6

Species-Level Strategies for Conserving Rare or Little-Known Species

Bruce G. Marcot and Curtis H. Flather

Conservation science is concerned, in part, with anticipating how natural or human-caused disturbance affects the pattern of commonness and rarity among the biota of a given area (Lubchenco et al. 1991; also see chap. 1). In this and the next chapter, we review various species- and system-level approaches proposed or applied to species conservation planning and assessment. We review the literature on approaches to conservation and management of individual species, groups of species, communities, and ecosystems, as published in peer-reviewed ecological outlets or in reports by land management agencies and nongovernmental organizations. We summarize conservation goals and objectives, discuss the scientific basis, and identify strengths and weaknesses of each approach for addressing conservation needs of rare or little-known (RLK) species of plants and animals. This review is intended to provide land and natural resource managers (hereafter, just "resource managers") access to this diverse literature and the basic information needed to select those approaches that best suit their conservation objectives and ecological context. A companion chapter (see chap. 8) outlines a process to evaluate how well each approach meets conservation objectives.

Categories of Approaches

In our review of the literature and drawing upon our knowledge and experience, we developed a classification system of approaches to conservation.

We classify the approaches in two broad categories. *Species approaches* result in conservation actions focused on providing for individual species or groups of species with common needs or common ecological characteristics. *System approaches* result in conservation actions focused on providing for community or ecosystem composition, structure, or function. These categories are artificial, in that species and system approaches are not necessarily mutually exclusive or independent. However, this dichotomy is useful for systematically presenting the numerous approaches appearing in the literature.

Other classifications of conservation approaches exist in the literature. One familiar to many resource managers is the dichotomy of coarse- and fine-filter management (Noss 1987). Although this classification has high recognition, it is also characterized by a high degree of ambiguity. For example, Overton et al. (2006) used these concepts to characterize landscape (coarse-filter) and microhabitat (fine-filter) use by pigeons (*Columba fasciata*) in western Oregon, whereas other authors use the concepts to refer to geographic scales of land-use planning and resource management. Schwartz (1999) defined a fine-filter approach as one where conservation efforts are focused on genetic, population, or species levels, and coarse-filter approaches are aimed at the community, ecosystem, or landscape levels. Under this simple distinction, coarse-filter approaches may correspond to what we refer to here as system approaches (e.g., Reyers et al. 2001; Armstrong et al. 2003), and fine-filter approaches may correspond to species approaches.

An underlying assumption in common among at least some of the species-level approaches is that meeting the needs of one or more species would serve, to a degree, to provide for other species and for broader ecological communities or systems. The operative phrase is *to a degree*, which begs for a risk analysis of threats to the species (i.e., a threats assessment; see chap. 4) and a risk management framework (see chap. 9) by which the likelihood of successfully meeting these assumptions is gauged.

As we have defined them, species approaches focus on meeting the needs of individual species or groups of species and include those focused on managing for the viability of individual species, surrogate species, or groups of species, plus geographically based approaches (table 6.1). For each of these approaches we describe the theory, concepts, and scientific basis for the approach and provide examples, where available, of their use in technical assessments, management or planning, and monitoring.

Table 6.1. *Summary of species approaches showing descriptions and main assumptions*

Name of Approach	Description	Main Assumptions
	VIABILITY APPROACH	
Conservation of individual species based on concepts of population viability	Likelihood of persistence of a population over a specified time period and geographic area	• The ecology, dynamics, demography, and/or genetics of a species is known well enough to estimate persistence probabilities • Viability analyses are realistic enough to incorporate and correctly represent the major factors influencing population size over time • Population isolation and fragmentation and lowering of genetic diversity have deleterious effects that can be modeled and predicted • Additional ecological considerations, including ecology and requirements of obligate symbionts, are known and accounted for
	SURROGATE SPECIES	
Umbrella species	A single species that represents the requirements of a portion or all of a species assemblage or community	• Single species somehow represent the requirements of other species or its biotic community • Successful conservation of an umbrella species confers protection of its species associates and its ecological community
Focal species	Target species that are identified for the purpose of guiding management of environments, habitats, and landscape elements in a tractable way	• Target species represent the response to stressors by other species • There is a greater degree of such representation if the focal species has stringent requirements of resource use and dispersal capability but that use the broadest area • Patterns of resource use, ecological processes, and aspects of system status associated with focal species are closely correlated with those of other species that the focal species is intended to somehow represent
Guild surrogates	One member or a subset of members serves as a surrogate for other members of the guild	• The status of one member or a subset of a guild is closely correlated with status of other members of the guild • Changes in environmental conditions affecting one or a subset of a guild would affect the other members in the same way or at least that population responses would be similar and significantly positively correlated

(continues)

Table 6.1. *Continued*

Name of Approach	Description	Main Assumptions
Habitat assemblage surrogates	One of a group of species that share common macrohabitats	• One or a subset of species of a species assemblage represents the full assemblage • All members of an assemblage respond the same to availability of macrohabitats and, more specifically, the response of one member is closely and positively correlated with responses of other members • Macrohabitat is the major factor affecting presence, distribution, and trend of associated species
Management indicator species	A variety of categories: federally listed species; species with special habitat needs; species that are hunted, fished, or trapped; species of special or social interest; and species for which population changes might indicate effects on or status of other species	• A set of species chosen as management indicator species somehow represent the array of management issues, conditions, objectives, and conservation concerns • Population status and trend of ecological indicator species represents others; that is, there is significant positive correlation in at least population trend (and possibly also size, distribution, and persistence likelihoods) between the indicator species and other species
Biodiversity indicator species	One or more species selected to indicate areas of high biodiversity or high biological productivity	• Indicator taxa are easier to locate and/or identify than indicated taxa • The number of species of indicator taxa serve as an index of the number of species of the indicated taxa • Locations of indicator taxa are highly correlated with locations of indicated taxa
Flagship species	Species that carry high public interest and garner social or cultural concern for their well-being	• Individual species can be identified in a given ecosystem and promoted in highly visible conservation programs that would entice positive interest and support by the general public
	Multiple Species	
Entire guilds	Species that share some common attribute of resource use or ecological role defining the guild	• All species of a guild would respond in like manner to presence or changes in environmental conditions
Entire habitat assemblages	Species that share common usage of some macrohabitat	• If the macrohabitat is provided, the requirements of the entire habitat group of species will be met

Name of Approach	Description	Main Assumptions
		• Species needs are mostly defined by the macrohabitat(s) they occupy
	GEOGRAPHICALLY BASED APPROACHES	
Locations of target species at risk	Known site locations of species at risk, which can include federally listed, heritage ranked, regionally sensitive, and other species	• Managing for known locations of target species likely contributes to ensuring persistence of the species and viability of its populations • Known locations are where the organism finds suitable resources and habitat conferring local persistence • Lack of "protection" of such sites confers a high likelihood of adverse anthropogenic stress on the organism and its environment, leading to local extirpation • Such organisms have low resilience or resistance, and high sensitivity, to disturbance
Species hot spots or concentrations of biodiversity	Global areas where a high number of endemic or other species are found in a relatively small geographic area	• Species' distributions are structured and nonrandom • Geographic patterns of species richness covary among at least some taxa • Hot spots represent ecological communities and ecosystems and their persistence
Reserves or protected areas	Areas delineated to represent, complement, or otherwise efficiently include species and biodiversity elements	• Protecting such areas from specific human activities or disturbances will provide for species or system persistence

Conservation of Individual Species Based on Concepts of Population Viability

This approach entails developing a conservation strategy for a given geographic area that incorporates quantitative or qualitative population viability analyses of the affected species. Such conservation strategies provide management guidelines based on an understanding of the life history, habitat needs, and factors that threaten the species. This usually includes recommendations for maintaining or restoring habitats necessary for various life functions such as breeding, foraging, roosting, dispersal, and migration. It may also include guidelines for managing conditions that would otherwise place a species at risk, such as discouraging the species' predators or competitors, or mitigating human disturbance factors (Holthausen et al. 1994). Such guidelines can take the form of designing reserves for particular species.

Methods for quantifying population viability of target species have become common in the conservation biology literature (see review by Beissinger and McCullough 2002). Population viability is generally defined as the likelihood of population persistence over a specified time period, over a specific geographic area, and over a specified population size. Population viability is thus seen as a probabilistic event, and viability outcomes are cast as odds or likelihoods. Quantitative population viability models can be useful in comparing relative population growth rates among populations (Beissinger and Westphal 1998) and can provide insights into how environmental factors induce fluctuations in rates of reproduction, mortality, and growth (Burgman et al. 1993).

A rather massive and relatively recent literature exists on metapopulation dynamics, modeling, demography, and viability evaluation of most taxonomic groups. This literature is, by definition, species-specific and based on autecological data or assumptions. Some examples include viability assessments of vascular plants (Satterthwaite et al. 2002), mollusks (Taylor et al. 2003), insects (Ranius 2000), fish (Ratner et al. 1997), reptiles (Doak et al. 1994), birds (Haig et al. 1993), and mammals (Armbruster and Lande 1993).

A main assumption of the population viability approach is that enough is known about the ecology, dispersal and colonization dynamics, demography, and/or genetics of a species to reasonably predict future sizes of populations and distributions of organisms, from which probabilities of persistence (or, conversely, extinction) can be estimated. Another important assumption is that viability analyses are realistic enough to incorporate and correctly represent the major factors influencing population size over time. Such factors include population isolation, fragmentation, and genetic diversity.

In practice, conducting a formal population viability analysis (PVA) entails knowing a lot about autecology, including demography and population genetics. For example, viability of plant and animal populations might be affected by obligate, or strongly facultative, mutualistic relationships (e.g., pollination, mychorrizal associations), dispersal agents and mechanisms, hybridization and out-crossing with nontarget species, episodic reproduction, and many other factors. Thus PVAs perform best with data-rich species.

Formal PVAs have come under some recent criticism in the conservation biology literature as the precision and performance of PVA models have been tested or compared. Like any modeling approach, PVAs may express the bias of the modeler (Brook 2000) and specific model structure (Pascual et al. 1997). Lindenmayer et al. (2000) tested the efficacy of the commercially available VORTEX PVA program to predict viability attributes of three marsupials in Australia and concluded that the model predicted reasonably well only when a rather unrealistic amount of detail was known about the spatial distribution and dynamics of habitat patches. They concluded that conservation biologists should proceed with caution when using PVA models to predict population responses in fragmented systems, even when the species is well known and has a relatively simple life history. We suggest that such caution would include disclosing key uncertainties regarding the species' habitat associations and population persistence estimates.

