assets (e.g., species, vegetation types) receive some level of conservation investment, and identify location-and-action combinations that complement those that have already received investment (Justus and Sarkar 2002). To apply complementarity we require information on the specific biodiversity content of locations, as opposed to summary statistics such as species richness. Furthermore, locations and actions must be evaluated collectively rather than independently.

Our objective when accounting for the principle of complementarity is to protect as many different, or "distinct," biodiversity assets as possible. As described earlier, the shape of our objective function reflects the functional relationship between assets protected and the level of investment. In the case of two actions A and B, we would consider not only an "A curve' and a "B curve' for assets only benefited by actions A and B, respectively, but also an "AB curve' for assets benefited by both actions A and B (Fig. 2). For three or more actions, we would consider a total of seven curves: A, B, C, AB, AC, BC, and ABC. The number of curves thus increases combinatorially as the number of actions increases. Using complementarity favors actions where for each dollar invested, the sum of the additional assets benefitted is greatest. Some actions might require other actions in order to

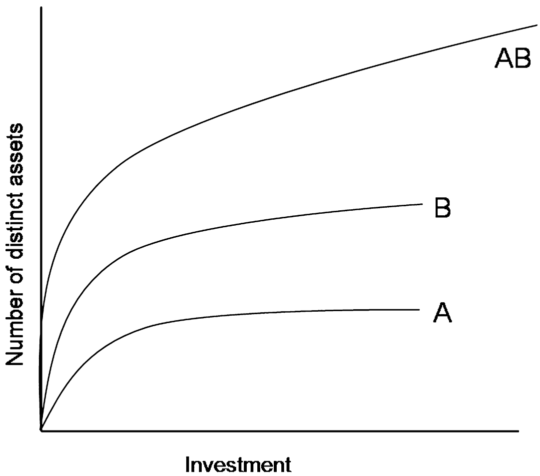

Figure 2. A hypothetical example of the curves required to account for complementarity between two actions (*A* and *B*). Curve *A* and curve *B* represent the number of distinct assets benefited by investing in action *A* and *B*, and curve *AB* represents overlapping assets benefited by investing in both actions *A* and *B* (Underwood *et al.* 2008).

be successful, and there can be negative interactions between actions if the action required for one species is detrimental to the conservation goals of another. Not accounting for these forms of complementarity will overestimate the conservation benefit of a set of actions.

Exploring Trade-Offs and Dealing with Multiple Objectives

A trade-off exists in conservation priority setting under two conditions: when two or more assets are not perfectly aligned, and/or when the budget or some other constraint prevents us from reaching our targets for all biodiversity assets. For example, we may be unable to immediately conserve adequate representations of each threatened species whose distributions do not overlap. Under such circumstances we have a trade-off between those targets that we meet now and those we may attempt to meet later. Trade-offs are unavoidable in priority setting when there are multiple considerations, which is almost always the case. Even in a simplistic prioritization problem involving only the identification of potential protected areas, protection of multiple assets will likely be sought and the implementation of protected areas will likely be incremental. We therefore need to make decisions about which areas should be protected first. Trade-offs can become even more complex when benefits other than biodiversity conservation are sought, such as the delivery of ecosystem services or the protection of sites of cultural significance.

In the case of ecosystem service conservation, we can assess the spatial and temporal congruence of ecosystem services and biodiversity assets. If such an analysis shows that areas of high biodiversity and ecosystem service value coincide, then conservation intervention will simultaneously benefit both (a “win-win” situation). Concordance between ecosystem service delivery and biodiversity conservation has been found at a variety of scales, suggesting that such opportunities exist (Turner *et al.* 2007). However, other analyses have found limited spatial congruence between important areas for ecosystem services delivery and biodiversity conservation (Chan *et al.* 2006; Naidoo *et al.* 2006). There are several reasons for such outcomes. It is likely that the provision of ecosystem services is best achieved by land-use options that are suboptimal for biodiversity conservation (for example, the growth and harvest of cereal crops may be more efficient in relatively species-poor ecosystems) (Cassman and Wood 2005). Many forms of biodiversity do not offer known benefit to humans, and some biodiversity is even detrimental to humans. In areas that are heavily cleared, habitat loss and fragmentation would reduce the ecosystem service value of the land, while increasing the biodiversity value of the remaining habitat (Turner *et al.* 2007). Finally, while patterns of richness in different assets may coincide, patterns of complementarity may not. We therefore need to explicitly account for the possibility that synergies between conservation and ecosystem services will not exist. The mathematical integration of multiple objectives depends on correct problem formulation, and again there is a distinction between maximal-coverage problems and minimum-set problems (Box 1).

