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Prioritizing choices in conservation

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The last word in ignorance is the man who says of an animal or plant: 'What good is it?' If the land mechanism as a whole is good, then every part is good, whether we understand it or not. If the biota, in the course of aeons, has built something we like but do not understand, then who but a fool would discard seemingly useless parts? To keep every cog and wheel is the first precaution of intelligent tinkering.
(Aldo Leopold, *Round River*, Oxford University Press, New York, 1993, pp. 145–6.)

Introduction

We are in the midst of a mass extinction in which at least 10%, and may be as much as 50%, of the world's biodiversity may disappear over the next few hundred years. Conservation practitioners face the dilemma that the cost of maintaining global biodiversity far exceeds the available financial and human resources. Estimates suggest that in the late twentieth century only US$6 billion per year was spent globally on protecting biodiversity (James et al. 1999), even though an estimated US$33 trillion per year of direct and indirect benefits were derived from ecosystem services provided by biodiversity, implying an asset worth US$330 trillion (Costanza et al. 1997). Together these crude estimates suggest that there could be a 500-fold underinvestment in conserving the world's biodiversity. However, even if these estimates are wildly wrong, the imbalance of funding is seriously inconsistent with best business practice in other sectors. In business, many companies spend about 10% of the value of their capital assets each year on maintaining those assets, although the figure varies depending on the type of asset. For example, 30% might be spent for computers compared with 5% for buildings: contrast that with 0.02% for biodiversity! Furthermore, the scale of underinvestment in biodiversity may be exaggerated by the effects of poor governance, sometimes even corruption, on achieving success in conservation (Smith et al. 2003). Given such problems, conservation scientists and non-government organizations (NGOs) supporting international conservation efforts are beginning to develop systems to more effectively target investment in biodiversity conservation (Johnson 1995; Kershaw et al. 1995; Olson & Dinerstein 1998; Myers et al. 2000; Possingham et al. 2001; Wilson et al. in press).

One fundamental resource allocation question facing conservation scientists and practitioners is whether conservation goals are best met by managing single species as opposed to whole ecosystems (Simberloff 1998). Efforts in conservation priority setting have historically concentrated on ecosystem-based priorities – determining where and when to acquire protected areas (Ferrier et al. 2000; Margules & Pressey 2000; Pressey & Taffs 2001; Meir et al. 2004). There has been comparatively little work on the question of how to allocate conservation effort between species. Despite the tension between ecosystem-based and species-based conservation, we believe there is merit in considering the issue of resource allocation between species because:

1. a 'fuzzy' idea such as ecosystem management holds little appeal for the general public, who prefer to grasp simpler messages conveyed by charismatic species such as tigers (Leader-Williams & Dublin 2000);
2. data on species, whether through direct counts of indicator species (Heywood 1995), or through assessments of threat (Butchart et al. 2005), provide some of the most readily available, repeatable and explicit monitoring and analytic systems with which to assess the success or otherwise of conservation efforts (Balmford et al. 2005).
3. in practice, almost regardless of their ultimate goal, conservation bodies often end up directing conservation actions to species and species communities (see e.g. figure 1 of Redford et al. 2003), probably because these are tangible and manageable components of ecosystems.

The topic of setting priorities for conservation is immense, so here we restrict ourselves to different methods for setting priorities between species. We explore the issues that a systematic approach should consider, and we show how simple scoring systems may lead to unintended consequences. We also recommend an explicit discussion of attributes of the species that make them desirable targets for conservation effort. Using a case study, we show how different perspectives will affect the outcome, and so as an alternative we present a method based on economic optimization. Ultimately, any decisions about 'what to save first' should include judgments that cannot be made by scientists or managers alone. Involving wider societal and political decision-making processes is vital to gain local support for, and ensure the ultimate success of, all conservation planning.

Single species approaches

Species-based conservation management approaches have, until fairly recently, concentrated on a single species, such as keystone species, umbrella species, indicator species or flagship species (see Leader-Williams & Dublin 2000). Keystone and umbrella species differ in the importance of their ecological role in an ecosystem:

1. **keystone species** have a disproportionate effect on their ecosystem, due to their size or activity, and any change in their population will have correspondingly large effects on their ecosystem (e.g. the sole fruit disperser of many species of tree);
2. **umbrella species** have such demanding habitat and/or area requirements that, if we can conserve enough land to ensure their viability, the viability of smaller and more abundant species is almost guaranteed.

In contrast, 'flagship species' encompass purely strategic objectives:

3. **flagship species** are chosen strategically to raise public awareness or financial support for conservation action.

Furthermore, definitions for indicator species can encompass both ecological and strategic roles:

4. **indicator species** are intended either to represent community composition or to reflect environmental change. With respect to the latter, indicator species must respond to the particular environmental change of con-

cern and demonstrate that change when monitored.

One species may be a priority species for more than one reason, depending on the situation or context in which the term is used. However, the terms 'keystone' and 'umbrella' are likely to remain more of a fixed characteristic or property of that species. In contrast, the term 'flagship' and, possibly to a lesser extent, 'indicator' may be more context-specific.

Promoting the conservation of a specific focal species may help to identify potential areas for conservation that satisfy the needs of other species and species assemblages (Leader-Williams & Dublin 2000). For example, the umbrella species concept (Simberloff 1998) can represent an efficient first step to protect other species. In addition, minimizing the number of species that must be monitored once a protected area has been created will reduce the time and money that must be devoted to its maintenance (Berger 1997).

Alternatively, conservation managers and international NGOs may choose to focus on the most charismatic 'flagship' species, which stimulate public support for conservation action, and that in turn may have spin-off benefits for other species. For example, use of the giant panda (*Ailuropoda melanoleuca*) as a logo by the World Wildlife Fund (WWF) has been widely accepted (Dietz et al. 1994) as a successful mechanism for conserving many other species across a wide variety of taxonomic groups. Furthermore, other mammalian and avian 'flagships' have been used to promote the conservation of large natural ecosystems (i.e. Mittermeier 1986; Goldspink et al. 1998; Downer 1996; Johnsingh & Joshua 1994; Western 1987; Dietz et al. 1994).

Nevertheless, the context of what may constitute a charismatic species can differ widely across stakeholders. For example, the tiger (*Panthera tigris*) is among the most popular flagship species in developed countries, but those in developing countries who suffer loss of life and livelihood because of tigers or other large predators have a different view (Leader-Williams & Dublin 2000). Such dissonance is best avoided by promoting locally supported flagship species (Entwistle 2000). For example, the discovery of a new species of an uncharismatic, but virus-resistant, wild maize, with its possible utilitarian value for human food production, highlighted the conservation value of the Mexican mountains in which it was found (Iltis 1988). This increase in local public awareness led to the establishment of a protected area that conserves parrots and jaguars (*Panthera onca*), orchids and ocelots (*Leopardus pardalis*), species that many consider charismatic. Hence this species of wild maize served as a strategically astute local flagship species. Another way of promoting local flagships is to prioritize those species that bring significant and obvious local benefits (Goodwin & Leader-Williams 2000), such as the Komodo dragon, *Varanus komodoensis* (Walpole & Leader-Williams 2001), which generates tourism. Similarly, species that can be hunted for sport, such as the African elephants (*Loxodonta africana*), may contribute directly to community conservation programmes (Bond 1994).

