Limits to the use of threatened species lists

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Threatened species lists are designed primarily to provide an easily understood qualitative estimate of risk of extinction. Although these estimates of risk can be accurate, the lists have inevitably become linked to several decision-making processes. There are four ways in which such lists are commonly used: to set priorities for resource allocation for species recovery; to inform reserve system design; to constrain development and exploitation; and to report on the state of the environment. The lists were not designed for any one of these purposes, and consequently perform some of them poorly. We discuss why, if and how they should be used to achieve these purposes.

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Governments and nongovernmental organizations produce threatened species lists for three main reasons: (1) to assess potentially adverse impacts on species; (2) to help inform conservation priorities, including reserve system design; or (3) as a component of State of the Environment Reports [1]. Moreover, in many countries there is a direct connection between threatened species lists and legislation (e.g. the Convention on International Trade in Endangered Species [2] and the US Endangered Species Act), leading to political and social considerations in the listing protocol. The ideal characteristics of these lists, and the protocols for generating them, will depend on the specific objectives. Regardless of who generated the list, there are three classes of user group: the public, governments and conservation organizations.

All protocols result in an assessment of threat, couched in words that reflect the probability of decline or loss of a taxon at different regional scales [3]. Some are designed to apply within a local region or state [4–6], whereas others have national [7–9], or international status [10,11], or are used at multiple political scales [12–14]. Some have also been developed for specific taxonomic groups [15–17]. These protocols share many attributes and use similar information, such as population size, extent, number of populations and trends in at least some of these variables.

Because of the variety of objectives and users, the interpretation of lists is variable, and most are used for more than one purpose, regardless of their original intent. Here, we critically assess four ways in which threatened species lists are commonly used. We argue that they are used for purposes beyond their original intent; furthermore, they perform some of these uses poorly. The protocols used to generate lists differ in the extent to which they consider management variables, taxonomic status, recoverability and assessments of past or future trends. In addition, they employ different logical systems to interpret data, treat missing data differently, and apply different weights to the variables. As a result, the level of correspondence among classifications resulting from different methods can be low, even when using the same data [18]. Given the widespread use of such lists to allocate scarce conservation resources, before application, users need to ask the question: what are the appropriate uses of threatened species lists?

Should lists of species at risk guide resource allocation for species recovery?

Threatened species classification systems provide a method for highlighting those species under higher extinction risk, so as to focus attention on conservation measures designed to protect them [10]. Most of the lists in Table 1 are used to prioritize species recovery and habitat protection activities with the aim of minimizing future extinctions [12,17,19]. Agencies in most countries must decide how to allocate resources among species, and often the only metric guiding these decisions is species conservation status. As a rule of thumb, more funding is allotted to species in the highest threat categories within a taxonomic group. For example, of the 18 completed Australian national recovery plans for plants, 17 were for species in the highest risk category [19].

It is inappropriate to use threatened species lists for resource allocation. Resources for conservation are limited. Spending the most money on species with the highest extinction probabilities is not the most efficient way of promoting recovery or minimizing global extinction rates, because some of the most highly ranked species require huge recovery efforts with a small chance of success, whereas other, less threatened taxa might be secured for relatively little cost. To minimize overall species loss, we should allocate resources to recovery actions such that the marginal rate of increase in viability is equalized across all threatened species. Status is only part of the information required for resource allocation, and this approach to threatened species recovery has been acknowledged in discussions of triage [20]. Figure 1 provides two examples of how optimal allocation of resources for threatened species leads to allocations that are very different to those based solely on threatened species rank. The examples show that the optimal allocation of funds does not mean that the most threatened species receive the most funding, although they rely on a crude

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Table 1. The intended purpose and actual uses of some threatened species list protocols

List protocol	Use				Intended primary purpose of list[a]	Refs
	A	B	C	D		
IUCN 1994	X	X	X	X	Assess risk	[10]
EPBC	X	X	X	X	Assess risk	[19]
USFWS	X	X	X	X	Assess risk	[12]
NatureServe/Heritage	X	X		X	Assess risk	[13]
SEMARNAT	X	X			Assess risk	[9]
CITES			X		Constrain trade	[2]
Lunney *et al.*	X	X	X		Set conservation priorities	[5]
Milsap *et al.*	X	X			Set conservation priorities	[4]
PIF	X	X			Set conservation priorities	[17]

[a]A, recovery resource allocation; B, reserve system design; C, constrain development; D, state of the environment reporting. An X marks every case in which we know how a list has been used.

