5

Special Considerations for the Science, Conservation, and Management of Rare or Little-Known Species

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Rare or little-known (RLK) species pose special problems to conservation management. It is difficult to detect and study rare species, and they do not lend themselves to experimentation (see chap. 2). Likewise, locations, numbers, and responses of species that are little known are difficult to impossible to predict (see chap. 4). This chapter addresses special considerations for the science, conservation, and management of RLK species, and also the further joint problem of species that may be both rare and little known, which constitutes the conditions of many inconspicuous taxa that contribute to the bulk of biological diversity.

Recall the Venn diagram in figure 1.1 (chap. 1) which depicts the overlap of the sets of species that are rare and those that are little known. We can expand on this overlap by suggesting three categories of each condition, as shown in table 5.1. Rare species (chap. 3) are those that (1) are scarce everywhere and have low density throughout their distributional range, even if total numbers are high; (2) have a small total population, which means that total global numbers are low, whether locally scarce or locally common; or (3) are genetically or behaviorally distinctive, including those that occur locally as a peripheral population, which means they occur on the edge of their overall distributional range, where local peripheral number or density is low even if the total population elsewhere is not (fig. 5.1). Little-known species are those for which we have (1) an incomplete taxonomic description, so that there is uncertainty over their taxo-

Table 5.1. *Categories and examples of rare and little-known species*

		Category of Rare		
		Scarce everywhere	Small total population	Distinctive or peripheral population
Category of Little-Known	Incomplete taxonomic description	Speckled chub, *Macrhybopsis aestivalis* species complex	Brown kiwi, *Apteryx australis*	White salmon pocket gopher, *Thomomys talpoides limosus*
	Limited species inventory	Sharp-tailed snake, *Contia tenuis*	Fuertes's parrot, *Hapalopsittaca fuertesi*	Bonobo, *Pan paniscus*, near Lac Tumba, western DR Congo
	Poor ecological understanding	Club mushroom, *Podostroma alutaceum*	Chevron skink, *Oligosoma homalonotum*	Dalles side-band snail, *Monadenia fidelis minor*

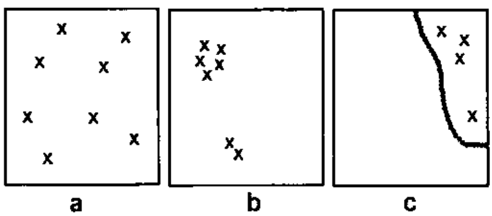

Figure 5.1. Three kinds of species rarity: (a) scarce everywhere, with low density throughout the range; (b) small total population, whether locally scarce or locally common; and (c) genetically or behaviorally distinctive, including peripheral populations on the edge of their distribution range (dotted line). X = occurrence of the species.

nomic status; (2) a limited species inventory, so that we have poor understanding of their basic occurrence and distribution; or (3) a poor ecological understanding, so that we do not know what environmental factors determine their occurrence or distribution.

Some examples illustrate the problems of species that are rare or little known in these various ways (see table 5.1). Among species that are probably scarce everywhere, one species for which we have incomplete taxonomic description is the rare fish speckled chub (*Macrhybopsis aestivalis*). This fish was previously considered a wide-ranging, polytypic species with six subspecies, but genetic and morphometric work has suggested that it may contain at least 10 distinct species, some of which are likely imperiled (Butler and Mayden 2003; Warren et al. 2000). In such cases, the distinct species may warrant separate attention and further inventory to determine their abundance and distribution.

Sharp-tailed snake (*Contia tenuis*), a small colubrid of western North America with limited inventory data (e.g., Leonard and Leonard 1998), has been considered scarce everywhere (but see further discussion later in chapter). Although it is well established taxonomically, its known habit of residing in well decayed downed logs makes it somewhat secretive and difficult to find, and thus it is difficult to determine its true abundance and population trend.