Several questions can arise from promoting conservation through single species (Simberloff 1998). One of these is how should individual species be prioritized? The common response is to begin with species that are most at risk of extinction, the critically endangered species. Many countries and agencies take this approach. However, there may be no known management for some of these species, and if there is, it may be risky and/or expensive. This can lead to a large share of limited conservation resources being expended with negligible or uncertain benefit (Possingham et al. 2002). On the other hand, when taking an ecosystem approach, managers might choose to focus on the keystone species that play the most significant role in the ecosystem. Unfortunately in many ecosystems we do not know the identity of keystone species. Often, after intensive study, they turn out to be invertebrates or fungi (Paine 1995), groups that are unlikely to attract public or government support unless ways can be found to make them locally relevant.

Another problem with single species conservation arises when the management of one focal species is detrimental to the management of another focal species. For example, in the Everglades of Florida, management plans for two charismatic, federally listed birds are in conflict. One species, the Everglades snail kite (*Rostrhamus sociabilis plumbeus)*, has been reduced to some 600 individuals by wetland degradation and agricultural and residential development. It feeds almost exclusively on freshwater snails of the genus *Pomacea* and requires high water levels, which increase snail production. The snail kite is thus an extreme habitat specialist (Ehrlich et al. 1992). The other species, the wood stork *Mycteria americana*, has been reduced to about 10,000 pairs by swamp drainage, habitat modification and altered water regimes. Ironically, the US Fish and Wildlife Service opposed a proposal by the Everglades National Park to modify water flow to improve stork habitat on the grounds that the change would be detrimental to the kite (Ehrlich et al. 1992) (an added thought-provoking detail is that both species are common in South America).

Another issue is that few studies have been carried out to assess the effectiveness of one focal species in adequately protecting viable populations of other species (Caro et al. 2004). For example, the umbrella-species concept is often applied in management yet rarely tested. The grizzly bear (*Ursus arctos*) has been recognized as an umbrella species but, had a proposed conservation plan for the grizzly bear in Idaho been implemented, taxa such as reptiles would have been underrepresented (Noss et al. 1996). Similarly, in a smaller scale study, the areas where flagship species, such as jaguar, tapir (*Tapirus terrestris*) and white-lipped peccary (*Tayassu pecari*), were most commonly seen did not coincide with areas of vertebrate species richness or abundance (Caro et al. 2004). Although these results may not hold true for all other protected areas based around flagship species, it does highlight the need for more field-based studies to determine the most appropriate approach for conserving the most biodiversity. As a result of problems associated with single species management, focus has been turning towards multiple species approaches.

Multispecies approaches

Methods based on several focal species, or protecting a specific habitat type, might be a more appropriate means of prioritizing protected areas (Lambeck 1997; Fleishman et al. 2000; Sanderson et al. 2002b). A frequent criticism of setting conservation priorities based on a single focal species is that it is improbable that the requirements of one species would encapsulate those of all other species (Noss et al. 1996; Basset et al. 2000; Hess & King 2002; Lindenmayer et al. 2002). Hence, there is a need for multispecies strategies to broaden the coverage of the protective umbrella (e.g. Miller et al. 1999; Fleishman et al. 2000, 2001; Carroll et al. 2001).

Among the different multispecies approaches, Lambeck's (1997) 'focal species' approach seems the most promising because it provides a systematic procedure for selecting several focal species (Lambeck 1997; Watson et al. 2001; Bani et al. 2002; Brooker 2002; Hess & King 2002). In Lambeck's (1997) innovative approach, a suite of focal species are identified and used to define the spatial, compositional and functional attributes that must be present in a landscape. The process involves identifying the main threats to biodiversity and selecting the species that is most sensitive to each threat. The requirements of this small and manageable suite of focal species guide conservation actions. The approach was extended by Sanderson et al. (2002a), who proposed the 'landscape species approach'. They defined landscape species by their 'use of large, ecologically diverse areas and their impacts on the structure and function of natural ecosystems . . . their requirements in time and space make them particularly susceptible to human alteration and use of wild landscapes'. Because landscape species require large, wild areas, they could potentially serve an umbrella function (*sensu* Caro & O'Doherty 1999) – meeting their needs would provide substantial protection for the species with which they co-occur. Like other focal-species approaches, this method of setting priorities carries inherent biases (Lindenmayer et al. 2002), and may be constrained by incomplete or inconsistent data.

Ecosystem and habitat-based approaches

Some conservation scientists believe that setting conservation priorities at the scale of ecosystems and habitats is more appropriate for developing countries with limited resources for conservation, inadequate information about single species and pressing threats such as habitat destruction. Logically, how much effort we place in conserving a particular ecosystem should take into account factors such as: how threatened it is, how well represented that ecosystem is in that country's protected area network, the number of species restricted to that ecosystem (endemic species), the cost of conserving the ecosystem and the likelihood that conservation actions will work. One can debate the relative importance of each of these factors – for example, some consider the the number of endemic species is paramount, whereas others prefer the notion of 'equal representation' whereby a fixed percentage of every habitat type is conserved.

The main goals of an ecosystem approach are to:

1. maintain viable populations of all native species *in situ*;
2. represent, within protected areas, all native ecosystem types across their natural range of variation;
3. maintain evolutionary and ecological processes;
4. manage over periods of time long enough to maintain the evolutionary potential of species;
5. accommodate human use and occupancy within these constraints (Grumbine 1994).

Although the financial efficiencies inherent in managing an ecosystem rather than several single species are attractive, this approach is also not without its problems. First, compared with a species, ecosystem boundaries are harder to define, so determining the location, size, connectivity and spacing of protected areas to conserve the full range of ecosystems, and variation within those ecosystems, is more difficult (Possingham et al. 2005). Second, individual species are more interesting to people and will attract greater emotional and financial investments than ecosystems. Third, although ecological services are provided by ecosystems, individual species often play pivotal roles in the provision of these services, particularly for direct uses such as tourism or harvesting. Finally, the main problem faced by managers wishing to implement an ecosystem approach is the lack of data available on how ecosystems function. This manifests itself in confusion about how much of each ecosystem needs to be conserved to protect biodiversity adequately in a region. In contrast, for the better known single species, the issue of adequacy can be dealt with using population viability analysis and/or harvesting models (Beissinger & Westphal 1998; this volume, Chapter 15).

Systematic conservation planning

Systematic conservation planning (or gap-analysis in the USA: Scott et al. 1993) focuses on locating and designing protected areas that comprehensively represent the biodiversity of each region. Without a systematic approach, protected area networks have the tendency to occur in economically unproductive areas (Leader-Williams et al. 1990), leaving many

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habitats or ecosystems with little or no protection (Pressey 1994). The systematic conservation planning approach can be divided into six stages (Margules & Pressey 2000).

