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FORUM

Measuring and Incorporating Vulnerability into Conservation Planning

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ABSTRACT / Conservation planning is the process of locating and designing conservation areas to promote the persistence of biodiversity *in situ*. To do this, conservation areas must be able to mitigate at least some of the proximate threats to biodiversity. Information on threatening processes and the relative vulnerability of areas and natural features to these processes is therefore crucial for effective conservation planning. However, measuring and incorporating vulnerability into conservation planning have been problematic. We develop a conceptual framework of the role of vulnerability assessments in conservation planning and propose a definition of vulnerability that incorporates three dimensions: exposure, intensity, and impact. We review and categorize methods for assessing the vulnerability of areas and the features they contain and identify the relative strengths and weaknesses of each broad approach. Our review highlights the need for further development and evaluation of approaches to assess vulnerability and for comparisons of their relative effectiveness.

The current rate of species extinction is estimated to be 100 to 1000 times that of the natural background rate in the absence of humans (Pimm and others 1995). The processes driving these losses and threatening biodiversity can be classified as either ultimate or proximate (Lambin and others 2001). Ultimate processes are the root causes of biodiversity loss that operate indirectly at global, national, or other broad levels and originate from societal, economic, demographic, technological, political, and cultural factors. Proximate processes are the physical expressions of ultimate processes and threaten biodiversity directly at regional or local scales. Proximate threatening processes include logging, clearing, agricultural expansion, urbanization, grazing, expansion of infrastructure, mining, invasion by exotic species, hydrological changes, and salinization. Factors that predispose areas to proximate threats include environmental variables such as climate, soil type and topography, as well as geographic variables such as proximity to population centers and infrastructure, including roads and irrigation systems.

Conservation planning is the process of locating and designing strict reserves and off-reserve management

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arrangements (hereafter collectively ''conservation areas'') to promote the persistence of biodiversity *in situ*. Conservation areas can be important in mitigating proximate threats arising from activities such as agriculture, logging, mining, or grazing of domestic livestock. In some cases, and depending on resources for management, conservation areas can also prevent or reduce the spread of exotic plants and animals and mitigate the adverse effects of changes to fire regimes and other natural disturbances. However, given the limitations of conservation areas in preventing all threats to biodiversity, conservation planning must operate as part of a broader conservation strategy that also addresses the ultimate causes of biodiversity loss through policy, legislation, education, and economic instruments.

Many methods have been developed to undertake conservation planning in a systematic manner (Margules and Pressey 2000). Since the early 1980s, there has been an increasing emphasis on meeting quantitative targets within a system of conservation areas that are complementary in terms of the features they contain. We define features here as the natural entities of concern to conservation planners. These include species, populations, species assemblages, and land types (e.g., vegetation units). A framework for systematic conservation planning can be described by several distinct stages (modified from Margules and Pressey 2000, Cowling and Pressey 2003):

1. Identify and involve stakeholders
2. Assess opportunities and constraints for conservation implementation
3. Identify goals for the planning process
4. Compile data on the planning region
5. Formulate conservation targets for biodiversity features
6. Review target achievement in existing conservation areas
7. Select additional conservation areas
8. Implement conservation actions in selected areas
9. Maintain the required values of conservation areas.

Threatening processes and the relative vulnerability of areas and features to these processes can influence every stage of this framework. For example, it might be important to involve those stakeholders whose activities represent proximate threats to biodiversity that can be mitigated by conservation planning (stage 1). Knowledge of ultimate and proximate threats helps in understanding the opportunities and constraints for planning (stage 2) and identifying the activities needed to complement and support conservation areas.

Goals for planning (stage 3) are most effectively selected after identifying the threatening processes that can be mitigated by conservation areas. At this stage, the benefits of conserving areas can be evaluated in terms of reduced risk to areas and the features they contain. If the reduction in risk from conserving an area is negligible, then other forms of threat mitigation will probably be necessary, with modification of goals as appropriate. For example, conservation areas might be ineffective in excluding exotic plants and animals or mitigating hydrological impacts from nearby developments unless complemented with intensive on-site management and changes in land use patterns throughout the planning region.

For those threatening processes that can be mitigated by conservation areas, spatially explicit data are required on their present and likely future distributions and, if possible, on their expected intensities, along with information on biodiversity (stage 4). Importantly, data on threats might need to be reviewed every few years to account for changes in their sources and patterns. Information on threatening processes can be used to guide the formulation of targets for features to be conserved (stage 5). Some recent studies have recommended that features that are most threatened should have larger targets (e.g., Burgman and others 2001, Pressey and others 2003).

A review of the existing conservation areas within the planning region is important to identify the extent to which targets have already been achieved (stage 6). The reasons for deficiencies in the coverage of existing conservation areas can often be understood in relation to spatial variation in the potential for extractive activities and the threats that they present. Conservation areas in many regions tend to be on land that has little extractive potential (e.g., Scott and others 2001).

If additional conservation areas are required to achieve targets (stage 7), information on the threatening processes operating in the planning region can assist conservation planning. Where options exist for target achievement, vulnerable areas can be avoided so that targets are achieved, as far as possible, in areas without liabilities for implementation and management (Wikramanayake and others 1998; Cowling and others 2003a). Considerations of defensibility, or avoiding vulnerable areas, can be especially important if resources are likely to be insufficient for effective management (Peres and Terborgh 1995).

When implementation of new conservation areas commences (stage 8), an important consideration in scheduling their implementation will often be their relative vulnerability. The more vulnerable areas might receive higher priority, especially if there are few or no alternative areas available to protect the features they contain (Pressey and Taffs 2001, Noss and others 2002, Lawler and others 2003). This strategy can minimize the extent to which targets are compromised by threatening processes during the frequently protracted process of establishing conservation areas on the ground (Pressey and others 2004). Analogous approaches have been used to prioritize regions or countries for conservation investment (Myers 1988, Dinerstein and Wikramanayake 1993, Balmford and Long 1994, Sisk and others 1994, Beissinger and others 1996, Myers and others 2000).

To maintain the values of areas conserved (stage 9), the effectiveness of their management in mitigating threats from external and internal sources can be monitored. Information from this stage might also feed back to earlier stages. Management problems might, for example, indicate the need for altered approaches to locating and designing conservation areas.

Clearly, information on threatening processes and the relative vulnerability of areas and features to these threats pervades the process of conservation planning. This is appropriate, because conservation planning is one of society's responses to threatening processes.