Podostroma alutaceum, a club mushroom, is likely scarce everywhere and is a species for which we have poor ecological understanding. This fungus is widely distributed in coastal and Cascade Range forests from northern Washington south to the San Francisco Bay area but is nearly always solitary in occurrence. Lack of understanding of its ecology means that it cannot be reliably predicted from habitat maps and environmental data.

Continuing with examples, among species with a total small population, one species for which we have an incomplete taxonomic understanding is a New Zealand endemic bird, the brown kiwi (*Apteryx australis*). Although three subspecies of brown kiwi are currently recognized (*A. a. mantelli* on North Island, *A. a. australis* on South Island, and *A. a. lawryi* on Stewart Island), recent genetic work suggests that the species should be split into two distinct species, the brown kiwi with distinct varieties occurring on North Island and near Okarito on South Island, and the tokoeka with distinct varieties near Haast and in Fiordland on South Island and on Stewart Island. The splitting of these species means that the total population of each is smaller than previously thought, which may mean that each has lower probabilities of persistence throughout a smaller range. Smaller populations also mean potentially greater harm to donor populations should removal be done for transplantation or captive breeding.

An example of a species with total small population and adequate taxonomic description but with limited inventory is Fuertes's parrot (*Hapalopsittaca fuertesi*), which, in 2002, was rediscovered after 90 years in the high Andes Mountains of Colombia. The parrot is known to associate with tall mature trees and to feed on berries in epiphyte-laden canopy branches, but surveys are needed to determine if and where more than just the small isolated population still exists. If removal of any of the parrots for captive breeding is to be done, lack of inventory data means great uncertainty as to the degree of potential adverse impact on the donor population. In a similar case, such inventories were also needed, and conducted, on the endangered Puerto Rican parrot before some individuals were removed for captive breeding (Vilella and Garcia 1995). However, in this case, dividing an already tiny wild population may lead to future problems of genetic bottlenecks and loss of genetic diversity (Wilson et al. 1994).

An example of a species with a small overall population for which we have poor ecological understanding is the chevron skink (*Oligosoma homalonotum*), listed by the government of New Zealand as a vulnerable species. Lost for more than 60 years, the chevron skink was first described in 1906, reported again in the 1970s, and rediscovered as a single subadult in 1991 on Little Barrier Island. Less than 250 sightings are known since 1906. It also currently occurs on Great Barrier Island but the apparent 60-year hiatus there was likely due to misidentification in museum records. Although locations and taxonomy are known, and a habitat study was conducted on the species during 1997–2002, little is known of its specific ecology. It was found to inhabit streamsides and damp places, and is likely vulnerable to predation by introduced Norway rats, but its specific habitat requirements are still very poorly known because the lizard is secretive and cryptic. Behavioral studies of a small captive population are ongoing.

Among species with a peripheral population, one taxon for which we have an incomplete taxonomic understanding is the white salmon pocket gopher (also called Columbia River pocket gopher; *Thomomys talpoides limosus*), a rare and locally endemic subspecies of the northern pocket gopher that occurs on the edge of its parent species range in southern Washington. According to the International Union for Conservation of Natural Resources (IUCN 2006), the taxonomy of *Thomomys talpoides* is still uncertain and may consist of several sibling species in the U.S. Pacific Northwest, possibly including the white salmon pocket gopher. The management implication is that, if these incipient sibling species are indeed elevated to species status, there may be conservation interests (or mandates) for some or each of them individually, including the white salmon pocket gopher.

An example of a species occurring as a rare peripheral population for which we have limited inventory is the bonobo or pygmy chimpanzee (*Pan paniscus*) near Lake Tumba in the western Democratic Republic of Congo. Bonobos have only recently been discovered in the old swamp forests of this area, are low in numbers and density there, and are clearly on the edge of their range. Inventories are being conducted under Bonobo Conservation International, but their local distribution, numbers, and population structure are still incompletely known. Discovery of local bonobo troops could mean changes in local forest use practices, such as potential designation of wildlife refuges or development of ecotourism centers from which to visit and view the animals.