1. Compile biodiversity data in the region of concern. This includes collating existing data, along with collecting new data if necessary, and if time and funds permit. Where biodiversity data, such as habitat maps and species distributions, are limited more readily available biophysical data may be used that reflect variation in biodiversity, such as mean annual rainfall or soil type.
2. Identify conservation goals for the region, including setting conservation targets for species and habitats, and principles for protected area design, such as maximizing connectivity and minimizing the edge-to-area ratio.
3. Review existing conservation areas, including determining the extent to which they already meet quantitative targets, and mitigate threats.
4. Select additional conservation areas in the region using systematic conservation planning software.
5. Implement conservation action, including decisions on the most appropriate form of management to be applied.
6. Maintain the required values of the conservation areas. This includes setting conservation goals for each area, and monitoring key indicators that will reflect the success of management (see below).

Ultimately, conservation planning is riddled with uncertainty, so managers must learn to deal explicitly with uncertainty in ways that minimize the chances for major mistakes (Margules & Pressey 2000; Araújo & Williams 2000, Wilson et al 2005), and be prepared to modify their management goals appropriately through adaptive management.

Systematic conservation planning can complement species-based approaches because it focuses on removing the threat of development and it compliments a long tradition of species recovery plans that concentrate on mitigating threats. The degree to which different countries use species-based planning as opposed to systematic conservation planning depends on historical, cultural and legislative influences. Even with systematic conservation planning, however, the better surveyed species or species groups often feature as the units for assessment. In other words, the conservation value of different areas is often assessed on the presence or conservation status of the species within it, simply because these are the best known elements of biodiversity. Systematic conservation planning approaches have become popular and widespread, partly because they are supported by several decision-support software packages (Possingham et al 2000, Pressey et al 1995, Williams et al 2000, Garson et al 2002).

Methods for setting conservation priorities of species

Prioritizing species, habitats and ecosystems by their perceived level of endangerment has become a standard practice in the field of conservation biology (Rabinowitz 1981; Master 1991; Mace & Collar 1995; Carter et al. 2000; Stein et al. 2000). The need for a priority-setting process is driven by limited conservation resources that necessitate choices among a subset of all possible species in any given geographical area, and distinct differences among species in their apparent vulnerability to extinction or need for conservation action. This need has led to the development of practical systems for categorizing and assessing the degree of vulnerability of various components of biodiversity, particularly vertebrates (e.g. Millsap et al. 1990; Mace & Lande 1991; Master 1991; Reed 1992; Stotz et al. 1996), and more recently ecoregions (Hoekstra et al. 2005).

Methods used for assessing the conservation status of species are varied but follow three general styles (Regan et al. 2004), rule-based, point scoring and qualitative judgement. Perhaps the best known system is that developed by the IUCN (International Union for the Conservation of Nature and Natural Resources) – The World Conservation Union – which uses a set of five quantitative rules with explicit thresholds to assign a risk of extinction (Mace & Lande 1991; IUCN 2001). Other methods adopt point-scoring approaches where points are assigned for a number of attributes and summed to indicate conservation priority (Millsap et al. 1990; Lunney et al. 1996; Carter et al. 2000). Other methods assess conservation status using qualitative criteria; judgements about a species' status are determined intuitively based on available information and expert opinion (Master 1991). One widely applied system is the biodiversity status-ranking system developed and used by the Natural Heritage Network and The Nature Conservancy (Master 1991; Morse 1993). This ranking system has been designed to evaluate the biological and conservation status of plant and animal species and within-species taxa, as well as of ecological communities.

Rule-based methods

Quantitative rule-based methods can be used to estimate the extinction risk of a species and thus contribute to determining priority areas for conservation action. For example, the IUCN Red List places species in one of the following categories: extinct (EX), extinct in the wild (EW), critically endangered (CR), endangered (EN), vulnerable (V), near threatened (NT) or least concern (LC), based on quantitative information for known life history, habitat requirements, abundance, distribution, threats and any specified management options of that species, and in a data deficient (DD) category if there are insufficient data to make an assessment (IUCN 2001). The IUCN system is based around five criteria (A to E) which reflect different ways in which a species might qualify for any of the threat categories (CR, EN, VU). A species is placed in a category if it meets one or more of the criteria – for example because there are less than 250 mature individuals of the Norfolk Island green parrot (*Cyanoramphus cookii*) in the wild it is immediately listed as endangered under criterion D of the IUCN Red List protocol. A similar species can meet a higher category of threat if it meets alternative criteria. For example, the orange-bellied parrot (*Neophema chrysogaster*) also has less than 250 mature individuals but it is listed as critically endangered, under criterion C2b, because the population is also in decline and all the individuals are in a single subpopulation. One conceptual problem with rule-based methods is that a species that just missed out on being listed as, say, endangered on several criteria would be ranked as vulnerable, equal with a species that may have only just met the criteria for being vulnerable.

The rule-based methods have the advantage that they are completely explicit about what feature of the species led to it being listed as threatened. In the IUCN system, assessors have to list the criteria whereby the species qualified for a particular category of threat, and also have to provide documentation to support this information – usually in the form of scientific surveys or field reports that detail the information used. As a result, listings may be continually updated and improved as new data become available. Normally this will allow a new consensus among experts, but in the exceptional cases where this is not agreed, the IUCN have a petitions and appeals process to resolve matters. For example, in 2001 some of the listings of marine turtle species were disputed among experts. On this occasion, IUCN implemented their appeal procedure and provided a new assessment (http://www.iucn.org/themes/ssc/redlists/petitions.html). The wide use of the IUCN system also means that there is an ever increasing resource of best-practice documentation and guidelines, which aid consistent and comparable approaches by different species assessors (see *http://www.iucn.org/themes/ssc/redlists.htm)*.

Point scoring method

The point scoring method for assigning conservation priority involves assigning a series of scores to each species based on different parameters relating to their ecology or conservation status, which together will determine their relative priority. One method of dealing with the scores is then to simply sum them to give an overall conservation priority, although this can be misleading. Beissinger et al. (2000) suggest that a categorical approach based on a combination of scores might be more accurate in determining overall conservation priority.

An example of a point scoring system is that developed by Partners in Flight (PIF) in 1995 in an effort to conserve non-game birds and their habitats throughout the USA (Carter et al. 2000). The PIF system involves assigning a series of scores to each species ranging from 1 (low priority) to 5 (high priority) for seven parameters that reflect different degrees of need for conservation attention. The scores are assigned within physiographical areas and the seven parameters are based on global and local information. Three of the parameters are strictly global and are assigned for the entire range of the bird: breeding distribution (BD), non-breeding distribution (ND) and relative abundance (AR). Other parameters are threats to breeding (TB), threats to non-breeding (TN), population trend (PT) and, locally, area importance (AI). The scores for each of these seven parameters are obtained independently (Carter et al. 2000). The PIF then uses a combination of approaches, including the summing of scores, to determine an overall conservation priority (Carter et al. 2000), with species that score highly on several parameters achieving high priority. Although this method of defining bird species of high conservation priority is thought to be reliable, like other methodologies, it is hindered by the lack of data on species distribution, abundance and populations trends, particularly in areas outside the USA to which many of these species migrate (Carter et al. 2000).

A problem with some point-scoring methods is that there is no explicit link to extinction risk, the weightings of each criteria, from 1 to 5 in the example above, are completely arbitrary, and there is an infinity of ways in which the scores could be combined: adding, multiplying, taking the product of the largest three values, and so forth. A related problem is that point-scoring methods can generate an artificially high ranking for a species when criteria are interrelated. For example, a system that prioritized species because they needed large home ranges, had slow reproductive rates and small litter sizes might end up allocating unreasonably high scores to any large-bodied species. All three of these traits are associated with relatively large body size, but they are not necessarily so much more vulnerable.