Conservation planning nonetheless lacks a consistent definition of vulnerability. Conservation planners also apply diverse methods to assess vulnerability depending on available or preferred sources of data, but without necessarily understanding the merits and limitations of alternative approaches. The aim of this review is to promote a more structured discussion of vulnerability and its applications in conservation planning. We propose a definition of vulnerability, describe the main methods for assessing it, discuss the relative strengths and weaknesses of each broad approach, and identify some important, unresolved issues for measuring and applying vulnerability.

Defining Vulnerability

Threatening Processes

The vulnerability of areas or features can be defined only in relation to one or more proximate threatening processes. A threatening process ''threatens or may threaten the survival, abundance or evolutionary development of a native species or ecological community'' (Commonwealth of Australia 1992, 1999). In itself, this broad definition does not identify the species and communities that are threatened or the levels of threat they face. For legislation or IUCN listing (e.g., IUCN 2001), these decisions are usually resolved through formal processes that include nomination, public and scientific consultation, assessment by experts against criteria, and revisions of official lists. In conservation planning, these formal listings are complemented or replaced by a variety of methods that interpret and map the distributions of threatening processes and the areas or features they might affect.

In this review, we consider proximate threatening processes whose extent can be mapped spatially within planning regions. These typically include transformation by urbanization, the expansion of infrastructure, the spread of invasive plants and animals, and extractive land uses such as agriculture, grazing, logging, and mining. For some regions, it might also be possible to apply our framework and definitions to altered disturbance regimes and climate change.

Three Dimensions of Vulnerability

Vulnerability is broadly defined in the risk analysis and hazard assessment literature as the potential for loss (Cutter 1996, Dilley and Boudreau 2001). Within the realm of conservation planning, Pressey and others (1996) defined vulnerability as the likelihood or imminence of biodiversity loss to current or impending threatening processes. We extend this definition by distinguishing three dimensions of vulnerability (Figure 1). Because conservation planning is fundamentally spatial, our definition and review primarily concern ''areas.'' These are any discrete part of the landscape considered by planners as units of evaluation and management, including ownership parcels, catchments, vegetation fragments, regular grids, or sometimes whole vegetation types or soil types. Two of our dimensions of vulnerability—exposure and intensity—apply to areas and consequently to the features they contain. The third dimension—impact—applies only to features. However, this shift in focus from areas to features does not apply in all cases. Vegetation types, for example, might serve both as areas of evaluation, to which exposure and intensity apply, and as the features that are impacted by the threatening processes, in this case through changes in structure and composition.

Exposure (Figure 1) can be measured either as the probability of a threatening process affecting an area over a specified time or the expected time until an area is affected. An analogous term for exposure from the risk analysis and hazard assessment literature is risk (Rowe 1977, Cutter 1996, Harwood 2000). The predisposition or sensitivity of an area to a threat is a component of exposure. If defined as a probability, exposure values would range from 0 (no vulnerability) to 1 (high vulnerability); if defined by time, exposure values would range from many years (low) to 0 years

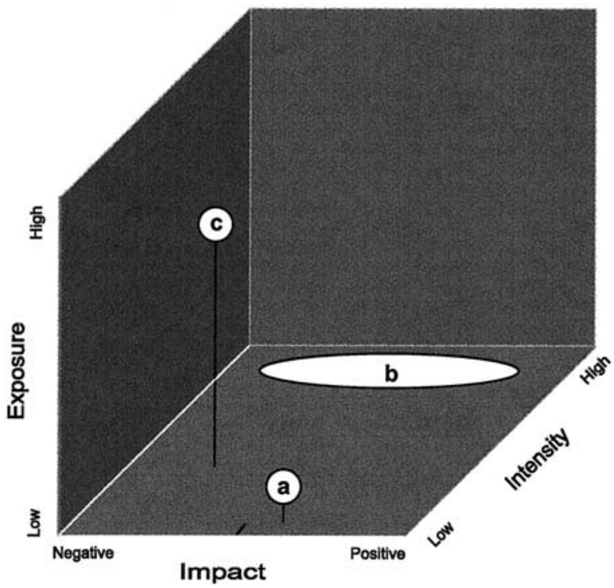

Figure 1. Three dimensions of vulnerability: exposure, intensity, and impact. The center of the impact axis indicates no impact. Areas and the features they contain are exposed to threatening processes of differing intensity. The impact of a threatening process on features is determined by its intensity. Therefore, impact varies only across the plane of intensity. The position of an area or a feature along the three axes will vary between threatening processes. Point a: An area with low exposure to a low-intensity threat might result in a positive impact (e.g., low-intensity selective logging in rainforest with the newly created gaps promoting the growth of an understory shrub). Point b: Areas presently inaccessible to grazing with low present exposure could have the potential to experience high-intensity grazing if infrastructure limitations (e.g., lack of watering points) were overcome. Expected impacts on plant species due to heavy grazing in these areas could vary from strongly negative to strongly positive. Point c: An area could have a high present exposure to logging due to its proximity to timber mills, but its terrain might limit the intensity of timber harvesting. The impact on animal species of interest is expected to be moderately negative.

(imminent or high). Both measures have analogies in population viability analyses (Shaffer 1990), which refer to both likelihood of decline or extinction and time to extinction or to some threshold population level. Exposure is commonly measured categorically, as in high, medium, or low suitability for agriculture (e.g., Pressey and Taffs 2001, Neke and du Plessis 2004) but has also been measured on continuous scales (e.g., Serneels and Lambin 2001). In most cases, measures of exposure implicitly refer to both probability of exposure and time until exposure. Methods for estimating exposure commonly produce maps of regions that show which areas are exposed to varying degrees (Figure 2).

Within risk assessments, measures of intensity (Figure 1) might include measures of magnitude, frequency, and duration (Harwood 2000). For biodiversity, the intensity of a threat can take many forms, including density of stock carried, cubic meters of timber extracted per hectare of a forest type, or the density of an invasive plant species (Rouget and others 2003). Intensity can also be estimated categorically, relative to observed intensities of threatening processes across the planning region or more broadly. Like exposure, intensity is amenable to mapping across whole planning regions (Figure 3).