Earlier "rules" regarding what constitutes viable populations have also been more recently viewed as too simplistic to rely on for real-world conservation programs. One such early rule was the "50–500 rule" whereby populations were deemed to have short-term viability if they consisted of at least 50 individuals, and long-term with at least 500. These numbers have appeared in some earlier land planning and habitat management documents. However, further scientific work suggested that they should apply to effective population sizes (of breeding individuals only, correcting overall population size for various factors of distribution, variability in population size, dispersal, sex ratio, and other things). The 50–500 rule was actually initially derived from laboratory-bred *Drosophila* (fruit flies) in controlled conditions and was based on minimal numbers of individuals needed to avoid genetic introgression or inbreeding depression. The "rule" is generally no longer used in real-world PVAs that typically entail estimating persistence or extinction likelihoods from population projections tailored to the life history of each species.

In a similar way, early contentions that there exist minimum viable population (MVP) sizes (e.g., Reed et al. 1986) that can be discretely calculated for a given species have also given way to more realistic projections of persistence likelihoods. The 50–500 rule was an early attempt at defining a blanket MVP size. The concept of MVP originally arose in the context of maintaining genetic diversity within a population and avoiding potentially deleterious effects of inbreeding depression. That is, an MVP is assumedly a size at or above which the population is secure from loss of genetic variation, and below which it is doomed to eventual extinction because of simplification of the gene pool caused by inbreeding depression, genetic drift, and related conditions.

Correctly calculating effective population size is one of the steps in estimating the potential loss of genetic variation as resulting from inbreeding depression or genetic drift in small, isolated populations, and thus in estimating the minimum size of a population that still maintains its genetic variability over time. There is no standardized method for calculating effective population size in all cases, and many variations exist (Waples 2002). Essentially, the main problem is that there are no fixed minimum population sizes below which a given species or population is doomed to extinction and above which it is assured of persistence. Although it still occasionally appears (e.g., Wielgus 2002), MVP analysis has given way in the scientific literature to more species-specific probabilistic approaches where the focus is on quantifying the degree of viability (likelihoods of persistence) given specified conditions, time periods, and geographic extents.

How well do PVAs perform in predicting population persistence? Clinchy et al. (2002) concluded, by using simulations, that recolonized habitat patches are often occupied ephemerally and thus that overall population persistence will be overestimated if static or declining patterns of patch occupancy are mistakenly attributed to dynamically stable metapopulations. However, in a test of Australian treecreepers, McCarthy et al. (2000) reported that PVA models underestimated occupancy of habitat patches.

Schiegg et al. (2005) tested the predictive accuracy of a stochastic, spatially explicit, individual-based population model of red-cockaded woodpeckers (*Picoides borealis*) by comparing simulation results with field data on population dynamics. They reported that their population models performed well and were reliable but could be improved by further including dynamics of dispersal and colonization. Johnson (2005) tested a patch occupancy model of metapopulation response of a tropical beetle species to synchrony of disturbance (flooding), and concluded that a simpler logistic regression approach provided adequate predictability over a more complicated Monte Carlo modeling approach.

McCarthy et al. (2000) used validation results to refine their initial models, an iterative procedure and information source not usually available in PVAs. McCarthy et al. (2001) also suggested using such an iterative process in modeling viability. In a similar vein, Foley (2000) recognized the uncertainty in PVA modeling and suggested using a sequential, empirical Bayesian approach in modeling population viability as new biological data are gathered on target species over time.

Coulson et al. (2001) pointed out that PVA modeling cannot predict catastrophes, and this failure creates great imprecision in viability projections. However, Brook et al.'s (2002) rebuttal to such criticism was that PVAs, despite their limitations, are still the most useful and complete methods for basing management on species demography.

Wahlberg et al. (1996) presented a successful prediction of the patchy distribution of an endangered butterfly by using a spatially realistic metapopulation model, which included first-order effects of patch area and isolation on local extinction and colonization within habitat patches. They concluded that such a model has utility for study and conservation of species in highly fragmented landscapes. Brook et al. (2000) reported that PVA predictions were "surprisingly accurate" when five PVA programs were tested on animal data sets to predict quasi-extinction probabilities and population sizes (quasi extinction occurs when a population size falls below some specified nonzero level [Ginzburg et al. 1982]). Their meta-analysis suggested that when PVAs are conducted using detailed demographic data and commercially available software, they can provide accurate and unbiased short-term projections of population viability.

In summary, results of validating formal PVA approaches to date suggest that predictions of population trend and persistence likelihoods (especially above some predefined quasi-extinction level) may be more accurate than predictions of time to extinction. The more detailed are the empirical data on biology and environmental conditions, the more accurate or precise and reliable are the viability projections. In the absence of such data, the accuracy and precision of PVA predictions may be, at best, uncertain, but PVAs could still be used to compare management strategies (e.g., Raphael and Holthausen 2002).

When data are lacking, as will be the case for little-known species, a "softer" population viability assessment approach may be used that relies more on expert judgment and qualitative ranking of potential stressor effects than it does on empirical data and modeling of population demographics, genetics, and other factors. Several decision-modeling approaches are applicable to data-poor situations, including use of Bayesian belief networks to incorporate expert judgment for evaluating population viability (e.g., Raphael et al. 2001). Levels of confidence can be expressed in expert-based viability assessments, and various sources of uncertainty can be described using a variety of methods. However, few if any such expert-based assessments of population viability have been formally tested with field validation research and experiments.

A twist to the formal PVA approach is in predicting level of viability or vulnerability of a species based on its general life history characteristics (Bernt-Erik and Bakke 2000). Some authors suggest that general life history characteristics can tell a lot about viability of a species and its populations (Verheyen et al. 2003). For example, the now-classic approach by Rabinowitz (1981) depicts rarity and potential viability risk of species according to their degree of habitat specificity, geographic extent, and local concentration (also see chap. 3). Some authors have integrated some life history characteristics into methods for ranking species vulnerability (e.g., Mace and Lande 1991). Other authors have used life history traits to predict invasiveness of plants (Higgins and Richardson 1999), birds (Cassey 2002), and other taxa. Davies et al. (2004) found that natural abundance and degree of specialization acted synergistically with beetle species so that rare and specialized beetles were especially vulnerable to extinction. They concluded that some combinations of traits increased vulnerability.

However, some authors have found that life history traits by themselves may be poor predictors of viability or risk levels of species (e.g., modeling validation analyses by Lehmkuhl et al. 2001; and simulation modeling by Wilcox and Possingham 2002) and therefore should be used with great caution. Several approaches have been proposed that explicitly incorporate uncertainty into ranking species vulnerability levels (e.g., Regan et al. 2000).

Conservation of Surrogate Species

This subcategory of species approaches is organized around the concept of surrogate species, where knowledge and conservation of one or a few species are presumed to provide for the needs of other species. This is a different focus than species indicating some aspect of ecosystem status, which is discussed in chapter 7 as a systems approach.

The value of using surrogate species is to reduce complexity of analysis, to provide a better-organized framework for management direction, to make monitoring tasks realistic, and to provide a way to draw inference to other species that cannot feasibly be monitored directly (Caro and O'Doherty 1999). The concept presumes that the surrogate species is one that is well known and easily sampled and provides a direct correlation with the species for which it is serving as surrogate. Surrogates are often relatively abundant species (Caro and O'Doherty 1999), but Warman et al. (2004) found some success in using rare Canadian species listed as threatened or endangered as surrogates to conserve terrestrial vertebrates.

Categories of surrogate species proposed in the literature include umbrella species, focal species, guild surrogates, management indicator species, biodiversity indicator species, and flagship species. Strictly speaking, the assumption that one species can be a surrogate to and represent others is by definition false based on the concept of the niche (Hutchinson 1978). Basic ecological theory says that each species is unique with respect to the environmental conditions occupied, its ecological role, and its resource-utilization functions. For this reason, no two species are entirely interchangeable or mutually replaceable (Caro et al. 2005). The surrogate species notion, however, is usually applied less stringently, looking at patterns of co-occurrence within some land area rather than niche overlap and, as such, the various categories of surrogate species explored here have varying degrees of utility, efficacy, and validity.

Umbrella Species

"Umbrella species" is a term that refers to a single species, typically a wide-ranging and widely distributed species, that represents the requirements of a portion or all of a species assemblage or community. It is the use of species with extensive spatial distributions that sets this approach apart from other surrogate species such that its area of occupancy on the landscape physically encompasses the occurrence of other, less widely distributed sympatric species (Caro and O'Doherty 1999). Suter et al. (2002) suggested that an umbrella species should have habitat requirements that are similar to those of other species, but that its range extent should be broader than other species it is intended to cover. The distinguishing, and often untested, assumption of the umbrella species approach is that satisfying conditions for the persistence of the umbrella species confers persistence to other co-occurring species of its biotic community.

Umbrella species have been used in a variety of conservation assessments and management plans. Launer and Murphy (1994) used a threatened butterfly as an umbrella species for conservation of a threatened grassland ecosystem. In the U.S. Pacific Northwest, the northern spotted owl (*Strix occidentalis caurina*) has been used as an umbrella species to further conservation of a wide array of old-growth species and communities. Invertebrates have been proposed both as umbrellas and as the benefactors of the umbrella approach. New (2004) suggested using velvet worms (phylum Onychophora) as potential umbrella species for other soil macroarthropods, while Rubinoff (2001) examined how well a vertebrate insectivore, California gnatcatcher (*Polioptila californica*), functioned as an umbrella for arthropods.

Several authors have explicitly tested the umbrella species concept. Berger (1997) tested the efficacy of using black rhinos (*Diceros bicornis*) as an umbrella species for conservation of other desert ungulates in Namibia. The study found that the space used by black rhinos alone would be unlikely to assure the existence of other ungulate populations since rhinos did not use all ungulate habitats during all seasons and rainfall levels. Similarly, Rowland et al. (2006) compared the distribution of their umbrella species, greater sage-grouse (*Centrocercus urophasianus*), to 39 other species also occurring as sagebrush obligates and found that their umbrella overlapped well for some species, such as the pygmy rabbit (*Brachylagus idahoensis*) and poorly for others, such as the lark sparrow (*Chondestes grammacus*). The poor overlap was again attributed to basic differences in habitat associations and overall geographic ranges.

Some studies have specifically looked at the value of rare species as umbrellas. In a test of the "umbrella effect" attributed to endemic and threatened bird species in South Africa and Lesotho, Bonn et al. (2002) found that a reserve network designed for these umbrella species may be insufficient to preserve overall species diversity.