Conservation status ranks method

Status ranks are based primarily on objective factors relating to a species' rarity, population trends and threats. Four aspects of rarity are typically considered: the number of individuals, number of populations or occurrences, rarity of habitat, and size of geographic range. Ranking is based on an approximately logarithmic scale, ranging from 1 (critically imperiled) to 5 (demonstrably secure). Typically species with ranks from 1 to 3 would be considered of conservation concern and broadly overlap with species that might be considered for review under the Endangered Species Act or similar state or international statutes.

The NatureServe system (Master 1991) is one example of a system that uses status ranks. Developed initially by The Nature Conservancy (TNC) and applied throughout North America, the NatureServe system uses trained experts who evaluate quantitative data and make intuitive judgements about species vulnerability. The aim of the NatureServe system is to determine the relative susceptibility of a species or ecological community to extinction or extirpation. To achieve this, assessments consider both deterministic and stochastic processes that can lead to extinction. Deterministic factors include habitat destruction or alteration, non-indigenous predators, competitors, or parasites, overharvesting and environmental shifts such as climate change. Stochastic factors include, environmental and demographic stochasticity, natural catastrophes and genetic effects (Shaffer 1981).

NatureServe assessments are performed on a basic unit called an element. An element can be any plant or animal species or infraspecific taxon (subspecies or variety), ecological community, or other non-taxonomic biological entity, such as a distinctive population (e.g. evolutionarily significant unit or distinct population segment, as defined by some agencies) or a consistently occurring mixed species aggregation of migratory species (e.g. shorebird migratory concentration area) (Regan et al. 2004). Defining elements in this way ensures that a broad spectrum of biodiversity and ecological processes are identified and targeted for conservation (Stein et al. 2000). This approach is believed to be an efficient and effective approach to capturing biodiversity in a network of reserves (e.g. Jenkins 1976, 1996). Assessment results in a numeric code or rank that reflects an element's relative degree of imperilment or risk of extinction at either the global, national or subnational scale (Master et al. 2000).

Back to basics – extinction risk versus setting priorities

The discussion above has reviewed methods for categorizing species according to their conservation priority. Running throughout is a tendency to equate conservation priority with extinction risk; yet these are clearly not the same thing (Mace & Lande 1991). Extinction risk is only one of a range of considerations that determine priorities for action or for conservation funding. The threat assessment is really an assessment of **urgency**, and an answer to the question of how quickly action needs to be taken. Hence, all other things being equal, the critically endangered species will be most likely to become extinct first if nothing is done. However, this is by no means the only consideration that should be used by a conservation planner. How then should extinction risk be used for priority-setting? It may be easier to make the analogy with a different system altogether. For example, the priority-setting systems used by Triage nurses in hospital emergency departments categorize people according to how urgently they need to be seen; those seen first are the ones that appear to have the most urgent and threatening symptoms. The symptoms can be very diverse, however, and some may turn out upon inspection and diagnosis to be less serious than might have been expected. Medical planning across the board would not use the triage system to allocate resources. The same is true for conservation planning. As with ill and injured people, our first sorting of cases should be according to urgency, and should also be precautionary (i.e. take more risks with listing species that are in fact not threatened than with failing to list those that really are). However, once the diagnosis is made, and the manager is reasonably sure that most critical cases are now known and diagnosed, a more systematic planning process should follow.

Variables other than risk

Now we consider a whole range of new variables other than risk. Table 2.1 shows a range of variables – grouped under headings of biological value (i.e. what biologists would consider), economic value, social and cultural value, urgency and practical issues. Under each of these headings are a range of attributes that might contribute to a species priority. The first three columns concern values, but the last two are rather different. Urgency is a measure that can be complicated to implement – i.e. high urgency may indicate that if nothing is done now, then it

Table 2.1 Classes and kinds of issues that are considered in priority-setting exercises for single-species recovery

Biological value	Economic value	Social and cultural value	Urgency	Practical issues
Degree of endemism	Cost of management or recovery	Scientific and educational benefits	Threat status = extinction risk	Feasibility and logistics
Relictual status	Direct economic benefits	Cultural status (e.g. ceremonial)	Time limitation, i.e. opportunities will be lost later	Recoverability, i.e. reversibility of threats, rate of species response
Evolutionary uniqueness	Indirect economic benefit	Political status (e.g. symbolic or emblematic)	Timeliness, i.e. likelihood of success varies with time	Popularity – will there be support from the community?
Collateral benefits to other species	Ecological services	Popularity		Responsibility, i.e. how much is this also someone else's responsibility?
Collateral costs to other species		Local or regional significance		Land tenure
Ecological uniqueness				Governmental/agency jurisdictions
Keystone species status				
Umbrella species status				

will be too late. This measure is not a value score that can easily be added to the others, and a moderate score has little meaning. Practical issues are also rather different, and will vary greatly in their nature and importance depending on the context. Some species that are considered urgent cases may be extremely impractical and/or costly to attend to. This set of considerations is probably not complete, but it does illustrate the point that there are more things to think about than extinction risk.

This initial classification by the value type is hard to manage in a priority-setting system. Therefore, in Table 2.2, we classify these into six criteria reflecting the nature of the attribute (importance, feasibility, biological benefits, economic benefits, urgency and chance of success). This classification has the advantage that the different questions are more or less independent of one another, and each addresses a question that public, policy-makers and scientists can all address, and for which they can provide at least relative scores.

Interestingly, the criteria that biologists commonly consider, and which form the basis of most formal decision-processes, fall under one heading (biological benefits). Yet in practice, the other five criteria (Table 2.2) also influence real decisions. Would it not, therefore, be preferable to incorporate these other criteria explicitly in the process of setting priorities?

Turning criterion-based ranks into priorities

A potential next step would be to add the scores from Table 2.2. By allocating a score of 1, 2 or 3 to each criterion and then adding the ranks, an overall priority could be calculated. We advise against this for several reasons. First, the different variables are not equal; we might for example wish to weight the biological issues more highly. Second, they are not additive: as mentioned earlier both urgency and chance of success are all or nothing decisions. For

Table 2.2 Criteria for setting prorities. The different kinds of considerations from Table 2.1 are classified into six criteria (rows), each of which can be qualitatively assessed for a particular species

Criterion	Explanation	Subcriteria	Scores
Importance	'Does anyone care?' A measure of how much support there is likely to be	Social and cultural importance (including charisma) Responsibility – how much of the species status depends on this project?	Important (I) Moderately important (M) Unimportant (U)
Feasibility	'How easy is this to achieve?' An assessment of the difficulty associated with this project	Logistical and political, source of funds, community attitudes Biological	Feasible (F) Moderately difficult (M) Difficult (D)
Benefits	'What good will it do?' A measure of how much good will result from the project.	Reduction in extinction risk, increase in population size, extent of occurrence Collateral biological benefits, to other species or processes	Highly beneficial (H) Moderately beneficial (M) Unclear benefits (U)
Costs	'What will it cost?' An assessment of the relative economic costs of the project (or gains). In this criterion there are both postive and negative aspects which have to be weighed against each other	Direct and indirect costs of project Direct and indirect social and economic costs and benefits that will flow from the project	Expensive Moderately costly Inexpensive
Urgency	'Can it be delayed?' A measure of whether the project is time-limited, or whether it can be delayed	Extinction risk, potential for loss of opportunity if delayed	Urgent Moderately urgent Less urgent
Chance of success	'Will it work?' An assessment of whether or not the project will work	Will it meet its specified objectives?	Achievable Uncertain Highly uncertain

example, if chance of success is nil we would not wish to invest in that species at all, so it would seem more logical to multiply other scores by the chance of success. Third, although we have sorted the issues into more-or-less independent categories, there still are associations between them. For example, the feasibility and chance of success are likely to be positively correlated, as are biological benefits and importance. Hence, simple scoring can lead to double-counting, which is not what was intended.