Impact (Figure 1) refers to the effects of a threatening process on particular features. In the risk assessment literature, impacts are sometimes referred to as outcomes or specific risks (Rowe 1977, Dilley and Boudreau 2001). Impact could indicate effects on distribution, abundance, or likelihood of persistence of species and will likely reflect life history traits (Elmqvist and others 2003, Pereira and others 2004). Animal species dependent on old growth forests, for example, respond differently to logging than species that inhabit a variety of post-logging stages. Impact might also depend on the spatial pattern of the threatening process, for example, whether sufficient connectivity is retained between old growth patches after logging operations. In Figure 1, large values to the left along the impact axis indicate strong negative impacts of the threatening process, which could be manifested as serious declines in abundance or by local or regional extinction (Figure 4). Large values to the right indicate strong positive impacts, for example, the increase in density of some native plant species under heavy stock grazing (James and others 1999, Figure 4). Values close to the middle of the axis indicate little impact (Figure 4). Some authors have subdivided impacts into initial effects and likelihood of recovery (e.g., Nilsson and Grelsson 1995). The impact of a threatening process is moderated by the level of threat mitigation offered by conservation areas.

Areas and the features they contain can probably be placed throughout much of the three-dimensional space in Figure 1. Exposure and intensity can be positively related. The proximity of natural areas to urban centers could determine both their likelihood of conversion to housing development and the intensity of that development during a specified period (Theobald and others 1997). In other situations, exposure and intensity might be unrelated. An example is the situation in which intensity of grazing is related to distance from watering points (Pringle and Landsberg 2004),

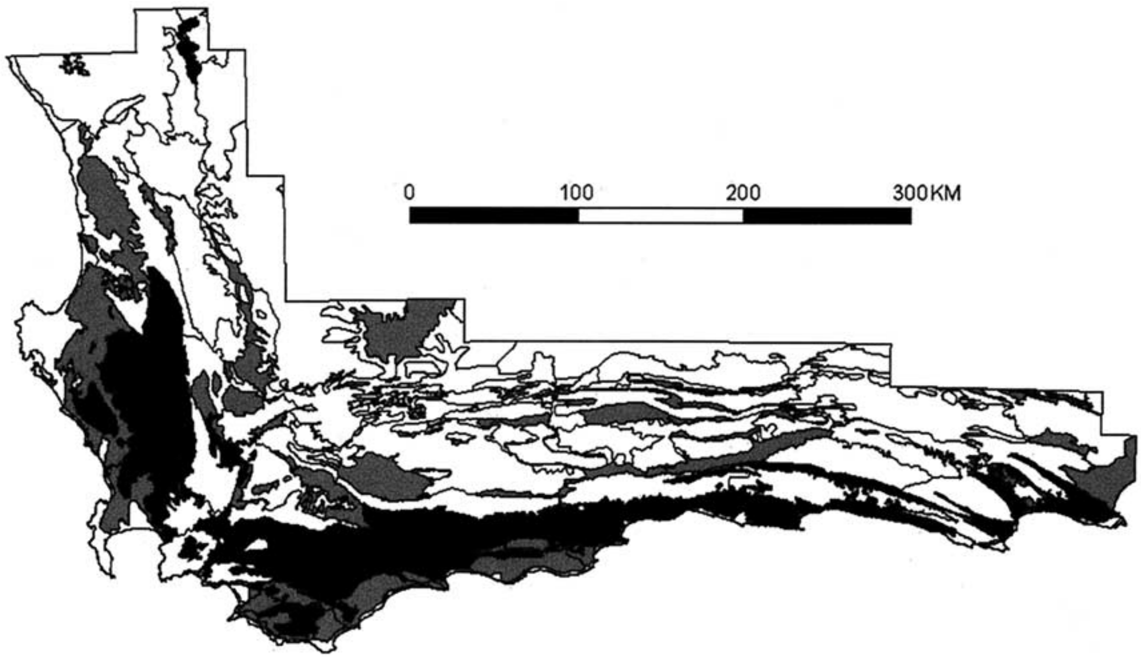

Figure 2. Categories of exposure (agricultural suitability) for 102 broad habitat units in the Cape Floristic Region of South Africa. Black areas have high exposure (high suitability), gray areas have moderate exposure, and white areas have low exposure (low suitability). See Cowling and Heijnis (2001) for methods used to map broad habitat units and see Rouget and others (2003) for derivation of categories of agricultural suitability.

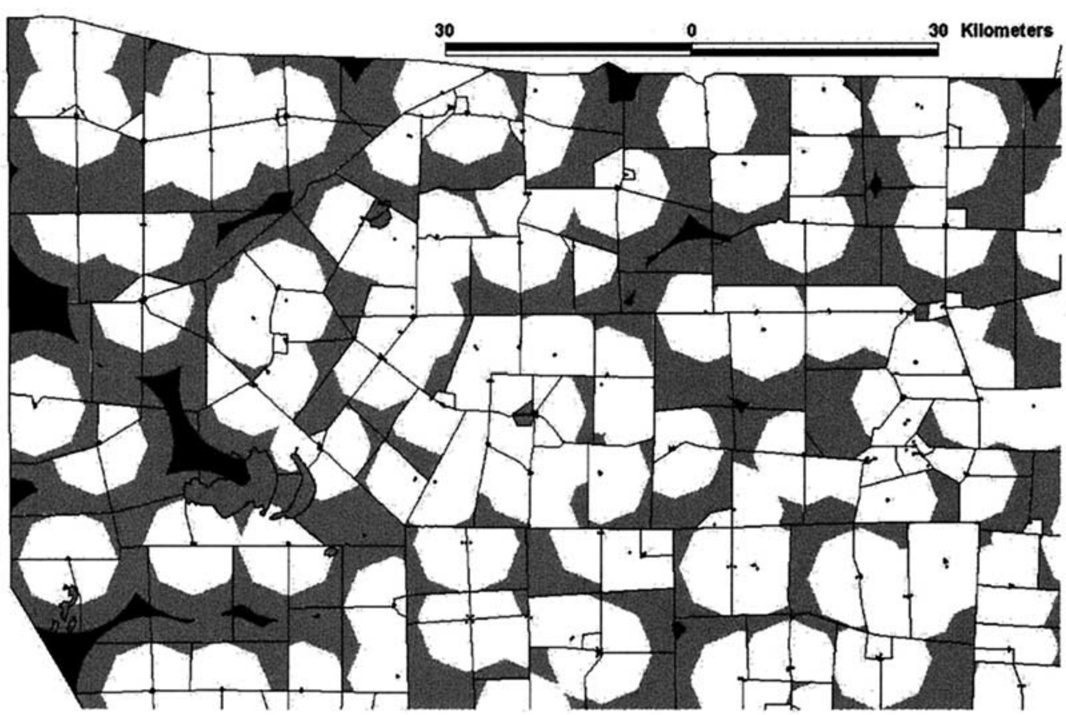

Figure 3. Intensity of stock grazing in South Australian rangelands as indicated by distance from watering points. The white regions represent areas of highest intensity (up to 6 km from a watering point). Gray areas have intermediate intensity (between 6 and 9 km from a watering point). Black areas have lowest intensity (greater than 9 km from a watering point) (Craig James and others, unpublished data).