The absence of strong evidence supporting the umbrella species approach across this admittedly small subset of studies is not atypical. In reviews of the umbrella species concept, Andelman and Fagan (2000), along with Roberge and Angelstam (2004), concluded that umbrella species too seldom overlap with other species of interest and rarely work to protect all species in the area of conservation concern.

Evidence of success in using the umbrella species concept is not completely absent in the literature. Bani et al. (2006) found that birds selected according to their sensitivity to habitat patch isolation, patch size, edge effect, and habitat structure were effective as umbrellas for other co-occurring birds. Similarly, Rondinini and Boitani (2006) found that amphibians and mammals (1654 species) together served as effective umbrellas for most species in the other taxa, and that developing reserves based on suitable habitat areas rather than overall distribution ranges improved the utility of one taxon serving as an umbrella for the other. Finally, Poiani et al. (2001) tested the umbrella species approach to conservation of natural grassland habitats in Minnesota using greater prairie chicken (*Tympanuchus cupido pinnatus*) as the umbrella species. They found that the umbrella species approach functioned well when used in conjunction with a conservation approach that also considered conservation of the largest native habitat patches. The combination of umbrella and habitat size criteria provided for greater conservation benefits than did either approach alone. Favorable evaluations of the umbrella species approach appear to be related to studies that have defined several (as opposed to a single) umbrella species, or have used the umbrella concept in combination with other conservation approaches.

Focal Species

"Focal species" refers to a set of target species that are identified for the purpose of guiding management of environments, habitats, and landscape elements in a tractable way. The Committee of Scientists (1999) defined a focal species as a species whose measurement provides substantial information beyond its own status. The key aspect of the focal species concept, at least as it relates to species conservation approaches, is that the distribution and abundance of focal species provide inference on the status of other species. Clearly the focal and umbrella species approaches are similar. What sets them apart is that focal species are defined using a variety of ecological attributes and are not constrained by the strong focus of umbrella species selection on species with large area requirements.

The seminal work on defining focal species has been attributed to Lambeck (1997). He defined focal species as those that are resource, area, or dispersal limited, and that respond to similar stressors. Moreover, focal species are a strict subset of the total species pool present in a landscape, and they possess the most stringent requirements. For example, focal species can be those whose distribution is restricted by specific disturbance events such as fire, who require the largest habitat patch sizes or landscape areas, and whose dispersal requirements are most limiting. In this way, the set of focal species should be those whose requirements collectively encompass the needs of all other species.

In addition to the common assumption for all surrogate approaches (i.e., reflecting the ecology of other species), the focal species approach assumes that a greater degree of such representation can be attained if the focal species has stringent requirements of resource use and dispersal capability. This characteristic assumption has received relatively little empirical testing among a variety of ecosystems.

There are many examples where focal species have been used to guide biodiversity conservation on public or private land. In Australia, Watson et al. (2001) used Lambeck's (1997) focal species approach to identify species that were sensitive to woodland fragmentation stressors in southeastern Australia. They identified a set of focal species based on those with the most restrictive requirements for minimum habitat patch size, habitat structural complexity, and habitat connectivity across the landscape. They concluded that the focal species approach was effective for efficiently developing landscape planning guidelines for woodland birds—noting that 95% of the resident woodland bird species in the region should be accommodated. However, they did not explicitly test this assertion with actual field trials and follow-on research.

Other authors have used the focal species approach to derive landscape designs. Snaith and Beazley (2002) used moose (*Alces alces*) as a focal species to derive a reserve design in Nova Scotia, Canada. Bani et al. (2002) proposed using the focal species approach to plan woodland ecological networks in Italy based on their central assumption that focal species encompass the structural and functional needs of entire ecological communities. They developed maps of habitat core areas and connections for their set of focal species. However, they did not explicitly test their main assumption.

Several authors have tested or reviewed the assumptions of the focal species approach. Rothley (2002) found that detailed descriptions of focal species and their conservation interests were needed to resolve conflicts among objectives. Lindenmayer et al. (2002) suggested that the underlying theoretical basis of the focal species approach is problematic, citing the limited utility of surrogate-species approaches in general. They also found that lack of data guiding selection of focal species sets is often a problem for practical implementation. They suggested using a mix of strategies in any given landscape to spread the risk of failure of any one approach. They cautioned that resource managers should be aware that the focal species approach might not result in the conservation of all biota in a landscape. In this spirit, Noss et al. (2002) used three complementary approaches—protecting special elements, representing environmental variation, and securing habitat for focal species—to identifying highest-priority "megasites" for conservation in the Greater Yellowstone Ecosystem.

Although the focal species concept has a foundation in the ecological literature, the varied criteria that can be used to define them has led to a diverse set of applications and considerable equivocation on the expected conservation benefits. For this reason it should not be surprising that some have cautioned against relying too heavily on the focal species approach to biodiversity conservation (Campbell et al. 2002). Confidence in this approach will remain reserved until more extensive testing has occurred.

Guild Surrogates

The guild surrogate approach entails one member or a subset of members serving as a surrogate for other members of the guild. It differs from an umbrella or focal species in that the surrogacy effect is restricted to guild members. Strictly speaking, the term "guild," as originated by Root (1967), refers to a set of species having a common diet and foraging mode. However, many other authors have appropriated and greatly extended the term to mean a set of species that share virtually anything in common, including habitat association, behavior, trophic orientation, diet, or resource selection patterns. We use this broader definition in our discussion of the guild surrogate species approach.

The main assumption of the guild approach is that the response of one member (or a small subset of members) to environmental conditions or changes is closely correlated with the response of all guild members. A corollary is that changes in environmental conditions affecting one or a subset of a guild would affect the other members in the same way. Consequently, population responses are expected to be similar in direction if not in magnitude—that is, significantly positively correlated.

In general, wildlife–habitat relationships databases can be used to generate lists of species sharing common attributes (Patton 1992). Any such list can be thought of as a guild in this broader sense of the term. Overall, the literature suggests that (1) correctly placing species into guilds that are assumed to contain species with similar ecological characteristics may necessitate detailed autecological, including behavioral, information; (2) guilds can be structured by a wide variety of mechanisms, including interspecific resource competition (mechanisms of resource and habitat overlap and partitioning) and common response to general habitat structure, but no one mechanism operates invariantly across all guilds; and (3) often, and even if guilds are defined based on detailed autecological information, guilds tend to consist of species that have disparate responses to changes in environmental and habitat conditions. Therefore, identifying guilds may require much ecological information, likely not available for RLK species.

In a study that sought to test the guild surrogate approach among bird communities, Block et al. (1987) concluded that one species of a guild does not represent the habitat specificity and population patterns of other members. Mannan et al. (1984) and Morrison et al. (2006) noted that individual species of a guild respond differently to changing environmental conditions, whereas the guild as a whole shows little or no variation in its presence, total species richness, or abundance. Morrison et al. (2006) concluded that grouping species into guilds may be useful for depicting species with similar functions or trophic relations, but the guild approach to management may not be useful for specifically predicting responses of individual species to environmental conditions and changes.

Based on these empirical tests, using guilds to predict common responses among all component species to environmental changes and management activities may not have a strong scientific foundation. Some have suggested that the lessons learned from the guild surrogate failures point to treating the entire guild as the response unit for management and monitoring—such an approach is discussed under the multiple species approaches.

Management Indicator Species

"Management indicator species" (MIS) is a broad term that has been used by the U.S. Department of Agriculture (USDA) Forest Service in its planning regulations for national forest and national grassland management. "MIS" refers to any species or set of species of management or conservation concern, and can include federally listed species; species with special habitat needs; species that are hunted, fished, or trapped; species of special interest; and species for which population changes are presumed to reflect the status of other species (R. Holthausen, pers. comm.). MIS have been used in U.S. national forest planning since the 1980s. For example, northern flying squirrels (*Glaucomys sabrinus*) have been used as a management indicator species of temperate rain forest (Smith et al. 2005). Milledge et al. (1991) suggested use of large owls and gliders as MIS in ash forests of Victoria, Australia.

For U.S. federal land management, habitat objectives are typically established for MIS, planning alternatives are evaluated based on their effects on MIS, and population trends of MIS are monitored and relations to habitat changes determined. It is difficult to derive a simple set of assumptions about the use of MIS, other than the following: a set of species chosen as MIS somehow represent the array of management issues, conditions, objectives, and conservation concerns. When used strictly to reflect the status of other species, a major assumption of the management indicator species approach is that the population status and trend of one species can represent others; that is, there is significant positive correlation in at least population trend (and possibly also size, distribution, and persistence likelihoods) between the indicator species and other species.

Niemi et al. (1997) examined the distribution of birds proposed as "management indicators" or "sensitive" species in a national forest in Wisconsin. They found that few bird species were consistently associated with habitats for which they were deemed to be indicators, and few sensitive species were positively or negatively associated with other species. Also, they found that most MIS were either too rare or too difficult to practically monitor.

Biodiversity Indicator Species

Biodiversity indicator species are single species or taxonomic groups used to identify areas of high species diversity or of high biological productivity (Caro and O'Doherty 1999). We extend the definition here to include relatively common species that are used to identify locations of rare species.

Kintsch and Urban (2002) tested the value of more common indicator plant species as predictors of the occurrence of rare plant species in the Amphibolite Mountains of North Carolina. (They termed these indicators focal species, but we believe they are more properly referred to as biodiversity indicators.) The indicators proved successful in predicting occurrences of the rare species, with 62 to 100% of the rare species locations predicted by the occurrence of the indicator species. However, there were also overprediction rates as high as 275%.

Fleishman et al. (2000, 2001) tested the value of a set of criteria that could be used to select biodiversity indicators. (They termed these species umbrellas, but again they seem better described as biodiversity indicators.) In one test Fleishman et al. (2000) used these criteria to identify two butterfly species as indicators of other butterflies. They found that protection of canyons where the two species occurred would protect sites of 97% of all other butterfly species. In a second test (Fleishman et al. 2001) they used the same criteria to select biodiversity indicators for both plant species and butterfly species in California and Ohio. Again they found that protection of sites of the indicators would protect a large proportion of sites for most other species within the same taxonomic group, and that protecting sites with the umbrella species was more efficient (few sites needed) than protecting random sites. They also tested the effectiveness of the indicators for cross-taxonomic protection. Here they found that the indicators were no more effective than random sites for cross-taxonomic protection.