And finally, a taxon that occurs as a peripheral population for which we have poor ecological understanding is the locally endemic Dalles sideband snail (*Monadenia fidelis minor*), a terrestrial mollusk found on the east side of Mount Hood in the Cascade Mountains and up the Columbia River Gorge east of the Cascade Mountains crest of northern Oregon. Its parent species is broadly distributed from Canada to California, but the specific ecology of this subspecies is poorly known. Some species, particularly more sessile ones such as mollusks, tend to develop unique habitat and environmental associations on their range peripheries; in some cases this may lead to speciation. Thus understanding the ecology of this species at its peripheral locations and comparing that to elsewhere in its range may be useful for determining whether more locally stringent or different sets of management guidelines might be needed for its overall conservation.

These examples of RLK species illustrate a wide array of conditions, taxa, and challenges for research and conservation. However, some of the attributes they do share include their inconspicuous nature and uncertain taxonomy or ecology, and their scarcity in part or all of their range. We next consider some of the conservation implications of being rare and little known.

Conservation Considerations

Several issues rise when considering conservation of RLK species: issues of scientific value and classification of such species, identification of species and taxa for conservation management, and setting conservation goals and objectives.

Science Issues

A scientific approach to identifying and conserving RLK species could first use the categories within "rare" and "little known" already discussed (see table 5.1). This would help set the stage for identifying what kind of further information (taxonomic, inventory, or ecological) is most critical, and how conditions contributing to rarity would guide conservation.

Further science issues of RLK species pertain to evaluating their persistence and their ecology. Persistence of species that are well known demographically or genetically can be modeled and assessed using population viability analysis (PVA). PVA consists of estimating persistence probabilities (or time to extinction), using quantitative calculations and models of effective population size, influence of genetic bottlenecks, and rate of change in population size. However, with species that are little known, a PVA approach would be fraught with uncertainty, so, at best, a more qualitative population viability evaluation (PVE) may be more in order. A PVE may consist of qualitative or categorical ranking of potential threats and the level of vulnerability of a species to environmental conditions and human activities, and not depend on genetic or demographic data on the population. PVEs have been conducted using expert panels and used to evaluate effects of alternative land management scenarios on many species for which demographic and genetic data are lacking (e.g., Raphael et al. 2001).

Many RLK species may play important ecological roles in their ecosystems (see chap. 4). To the extent possible, the species could be treated in functional groups (see chap. 7) such as soil arthropod decomposers, or in substrate groups such as fungi associated with decaying wood, even if knowledge of individual species is poor. Other species and system-level approaches to representing or indicating RLK species are discussed in chapters 6 and 7. Other science issues of RLK species also pertain to problems of sampling, detectability, and modeling, and depicting and dealing with uncertainty in management (see chap. 12).

Identifying Taxa for Conservation Management

Selecting taxa on which to focus conservation management activities is fundamental to developing and implementing an overall species conservation program. Species conservation has traditionally focused on a limited number of often well-studied vertebrates, plants, and showy arthropods (e.g., butterflies). Many of these species are referred to as charismatic, reflecting a societal and science bias toward certain taxa (Cassidy and Grue 2001). RLK species can also include a diverse assemblage of taxa less familiar and less charismatic to society and conservation science in general. Limitations of the charismatic species approach to RLK species conservation are discussed further in chapter 6.

Conservation Goals and Objectives

Chapter 2 outlined some goals and objectives typical of conservation planning. Careful consideration and clear statements of goals and objectives are critical because they will determine how and what information is collected, analyzed, and used in decision making.