Multicriteria decision-analysis is one decision-making tool for choosing between priorities that rate differently for separate criteria. There are innumerable ways of carrying out a multicriteria analysis, and the process can be complex and may lead to ambiguous results. An expedient process at this stage is to invite a range of experts representing different perspectives to rate the priorities explicitly. For example, given the possible set of scores in Table 2.2, what set would they most wish to see in the top priorities versus those lower down? This sounds complicated but in practice we think it is feasible.

A good example of this approach was developed for UK birds by the Royal Society for the Protection of Birds (Avery et al. 1995). Three criteria were used: global threat, national decline rates and national responsibility, and each was rated high, medium or low. However, by simply adding these scores, globally endangered species that are stable, and for which the UK has medium responsibility, had the

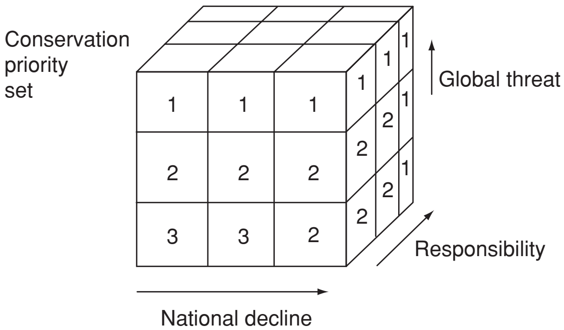

Fig. 2.1 The conservation cube. (From Avery et al. 1995.)

same priority as globally secure species exhibiting slow decline in the UK. This would not be most people's choice; whatever their status in the UK, a globally endangered species probably should be in the category of highest priority. Hence, Avery et al. (1995) set priorities using their conservation cube (Fig. 2.1). Here they evaluated each of the 27 possible circumstances into three categories for priority. In their system, any globally threatened species and any species declining at a high rate nationally are the highest priority.

This approach can be taken more generally using the six criteria in Table 2.2. By asking what would be the criteria associated with top priority species, it is possible to assemble a profile. For example, whereas a species conservation 'idealist' might choose to ignore importance, feasibility, economic benefits and chance of success, and to focus just on the most urgent and most threatened forms, a more 'political' approach would be to maximize importance and economic benefits and minimize risk of failure. Hence the two profiles would look quite different (Fig. 2.2). Figure 2.2 illustrates the different approaches – see how you would score the criteria in Table 2.2 to make your own set!

Here we are effectively creating a complex rule set that maps any species into one of three categories without adding or multiplying the scores for different criteria. The method suffers from its somewhat arbitrary nature. Below we suggest that optimal allocation of funds between species can be achieved more rigorously if we place the problem within an explicit framework in which we can apply decision theory.

A decision theory approach – optimal allocation

A major problem with using scores or ranks for threatened species to determine funding and action priorities is that these methods were not designed for that task – they were designed to determine the relative level of threat to a suite of species (Possingham et al. 2002). Hence they cannot provide the solution to the problem of optimal resource allocation between species – this problem should be formulated then solved properly (Possingham et al. 2001).

Optimal allocation is one simple and attractive approach to prioritization that could inform decisions about how to allocate resources between species. It requires information about

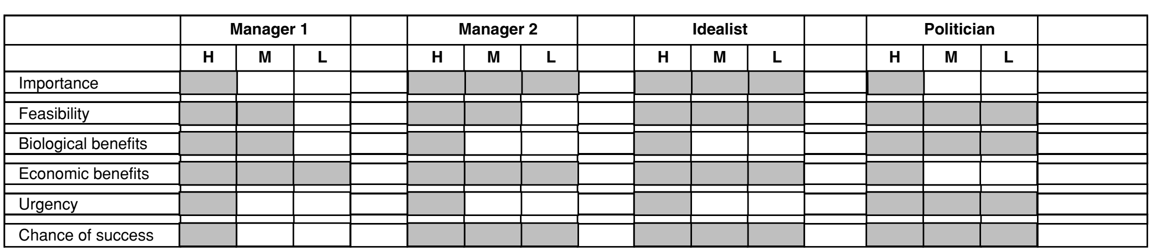

Fig. 2.2 Priority sets for four different people. The blocked out cells indicate the conditions under which assessors would choose to include species in their priority set, according to how they scored on the variables in Table 2.2 as H, high; M, medium; L, low.

the relationship between the resources allocated to the species and the reduction in probability of extinction. Here we use expert opinion and/or population models to estimate the relationship between percentage recovery (measured, for example, in terms of probability of not becoming extinct) and the funds allocated to that species.

For poorly known taxa the curves showing this relationship would very much be a reflection of expert opinion, garnered by asking questions about how much it might cost to give a particular species a 90% chance of not becoming extinct in the long term. Given a set amount of money for a set period in the conservation budget, the optimal allocation of