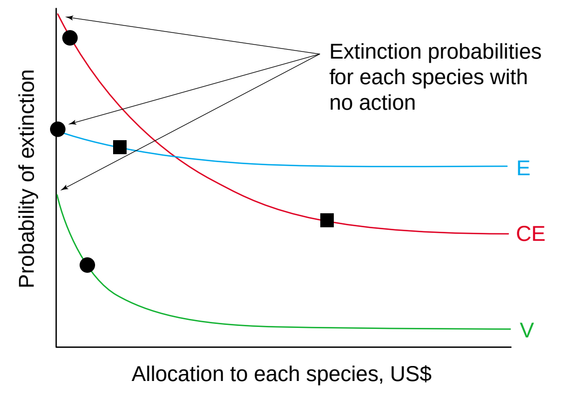

Fig. 1. Two solutions to the problem of allocating resources to recover threatened species. The lines represent the way in which the risk of extinction declines for each species as money is spent on that species (in this simple example, actions are assumed to affect species independently). Assume one species is listed as Critically Endangered (CE, red), one Endangered (E, blue) and the other as Vulnerable (V, green). With no allocation of resources to any species, the CE species has a higher extinction probability than does the E species, which, in turn, has a higher extinction probability than does the V species. Allocation of funds is assumed to reduce extinction probabilities differently. If the objective is to minimize loss of species then the optimal allocation of funds is found when the marginal rate of gain for each species is the same. With a small allocation of funds (circle) the optimal strategy allocates most funding to the V species, less to the CE and none to the E species. With a larger budget (square) most of the money should be spent on the CE species, some on the V species and little on the E species.

understanding of how recovery actions influence probability of extinction.

Should lists of species at risk be used to set priorities for reserve selection?

Reserve selection algorithms seek to conserve biodiversity as efficiently and completely as possible [21,22]. Conserving populations of threatened species often drives and simplifies reserve system planning. For example, Ceballos *et al.* [23] and Noss [24] recommend using threatened species lists as one of several factors identifying high-risk ecosystems. Given the social and legal importance of threatened species, protecting such species might take precedence over other criteria.

Although simplifying complex problems makes sense, there is no biological justification for using threatened species alone as an umbrella group for all biodiversity. The use of threatened species as surrogates for biodiversity is limited, because most invertebrate animals and nonvascular plants do not appear on any threatened species lists. The use of single threatened species as umbrella species for biodiversity conservation is particularly problematic [25–27]. For example, in southern California, the presence of the California gnatcatcher *Polioptila c. californica*, an endangered species, is used to prioritize coastal sage scrub habitat for conservation. However, Rubinoff [27] found that the presence of the gnatcatchers was a poor indicator of habitat value for coastal sage scrub biota, and prioritizing acquisition based on patch size protected more biodiversity.

Furthermore, not all listed species will benefit from inclusion in a reserve system. Such species, regardless of their status, should not influence reserve design. Using listed species to design reserve systems runs into the same logical flaws as the use of lists to set resource allocation priorities. A high risk of extinction does not imply a high priority for an action, in this case reserve dedication. We need to set priorities for reserve selection using a decision theory approach so that the reserve system maximizes a specific objective, such as securing as many species as possible within socioeconomic constraints [28]. For example, Araujo and Williams [29] maximize the probability of persistence across species to identify networks of conservation areas.

Should threatened species lists be used to constrain development or exploitation?

In most countries, environmental impact processes evaluate the likelihood that development will affect any threatened species. Most environmental regulations are implemented in a way that assumes development activities are benign, until proven otherwise. This policy places the burden of proof on stakeholders who seek to protect public good or public values [30]. Threatened species lists are one of the few tools at the disposal of regulatory agencies and the public to limit adverse environmental impacts of development. When a proposed development action is judged to increase risks to threatened species, that activity might be modified or postponed. If there is no evidence that listed species are present, or impacts are negligible, development can proceed. There are many examples in which the presence of a threatened species has been a serious impediment to some human activity (e.g. dam construction on the snail darter *Percina tanasi*, development on the California gnatcatcher and spectacled flying fox *Pteropus conspicullatus*) and society might naïvely believe that this process is adequate for biodiversity protection. The pros and cons of endangered species legislation have been extensively debated, and we add only three points.