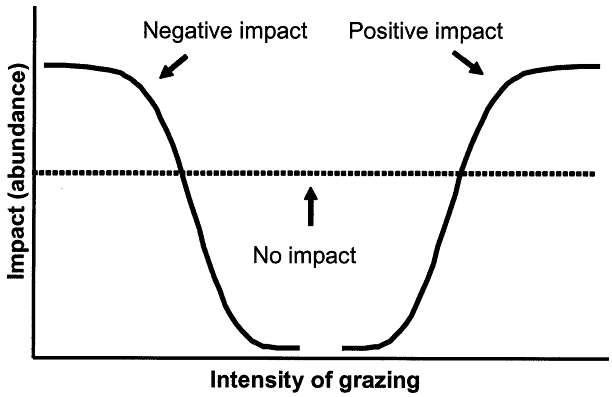

Figure 4. Differing impacts on species due to different levels of grazing intensity in Australian rangelands (modified from Biograze 2000). The intensity of grazing is inversely related to distance from artificial watering points. Grazing has positive impacts on species that are most abundant in areas of high intensity (close to watering points; right end of horizontal axis). Grazing has negative impacts on species that are most abundant in areas of low intensity (distant from watering points; left end of horizontal axis). Some species show no consistent change in abundance along the gradient of grazing intensity.

but all areas within a maximum radius of a watering point are equally likely to be exposed to grazing at some level in the course of a year. Negative relationships are also feasible (Figure 1). An area could have a high present exposure to logging because of its proximity to timber mills but its terrain or tree species might limit the intensity of timber harvesting. Alternatively, distant or inaccessible areas with low present exposure to logging could have the potential for highly intense extraction if limitations on access were overcome. The impacts on features are elicited by intensity, not by exposure. Impacts from low-intensity threats are likely to be generally small. Stronger negative or positive impacts will probably arise from higher levels of intensity. Conceivably, impacts on the same species could shift from positive to negative, and vice versa, as intensity increases. An example would be an understory shrub in dense forest that is more abundant under low-intensity logging but eliminated by high-intensity logging.

Areas of particular concern for conservation planners will have high exposure to highly intense threatening processes. Features of concern will be those occurring in such areas and experiencing strongly negative impacts, especially those with distributions similar to or smaller than the expected extent of the threatening process.

Mapping Vulnerability

An important task of conservation planners is to produce spatially explicit data on these three dimensions of vulnerability. For exposure (Figure 2), this requires spatial predictions of the future distribution of threatening processes. This may be based on the current distributions of threats and knowledge of variables that could predispose areas or features to those threats. In the case of vegetation clearing, these variables often include agricultural suitability, terrain, and proximity to infrastructure or population centers (Mertens and Lambin 1997, Geoghegan and others 2001, Serneels and Lambin 2001). The exposure of areas or the features they contain can be used to inform one another (Kershaw and others 1995, Beissinger and others 1996, Flather and others 1998, Troumbis and Dimitrakopoulos 1998, Brooks and others 2001). For example, the exposure of areas to agricultural clearing has been predicted from the suitability for agriculture of the land types occurring within them (Pressey and Taffs 2001). In reverse, the exposure of land types and species to urbanization and invasive alien plants has been predicted from threat profiles of the areas in which they occurred (Rouget and others 2003).

Spatial predictions of intensity are less common than predictions of exposure because they require information to discriminate between areas likely to be affected at different levels (Laurance and others 2001). For some threats, such as clearing, conservation planners must often assume that intensity is binary. Areas are considered to be either cleared or not (Mertens and Lambin 1997, Geoghegan and others 2001, Schneider and Pontius 2001), intensity is assumed to be either high or low, and the primary concern is exposure. Although this might sometimes be the case, this approach is probably more often necessitated by limitations of data, including the coarse resolution of maps. Clearing at scales finer than the resolution of mapping and the selective removal of plant species cannot be identified from remote sources, making both mapping of present intensity and spatial prediction of future intensity difficult (Lambin 1999). Although a binary approach might sometimes also be necessary for other threatening processes, intensity is a key consideration for threats, such as stock grazing and logging, which vary widely in intensity. The intensity of grazing, for example, is known to vary according to factors such as proximity to watering points, distance from population centers, timing of stock introduction, and the traditions and socioeconomic status of communities (James and others 1999, Lambin and others 2001). If variations in these factors can be reliably linked to variations in grazing intensity (e.g., Pringle and Landsberg 2004), then gradients of intensity can be predicted and mapped (Figure 3).

Probably the most difficult dimension of vulnerability for planners to deal with is impact. This often requires feature-specific information on the effects of different levels of intensity, spatial information on features relative to variations in intensity, and ways of integrating this information across assemblages of species, sets of vegetation types, or other groups of features (Schumaker and others 2004). Impact also has a strong temporal element, because it might be either immediately apparent or delayed. Vegetation clearing is often assumed to produce a binary impact. Areas are either uncleared, with their biodiversity initially intact (no impact if fragmentation effects are ignored), or cleared, with their biodiversity eliminated (the impact is strongly negative). Assumptions of binary impacts are most problematic for threatening processes that alter the structure and composition of areas. For threats like logging, grazing, and altered fire regimes, the native species in an area are likely to range across the negative and positive halves of the impact axis in Figure 1.

Vulnerability and Uncertainty

There is commonly some uncertainty around assessments of exposure, intensity, and impact, and there are benefits in exploring this uncertainty (Possingham 1996). Where data are available, uncertainty estimates might be obtained using statistical methods (e.g., upper and lower confidence limits on vulnerability predictions) or through summarizing the opinions of experts as a probability distribution (Figure 5). If it is not feasible to obtain this information for every area in a planning region, it could be derived for broad subdivisions of the region considered to have roughly equal vulnerability. Such assessments could encourage the refinement of vulnerability predictions with the aim to minimize the misallocation of conservation effort. If vulnerability is overestimated, scarce resources could be allocated to areas that do not, in fact, need protection. Conversely, if vulnerability is underestimated, areas that are, in fact, threatened could be overlooked and have their conservation values reduced or eliminated. Analysis of uncertainty could also help to identify the sensitivity of predictions to input data and assumptions and the sensitivity of planning outcomes to errors and uncertainties associated with predictions.