In maximal-coverage problems actions are selected that maximize some benefit without exceeding a conservation budget. When there are multiple objectives, we can modify the relative weighting of each objective: for example, the conservation of biodiversity and the provision of ecosystem services. In effect, the overall objective becomes the selection of actions that optimize the weighted sum off these subobjectives. Weighting objectives can involve some level of arbitrariness, as previously discussed, but also provides flexibility to explore trade-offs by investigating a range of weights (Arthur *et al*. 2004).

A trade-off curve can be helpful in priority setting for multiple objectives, by highlighting sets of options that achieve different amounts of each objective. When we vary the relative importance of protecting biodiversity versus protecting ecosystem services through its full spectrum, we define a "Pareto-frontier": the best landscape outcomes achievable for any two-criteria optimization (Nalle *et al*. 2004). An example of this frontier is shown in Figure 3. The highest and lowest points in this figure correspond to landscapes planned exclusively for ecosystem services and biodiversity, respectively, with the points in between representing compromises across both objectives. When aiming to achieve two disparate objectives, any set of actions that exists inside the Pareto-optimal line is suboptimal.

In minimum-set problems actions are selected that meet conservation targets at a minimal overall cost. We can include targets for a virtually unlimited number of assets. In this type of problem formulation the achievement of targets for each asset is a kind of subobjective, where the overall objective is to meet all targets. For example, Chan *et al*. (2006) prioritized places that protect variable targets for a set of assets, and that store at least 50% of the carbon within the entire region. This target-based approach has the benefit of avoiding the need for arbitrary weights (although targets can involve arbitrary elements) and of identifying sets of priorities that can be evaluated against quantitative targets. The target-based approach itself does not explicitly address trade-offs, but trade-offs can be investigated by varying the targets and evaluating the difference in the overall cost (Egoh *et al*. 2007).

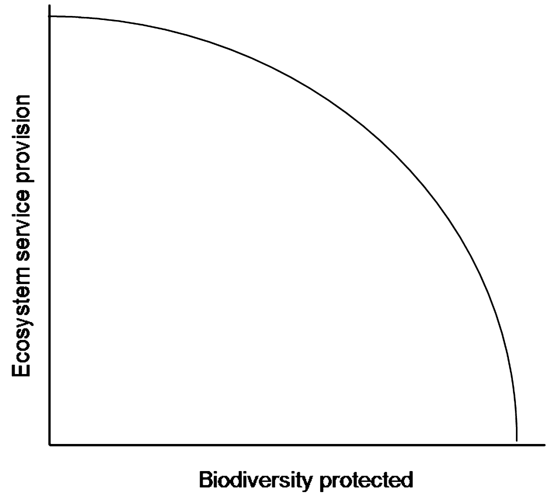

Figure 3. Pareto-frontier for biodiversity and ecosystem service protection for a hypothetical landscape.

How Can We Better Set Conservation Priorities?

Account for the Sociopolitical–Economic–Ecological Context

The probability of success of different conservation actions is influenced by numerous sociopolitical factors, including political stability and corruption, budget continuity, governance, and stakeholder willingness to be involved in conservation initiatives (Barrett *et al*. 2001; Smith *et al*. 2003; Knight and Cowling 2007). Although uncommon, we can account for the probability that a conservation action will succeed by modifying the expected biodiversity benefit (Hobbs and Kristjanson 2003; McBride *et al*. 2007; Joseph *et al*. In Press-b).

In global and even regional problems, there can be much variation in the political and economic systems and in societal needs, which affect the suitability of different conservation actions. Inadequate handling of such issues has led to conservation failures. For example, the implementation of the Natura 2000 network in Finland was problematic due to inadequate consideration of the needs and desires of landholders in the prioritization process, and the availability of sufficient funds to compensate landholders for the loss of land rights (Hiedanpää 2002). Such an example points to the importance of ensuring that the conservation instruments employed are suited to the local context.