First, the binary nature of lists can lead to the illogical outcome that developments with small impacts might be curtailed by a listed species, but developments with large impacts on one or more nonlisted species proceed with no mitigation requirements. An alternative approach is to consider the impact of developments on a much wider array of elements of biodiversity at a site and ask the question: are any of those impacts, or the cumulative impacts across all species, unacceptably large? This would disentangle the listing process from the Environmental Impact Assessment process.

Second, the high-profile application of threatened species lists to constrain development affects the listing process itself. Even in instances where listing decisions are purported to be based on criteria of extinction risk, economic or social criteria can override scientifically based criteria (S.J. Andelman *et al.*, unpublished). This can lead to subjectivity in the lists.

Third, listing might increase threats to a species. When the presence of a threatened species in an area is viewed as an impediment to a particular land use, land managers might destroy habitat, deny the presence of the species or deny access to the area for researchers or government officials. This is an unintended consequence of a threatened species list when incentives for landowners to conserve threatened species on their properties are lacking.

Should threatened species lists be used to indicate changes in the status of biodiversity?

Threatened species lists are used to report on the state of the environment, based on statistics derived from: the number of threatened/extinct species per taxon; the total number of threatened/extinct species; the proportion of threatened/extinct species per taxon; and changes in the number of threatened species per taxon.

There are many examples of the application of these statistics. At an international level, they include the Montreal criteria [31], Article 7 of the Convention on Biological Diversity, and the global assessments of impact provided by organizations such as the World Resources Institute [32]. At the national level, in the USA, the US Fish and Wildlife Service, the National Marine Fisheries Service and the Heinz Center provide annual reports on the status of species or ecosystems based on these lists [33]; and, in Australia, the State of the Environment Report used them to provide an assessment of human impacts on the environment [1]. Such measures influence public awareness of conservation objectives. In theory, they provide a benchmark for the extent and severity of human impacts, and a basis for comparing impacts between places, between taxa, and over time.

In reality, threatened species lists might have limited value as indicators of changes in the state of the environment because of: uneven taxonomic treatment; variation in observational effort; and the fact that changes in the lists more often reflect change in knowledge of status rather than change in status itself [34].

Almost all lists, both official and unofficial, substantially under-represent some taxa, particularly insects and fungi. In Australian, US Federal and international lists, there is significant bias towards large species and those that are close to humans in evolutionary terms [35,36]. Master *et al.* [37] report equivalence in numbers of species of reptiles, mammals and birds between the US Federal list and Heritage assessments of conservation status, and substantially fewer species on the Federal list among all other taxa. Any biases resulting from the personal interests of the experts making the decisions might result in inflated lists of species at risk for a few taxa, eroding the credibility of the entire list, and of changes in those lists from one period to another. However, in groups that are thoroughly evaluated, the relative differences in number among taxa might actually reflect differences in threat.

Comparisons of lists among areas, among taxa, or through time are also compromised by differences in survey intensity. For example, the list of threatened species in South Africa is much longer than that from any other African nation, in part, because the survey effort in that country has been higher than in its neighbors [38]. To eliminate survey effort bias, metrics from threatened species lists should be scaled to account for the number of species that have been assessed, and for the number for which there are sufficient data to make an assessment.

Most changes in the composition of threatened species represent changes in knowledge. When the causes of status change are not apparent, changes in knowledge and taxonomy mask true changes in conservation status. For example Master *et al.* [37] reported increases in the number of listed species in the USA from ~200 in 1974 to ~1200 in 1998. The primary cause of the increase was the addition of numerous plants, reflecting backlogs in processing information through formal channels. The number of species listed at risk in Australia increased from 1027 in 1993 to 1442 in 2000, most of which were due to the increasing numbers of assessments [19,39]. BirdLife International noted a total of 1030 bird species threatened with global extinction in 1988 [40]. This figure increased to 1111 in 1994, most of which was attributed to taxonomic changes and improved knowledge. There were >200 differences between the two lists, but few reflected true changes in underlying status.

We do not generally recommend using threatened species lists as they are currently constructed for indicating changes in the state of the environment, except where comprehensive data are maintained on well-studies groups (e.g. birds and mammals) allowing robust comparisons over time and space [37,41].