Butchart et al. (2006) and Quayle and Ramsay (2006) suggested using trends in conservation status of species as indicators of biodiversity conditions. This approach would use selected species with particular trends in distribution or abundance as indexes to overall biodiversity conditions but would not necessarily extend reliable inference to individual species, particularly rare or little-known species. Oertli et al. (2005) tested the overlap in distribution of bees, grasshoppers, and wasps in the Swiss Alps and determined that none of the test taxa reflected species richness or community similarity of the other taxa well enough to be used as biodiversity indicator species.

Flagship Species

Flagship species are those that carry high public interest and garner social or cultural concern for their well-being. They are rallying points for conservation of broader ecological communities and systems (Walpole and Leader-Williams 2002). The main assumption of the flagship species approach is that individual species can be identified in a given ecosystem and promoted in highly visible conservation programs that would entice positive interest and support by the general public. Flagships are often claimed to have surrogacy benefits, but their distinction is in being able to elicit strong public support for conservation (Caro and O'Doherty 1999).

Examples of flagship species include the northern spotted owl, grizzly bear (*Ursus arctos horribilis*), bald eagle (*Haliaeetus leucocephalus*), and tiger (*Panthera tigris*), and in southern Africa the "Big Five"—African elephant (*Loxodonta africana*), leopard (*P. pardus*), lion (*P. leo*), African buffalo (*Syncerus caffer*), and rhino (*Diceros bicornis* or *Ceratotherium simum*) are touted as such in many African ecotourism and conservation activities. In India, entire national conservation programs have been founded on flagship species, including Project Tiger and more recently Project Elephant, the latter for the Asian elephant (*Elephas maximus*). Clark et al. (2003) proposed using Baird's tapir (*Tapirus bairdii*) as a flagship species in Costa Rica in an interdisciplinary approach to resolve questions about conservation policies entailing reasonableness, political practicality, and moral justification.

In summary, flagship species are used to rally conservation support. They are useful for spurring social concern and public interest for conservation, usually of large-bodied, charismatic species (usually birds or mammals) and their habitats. They might also be useful for educating the public about broader concerns for ecosystem health. For example, in a general news article on primate conservation, the Canadian Broadcasting Corporation reported that "Primates act as a flagship species that indicate the health of their surrounding ecosystem" (CBC 2002). To indicate or include conservation of RLK species, flagship species may be useful to initially garner some general protection of habitats or geographic locations.

However, relying on flagships to fully provide for all RLK species may produce some of the same problems as experienced with other surrogate approaches: lack of specificity of resources, environmental conditions, and habitats selected by the RLK species; lack of assurance that sites used by RLK species will be protected; and lack of assurance that natural disturbances and human activities will not harm RLK species. Flagships may generally not serve as reliable indicators of other species, particularly RLK species. Further, many of the species identified in the literature as umbrella species, focal species, or management indicator species might be better thought of as flagship species. That is, the motivation for choosing them has large elements of public concern rather than ecological representation or sensitivity to environmental change.

Conservation of Multiple Species

In this variant of the species approach, the aim is to ensure the viability or continuance of multiple species as an assemblage or selected subset of the ecological community. The two main categories pertinent to this are the use of entire guilds and the use of entire habitat groups.

Entire Guilds

As defined in the "Guild Surrogate" section, a guild is a set of species that share some common attribute of resource use or ecological role. In the entire guild approach, the focus is not on a species selected to represent the rest of the guild, but rather on all species of the guild. The guild as a whole becomes the response that is managed and monitored to determine the success or failure of conservation management. The main assumption of the entire guild approach is that conservation of all guild members can be judged by monitoring the total number of individuals making up the guild. One of the strengths of this approach is derived from pooling the counts from all guild member species, thus increasing the resource manager's power to detect trends in the guild as a whole (Verner 1984).

Entire guilds have been used for assessing habitat disturbance effects and, in some cases, guiding management. For example, federal land resource managers have used guilds of avian cavity excavators and Neotropical migrants as a means of focusing forest and grassland management for all species of these guilds. Knopf et al. (1988) studied the effects of seasonal cattle grazing on the guild structure of a riparian avifauna. Maurer et al. (1981) studied effects of logging on the guild structure of a forest bird community in West Virginia. Rewa and Michael (1984) used habitat evaluation procedures (HEPs) to assess habitat values of wildlife guilds. O'Connell et al. (2000) used 16 bird guilds based on their behavioral and physiological response as indicators of ecological conditions in the central Appalachians.

However, few studies have tested the basic assumption that the status of all members of a guild could be used to infer the conservation status of guild members, and that conditions providing for or affecting the guild as a whole would equally influence all members of the guild. Steffan-Dewenter et al. (2002) observed that the correlation of various bee pollinator guilds with landscape attributes varied depending on the scale of the analysis. They reported that disruption of local landscape attributes affected solitary wild bees more than social bees.

Lopez et al. (2002) reported that guilds of wetland plant species responded differentially to changes in landscape habitat attributes. In their detailed study of ungulates in East and Southern Africa, Fritz et al. (2002) identified a series of herbivore guilds based on their common trophic relations and responses to environmental variations. They discovered that the guilds respond differently to competitive interaction with large herbivores and to soil and climatic factors.

In a Brazilian Atlantic rainforest, Aleixo (1999) found that forest bird guilds delineated on the basis of foraging behavior were similar in species richness and diversity but strongly differed in species composition between primary and selectively logged forest, and that within guilds, understory and terrestrial insectivore birds were most sensitive to habitat changes. Bishop and Myers (2005) determined that there are tight associations among bird species with common primary habitat, area sensitivity, migratory status, and nest placement, and that species so grouped could serve usefully to identify areas of high bird species richness as candidate areas for conservation.

The lesson, like the individual-species indicator approach to the use of guilds (discussed earlier), is that species within a guild typically respond differently to environmental change, particularly anthropogenic stressors. Thus, even if a guild as a whole remains unchanged or shows some group responses, one cannot assert generally that this response pertains to each component species of the guild.

Entire Habitat Assemblages

In the entire habitat assemblage approach, species are identified that belong to a habitat group; that is, they share common usage of some macrohabitat, such as a general vegetation type and vegetation structural or successional stage. The approach focuses on providing for these macrohabitat types under the assumption that if the macrohabitat is provided, the requirements of the entire habitat group of species will be met. Implicit in this approach is the assumption that species needs are mostly defined by the macrohabitat(s) they occupy. It is distinguished from the entire guild approach in that this approach focuses on monitoring macrohabitat status rather than monitoring species.

A major effort to identify and evaluate entire habitat groups has been the "source habitats" analysis by Wisdom et al. (2000) for the interior Columbia River basin. Source habitats of terrestrial vertebrate species, as analyzed in this study, are primarily vegetation conditions that assumedly provide for the requirements of the organisms. In other analyses, Fauth et al. (2000) modeled landscape patterns to identify source habitats of Neotropical migrants in the U.S. Midwest, and Dyer et al. (2001) explored definitions of functional groups (see later discussion) of plants based on their source habitats and growing conditions.

The validity of the basic underlying assumptions to the entire habitat group approach has not been explicitly tested in the field and reported in the literature. Many approaches to depicting and researching wildlife–habitat relationships suggest, however, that habitat is defined for each species individually and with multiple parameters at various scales of space and time, including at macro- as well as microhabitat levels.

However, for habitats that are discrete and highly limited geographically, such as caves, ocean-floor thermal vents, vernal pools, and thermal hot springs, the tenet underlying this approach may be entirely valid. That is, conserving such extreme and isolated habitats may conserve its full biota, including RLK species.

Geographically Based Approaches

A common strategy used to conserve species, regardless of their degree of commonness or rarity, is to establish management areas (e.g., parks, reserves, refuges) where the overarching objective is the protection or restoration of natural conditions and biological diversity (Beazley et al. 2005). Numerous approaches now exist to assist conservation scientists in the identification and prioritization of areas that should be the focus of resource management. A characteristic common to these approaches is that they are geographically explicit—that is, all lead ultimately to the delineation of geographic areas on the landscape that warrant conservation focus. Furthermore, all are based on the key assumption that by managing the delineated areas appropriately, the species, their biophysical environments, and the ecological system's characteristic processes will be preserved (Flather et al. 1997).

Three broad classes of geographically based approaches may be described: (1) approaches based on management for locations of target species at risk, (2) approaches that focus on the identification of so-called hot spots or centers of concentration of rare species, and (3) approaches based on reserves or protected areas established for biodiversity conservation. The difference among the categories is often subtle, and membership in each category is not necessarily mutually exclusive. Often, geographically based approaches are applied at a coarse scale with the intent of identifying broad geographic areas that compel conservationists to take a finer-scale examination of the biodiversity pattern within these areas (Shriner et al. 2006).

Locations of Target Species at Risk

Management for locations of target species at risk refers to providing guidelines for protecting sites known to be occupied by individuals of a species deemed to be at risk. This is the basic concept behind the Survey and Manage species program of the Forest Service and the U.S. Department of the Interior Bureau of Land Management in the Pacific Northwest (Molina et al. 2006). In that program, locations of target species have been the basis of local protection and "known site surveys" in the Survey and Manage species program of the interagency Northwest Forest Plan (see chap. 4). This approach differs from the PVA approach in that the PVA approach entails modeling population demography or genetics, whereas managing for locations of target species at risk does not and is based primarily on known occupancy patterns.

The main tenet of this approach is that managing for known locations of target species contributes to ensuring persistence of the species. That is, it is presumed that known locations are where the organism finds suitable resources and habitat conferring local persistence. This is an important assumption because management guidelines for sustaining the species will focus on sustaining habitats where the species is known to occur. In general, the validity of this assumption is largely untested and unknown for many of the rare and little-known taxa.

This presents two potential problems. First, if known locations represent suboptimal habitat, managing to sustain these habitats may do little to sustain the species. Second, even if locations where the species currently occur do, in fact, represent optimal habitat, simply protecting these locations may not sustain the habitat—an understanding of the processes behind habitat formation is required (Everett and Lehmkuhl 1999; Ferrier 2002). Sustaining habitats then becomes linked with restoring various natural disturbance regimes (see chap. 7).

For RLK species of fungi, cryptogams, vascular plants, and invertebrates, this approach may rely on site location information stored in State Heritage Program databases and other agency location data. The approach thus relies on the completeness, accuracy, and recency of such location data; on being able to identify land-disturbing management activities to be avoided at each location; and on an appropriate protection buffer around each location (Aubry and Lewis 2003). Unfortunately, species databases are often incomplete for many taxa, extremely sparse for many regions, and characterized by survey biases toward easily accessible sites (Ferrier 2002). Although managing for locations of target species at risk can be a critical component of planning, such data are lacking for many if not most RLK species.