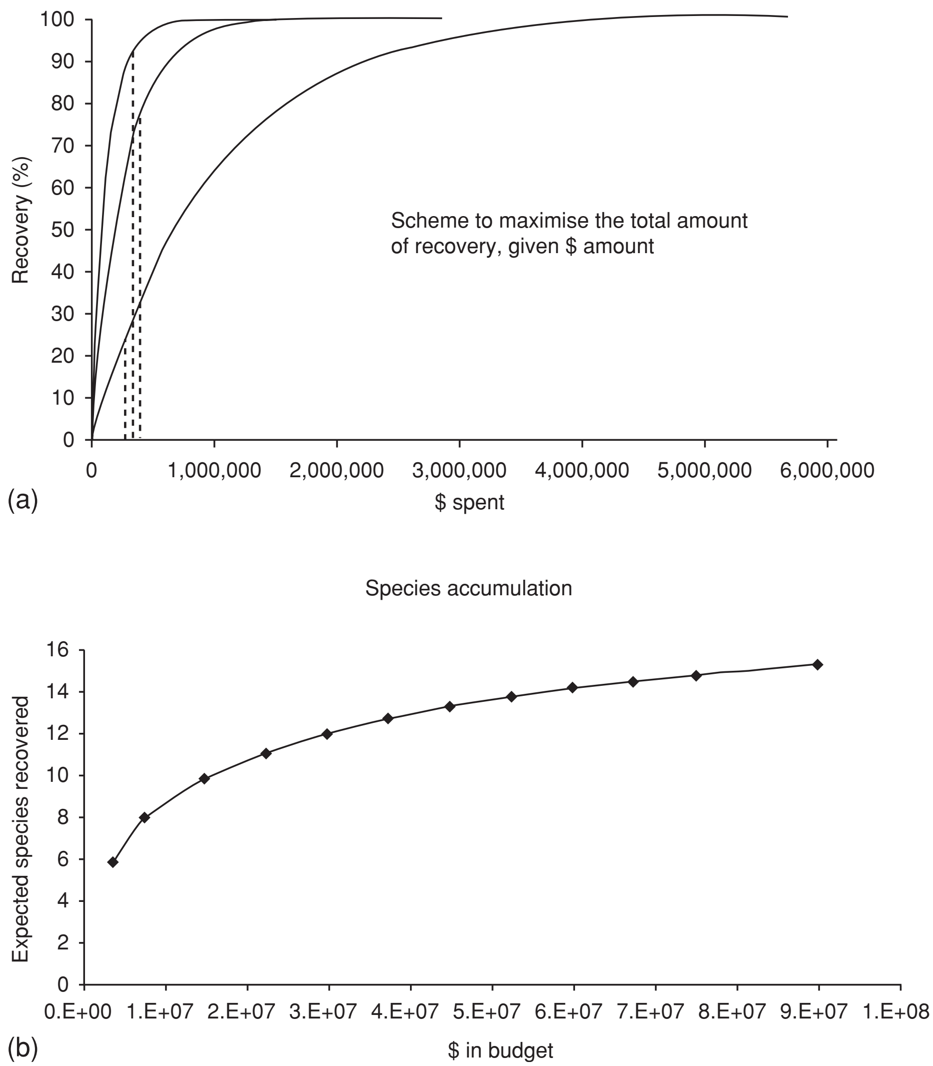

Fig. 2.3 Optimal allocation. (a) Three curves show the expected recovery for three different species given certain amounts of investment. If the manager has a specified budget (in this case $1 million), the optimal allocation among species that achieves the greatest total amount of recovery will result if funds are allocated as shown by the vertical dotted lines (see Possingham et al. 2002). (b) Increasing investment leads to gradually increasing numbers of species recovered.

funds can be determined between species. This occurs when the rate of gain of recovery for each species is equal, such that there is no advantage in shifting resources from one species to another (see Fig. 2.3 and Possingham et al. 2002). The implicit objective is to maximize the mean number of recovered species given a fixed budget and assuming all species are of equal 'value'.

Using the set of species plotted in Fig. 2.3, we estimate the costs of recovery, and then find the optimal allocation of funds per species. The species accumulation curve shows the total expected number of species that can be recovered given a conservation budget. The algorithm will tend first to select species that show large recovery for relatively low costs. Slow responders will be conserved later. Given an annual budget basis, the more intractable conservation problems may never be funded because the selection process will always favour allocation of resources to the species that provide the greatest gains for the smallest costs (the low-hanging fruit).

So how would these two approaches: criteria-based prioritization and optimal allocation of resources differ in practice? Obviously there is no general answer to this, other than a priori we do not expect them to be the same. The outcome of a small case study, based on real species and the expertise of two real conservation managers is shown in Fig. 2.4.

When species are rated highly by the criteria the two approaches give similar results, but at low criterion scores there can be much variability. Perhaps the only general conclusion here is that inevitably the optimal allocation approach will favour some species that, on the basis of the criteria, would not be given high priority. In practice, sensible management could use both approaches – the criteria to select high-priority sets and the financial algorithm to then maximize the benefits from the finite resources available to conservation.

Conclusions

Priority setting needs to consider a range of variables, and although this undoubtedly occurs, it is not always transparent. Although much effort has gone into biologically based systems, in practice other societal value judgments are often included. We suggest that, if conservation goals are to be achieved, it is vital to be explicit about what these are, and to decide upon them in an open and consultative manner before choices are made.

Different people and organizations, and different sectors in society, will make different choices in their value judgments. Approaches to understanding these choices are important so we can interpret the differences in setting priorities.

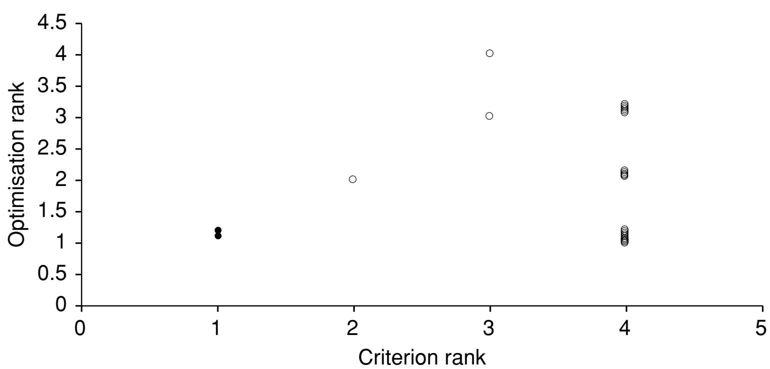

Fig. 2.4 Comparison of priority ranks for18 species using the criteria-based method versus optimal allocation of funds.

We recommend using more than one method to set priorities, and the comparison can be informative. We also recommend that decisions about resource allocation be formulated more explicitly in terms of objectives, constraints and costs.

For if one link in nature's chain might be lost, another might be lost, until the whole of things will vanish by piecemeal.
(Thomas Jefferson (1743–1826) in Charles Miller, *Jefferson and Nature*, 1993.)

References

Araújo, M.B. & Williams, P.H. (2000) Selecting areas for species persistence using occurrence data. *Biological Conservation* **96**: 331–45.

Avery, M., Gibbons, D.W., Porter, R., Tew, T., Tucker, G. & Williams, G. (1995) Revising the British Red Data List for birds: the biological basis of UK conservation priorities. *Ibis* **137**(S1): 232–9.

Balmford, A., Bennun, L., ten Brink, B., et al. (2005) Science and the Convention on Biological Diversity's 2010 target. *Science* **307**: 212–13.

Bani, L., Baietto, M., Bottoni, L. & Massa, R. (2002) The use of focal species in designing a habitat network for a lowland area of Lombardy, Italy. *Conservation Biology* 16: 826–31.

Basset, Y., Novotny, V., Miller, S.E. & Pyle, R. (2000) Quantifying biodiversity: experience with parataxonomists and digital photography in Papua New Guinea and Guyana. *Bioscience* **50**(10): 899–908.

Beissinger, S.R. & Westphal, M.I. (1998) On the use of demographic models of population viability in endangered species management. *Journal of Wildlife Management* **62**: 821–41.

Beissinger, S.R., Reed, J.M., Wunderle, Jr, J.M., Robinson, S.K. & Finch, D.M. (2000) Report of the AOU Conservation Committee on the Partners in Flight species prioritization plan. *Auk* **117**: 549–61.

Berger, J. (1997) Population constraints associated with the use of black rhinos as an umbrella species for desert herbivores. *Conservation Biology* **11**: 69–78.

Bond, I. (1994) Importance of elephant sport hunting to CAMPFIRE revenue in Zimbabwe. *TRAFFIC Bulletin* **14**: 117–19.

Brooker L. (2002) The application of focal species knowledge to landscape design in agricultural lands using the ecological neighbourhood as a template. *Landscape and Urban Planning* **60**: 185–210.