Conclusions

There is no doubt that threatened species lists fulfil important political, social and scientific needs. For example, lists of endangered species are very useful in explaining to people the importance of recovery or loss of species. In most circumstances, they are the only tools available that have a clear social mandate and that rest on substantial sound ecological knowledge. However, these lists have many limitations in dealing effectively with the problems that they are used to solve, and unthinking application carries risks of its own [20,42]. There are, however, some remedies:

- Create new tools for allocating resources to recovery and for mandating constraints on development or trade that include reductions in ecological risk and socioeconomic costs and benefits [28].
- Use additional criteria besides threatened species in reserve selection for biodiversity conservation, including criteria related to other targets (e.g. ecological systems), viability of 'targets', the urgency and degree of threats to the targets, and the feasibility or opportunity to abate threats [43].
- In compiling reports on the state of the environment, record changes in knowledge and trends in populations and range separately from changes in status, and only use comprehensive and systematic assessments.

Setting conservation priorities involves value judgements and is part of a broader social debate. The relative value of biodiversity is culturally sensitive and includes decisions about the equitable distribution of costs and the social importance of adverse outcomes. It is naïve and counterproductive from all points of view to use threatened species lists alone to allocate resources for recovery, to guide reserve planning, or to constrain the use of the natural environment. Other tools are necessary for these tasks, and threatened species lists should be a part of the contributing information. The discussion here will go someway to resolving the current misuse of lists of species at risk of extinction.

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Trophic strategies, animal diversity and body size

Kevin D. Lafferty and Armand M. Kuris

A primary difference between predators and parasites is the number of victims that an individual attacks throughout a life-history stage. A key division within natural enemies is whether a successful attack eliminates the fitness of the prey or the host. A third distinctive axis for parasites is whether the host must die to further parasite development. The presence or absence of intensity-dependent pathology is a fourth factor that separates macroparasites from microparasites; this also distinguishes between social and solitary predators. Combining these four dichotomies defines seven types of parasitism, seven corresponding parasites, three forms of predation and, when one considers obligate and facultative combinations of these forms, four types of predator. Here, we argue that the energetics underlying the relative and absolute sizes of natural enemies and their victims is the primary selective factor responsible for the evolution of these different trophic strategies.

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Although reproduction is the overriding theme of the drama of life, feeding and being fed on, are the key subplots. For this reason, ecologists and evolutionary biologists are well aware of the importance of predation and are increasingly considering the role of parasitism in the control of populations and the structure of communities. A barrier to fully incorporating concepts about parasitism has been the daunting diversity of parasitic strategies that exist; this can impair our modeling efforts, blur theoretical predictions and retard basic communication about parasitism.

Early population modelers recognized a need to define four types of distinct NATURAL ENEMY (see Glossary): PREDATORS, PARASITOIDS, MICROPARASITES and MACROPARASITES. It is about this diversity of trophic strategies that we ask: are these distinct trophic strategies? Are there any other strategies that these modeling categories do not adequately cover? What life-history traits of a natural enemy define the axes of the adaptive landscape of trophic strategy? In addition, what are the ecological consequences and corollaries of these strategies? Our conclusion morphs the old adage 'you are what you eat' into 'you are how you eat'.

Evolutionary concepts of trophic strategies

Natural enemies take nourishment from a victim (host) or victims (prey) using a variety of trophic strategies. The particular strategy that an individual natural enemy uses can vary from one victim to the next (we treat asexually produced progeny in a host as equivalent to a single genetic individual). For instance, a PARASITE might interact very differently with an intermediate host compared with a definitive host. Our goal is to determine the underlying key factors that distinguish different trophic strategies used by natural enemies. We do this with a factorial application of four dichotomies, each of which describes an aspect of enemy–victim interactions (expanding here on earlier work [1]*). A logical trophic strategy emerges for ten of the potential 16 cells (Fig. 1). These dichotomies apply to all taxa, including animals, plants, PATHOGENS, phages, bacteria, microbes and helminths, enabling us, for example, to evaluate herbivores as types of parasites or predators of plants. Nonetheless, our approach does not supplant existing terminologies based on the taxonomy of the victim (herbivore or carnivore). We intentionally avoid creating new terminology by adapting existing terms that, although they might refer traditionally to specific taxa, apply conceptually to other organisms.

The first dichotomy: does the enemy attack more than one victim?

How do parasites differ from predators? In his authoritative work on parasite ecology and evolution, Combes [2] considers that it is the durability of the

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