Market-based instruments take advantage of market forces and social responses and are often an efficient conservation prioritization tool. Reverse auctions is a market-based instrument that involves the efficient selection of "bids" placed by landowners for the financial compensation they will accept to manage their land for conservation outcomes (Table 2). Reverse auctions have been carried out successfully in the United States (Reichelderfer and Boggess 1988) and in parts of Australia (Grieve and Uebel 2003; Stoneham *et al*. 2003; Hajkowicz *et al*. 2007). Reverse auctions differ from more traditional priority-setting approaches because they only consider parcels of land that are available for conservation intervention (only amenable landholders will place bids), and they elicit real costs, thus overcoming the difficulty of predicting the costs of conservation. Reverse auctions are, however, unable to assess the potential contribution of all parcels of land to an overall conservation goal and are not suitable for strategic planning over large regions.

It is also important to consider the ecological context and the extent that an asset is likely to be benefited by a particular action. In some cases, an action implemented alone is unlikely to achieve the conservation goal and the interdependencies between actions must be considered. A simple example is the purchase and management of land for the establishment of protected areas. If the key threat facing an area is native habitat loss due to conversion to another land use, then purchase of the land might abate this immediate threat. However, if the biodiversity values of the land are also threatened by weed invasion and habitat degradation, then land purchase without accounting for the costs of restoration and management to ensure the persistence and health of the biodiversity it contains will be insufficient. In such cases the costs of all required conservation actions (acquisition and management) and their relative benefits should be considered from the outset.

Explicitly Account for Uncertainty and Risk

There are many forms of uncertainty associated with biodiversity conservation. These include the likelihood of success of conservation actions, uncertainty about our objectives (have we been asking the right question?), and uncertainty associated with the data used to parameterize our analyses. Uncertainty (randomness with unknowable probabilities) differs from risk (randomness with knowable probabilities) (Knight 1921).

First, we can assess whether the uncertainty in a particular parameter will be naturally reduced or resolved during the process of implementing the conservation action. For example, there is often uncertainty associated with both the cost of purchasing a parcel of land and whether the assets we wish to conserve are present at the site. However, before establishing a protected area, we will be exposed to updated information about the land. For example, we will know the price at which a landowner is willing to sell and we are likely to undertake a biological survey to ensure that the land contains the assets we wish to conserve. For these kinds of uncertain parameters an adaptive approach is essential, where priorities are updated as new information becomes available (see below).

If the uncertainty will not be naturally resolved, we might aim to reduce the uncertainty by collecting more information, for example, through improving our knowledge of the system and its dynamics (Haight *et al*. 2005; Armsworth *et al*. 2006). However, obtaining more information can come at a cost, both financially and through opportunities that might be lost as we acquire further information. For example, it has been argued that new biodiversity surveys lead to more efficient conservation plans and therefore the surveys represent good investments (Balmford and Gaston 1999). Grantham (2008) found diminishing returns on surveys of Protea species in South Africa, however, and identified thresholds beyond which further investments in surveys would not deliver more effective protection of Protea species (despite a much better understanding of the distribution of the species). This study indicates the importance of accounting for the return on investments in different data, the benefits the additional data deliver to our conservation outcomes, and the opportunity costs associated with the investments.

We can also explore the range of possible values for an uncertain parameter, noting how decisions may change depending upon the parameter value. Such sensitivity analyses can be useful to identify the parameters that have the greatest impact on the outcomes, and determine which parameters to collect more information for, and thus reduce uncertainty. However, in many situations, risks and uncertainty cannot be removed.

We might aim to set conservation priorities that are robust to risk and uncertainty (Ben-Haim 2001; Nicholson and Possingham 2007). Here we need to know (or estimate) the likelihood that an unplanned but conservation-relevant event may occur, such as the risk of a hurricane, fire, or coral bleaching event, or the risk that a conservation action will not be carried out correctly (the inverse of its likelihood of success). We can then either prioritize actions (or locations to carry out an action) that meet conservation targets while minimizing some combination of risk and cost (yet another trade-off) (Game *et al*. 2008), or prioritize actions that maximize the expected or likely conservation benefits for a fixed budget (Joseph *et al*. In Press-b). Note that these solutions represent modifications of Equations (1) and (2), respectively.

Allow for Opportunities and Feedbacks

Like all actions, conservation actions create reactions and these reactions can be in the form of potentially beneficial opportunities or negative feedbacks. In some cases, opportunities and feedbacks can be predicted, particularly where there is a sound knowledge of the system or we have been able to learn from past experiences. Where possible, predictions of both positive and negative effects of conservation actions should be explicitly incorporated into the priority setting processes.