Vulnerability and Scale

Decisions about geographic scale have important implications for analyses of vulnerability. Some scale

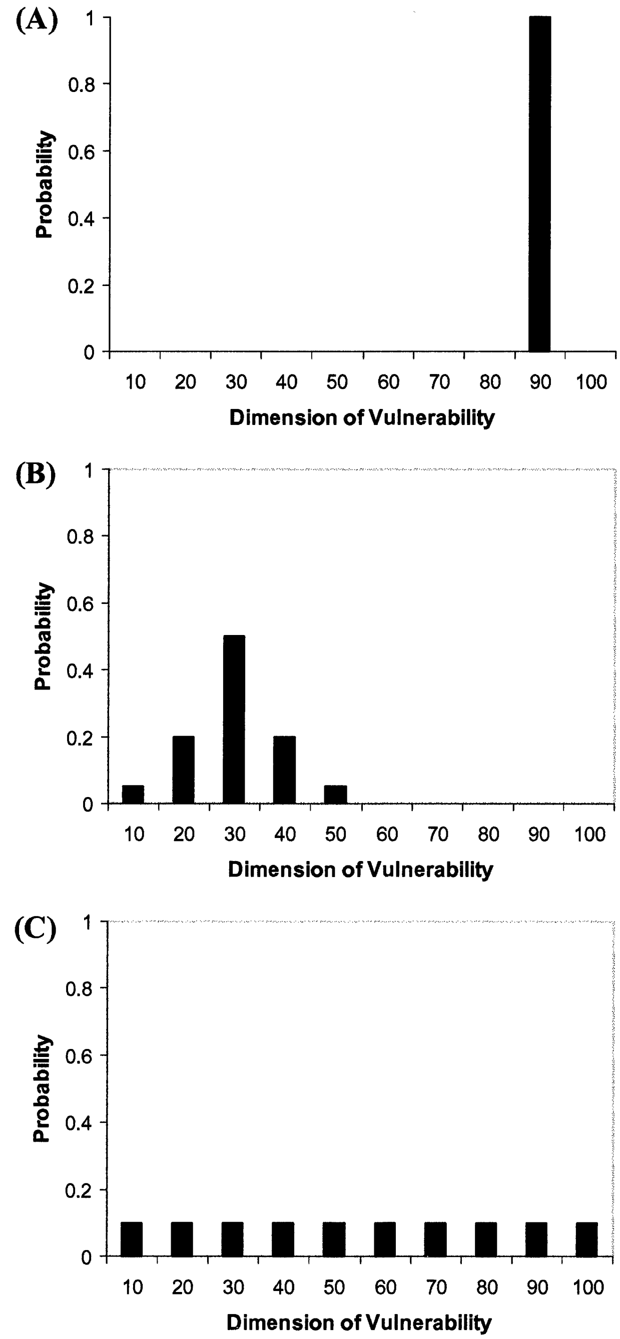

Figure 5. Levels of confidence in estimating dimensions of vulnerability. The dimension of vulnerability, in this case either exposure or intensity, is measured on a notional scale from 1 to 100, relative to some baseline. Equivalent graphs for impact would have both positive and negative values on the horizontal axes. (A) High confidence that an area has high exposure to a threatening process or will be affected at high levels of intensity. (B) Moderate confidence that exposure or intensity is low. (C) No confidence in predictions of exposure or intensity.

effects relate to the resolution with which features are mapped. Rouget (2003) compared a broad-scale assessment of habitat transformation (1:250,000) with another assessment at the scale of individual land parcels (1:10,000). Transformation of heavily fragmented vegetation types was overestimated at the broad scale because small remnants were not detected. These remnants were identified with fine-scale mapping as high priorities for conservation action, partly due to their vulnerability to clearing.

It is often assumed that vulnerability is homogeneous within areas and across the features contained in each area (e.g., ratings for exposure to urbanization and alien plants by Rouget and others 2003). Even without this assumption, it can still be useful to allocate single vulnerability ratings to whole areas based on the different values of the features they contain (e.g., ratings for exposure to agricultural clearing by Rouget and others 2003), with the ratings then used in planning (e.g., Cowling and others 2003a). These approaches inevitably obscure spatial variation in vulnerability within areas that can be revealed by more detailed analyses. Importantly, finer-scale analyses are likely to show that features within single areas differ in exposure to and expected intensity of threatening processes. A precautionary approach is therefore to work with maximum values when estimating the vulnerability of whole areas (Rouget and others 2003).

The effects and limitations of spatial generalization of vulnerability are most apparent at the scale of whole planning regions. When regions are used as units of conservation prioritization across continents or countries, two types of error can occur. First, and most obviously, high-priority regions will likely contain areas of low priority, based on their vulnerability and biota. Second, and of greater concern, some low-priority regions can contain areas of high priority in national or continental contexts, due to their high vulnerability and high conservation value (Pressey and others 2000, Bates and Demos 2001, Veech 2003). These priority areas are obscured and easily overlooked in global or national priority settings that use regions as their resolution.

Methods to Assess Vulnerability

We reviewed methods that have been used to assess vulnerability and categorized them into four groups (Table 1), based mainly on the types of data used. The main assumptions of these methods are listed in Table 2. All methods estimate exposure, but some deal also with intensity and impact (Table 1). A limitation of all approaches is that they cannot account for ultimate factors that mitigate or increase proximate threats. These factors are both difficult to interpret spatially and highly dynamic. Examples include changes in global markets and government policies, such as subsidies and tax incentives, which can make the clearing of previously unsuitable land economically viable. In some cases, vulnerability depends on the attitudes of individual landholders. Attitudes can be understood through approaches such as household surveys (Geoghegan and others 2001) but are prone to change as people move and can be difficult to depict across large regions.

All the methods in Table 1 have been used at a variety of spatial scales and resolutions and in countries with differing levels of development, even in those typically regarded as data-poor, such as Ecuador, Honduras, Brazil, India, Mexico, and countries within sub-Saharan Africa. The data underpinning many of the methods are globally available and so most methods are applicable anywhere, at least at a coarse scale.