Species "Hot Spots" or Concentrations of Biodiversity

Distributions of plant and animal species are inherently heterogeneous. One manifestation of this heterogeneity is that some areas (whether examined at a global, continental, regional, or local scale) support more species than other areas. This pattern is observed whether one is counting (or estimating) all species that occur within some locale, or is counting a subset of species that share some attribute of conservation interest (e.g., rare species, threatened species, or endemic species). Given spatial structure in species' distributions, resource managers are faced with deciding which geographic areas should be the focus for biodiversity conservation.

The term "hot spot" was coined by Myers (1989) to denote those global areas where a high number of endemic species were found in a relatively small geographic area. Myers argued that greater conservation benefits could be realized if efforts were focused in those areas where endemics were concentrated—that is, the hot spots. Since Myers' specific usage with endemism, the term "hot spot" has become generalized to refer to an area, or set of areas, that ranks high on any number of ecosystem attributes, including species richness (Scott et al. 1993), threatened or endangered species (Flather et al. 1998), imperiled species (Chaplin et al. 2000), or indicators of ecosystem condition (Hof et al. 1999).

The concepts underlying hot spot analysis have been implemented in the United States under an approach generally referred to as gap analysis (Burley 1988). Gap analysis uses cartographic techniques to identify underrepresented elements of biodiversity in an existing network of protected areas—that is, hot spots that lie outside existing protected areas. As described by Scott et al. (1993), in gap analysis vegetation maps and animal distributions are used to determine if vegetation types are inadequately represented within protected areas, or if centers of species richness fall outside such areas currently managed to conserve biodiversity. Those areas of vegetation types or high species richness that fall outside the existing conservation network thus define the geographic areas that could be considered for future conservation efforts.

The appeal of mapping the occurrence pattern of plants and animals as a way of setting conservation priorities is derived largely from the simplicity of the approach. This simplicity, however, belies an important constraint associated with this approach—distributional data for most taxa are incomplete or unavailable (Ferrier 2002). One means of overcoming this data constraint is to assume that the diversity pattern of relatively well studied taxa indicate the pattern among other taxonomic groups (Reid 1998). Although there is some evidence that the geographic pattern of species richness covary among some taxa, this pattern is certainly not generally true. Empirical evidence to date cautions conservation planners against using hot spots for a few taxa to indicate where the overall diversity hot spots would occur if we had taxonomically comprehensive and spatially extensive inventories (Flather et al. 1997).

Another consideration in mapping hot spots or centers of concentration of any set of species is errors in species occurrence. This is particularly important with some RLK species that may be difficult to detect and thus may appear absent in seemingly suitable habitat (see chap. 5 for further discussion of detectability). Since such errors apply to each species, the overlaying of individual species occurrence maps can cause these errors to compound (Dean et al. 1997), leading to suboptimal conservation decisions

(Conroy and Noon 1996). Also, areas of high species richness might represent ecotones where edges of species' ranges meet, rather than more optimal conditions toward the center of species' ranges and ecosystems (Araújo and Williams 2001). Thus focusing conservation on those hot spots might overlook conservation of optimal habitat conditions for some species.

Apart from the problems associated with the hot spot approach caused by limited distributional data and locational errors, there is also concern that hot spot analysis focuses too much on ecological pattern at the expense of the ecological processes that have generated the pattern. This could be a problem insofar as conservation of a specific pattern would fail to provide for the future dynamics of the processes, such as future wildfires, that would produce mosaics of vegetation age classes outside of current locations of such mosaics. As currently applied, hot spot analysis implicitly assumes that geographic delineation of areas based on patterns of species occurrence will also "capture" the important processes that have caused, and will presumably maintain, that pattern. However, theoretical and empirical support for this assumption is weak (McNeely 1994).

A final limitation of the hot spot approach is that it often focuses on counts of species rather than species composition. Because areas that rank high on species counts may share many species, the typical hot spot analysis does not permit the teasing out of the relative contribution of each area to the preservation of some overall species pool. Consequently, it does not necessarily follow that conservation priority should always be given to the most species-rich areas. A solution to this problem is to consider measures that take into account how completely the species pool within a geographic area is represented within a reserve system (Vane-Wright et al. 1991). Over the last couple of decades, a number of methods have been developed that attempt to efficiently (greatest conservation benefit for the least cost) identify priority conservation areas (Margules and Pressey 2000). These approaches are the subject of the next section.

Reserves or Protected Areas Established for Biodiversity Conservation

One of the next great challenges for conservation science is the design and implementation of comprehensive and ecologically adequate reserve networks (Ferrier 2002). Although reserves have been used to address systemwide conservation needs, they are often created for one or a few species that are commonly RLK or in peril. We have considered reserves in this species context and discuss reserves as a species-level strategy rather than as one of the system-level strategies that are reviewed in chapter 7. Worldwide, the proportion of land that is strictly managed for the conservation of biological diversity is small—about 3% of the terrestrial land base (McNeely 1994).

Because conservation reserve areas are rare (at least in areal extent) and human impacts on natural ecosystems are increasing, there is a growing realization that the choice should be optimal when there are alternatives on where to locate reserve lands (Flather et al. 2002). A major assumption of the reserve or protected area approach is that protecting such areas from specific human activities or disturbances will provide for species or system persistence over time. In this approach, it is important to clarify specifically what "protection" refers to; that is, what elements or aspects of ecological communities and ecosystems are to be protected, and what conditions or (usually anthropogenic) stressors they are to be protected from. Most of the literature on protected area approaches, however, fails to specify or clarify these basic criteria.

Much of the recent literature on optimal reserve designs has focused on species assemblages. As such, the metric of interest concerns a collection of species and often attempts to maximize the number of different species that are conserved within some fixed or minimal area or set of areas (e.g., Camm et al. 1996). However, the reserve design problem also has important aspects that are focused on the conservation of individual species. The establishment of wildlife refuges in the United States has often been tied directly to conservation of a single species. Moreover, reserve design criteria for a single target species of conservation concern, such as the black-footed ferret (*Mustela nigripes*) (Bevers et al. 1997), have been used to address resource management conflicts on lands managed for multiple uses. Reserve designs for single species often consider how population viability varies as a function of habitat size and layout (Hof and Flather 1996), and often account for metapopulation dynamics (Holloway et al. 2003).

Early reserve design principles directed at the conservation of species assemblages were derived largely from the equilibrium theory of island biogeography (MacArthur and Wilson 1967), where the number of species expected to be supported on an island or habitat patch is predicted by the interplay between colonization and extinction rates. Key design principles ostensibly derived from this theory can be summarized as follows: (1) a single large reserve is better than several small ones of equal total area (the so-called SLOSS [single large or several small] debate); (2) reserve shapes that are compact are better than those that are convoluted or elongated; (3) multiple reserves that are close together are better than multiple reserves that are far apart; and (4) reserve units that are connected via corridors are better than reserve units that are isolated (Wilson and Willis 1975, 529). However, the SLOSS debate may have revealed that none of these principles followed strictly from the theory of island biogeography, and that most are empirically untrue in many of the archipelagoes and reserve systems for which there are adequate data, although these design rules live on (J. Quinn, pers. comm.).

The applicability of these reserve design recommendations based on island-biogeographic principles has been questioned for a number of reasons, including unrealistic simplifying assumptions and the inability to track species identity. Island-biogeographic design recommendations assume that habitat is identical from place to place—an assumption that will never be met in practice (Soulé and Simberloff 1986). Habitats vary spatially and this heterogeneity almost certainly accounts for the fact that many empirical studies have found that species counts from several dispersed sites are at least as large as counts from a single contiguous site of equal total area (Simberloff 1998). Island biogeography theory predicts the number of species that could be supported in habitat of a certain size. It does not predict the identity of those species. For this reason, it is not possible to predict the total number of species in a complex of multiple habitat patches because the number of species shared among units is indeterminate. More contemporary reserve selection algorithms that explicitly account for spatial variation in habitat and species composition address these two shortcomings (Higgs and Usher 1980).

The reserve selection problem can be simply stated as a question: How do we best locate reserve units on the landscape such that they contain the greatest number of biodiversity elements as possible (Pressey et al. 1993)? Three concepts have emerged as key in addressing that problem: representativeness, complementarity, and efficiency (for reviews see Margules and Pressey 2000; Sarkar et al. 2006). Under the concept of representativeness, conservation is focused on ensuring that some target set of species are adequately addressed in the conservation plan. The concept of complementarity is closely related to representativeness and measures the gain in new species that receive coverage as individual reserve units are added to an existing or candidate reserve network. The notion of efficiency is important because it recognizes that resources available to conservation efforts are limited, and therefore reserve designs need to maximize representation for the least cost. These concepts have emerged as important reserve design principles since scoring procedures that rank conservation priority based on simple counts of biodiversity elements (e.g., species hot spots) may rank areas with high counts as having high conservation priority even if many of the species are shared among the reserve units making up the network.

Although the literature is characterized by what may seem to be an overwhelming variety of reserve selection algorithms, the approaches do fall into two broad classes—those that define reserve networks based on iterative or stepwise algorithms, and those that are based on optimization techniques from operations research. Both approaches generally attempt to define the set of reserve units that will meet the biodiversity targets (e.g., occurrence of all species, rare species, or endemic species at least *n* times in the reserve network) for the minimum amount of total reserve area or reserve acquisition cost. They differ fundamentally in that iterative procedures are often referred to as heuristic since they can only approximate a maximally efficient design (Underhill 1994); operations research approaches do prescribe a truly optimal design.

The literature is dominated by heuristic reserve design algorithms (for review see Williams 1998) and their frequent use is related to the fact that they are intuitive and simple, they seem to be applicable to a broad and realistic set of conservation problems, and they seem to provide reasonably good solutions when compared to optimization approaches (Pressey et al. 1997). Attempts to use optimization methods sometimes fail or take too long to solve reserve selection problems, although this limitation may be overcome by creative formulation or software advances (see Rodrigues and Gaston 2002).

Since we are primarily concerned with RLK species, a logical question is how well reserve selection algorithms perform if rarity, rather than simple richness, is the biodiversity element of interest. A few investigators have compared the relative efficiency of simple richness-based algorithms with rarity-based algorithms. They found that rarity-based algorithms tended to be more efficient at finding the minimum reserve area necessary to represent all species at least once (Kershaw et al. 1994). However, this finding is hardly general (Csuti et al. 1997) and appears to be contingent on the size of the species pool, scale (i.e., size of individual reserve selection units), and degree of species endemism (Rodrigues and Gaston 2001). Consequently, differences in efficiency between richness- and rarity-based algorithms are not generalizable and depend on the distribution of species abundances, the size of reserve units considered, and the pattern of species occurrence on the landscape (Williams 1998).