Establishing a protected area in a region can have positive effects. For example, the livelihood of residents might be improved through the provision of opportunities for ecotourism ventures, which can potentially reduce the dependence of local communities on natural resources. Successful conservation outcomes in a region can also be the key that unlocks further funds for conservation. In order to account for such leverage, we can modify Equation (1) to include an increased budget to represent additional funds, and we can modify Equation (2) to include the additional benefits that have been gained.

Negative feedbacks can also occur. For example, the purchase of properties for a new protected area may drive land prices up due to increased demand for land for other purposes. This can have perverse effects on local communities, and also make it more expensive to establish additional protected areas (Costello and Polasky 2004). The protection of a particular parcel of land because it is biologically valuable and also threatened by conversion, may simply displace the threat to nearby areas. There exist approaches to predicting and incorporating negative feedbacks in dynamic conservation planning situations (Armsworth *et al*. 2006). Whenever a parcel of land is purchased, the relative price of other parcels of land is altered using knowledge of the dynamics of the local market. Likewise, the likelihood of success can be updated in a similar manner.

It is important to acknowledge that the economic and sociopolitical reaction of systems and people to conservation cannot always be predicted. Unforeseen reactions can occur, leading to perverse conservation and or social outcomes. Hence, rather than being top-down and inflexible, conservation priority setting should be modest and flexible to allow for adaptation to or mitigation of perverse outcomes. The financial and ecological impact of unforeseen opportunities or reactions should be explicitly accounted for as they arise and assessed within the prioritization framework. We can then learn from such evaluations to better respond to opportunities and negative feedbacks in the future.

Evaluate, Learn, and Improve

There has been growing concern among the conservation community surrounding the limited availability of evidence that investments have achieved sound conservation outcomes (Ferraro and Pattanayak 2006). This situation reflects a general tendency for only successful conservation stories to be reported (Redford and Taber 2000; Knight 2006) and the limited resources allocated to monitoring and evaluation (Field *et al*. 2007). This is problematic, since a lack of evaluation and honest reporting means that the process of learning and improvement is hindered and past mistakes may be repeated. There is a clear need for better evaluation procedures, a systematic process for learning from past successes and failures, improved communication to extend this knowledge, and adaptive priority setting.

Adaptive priority setting is a relatively simple process, requiring the fluid updating of priorities and priority-setting approaches in response to new information as it becomes available (Higgins and Duane 2008). Again flexibility is essential, both in the mentality of decision-makers and the tools deployed. Broadly speaking, there are three approaches to adaptive management: passive adaptive management, where we simply learn by analyzing the outcomes of past interventions (this is the most common approach); experimental adaptive management, where we actively manipulate the system to maximize how fast we learn; and active adaptive management, where we optimally manage taking into account the possibility for learning. The latter is optimal, but technically complex (Walters 1986; McCarthy and Possingham 2007; Hauser and Possingham 2008). A properly formulated active adaptive management problem leads to solutions to conservation problems that properly integrate managing, monitoring, and learning, and hence leads to a reduction of uncertainty.

Conclusions

A broad range of priority-setting approaches is available, matching the myriad types of conservation decisions faced by conservation scientists and practitioners. By distilling conservation prioritization problems into the components of age-old decision theory, all problems become comprehensible and solvable. While acknowledging the variety and complexity of conservation problems, we have presented a framework and summarized general rules that can help to guide most conservation prioritization tasks. First, problem formulation is critical. Without a clear definition of goals, as well as the identification of actions and their costs and likely benefits, decisions are unlikely to be cost-effective, and outcomes cannot be evaluated. Second, considering the conservation actions to be undertaken in a particular location to protect key assets is the most comprehensive form of conservation prioritization. Third, conservation decisionmakers need not shy away from problems with multiple objectives, or those that are prone to uncertainty and risk—rather, this information can be explicitly included in the priority-setting process to improve conservation decision-making. And finally, an adaptable and flexible approach to priority setting is essential to ensure we can effectively respond to opportunities, feedbacks, and improved knowledge. The principles and examples we present should only be applied with flexibility, humility, and a sound knowledge of the socioecological system.

Acknowledgments

The authors acknowledge funding from the Australian Research Council, Conservation International, The Nature Conservancy, the Applied Environmental Decision Analysis Commonwealth Environmental Research Facility, the Australian Department of Environment, Water, Heritage and the Arts, and the Commonwealth Scientific and Industrial Research Organization (CSIRO). We are also grateful for many stimulating conversations and discussions with collaborators and colleagues from around the world that have provided a diverse array of theories, planning approaches, data, and applications on which we could base this review.

Conflicts of Interest

The authors declare no conflicts of interest.

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