Group 1: Methods Based on Tenure and Land Use

The first method in this group (Table 1) infers the vulnerability of features from their relative amounts within conservation areas. The result is sometimes a binary classification of vulnerability, with features identified as either adequately protected (not vulnerable) or inadequately protected (vulnerable). However, the frequent assumption of uniform exposure and intensity outside conservation areas (Table 2) is often invalid (MacDougall and Loo 2002) because some other land uses can mitigate threats (Carroll and others 2004). This is acknowledged in the second method in this group that uses information on permitted or projected land uses (Table 1). Such methods often result in a categorical classification of vulnerability (Fearnside and Ferraz 1995; Jennings 2000; Stoms 2000). Information on the expected intensity of a threat might also be obtained from such analyses if, for example, a land use plan stipulates limits on development or commercial activities (e.g., the permitted density of grazing stock). Land use plans depicting permitted or projected land uses are available at fine scales, but their geographic coverage is generally limited.

Many studies in this group assume that conservation areas are effective and not themselves exposed to threatening processes (Table 2). For some threats, such as hunting and weed invasion, this will often not be so. In some regions, conservation areas are not even secure from vegetation clearing (Peres and Terborgh

Table 1. Four general types of methods used to assess vulnerability

Concept	Method of measurement	Dimension of vulnerability	Examples of use
Group 1: Methods based on tenure and land use	(a) Vulnerability is estimated from coverage in existing conservation areas	(a) Exposure	(a) Dinerstein and Wikramanayake 1993; Beissinger and others 1996; Castley and Kerley 1996; Maddock and Benn 2000; Reyers and others 2001; MacDougall and Loo 2002
	(b) Vulnerability is estimated from permitted or projected land uses	(b) Exposure and intensity	(b) Scott and others 1993; Fearnside and Ferraz 1995; Abbitt and others 2000; Fairbanks and Benn 2000; Jennings 2000; Stoms 2000; MacDougall and Loo 2002; Ricketts and Imhoff 2003
Group 2: Methods based on environmental or spatial variables	(a) The past impacts of threatening processes are used to indicate the vulnerability of features. These values are then given to presently unaffected areas that contain the same features.	(a) Exposure	(a) Myers 1988; Dinerstein and Wikramanayake 1993; Balmford and Long 1994; Sisk and others 1994; Awimbo and others 1996; Pressey and others 1996; Mittermeier and others 1998; Kremen and others 1999; Abbitt and others 2000; Myers and others 2000; Pressey and others 2000; Reyers and others 2001; MacDougall and Loo 2002; Sierra and others 2002
	(b) Characteristics of areas or features exposed to threats in the past are used in qualitative or informal quantitative analyses to predict vulnerability.	(b) Exposure and intensity	(b) Sisk and others 1994; Beissinger and others 1996; Theobald and others 1997; Lathrop and Bognar 1998; Moyle and Randall 1998; Thompson and Jones 1999; Abbitt and others 2000; Maddock and Benn 2000; Stoms 2000; Wickham and others 2000; Reyers and others 2001; Theobald and others 2001; Sierra and others 2002; Villa and McLeod 2002; Pressey and others 2003; Rouget and others 2003; Veech 2003; Jackson and others 2004; McKee and others 2004; Reyers 2004
	(c) Characteristics of areas or features exposed to threats in the past are used in spatially explicit, quantitative models to predict vulnerability.	(c) Exposure and intensity	(c) Allen and Barnes 1985; Ludeke and others 1990; Treweek and Veitch 1996; Veldkamp and Fresco 1996; White and others 1997; Higgins and others 1999; Higgins and others 2000; Menon and others 2001 ; Barredo and others 2003; Kline and others 2003; Rouget and others 2003; Dark 2004; Gido and others 2004; Jackson and others 2004; Linkie and others 2004; McConnell and others 2004; Soares-Filho and others 2004; Wooldridge and Done 2004; Wilson and others 2005
Group 3: Threatened species are used to indicate vulnerability	The number of threatened species and their relative threat ratings are combined to indicate vulnerability.	Exposure, intensity, and impact	Kershaw and others 1995; Beissinger and others 1996; Freitag and van Jaarsveld 1997; Ceballos and others 1998; Flather and others 1998; Troumbis and Dimitrakopoulos 1998; Brooks and others 2001; MacDougall and Loo 2002; Sierra and others 2002; Andelman and Willig 2003; Danielsen and Treadaway 2004; McKee and others 2004
Group 4: Experts decide on relative vulnerability	Opinions are sought from experts on the relative vulnerability of areas or features.	Exposure, intensity, and impact	Richter and others 1997; Ricketts and others 1999; Dinerstein and others 2000; Noss and others 2002; Nel and others 2004

Table 2. Main assumptions of the four general types of methods used to assess vulnerability

Assumption	Group 1	Group 2	Group 3	Group 4
Vulnerability is uniform outside conservation areas	✓			
Conservation areas are not vulnerable	✓	✓[a]		
Survey effort is not biased relative to conservation areas	✓			
Patterns of past threats indicate future patterns		✓	✓	
Estimates of the historical and current extent of a threat are accurate		✓		
Variables chosen for analysis are related to the relative vulnerability of areas		✓[b]		
Relative weightings of variables are accurate		✓[b]		
Variables operate independently		✓[c]		
Threatened species indicate vulnerable areas			✓	
Similar combined scores derived from the number and threat level of species in different areas are equivalent			✓	
Information from experts is accurate				✓

[a]When existing conservation areas are excluded from predictions.
[b]Groups 2b and c only.
[c]Group 2b only.

1995; Menon and others 2001). This assumption also depends on the categories of conservation areas recognized in the analysis and the extractive uses they permit (IUCN 1994). Another assumption of group 1 methods is that survey effort for species is not biased toward or away from conservation areas (Table 2). If it is biased toward them, then the vulnerability of species might be underestimated because there will be relatively many unrecorded occurrences outside formal protection.

Group 2: Methods Based on Environmental and Spatial Variables

Methods in this group use the same types of data, but we have split them into three subgroups according to their ways of converting observations into predictions (Table 1).

Methods in group 2(a) identify the extent of past impacts on features such as species or vegetation types and use these data to predict future impacts on the same features. The predictive ability of these methods is limited by the need for a consistent map of features across the study region. Typically, these methods have assessed exposure.