Data limitations have long been recognized as an important constraint associated with geographically based conservation efforts (Botsford et al. 2003). Like hot spot analysis approaches, there have been attempts to quantify the degree to which reserve designs based on well-studied taxa also meet the conservation needs of poorly known groups (Flather et al. 1997; Reid 1998). For example, Launer and Murphy (1994) found that of all sites where a rare butterfly occurred, in an attempt to conserve genotypic representation, greater than 98% of native spring-flowering forbs also received protection. Similarly, Csuti et al. (1997) indicated that reserve designs resulting in complete representation of one major taxon may adequately represent occurrence patterns of other unrelated taxa. Although these findings do offer some hope that biodiversity benefits from reserves that are designed based on well-known taxa will transfer to little-known and unmeasured taxa, opposing evidence does exist in the literature. For example, in deciduous woodlands of western Norway, Sætersdal et al. (1993) found that among the 32 reserve units with complete representation of plants, only 5 units were shared among the set of 12 units with complete representation of birds. Such contradictory findings in the literature point to an important research need—namely, to identify those ecological circumstances, if any, when it is tenable to use occurrence patterns of one or a few taxa to represent the pattern for other taxa.

Another important limitation with reserve selection algorithms is that the results are contingent upon the pattern of species occurrence across the region where the reserve design is being planned. Since most algorithms consider only presence/absence data, it is difficult to determine whether the reserve will conserve the species pool as land use activities change the landscape context within which the reserve is embedded (Soulé and Simberloff 1986).

Potential approaches to address this concern include consideration of land suitability and anthropogenic threats (Margules and Pressey 2000) or explicit measures of persistence probability (Williams and Araújo 2000; Gaston et al. 2002). However, estimates of land suitability and persistence are often based on current conditions or recent historical data, and it is not clear that reserve designs will be robust (i.e., continue to conserve species) to future land use changes in surrounding nonreserve lands. Incorporation of such concerns will require more mechanistic models of species persistence which acknowledge that reserve units are not independent of each other, nor are they independent of the landscape matrix within which they reside.

Conclusion

A plethora of species-level approaches to conservation has been advanced, dealing with population viability, species surrogacy, multiple species groups, and geographic targets. Most of these approaches were developed to help streamline difficult, and at times intractable, tasks of simultaneously managing for many species, although the viability approaches usually pertain to single species.

The approaches are not clearly independent. Thus any attempt at categorizing these approaches is not clean because of vague and ambiguous definitions. For example, the literature has many examples of where authors have mixed and overlapped the definitions and use of many of the species approaches we reviewed here. We suggest that whatever names are used, the purpose of, and assumptions behind any given approach be clearly articulated. In this way, the pertinence and applicability to conservation of RLK species can be more accurately assessed.

Few approaches have had their fundamental assumptions rigorously tested. This is particularly problematic when extending inference to conservation of RLK species. Objectives, assumptions, implementation, and actual performance of any given approach for RLK species conservation should not be conflated but should be evaluated individually. For example, the objective of some surrogate species approach, such as the use of indicator species, could be to focus management on a species that is not rare and is well known, to indicate the status and conservation of some RLK species. The assumption would be that if the indicator species is provided for, then the RLK species will be provided for. The implementation of the approach might be feasible and economical. However, the veracity and actual performance of the approach might be unknown or suspect; that is, whether providing for the target indicator species truly serves to provide for the RLK species. Research and monitoring can help determine the degree of veracity and performance, which should not be presumed simply from the stated objective and assumption. Likewise, the implications of uncertainty (i.e., errors and lack of basic information on RLK species) need to be admitted and their influence on performance of any given conservation approach evaluated.

In general, RLK species have characteristics that make demonstrably successful implementation of any of these approaches challenging. To a certain degree, perhaps the best we can reasonably expect is to use research and monitoring to determine which approaches, alone or in combination, could best meet conservation objectives for some management situations or geographic areas or species groups, and then extend inference of those results elsewhere. It may not be reasonable to conduct enough study on RLK species to fully determine their true rarity and make them well known.

The next chapter continues a review of system-level approaches and will draw fuller conclusions on application of both species- and system-level approaches.

Acknowledgments

We thank Carolyn Sieg for suggestions, discussions, and contributions. We thank Richard Holthausen, Randy Molina, Dede Olson, and Martin Raphael for helpful discussions. We appreciate the useful reviews of the manuscript by Chris Iverson, Nancy Molina, and Jim Quinn.

References

Aleixo, A. 1999. Effects of selective logging on a bird community in the Brazilian Atlantic forest. *Condor* 101:537–48.

Andelman, S. J., and W. F. Fagan. 2000. Umbrellas and flagships: Efficient conservation surrogates, or expensive mistakes? *Proceedings of the National Academy of Sciences (USA)* 97:5954–59.

Araújo, M. B., and P. H. Williams. 2001. The bias of complementarity hotspots toward marginal populations. *Conservation Biology* 15:1710–20.

Armbruster, P., and R. Lande. 1993. A population viability analysis for African elephant (*Loxodonta africana*): How big should reserves be? *Conservation Biology* 7:602–10.

Armstrong, G. W., W. L. Adamowicz, J. A. Beck, S. G. Cumming, and F. K. A. Schmiegelow. 2003. Coarse filter ecosystem management in a nonequilibrating forest. *Forest Science* 49:209–23.

Aubry, K. B., and J. C. Lewis. 2003. Extirpation and reintroduction of fishers (*Martes* *pennanti*) in Oregon: Implications for their conservation in the Pacific states. *Biological Conservation* 114:79–90.

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

Bani, L., D. Massimino, L. Bottoni, and R. Massa. 2006. A multiscale method for selecting indicator species and priority conservation areas: A case study for broadleaved forests in Lombardy, Italy. *Conservation Biology* 20:512–26.

Beazley, K., L. Smandych, T. Snaith, F. MacKinnon, P. Austen-Smith, and P. Duinker. 2005. Biodiversity considerations in conservation system planning: Map-based approach for Nova Scotia, Canada. *Ecological Applications* 15:2192–2208.

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

Beissinger, S. R., and D. R. McCullough, eds. 2002. *Population viability analysis.* Chicago: University of Chicago Press.

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

Bernt-Erik, S., and O. Bakke. 2000. Avian life history variation and contribution of demographic traits to the population growth rate. *Ecology* 81:642–53.

Bevers, M., J. Hof, D. W. Uresk, and G. L. Schenbeck. 1997. Spatial optimization of prairie dog colonies for black-footed ferret recovery. *Operations Research* 45:495–507.

Bishop, J. A., and W. L. Myers. 2005. Associations between avian functional guild response and regional landscape properties for conservation planning. *Ecological Indicators* 5:33–48.

Block, W. M., L. A. Brennan, and R. J. Gutierrez. 1987. Evaluation of guild-indicator species for use in resource management. *Environmental Management* 11:265–69.

Bonn, A., A. S. L. Rodrigues, and K. J. Gaston. 2002. Threatened and endemic species: Are they good indicators of patterns of biodiversity on a national scale? *Ecology Letters* 5:733–41.

Botsford, L. W., F. Micheli, and A. Hastings. 2003. Principles for the design of marine reserves. *Ecological Applications* 13:S25–S31.

Brook, B. W. 2000. Pessimistic and optimistic bias in population viability analysis. *Conservation Biology* 14:564–66.

Brook, B. W., J. J. O'Grady, A. P. Chapman, M. A. Burgman, H. R. Akçakaya, and R. Frankham. 2000. Predictive accuracy of population viability analysis in conservation biology. *Nature* 404:385–87.

Brook, B. W., M. A. Burgman, H. R. Akçakaya, J. J. O'Grady, and R. Frankham. 2002. Critiques of PVA ask the wrong questions: throwing the heuristic baby out with the numerical bath water. *Conservation Biology* 16:262–63.

Burgman, M. A., S. Ferson, and H. R. Akçakaya. 1993. *Risk assessment in conservation biology*. London: Chapman and Hall.

Burley, F. W. 1988. Monitoring biological diversity for setting priorities in conservation. Pp. 227–30 in *Biodiversity*, ed. E. O. Wilson. Washington, DC: National Academy Press.

Butchart, S. H., H. R. Akçakaya, E. Kennedy, and C. Hilton-Taylor. 2006. Biodiversity indicators based on trends in conservation status: Strengths of the IUCN Red List Index. *Conservation Biology* 20:579–81.

Camm, J. D., S. Polasky, A. Solow, and B. Csuti. 1996. A note on optimal algorithms for reserve site selection. *Biological Conservation* 80:83–97.

Campbell, S. P., J. A. Clark, L. H. Crampton, A. D. Guerry, L. T. Hatch, P. R. Hosseini, J. J. Lawler, and R. J. O'Connor. 2002. An assessment of monitoring efforts in endangered species recovery plans. *Ecological Applications* 12:674–81.

Caro, T. M., and G. O'Doherty. 1999. On the use of surrogate species in conservation biology. *Conservation Biology* 13:805–14.

Caro, T., J. M. Eadie, and A. Sih. 2005. Use of substitute species in conservation biology. *Conservation Biology* 19:1821–26.

Cassey, P. 2002. Life history and ecology influences establishment success of introduced land birds. *Biological Journal of the Linnean Society* 76:465–80.

Canadian Broadcasting Corporation (CBC). 2002. Great apes among species threatened by extinction: Red list. CBC News Online Staff. CBC News on-line, Tuesday, 8 October 2002. Toronto: Canadian Broadcasting Centre. http://www.cbc.ca/storyview/CBC/2002/10/08/endangered021008.

Chaplin, S. J., R. A. Gerrard, H. M. Watson, L. L. Master, and S. R. Flack. 2000. The geography of imperilment. Pp. 159–99 in *Precious heritage: The status of biodiversity in the United States*, ed. B.A. Stein, L. S. Kutner, and J. S. Adams. New York: Oxford University Press.

Clark, T. W., M. Wishnie, and G. Gorman. 2003. An interdisciplinary approach to natural resources conservation: A flagship species example from Costa Rica. *Journal of Sustainable Forestry* 16:161–90.

Clinchy, M., D. T. Haydon, and A. T. Smith. 2002. Pattern does not equal process: What does patch occupancy really tell us about metapopulation dynamics? *American Naturalist* 159:351–62.