Butchart, S.H.M., Stattersfield, A.J., Bennun, L.A., et al. (2005) Measuring global trends in the status of biodiversity: Red List Indices for birds. *Public Library of Science Biology* **2**: 2294–304.

Carroll, C., Noss, R.F. & Paquet, P.C. (2001) Carnivores as focal species for conservation planning in the Rocky Mountain region. *Ecological Applications* **11**: 961–80.

Caro, T. & Doherty, G. (1999) On the use of surrogate species in conservation. *Conservation Biology* **13**: 805–14.

Caro, T., Engilis, A., Fitzherbert, E. & Gardner, T. (2004) Preliminary assessment of the flagship species concept at a small scale. *Animal Conservation* **7**: 63–70.

Carter, M.F., Hunter, W.C., Pashley, D.N. & Rosenberg, K.V. (2000) Setting conservation priorities for landbirds in the United States: the Partners in Flight Approach. *Auk* **117**: 541–8.

Costanza, R., d'Arge, R., de Groot, R., et al. (1997) The value of the world's ecosystem services and natural capital. *Nature* **387**: 253–60.

Dietz, J.M., Dietz, L.A. & Nagagata, E.Y. (1994) The effective use of flagship species for conservation of biodiversity: the example of lion tamarins in Brazil. In *Creative Conservation: Interactive Management of Wild and Captive Animals* (Eds P.J.S. Olney, G.M. Mace & A.T.C. Feistner), pp. 32–49. Chapman & Hall, London.

Downer, C. (1996) The mountain tapir, endangered 'flagship' species of the high Andes. *Oryx* **30**: 45–58.

Ehrlich, P.R., Dobkin, D.S. & Wheye, D. (1992) *Birds in Jeopardy: the Imperiled and Extinct Birds of the*

United States and Canada, including Hawaii and Puerto Rico. Stanford University Press, Stanford, CA.

Entwistle, A. (2000) Flagships for the future? *Oryx* **34**: 239.

Ferrier, S., Pressey, R.L. & Barrett, T.W. (2000) A new predictor of the irreplaceability of areas for achieving a conservation goal, its application to real-world planning, and a research agenda for further refinement. *Biological Conservation* **93**: 303–25.

Fleishman, E., Murphy, D.D. & Brussard, P.F. (2000) A new method for selection of umbrella species for conservation planning. *Ecological Applications* **10**: 569–79.

Fleishman, E., Blair, R.B. & Murphy, D.D. (2001) Empirical validation of a method for umbrella species selection. *Ecological Applications* **11**: 1489–501.

Garson, J., Aggarwal, A. & Sarkar, S. (2002) Birds as surrogates for biodiveristy: an analysis of a data set from southern Québec. *Journal of Biosciences* **27**:347–60.

Goldspink, C.R., Holland, R.K., Sweet, G. & Stjernsredt, R. (1998) A note on the distribution of the puku, Kobus vardoni, in Kasanka National Park, Zambia. *African Journal of Ecology* **36**: 23–33.

Goodwin, H.J. & Leader-Williams, N. (2000) Tourism and protected areas: distorting conservation priorities towards charismatic megafauna? In *Priorities for the Conservation of Mammalian Diversity: Has the Panda had its Day?* (Eds A. Entwistle & N. Dunstone), pp. 257–75. Cambridge University Press, Cambridge.

Grumbine, R.E. (1994) What is ecosystem management? *Conservation Biology* **8**: 27–38.

Hess, G.R. & King, T.J. (2002) Plannng for wildlife in a suburbanizing landscape I: selecting focal species using a Delphi survey approach. *Landscape and Urban Planning* **58**: 25–40.

Heywood, V.H. (1995) *Global Biodiversity and Assessment*. Cambridge University Press, New York.

Hoekstra, J.M., Boucher, T.M., Ricketts, T.H. & Roberts, C. (2005) Confronting a biome crisis: global disparities of habitat loss and protection. *Ecology Letters* **8**: 23–9.

Iltis, H.H. (1988) Serendipity in the exploration of biodiversity: what good are weedy tomatoes? In *Biodiversity* (Ed. E.O. Wilson), pp. 98–105. National Academy Press, Washington, DC.

IUCN (2001) *IUCN Red List Categories: Version 3.1*. Species Survival Commission, International Union for the Conservation of Nature and Natural Resources, Gland, Switzerland.

James, A.N., Gaston, K.J. & Balmford, A. (1999) Balancing the earth's accounts. *Nature* **401**: 323–4.

Jenkins, R.E. (1976) Maintenance of natural diversity: approach and recommendations. In *Transactions of the 41st North American Wildlife and Natural Resources Conference* 21–25 March, Washington, DC (Ed. K. Sabol), pp. 441–51.

Jenkins, R.E. (1996) Natural Heritage Data Centre Network: managing information for managing biodiversity. In *Biodiversity in Managed Landscapes: Theory and Practice* (Eds R.C. Szaro & D.W. Johnston), pp. 176–92. Oxford University Press, New York.

Johnson, N.C. (1995) *Biodiversity in the Balance: Approaches to Setting Geographic Conservation Priorities*. Biodiversity Support Program, World Wildlife Fund, Washington, DC.

Johnsingh, A.J.T. & Joshua, J. (1994) Conserving Rajaji and Corbett National Parks: the elephant as a flagship species. *Oryx* **28**: 135–40.

Kershaw, M., Mace, G.M. & Williams, P.H. (1995) Threatened status, rarity and diversity as alternative selection measures for protected areas: a test using Afrotropical antelopes. *Conservation Biology* **9**: 324–34.

Lambeck, R.J. (1997) Focal species: a multi-species umbrella for nature conservation. *Conservation Biology* **11**: 849–56.

Leader-Williams, N. & Dublin, H.T. (2000) Charismatic megafauna as 'flagship species'. In *Priorities for the Conservation of Mammalian Diversity: has the Panda had its Day?* (Eds A. Entwistle & N. Dunstone), pp. 53–81. Cambridge University Press, Cambridge.

Leader-Williams, N., Harrision, J. & Green, M.J.B. (1990) Designing protected areas to conserve natural resources. *Science Progress* **74**: 189–204.

Lindenmayer, D.B., Manning, A.D., Smith, P.L., et al. (2002) The focal-species approach and landscape restoration: a critique. *Conservation Biology* **16**: 338–45.

Lunney, D., Curtin, A., Ayers, D., Cogger, H.G. & Dickman, C.R. (1996) An ecological approach to identifying the endangered fauna of New South Wales. *Pacific Conservation Biology* **2**: 212.

Mace, G.M. & Collar, N.J. (1995) Extinction risk assessment for birds through quantitative criteria. *Ibis* **137**: 240–46.

Mace, G.M. & Lande, R. (1991) Assessing extinction threats: toward a reevaluation of IUCN threatened species categories. *Conservation Biology* **5**: 148–57

Margules, C.R. & Pressey, R.L. (2000) Systematic conservation planning. *Nature* **405**: 243–53.

Master, L.L. (1991) Assessing threats and setting conservation priorities. *Conservation Biology* **5**: 559–63.