The other methods in this group differ from those in 2(a) by identifying the underlying spatial (e.g., proximity to infrastructure, urban expansion) and intrinsic environmental characteristics (e.g., soil type, slope, and climate) believed to have predisposed areas to threatening processes in the past. Presently unaffected areas that share these characteristics are then identified to predict the exposure, and potentially the intensity, of future threats in these areas. These methods can therefore resolve spatial differences in vulnerability within features, for example, in relation to slope or proximity to urban centers. Variables such as housing density and proximity to roads can provide information regarding the intensity of threats as well as the likelihood of exposure.

Group 2(b) methods include rule-based methods, spatial overlay analysis, and correlation. The results are informal (i.e., not using formal modeling procedures) spatially explicit predictions of relative vulnerability. Group 2(b) methods might improve on those in group 2(a) in two ways: (i) by focusing not just on past impacts but on the potential drivers or predictors of exposure, and (ii) by identifying combinations of factors such as land capability and proximity to irrigation infrastructure that could improve predictive accuracy.

Methods in group 2(c) also use information on the distribution of past impacts but, unlike those in 2(b), base their predictions about future vulnerability on more systematic and rigorous quantitative analyses. Analytical approaches used might include logistic regression (Ludeke and others 1990, Higgins and others 1999, Schneider and Pontius 2001, Serneels and Lambin 2001, Turner and others 2001), decision-tree modeling (Rouget and others 2003), and cellular automata (Barredo and others 2003). These methods can develop quantitative models that relate vulnerability, usually on a continuous scale, to several potential predictor variables concurrently. They differ from group 2(a) methods, which do not separate predictor variables. They differ in three main ways from group 2(b) methods. First, they can improve understanding of variation in vulnerability relative to variations in values of each predictor variable through the development of linear and nonlinear models. Second, they can combine variables and identify their relative influence on vulnerability by empirically determining weights. Third, they can identify interactions between variables, indicating, for example, whether the influence of soil type on exposure to clearing is greater in high rainfall areas than low rainfall areas. These methods have been used to assess exposure but could be extended to assess intensity by specifying a dependent variable on a scale representing the probability of the threatening process occurring at different levels of intensity (e.g., through use of a multinomial model; Augustin and others 2001).

Group 2 methods are based on several assumptions (Table 2). They assume that the pattern of past exposure to threatening processes is indicative of future patterns. Factors that undermine this assumption include extrinsic forces, such as changes in global markets, the development of new crops, technological advances, or the exhaustion of highly suitable areas and the consequent shifting of threats to less suitable areas. For these and other reasons, the reliability of predictions is likely to decrease as they are projected further into the future as the relationships between predictor variables and vulnerability change (Linkie and others 2004). Another factor is scale of mapping. The remaining areas of a broadly defined vegetation type that has been largely cleared might be on residual, rocky areas unsuitable for agriculture but too small to be distinguished. The exposure of such areas would be overestimated by a measure based only on previous vegetation loss. Similarly, minor clearing of a coarsely mapped vegetation type could result in underestimation of the vulnerability of pockets of vegetation on arable soils within it. These problems can be reduced by using data on past impacts that best reflect the time frame, geographical scale, and resolution of the contemporary factors believed to be driving the threatening processes of concern.

Group 2 methods also assume that estimates of the historical and current extent of threatening processes are accurate (Table 2). These estimates are, however, prone to error, as shown by the difficulties of estimating historical distributions of vegetation types (Allen and Barnes 1985, Noss 1985, Downton 1995, Lambin and Ehrlich 1997, Menon and Bawa 1998). A second and related issue, raised by Gaston and others (2002), is the need for a careful definition of "loss." At what stage does the alteration of a vegetation type (e.g., due to logging and drainage) equate to its loss?

Methods 2b and 2c assume that the correct predictive variables have been selected and that their relative weightings are accurate (Table 2). An assumption of group 2(b) methods is that multiple predictors operate independently. Their combination (through weighting and aggregation) to obtain an overall vulnerability value is likely to be largely subjective and arbitrary (Villa and McLeod 2002). For example, Neke and du Plessis (2004) averaged values for exposure to mining of coal and minerals to obtain an overall threat map for mining. The assumption of independence between predictor variables and the problem of aggregation can be avoided by methods in group 2(c).

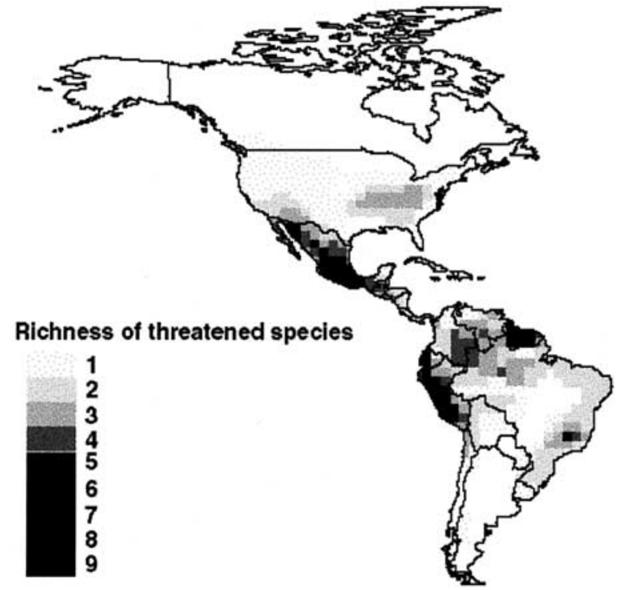

Figure 6. Information on the distribution of threatened bat species used to prioritize areas for conservation investment in the continental Western Hemisphere (modified from Andelman and Willig 2003). Threatened bat species are those listed as critically endangered, endangered, or vulnerable on the IUCN Red List. [Reproduced by Andelman and Willig 2003, with permission].