Committee of Scientists. 1999. Sustaining the people's lands: Recommendations for stewardship of the national forests and grasslands into the next century. *Committee of Scientists Report*, March 15, 1999. Washington, DC: U.S. Department of Agriculture, 193 pp.

Conroy, M. J., and B. R. Noon. 1996. Mapping of species richness for conservation of biological diversity: Conceptual and methodological issues. *Ecological Applications* 6:763–73.

Coulson, T., G. M. Mace, E. Hudson, and H. Possingham. 2001. The use and abuse of population viability analysis. *Trends in Ecology and Evolution* 16:219–21.

Csuti, B., S. Polasky, P. H. Williams, R. L. Pressey, J. D. Camm, M. Kershaw, A. R. Kiester, et al. 1997. A comparison of reserve selection algorithms using data on terrestrial vertebrates in Oregon. *Biological Conservation* 80:83–97.

Davies, K. F., C. R. Margules, and J. F. Lawrence. 2004. A synergistic effect puts rare, specialized species at greater risk of extinction. *Ecology* 85:265–71.

Dean, D. J., K. R. Wilson, and C. H. Flather. 1997. Spatial error analysis of species richness for a gap analysis map. *Photogrammetric Engineering and Remote Sensing* 63:1211–17.

Doak, D., P. Kareiva, and B. Klepteka. 1994. Modeling population viability for the desert tortoise in the western Mojave desert. *Ecological Applications* 4:446–60.

Dyer, A. R., D. E. Goldberg, R. Turkington, and C. Sayre. 2001. Effects of growing conditions and source habitat on plant traits and functional group definition. *Functional Ecology* 15:85–95.

Everett, R. L., and J. F. Lehmkuhl. 1999. Restoring biodiversity on public lands through disturbance and patch management irrespective of land-use allocation. Pp. 87–105 in *Practical Approaches to the Conservation of Biological Diversity*, ed. R. K. Baydack, H. Campa III, and J. B. Haufler. Washington, DC: Island Press.

Fauth, P. T., E. J. Gustafson, and K. N. Rabenold. 2000. Using landscape metrics to model source habitat for Neotropical migrants in the midwestern U.S. *Landscape Ecology* 15:621–31.

Ferrier, S. 2002. Mapping spatial pattern in biodiversity for regional conservation planning: where do we go from here? *Systematic Biology* 51:331–63.

Flather, C. H., K. R. Wilson, D. J. Dean, and W. C. McComb. 1997. Identifying gaps in conservation networks: Of indicators and uncertainty in geographic-based analyses. *Ecological Applications* 7:531–42.

Flather, C. H., M. S. Knowles, and I. A. Kendall. 1998. Threatened and endangered species geography: Characteristics of hot spots in the conterminous United States. *BioScience* 48:365–76.

Flather, C. H., M. Bevers, and J. Hof. 2002. Prescribing habitat layouts: Analysis of optimal placement for landscape planning. Pp. 428–53 in *Applying landscape ecology in biological conservation*, ed. K. J. Gutzwiller, New York: Springer-Verlag.

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

Fleishman, E., R. B. Blair, and D. D. Murphy. 2001. Empirical validation of a method for umbrella species selection. *Ecological Applications* 11:1489–1501.

Foley, P. 2000. Problems in extinction model selection and parameter estimation. *Environmental Management* 26:55–73.

Fritz, H., P. Duncan, I. J. Gordon, and A. W. Illius. 2002. Megaherbivores influence trophic guild structure in African ungulate communities. *Oecologia* 131:620–25.

Gaston, K. J., R. L. Pressey, and C. R. Margules. 2002. Persistence and vulnerability: Retaining biodiversity in the landscape and in protected areas. *Journal of Biosciences* 27 (4 Suppl 2): 361–84.

Ginzburg, L. R., L. B. Slobodkin, K. Johnson, and A. G. Bindman. 1982. Quasi-extinction probabilities as a measure of impact on population growth. *Risk Analysis* 2:171–81.

Haig, S. M., J. R. Belthoff, and D. H. Allen. 1993. Population viability analysis for a small population of red-cockaded woodpeckers and an evaluation of enhancement strategies. *Conservation Biology* 7:289–301.

Higgins, S. I., and D. M. Richardson. 1999. Predicting plant migration rates in a changing world: The role of long-distance dispersal. *American Naturalist* 153:464–75.

Higgs, A. J., and M. B. Usher. 1980. Should nature reserves be large or small? *Nature* 285:568–69.

Hof, J., and C. H. Flather. 1996. Accounting for connectivity and spatial correlation in the optimal placement of wildlife habitat. *Ecological Modelling* 88:143–55.

Hof, J., C. Flather, T. Baltic, and S. Davies. 1999. Projections of forest and rangeland condition indicators for a national assessment. *Environmental Management* 24:383–98.

Holloway, G. J., G. H. Griffiths, and P. Richardson. 2003. Conservation strategy maps: A tool to facilitate biodiversity action planning illustrated using the heath fritillary butterfly. *Journal of Applied Ecology* 40:413–21.

Holthausen, R. S., M. J. Wisdom, J. Pierce, D. K. Edwards, and M. M. Rowland. 1994.

Using expert opinion to evaluate a habitat effectiveness model for elk in western Oregon and Washington. Research Paper PNW-RP-479. Portland, OR: US Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Hutchinson, G. E. 1978. *An introduction to population ecology*. New York: Yale University Press.

Johnson, D. M. 2005. Metapopulation models: An empirical test of model assumptions and evaluation methods. *Ecology* 86:3088–98.

Kershaw, M., P. H. Williams, and G. C. Mace. 1994. Conservation of Afrotropical antelopes: consequences and efficiency of using different site selection methods and diversity criteria. *Biological Conservation* 3:354–72.

Kintsch, J. A., and D. L. Urban. 2002. Focal species, community representation, and physical proxies as conservation strategies: A case study in the Amphibolite Mountains, North Carolina, U.S.A. *Conservation Biology* 16:936–47.

Knopf, F. L., J. A. Sedgwick, and R. W. Cannon. 1988. Guild structure of a riparian avifauna relative to seasonal cattle grazing. *Journal of Wildlife Management* 52:280–90.

Lambeck, R. J. 1997. Focal species: A multispecies umbrella for nature conservation. *Conservation Biology* 11:849–56.

Launer, A. E., and D. D. Murphy. 1994. Umbrella species and the conservation of habitat fragments: A case of a threatened butterfly and a vanishing grassland ecosystem. *Biological Conservation* 69:145–53.

Lehmkuhl, J. F., B. G. Marcot, and T. Quinn. 2001. Characterizing species at risk. Pp. 474–500 in *Wildlife–habitat relationships in Oregon and Washington*, ed. D. H. Johnson and T. A. O'Neil. Corvallis: Oregon State University Press.

Lindenmayer, D. B., R. C. Lacy, and M. L. Pope. 2000. Testing a simulation model for population viability analysis. *Ecological Applications* 10:580–97.

Lindenmayer, D. B., A. D. Manning, P. L. Smith, H. P. Possingham, J. Fischer, I. Oliver, and M. A. McCarthy. 2002. The focal-species approach and landscape restoration: A critique. *Conservation Biology* 16:338–45.

Lopez, R. D., C. B. Davis, and M. S. Fenessy. 2002. Ecological relationships between landscape change and plant guilds in depressional wetlands. *Landscape Ecology* 17:43–56.

Lubchenco, J., A. M. Olson, L. B. Brubaker, S. R. Carpenter, M. M. Holland, S. P. Hubbell, S. A. Levin, et al. 1991. The sustainable biosphere initiative: An ecological research agenda. *Ecology* 72:318–25.

MacArthur, R. H., and E. O. Wilson. 1967. *The theory of island biogeography*. Princeton: Princeton University Press.

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

Mannan, R. W., M. L. Morrison, and E. C. Meslow. 1984. Comment: The use of guilds in forest bird management. *Wildlife Society Bulletin* 12:426–30.

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

Maurer, B. A., L. B. McArthur, and R. C. Whitmore. 1981. Effects of logging on guild structure of a forest bird community in West Virginia. *American Birds* 35:11–13.

McCarthy, M. A., D. B. Lindenmayer, and H. P. Possingham. 2000. Testing spatial PVA models of Australian treecreepers (Aves: Climacteridae) in fragmented forest. *Ecological Applications* 10:1722–31.

McCarthy, M. A., H. P. Possingham, J. R. Day, and A. J. Tyre. 2001. Testing the accuracy of population viability analysis. *Conservation Biology* 15:1030–38.

McNeely, J. A. 1994. Lessons from the past: Forests and biodiversity. *Biodiversity and Conservation* 3:390–405.

Milledge, D., C. Palmer, and J. Nelson. 1991. "Barometers of change": The distribution of large owls and gliders in mountain ash forests of the Victorian Central Highlands and their potential as management indicators. Pp. 53–66 in *Conservation of Australia's forest fauna*, ed. D. Lunney. New South Wales, Australia: Royal Zoological Society of New South Wales.

Molina, R., B. G. Marcot, and R. Lesher. 2006. Protecting rare, old-growth forest associated species under the survey and manage guidelines of the Northwest Forest Plan. *Conservation Biology* 20:306–18.

Morrison, M. L., B. G. Marcot, and R. W. Mannan. 2006. *Wildlife–habitat relationships: Concepts and applications*. 3rd ed. Washington, DC: Island Press.

Myers, N. 1989. Threatened biotas: "Hotspots" in tropical forests. *Environmentalist* 8:1–20.

New, T. R. 2004. Velvetworms: Charismatic invertebrates for conservation. *Wings* 27:12–15.

Niemi, G. J., J. M. Hanowski, A. R. Lima, T. Nicholls, and N. Weiland. 1997. A critical analysis on the use of indicator species in management. *Journal of Wildlife Management* 61:1240–52.

Noss, R. 1987. From plant communities to landscapes in conservation inventories: A look at the Nature Conservancy (USA). *Biological Conservation* 41:11–37.

Noss, R. F., C. Carroll, K. Vance-Borland, and G. Wuerthner. 2002. A multicriteria assessment of the irreplaceability and vulnerability of sites in the Greater Yellowstone Ecosystem. *Conservation Biology* 16:895–908.

O'Connell, T. J., L. E. Jackson, and R. P. Brooks. 2000. Bird guilds as indicators of ecological condition in the central Appalachians. *Ecological Applications* 10:1706–21.

Oertli, S., A. Müller, D. Steiner, A. Breitenstein, and S. Dorn. 2005. Cross-taxon congruence of species diversity and community similarity among three insect taxa in a mosaic landscape. *Biological Conservation* 126:195–205.