Master, L.L., Stein, B.A., Kutner, L.S. & Hammerson, G. (2000) Vanishing assets: conservation status of US species. In *Precious Heritage. The Status of Biodiversity in the United States* (Eds B.A. Stein, L.S. Kutner & J.S. Adams), 93–118. Oxford University Press, New York.

Meir, E., Andelman, S. & Possingham, H.P. (2004) Does conservation planning matter in a dynamic and uncertain world? *Ecology Letters* **7**: 615–22.

Miller, B., Reading, R., Strittholt, J., et al. (1999) Using focal species in the design of nature reserve networks. *Wild Earth* **8**: 81–92.

Millsap, B.A., Gore, J.A., Runde, D.E. & Cerulean, S.I. (1990) Setting priorities for conservation of fish and wildlife species in Florida. *Wildlife Monographs* **111**: 1–57.

Mittermeier, R.A. (1986) Primate conservation priorities in the Neotropical region. In *Primates: The Road to Self-sustaining Populations* (Ed. K. Benirschke), pp. 221–40. Springer-Verlag, New York.

Morse, L.E. (1993) Standard and alternative taxonomic data in the multi-institutional Natural Heritage Data Center Network. In *Designs for a Global Plant Species Information System* (Eds. F.A Bisby, G.F. Russell & R. Pankhurst), pp. 127–56. Oxford University Press, Oxford.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B. & Kent, J. (2000) Biodiversity hotspots for conservation priorities. *Nature* **403**: 853–8.

Noss R.F., Quigley, H.B., Hornocker, M.G., Merrill, T. & Paquet, P.C. (1996) Conservation biology and carnivore conservation in the Rocky Mountains. *Conservation Biology* **10**: 949–63

Olson, D.M. & Dinerstein, E. (1998) The global 200: a representation approach to conserving the Earth's most biologically valuable ecoregions. *Conservation Biology* **12**: 502–15.

Paine, R.T. (1995) A conversation on refining the concept of keystone species. *Conservation Biology* **9**: 962–64.

Possingham, H., Ball, I. & Andelman, S. (2000) Mathematical methods for identifying representative reserve networks. In *Quantitative Methods for Conservation Biology* (Eds S. Ferson & M. Burgman), pp. 291–305. Springer-Verlag, New York.

Possingham, H.P., Andelman, S.J., Noon, B.R., Trombulak, S. & Pulliam, H.R. (2001) Making smart conservation decisions. In *Research Priorities for Conservation Biology* (Eds G. Orians & M. Soulé), pp. 225–44. Island Press, Covelo, California.

Possingham, H.P., Andelman, S.J., Burgman, M.A., Mendellin, R.A., Master, L.L. & Keith, D.A. (2002) Limits to the use of threatened species lists. *Trends in Ecology and Evolution* **17**: 501–7.

Possingham, H.P., Franklin, J., Wilson, K. & Regan, T.J. (2005) The roles of spatial heterogeneity and ecological processes in conservation planning. In *Ecosystem Function in Heterogeneous Landscapes* (Eds G.M. Lovett, C.G. Jones, M.G. Turner & K.C. Weathers), pp. 389–406. Springer-Verlag, New York.

Pressey, R.L. (1994) *Ad hoc* reservations: forward or backward steps in developing representative reserve systems? *Conservation Biology* **8**: 662–8.

Pressey, R.L. & Taffs, K.H. (2001) Sampling of land types by protected areas: three measures of effectiveness applied to western New South Wales. *Biological Conservation* **101**: 105–17.

Pressey, R.L., Ferrier, S., Hutchinson, C.D., Sivertsen, D.P. & Manion, G. (1995) Planning for negotiation: using an interactive geographic information system to explore alternative protected area networks. In *Nature Conservation 4: the Role of Networks* (Eds D.A. Saunders, J.L. Craig & E.M. Mattiske), pp. 23–33. Surrey Beatty and Sons, Sydney.

Rabinowitz, D. (1981) Seven forms of rarity. In *The Biological Aspects of Rare Plant Conservation* (Ed. H. Synge), pp. 205–17. Wiley, Chichester.

Redford, K.H., Coppolillo, P., Sanderson, E.W., et al. (2003) Mapping the conservation landscape. *Conservation Biology* **17**: 116–31

Reed, J.M. (1992) A system for ranking conservation priorities for neotropical migrant birds based on relative susceptibility to extinction. In *Ecology and Conservation of Neotropical Migrant Landbirds* (Eds J.M.H. Hagan III & D.W. Johnston), pp. 524–36. Smithsonian Institution Press, Washington, DC.

Regan, T.J., L.L. Master & G.A. Hammerson. (2004) Capturing expert knowledge for threatened species assessments: a case study using Natureserve conservation status ranks. *Acta Oecologica* **26**: 95–107.

Sanderson, E.W., Redford, K.H., Chetkiewicz, C.L.B., et al. (2002a) Planning to save a species: the jaguar as a model. *Conservation Biology* **16**: 58–72.

Sanderson, E.W., Redford, K.H., Vedder, A., Ward, S.E. & Coppolillo, P.B. (2002b) A conceptual model for conservation planning based on landscape species requirements. *Landscape and Urban Planning* **58**: 41–56.

Scott, J.M., Davis, F., Csuti, B., et al. (1993) Gap analysis: a geographic approach to protection of biological diversity. *Wildlife Monographs* **123**: 1–41.

Shaffer, M.L. (1981) Minimum population sizes for species conservation. *BioScience*, 31, 131-134

Simberloff, D. (1998) Flagships, umbrellas, and keystones: is single-species management passe' in the landscape era. *Biological Conservation* **83**: 247–57.

Smith, R.J., Muir, R.D.J., Walpole, M.J., Balmford, A. & Leader-Williams, N. (2003) Governance and the loss of biodiversity. *Nature* **426**: 67–70.

Stein, B.A., Kutner, L.S. & Adams, J.S. (2000) *Precious Heritage. The Status of Biodiversity in the United States*. Oxford University Press, New York.

Stotz, D.F., Fitzpatrick, J.W., Parker, III, T.A. & Moskovits, D.K. (1996) *Neotropical Birds: Ecology and Conservation*. University of Chicago Press, Chicago, IL

Walpole, M.J. & Leader-Williams, N. (2002) Ecotourism and flagship species in conservation. *Biodiversity and Conservation* **11**: 543–547.

Watson, J., Freudenberger, D. & Paull, D. (2001) An assessment of the focal species approach for conserving birds in varigated landscapes in southeastern Australia. *Conservation Biology* **15**: 1364–73.

Western, D. (1987) Africa's elephants and rhinos: flagships in crisis. *Trends in Ecology and Evolution* **2**: 343–6.

Williams, P.H., Burgess, N.D. & Rahbek, C. (2000) Flagship species, ecological complementarity and conserving the diversity of mammals and birds in sub-Saharan Africa. *Animal Conservation* **3**: 249–60.

Wilson, K.A., Westphal, M.I., Possingham, H.P. & Elith, J. (2005) Sensitivity of conservation planning to different approaches to using predicted species distribution data. *Biological Conservation* **122**: 99–112.

Wilson, K.A., McBride, M., Bode, M. & Possingham, H.P. (In press) Prioritising the allocation of conservation resources between biodiversity hotspots. *Nature*.