Group 3: Methods Based on Threat Status of Species

Several studies have identified vulnerable areas as those with high concentrations of taxa with high probabilities of extinction (Table 1, Figure 6). This information is mostly inferred from threat categories such as those in the IUCN Red Lists (IUCN 2001) or from other predictions of risk of extinction (Kohira and Ninomiya 2003, Lips and others 2003, Henle and others 2004, O'Grady and others 2004). The IUCN threat categories could be regarded as descriptive to the extent that they reflect previous or current patterns of exposure and intensity within species' ranges (e.g., a recorded reduction in distribution or abundance). Nonetheless, these methods can be predictive. Previous declines of species can signal further declines in the same region in the absence of effective threat abatement, and regional threat assessments (Gärdenfors and others 2001) might be reliable guides to vulnerability in other regions yet to be exposed to the same threatening processes. Furthermore, important reasons for including species on IUCN Red Lists are their naturally small ranges or population sizes that make negative impacts of future threats more likely. Importantly, one of the IUCN criteria is explicitly predictive. Species can be listed as endangered or vulnerable if there is a predicted population size reduction within the next 10 years or three generations, whichever is longer. Threat status of species usually integrates information on exposure, intensity, and impact (Table 1).

Group 3 methods have the advantage of identifying impacts of threatening processes, such as hunting and introduced species, that are difficult to map from remote sources and therefore seldom considered by methods in group 2. Group 3 methods are, however, often limited by a lack of data on the distribution of threatened species (MacDougall and Loo 2002). For some regions, data might be available only at a resolution much larger than the areas being considered for conservation action (Pressey and others 2003). Where adequate data are available, caution is needed in using the distribution of threatened species to assess vulnerability (for a discussion see Gaston and others 2002). There might be a mismatch between the distribution of threatened species and vulnerable areas, particularly if the species' threat categories reflect processes occurring in only part of the region, or even outside its boundaries. In addition, the vulnerability of an area is usually calculated from the weighted number of co-occurring threatened taxa, with weights reflecting threat level. Equivalent scores can therefore mean different things (Table 2). For example, an area with many moderately threatened taxa might have an overall vulnerability rating similar to that of an area containing a few taxa at imminent risk of extinction.

Group 4: Methods Using Expert Judgment

Opinions can be sought from experts to estimate exposure to threatening processes, and perhaps also intensity and impact. One of the major benefits is the potential to capture important information that is not available in regional datasets. Disadvantages of expert knowledge, if used without the support of other information, include its inevitable geographic and taxonomic biases (Cowling and others 2003b), lack of transparency and repeatability, and unknown inaccuracies. Pearce and others (2001) compared the accuracy of expert predictions of species distributions with those from statistical models. They found that models developed using only expert knowledge performed significantly worse than statistical models and related this difference to the differing resolutions of the assessments. Similar problems are likely for predictions of vulnerability. Mixed planning strategies are therefore desirable, combining the benefits of explicit analysis of available spatial data with the knowledge of experts (Pressey and Cowling 2001, Cowling and Pressey 2003).

There are many profitable ways of designing mixed strategies, and expert knowledge can improve vulnerability assessments throughout the planning process. It can assist, for example, in the following: identifying the important threatening processes operating in the planning region (Richter and others 1997, WWF and ICIMOD 2001, Noss and others 2002); determining the distribution of threatening processes (e.g., expert opinion might be crucial to distinguish between deforested and naturally unforested areas) (Rouget and others 2003); determining the intensity of threats (WWF and ICIMOD 2001); identifying important predictor variables and specifying their likely functional relationship for methods in group 2 (Pearce and others 2001, Wooldridge and Done 2004); validating and refining predictions of vulnerability; and advising on threat status of species (IUCN 2001).

Conclusions

Information on threatening processes and the relative vulnerability of areas and features to these processes is imperative for conservation planning. We have defined vulnerability according to three dimensions: exposure, intensity, and impact. Four broad methods have been used to assess vulnerability, classified according to the types of data used. Our classification of methods is intended to aid discussion of their relative merits and weaknesses. There are overlaps between the groups, and expert judgments (group 4) will potentially inform the other three. In many instances, a combination of approaches to assessing vulnerability will be appropriate to yield the best overall result, given restrictions on data availability (e.g., Jackson and others 2004).

There have been few comparisons of the results from the different methods, and as far as we are aware, none that validate alternative methods and compare their accuracies against ''true'' values. The comparative studies show, not surprisingly, that different methods give different results. Pressey and others (2000) found a weak correlation between the exposure of areas based on the extent of past clearing (Group 2a) and land capability (Group 2b). Rouget and others (2003) compared rule-based approaches (Group 2b) and statistical models (Group 2c) for agriculture and the spread of alien plants and found similar overall patterns of predicted exposure, but some substantial local differences. Without information on the relative accuracy of the predictions from the various methods, there is an unknown risk of prioritizing inappropriate areas, resulting in opportunity costs for biodiversity conservation.

Analyses are required to compare the methods and determine, through validation studies, which are most accurate. Ideally, this would involve a wide range of datasets to prevent the idiosyncrasies of particular regions influencing the results and to understand situations in which particular methods are superior. The datasets used to validate predictions should be independent of the datasets used to generate them; otherwise, an overly optimistic impression of predictive accuracy will result. Where it is not feasible to obtain completely independent data, a validation dataset could be extracted from the original dataset using data-splitting techniques (Power 1993). Alternatively, predictions could be validated at short-term intervals (e.g., 3 to 5 years). This approach need not hinder the planning process. Rather, it will likely take advantage of expansions of threatening processes proceeding in parallel with any conservation actions motivated by planning. Another alternative is to make use of historical datasets (e.g., land cover maps from 20 years ago) to predict present-day patterns of threats (see Brook and others 1997 for an equivalent validation of a population viability model). If predictions are acceptably accurate, then one would expect that, with some adjustment of the approach to account for contemporary proximate variables, future threats could be confidently predicted.

A comprehensive assessment of vulnerability would consider all of the threats affecting an area and also the dynamic responses of threats to conservation actions. Combining vulnerability scores for multiple threats is analytically tractable and could be achieved by differentially weighting threats to reflect their relative importance, ideally informed by their respective impacts. For example, Neke and du Plessis (2004) weighted the predicted exposure of grassland to forestry, agriculture, grazing, mining, and urban development according to their expected relative impacts on grassland biodiversity. The overall vulnerability of each area was then determined by the maximum score across all potential threats. Incorporating dynamics into assessments of vulnerability would require recognition that two commonly held assumptions might be false: that threats respond statically to conservation actions, and that mitigation measures will successfully eliminate the source of threats. For example, there is potential for conservation action to shift threats elsewhere (e.g., the redirection of a housing development to a nearby area). Conversely, conservation action that eliminates a source of alien plants could positively affect nearby areas by preventing spread. Developing a case-by-case understanding of how a threat will respond to different conservation actions will allow planners to anticipate both the positive and negative consequences of each action, thereby making conservation planning more effective.

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