Overton, C. T., R. A. Schmitz, and M. L. Casazza. 2006. Linking landscape characteristics to mineral site use by band-tailed pigeons in western Oregon: Coarse-filter conservation with fine-filter tuning. *Natural Areas Journal* 26:38–46.

Pascual, M. A., P. Kareiva, and R. Hilborn. 1997. The influence of model structure on conclusions about the viability and harvesting of Serengeti wildebeest. *Conservation Biology* 11:966–76.

Patton, D. R. 1992. *Wildlife–habitat relationships in forested ecosystems*. Portland, OR: Timber Press.

Poiani, K. A., M. D. Merrill, and K. A. Chapman. 2001. Identifying conservation-priority areas in a fragmented Minnesota landscape based on the umbrella species concept and selection of large patches of natural vegetation. *Conservation Biology* 15:513–22.

Pressey, R. L., C. J. Humphries, C. R. Margules, R. I. Vane-Wright, and P. H. Williams. 1993. Beyond opportunism: Key principles for systematic reserve selection. *Trends in Ecology and Evolution* 8:124–28.

Pressey, R. L., H. P. Possingham, and J. R. Day. 1997. Effectiveness of alternative heuristic algorithms for identifying indicative minimum requirement for conservation reserves. *Biological Conservation* 80:207–19.

Quayle, J. F., and L. R. Ramsay. 2006. Biodiversity indicators based on trends in conservation status: advancing the science. *Conservation Biology* 20:582–83.

Rabinowitz, D. 1981. Seven forms of rarity. Pp. 205–17 in *The biological aspects of rare plant conservation,* ed. H. Synge. Chichester: Wiley.
Ranius, T. 2000. Minimum viable metapopulation size of a beetle, *Osmoderma eremita,* living in tree hollows. *Animal Conservation* 3:37–43.
Raphael, M. G., M. J. Wisdom, M. M. Rowland, R. S. Holthausen, B. C. Wales, B. G. Marcot, and T. D. Rich. 2001. Status and trends of habitats of terrestrial vertebrates in relation to land management in the interior Columbia River basin. *Forest Ecology and Management* 153:63–87.
Raphael, M. G., and R. S. Holthausen. 2002. Using a spatially explicit population model to analyze effects of habitat management on northern spotted owls. Pp. 701–712 in *Predicting species occurrences: issues of scale and accuracy,* ed. J. M. Scott, P. J. Heglund, M. L. Morrison, J. B. Haufler, M. G. Raphael, W. A. Wall, and F. B. Samson. Washington, DC: Island Press.
Ratner, S., R. Lande, and B. B. Roper. 1997. Population viability analysis of spring chinook salmon in the South Umpqua River, Oregon. *Conservation Biology* 11:879–89.
Reed, J. M., P. D. Doerr, and J. R. Walters. 1986. Determining minimum population sizes for birds and mammals. *Wildlife Society Bulletin* 14:255–61.
Regan, H. M., M. Colyvan, and M. A. Burgman. 2000. A proposal for fuzzy International Union for the Conservation of Nature (IUCN) categories and criteria. *Biological Conservation* 92:101–108.
Reid, W. V. 1998. Biodiversity hotspots. *Trends in Ecology and Evolution* 13:275–80.
Rewa, C. A., and E. D. Michael. 1984. Use of habitat evaluation procedures (HEP) in assessing guild habitat value. *Transactions of the Northeast Section of the Wildlife Society* 41:122–29.
Reyers, B., D. H. K. Fairbanks, A. S. van Jaarsveld, and M. Thompson. 2001. Priority areas for conservation of South African vegetation: A coarse filter approach. *Diversity and Distributions* 7:79–95.
Roberge, J. M., and P. Angelstam. 2004. Usefulness of the umbrella species concept as a conservation tool. *Conservation Biology* 18:76–85.
Rodrigues, A. S. L., and K. J. Gaston. 2001. How large do reserve networks need to be? *Ecology Letters* 4:602–609.
———. 2002. Optimisation in reserve selection procedures: Why not? *Biological Conservation* 107:123–29.
Rondinini, C., and L. Boitani. 2006. Differences in the umbrella effects of African amphibians and mammals based on two estimators of the area of occupancy. *Conservation Biology* 20:170–79.
Root, R. B. 1967. The niche exploitation pattern of the blue-gray gnatcatcher. *Ecological Monographs* 37:317–50.
Rothley, K. D. 2002. Dynamically-based criteria for the identification of optimal bioreserve networks. *Environmental Modeling and Assessment* 7:123–28.
Rowland, M. M., M. J. Wisdom, L. H. Suring, and C. W. Meinke. 2006. Greater sage-grouse as an umbrella species for sagebrush-associated vertebrates. *Biological Conservation* 129:323–35.
Rubinoff, D. 2001. Evaluating the California gnatcatcher as an umbrella species for conservation of southern California coastal sage scrub. *Conservation Biology* 15:1374–83.
Sætersdal, M., J. M. Line, and H. J. B. Birks. 1993. How to maximize biological diver-

sity in nature reserve selection: Vascular plants and breeding birds in deciduous woodlands, western Norway. *Biological Conservation* 66:131–38.

Sarkar, S., R. L. Pressey, D. P. Faith, C. R. Margules, T. Fuller, D. M. Stoms, A. Moffett, et al. 2006. Biodiversity conservation planning tools: Present status and challenges for the future. *Annual Review of Environment and Resources* 31:123–59.

Satterthwaite, W. H., E. S. Menges, and P. F. Quintana-Ascencio. 2002. Assessing scrub buckwheat population viability in relation to fire using multiple modeling techniques. *Ecological Applications* 12:1672–87.

Schiegg, K., J. R. Walters, and J. A. Priddy. 2005. Testing a spatially explicit, individual-based model of red-cockaded woodpecker population dynamics. *Ecological Applications* 15:1495–1503.

Schwartz, M. W. 1999. Choosing the appropriate scale of reserves for conservation. *Annual Review of Ecology and Systematics* 30:83–108.

Scott, J. M., F. Davis, B. Csuti, R. Noss, B. Butterfield, C. Groves, H. Anderson, et al. 1993. Gap analysis: A geographic approach to protection of biological diversity. *Wildlife Monographs* 123:1–41.

Shriner, S. A., K. R. Wilson, and C. H. Flather. 2006. Reserve networks based on richness hotspots and representation vary with scale. *Ecological Applications* 16:1660–73.

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

Smith, W. P., S. M. Gende, and J. V. Nichols. 2005. The northern flying squirrel as an indicator species of temperate rain forest: Test of any hypothesis. *Ecological Applications* 15:689–700.

Snaith, T. V., and K. F. Beazley. 2002. Moose (*Alces alces americana* [Gray Linnaeus Clinton] Peterson) as a focal species for reserve design in Nova Scotia, Canada. *Natural Areas Journal* 22:235–40.

Soulé, M. E., and D. Simberloff. 1986. What do genetics and ecology tell us about the design of nature reserves? *Biological Conservation* 35:19–40.

Steffan-Dewenter, I., U. Münzenberg, C. Bürger, C. Thies, and T. Tscharntke. 2002. Scale-dependent effects of landscape context on three pollinator guilds. *Ecology* 83:1421–32.

Suter, W., R. F. Graf, and R. Hess. 2002. Capercaillie (*Tetrao urogallus*) and avian biodiversity: Testing the umbrella-species concept. *Conservation Biology* 16:778–88.

Taylor, R. J., T. Regan, H. Regan, M. Burgman, and K. Bonham. 2003. Impacts of plantation development, harvesting schedules and rotation lengths on the rare snail *Tasmaphena lamproides* in northwest Tasmania: A population viability analysis. *Forest Ecology and Management* 175:455–66.

Underhill, L. G. 1994. Optimal and suboptimal reserve selection algorithms. *Biological Conservation* 70:85–87.

Vane-Wright, R. I., C. J. Humphries, and P. H. Williams. 1991. What to protect? Systematics and the agony of choice. *Biological Conservation* 55:235–54.

Verheyen, K., O. Honnay, G. Motzkin, M. Hermy, and D. R. Foster. 2003. Response of forest plant species to land-use change: A life-history trait-based approach. *Journal of Ecology* 91:563–77.

Verner, J. 1984. The guild concept applied to management of bird populations. *Environmental Management* 8:1–14.

Wahlberg, N., A. Moilanen, and I. Hanski. 1996. Predicting the occurrence of endangered species in fragmented landscapes. *Science* 273:1536–38.

Walpole, M. J., and N. Leader-Williams. 2002. Tourism and flagship species in conservation. *Biodiversity and Conservation* 11:543–47.

Waples, R. S. 2002. Definition and estimation of effective population size in the conservation of endangered species. Pp. 147–68 in *Population viability analysis*, ed. S. R. Beissinger and D. R. McCullough. Chicago: University of Chicago Press.

Warman, L. D., D. M. Forsyth, A. R. E. Sinclair, K. Freemark, H. D. Moore, T. W. Barrett, R. L. Pressey, and D. White. 2004. Species distributions, surrogacy, and important conservation regions in Canada. *Ecology Letters* 7:374–79.

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

Wielgus, R. B. 2002. Minimum viable population and reserve sizes for naturally regulated grizzly bears in British Columbia. *Biological Conservation* 106:381–88.

Wilcox, C., and H. Possingham. 2002. Do life history traits affect the accuracy of diffusion approximations for mean time to extinction? *Ecological Applications* 12:1163–79.

Williams, P. H. 1998. Key sites for conservation: area-selection methods for biodiversity. Pp. 211–49 in *Conservation in a changing world*, ed. G. M. Mace, A. Balmford, and J. R. Ginsberg. Cambridge: Cambridge University Press.

Williams, P. H., and M. B. Araújo. 2000. Using probability of persistence to identify important areas for biodiversity conservation. *Proceedings of the Royal Society of London, Series B* 267:1959–66.

Wilson, E. O., and E. O. Willis. 1975. Applied biogeography. Pp. 522–34 in *Ecology and evolution of communities*, ed. M. L. Cody and J. M. Diamond. Cambridge, MA: Belknap Press.

Wisdom, M. J., R. S. Holthausen, B. C. Wales, C. D. Hargis, V. A. Saab, D. C. Lee, W. J. Hann, et al. 2000. *Source habitats for terrestrial vertebrates of focus in the Interior Columbia Basin: broad-scale trends and management implications.* 3 vols. General Technical Report PNW-GTR-485. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.