As noted in previous sections, the initial choice of species and the criteria used to designate them are important scientific and programmatic considerations for setting goals and objectives. Once species are designated, selecting the science-based metrics to gauge species viability and population persistence also becomes a critical consideration for setting conservation objectives. Maintaining "viability" typically relies on use of population viability metrics (viz., population demographic and genetic data) as used in classic population viability analyses (Shaffer 1990). Given the potential number of RLK species, the difficulties of obtaining such information, and ultimately the expense of such an undertaking, it may not be realistic to use such viability metrics.

For example, to protect rare, late-seral forest species on federal public lands in the Pacific Northwest, the Northwest Forest Plan (as described in the FEMAT [1993] analysis) set and evaluated viability objectives within a context of species distribution in reserve and matrix land allocations, and effects of other management guidelines. Initial viability assessments centered on "understanding how provision of habitat on federal lands under each [management] option could contribute to population persistence and distribution over a century" (FEMAT 1993, II-99). Provision of habitat included the amount, quantity, and distribution of habitat. Meeting this objective required combining guidelines for conservation of individual target species with guidelines for geographically based conservation (viz., reserved areas) as later implemented in the Survey and Manage program under the Northwest Forest Plan of the Pacific Northwest as discussed in chapter 4.

The geographic scale at which conservation planning decisions are made also has a major effect on meeting goals and objectives for conserving RLK species (Kintsch and Urban 2002; Schwartz 1999). Monitoring and maintaining the persistence of RLK species at a local (e.g., federal Forest Service district, approximately 100,000 to 200,000 acres) planning level (as in many Forest Service and Bureau of Land Management [BLM] sensitive species programs) differs from maintaining persistence at a large regional scale that might include many local planning units. At the local scale, there is more intimate knowledge that contributes to regional planning, but local resource specialists can have a hard time seeing how they fit into the big picture. That is, resource specialists are most familiar with the planned use of their managed land base, as well as species distribution and location of specific habitat types therein; given that familiarity, conservation strategies can be integrated with local management plans. When regional-scale conservation objectives are chosen, a high degree of regional planning and oversight is needed to both collect needed information and develop management plans that can cross multiownership boundaries. Communicating results and decisions of a regional plan to field units or different landowners and coordinating and monitoring the conservation program becomes a complex task that requires strong interagency or multiowner cooperation.

For example, conducting surveys to determine distribution, abundance, and vegetation associations of RLK species across a broad regional area may entail establishing a stratified random or multistage sampling design (Cutler et al. 2001; Philippi et al. 2001), such that the aggregate of local samples provides a desired level of confidence or power in determining presence and association with specific vegetation conditions at the wider scale of an ecoprovince. Individual administrative units, such as management districts of national forests, may play critical roles in gathering such information even though the statistical inference is to be made to an area broader than their individual administrative boundaries. It is critical that participants understand the broader-scale context and value of such efforts that may, otherwise, seem expensive and pointless at the scale of an individual resource management project or field administrative unit. It is statistically inappropriate to draw inferences to finer scales from coarse-scale data (Plotkin and Muller-Landau 2002). There may also be opportunities to establish multiscale sampling designs and modeling schemes that provide estimates of species distribution and abundance at given confidence levels and at two or more geographic scales or levels of spatial resolution (Borcard et al. 2004).

In fact, determining the appropriate scale for setting conservation goals and objectives; for designing inventories, surveys, and studies; and for developing and applying species–habitat relationships models of rare and little-known species can be a vital early step in the process. Remember that "scale" means several things: geographic extent, spatial resolution, and level of detail on maps and in geographic information systems (GIS) and other databases. Appropriate scales should be determined by (1) clearly identifying conservation issues, including environmental conditions and species (e.g., preservation of an old-growth stand, restoration of a watershed, conservation of overall biodiversity, protection of specific RLK species, etc.); (2) evaluating available data on distribution of environmental conditions and species of conservation interest, to determine their geographic extent and spatial resolution; and (3) consulting any pertinent models or experts to provide further understanding of geographic locations, areal extent, and spatial resolution pertinent to the species' population distribution and potential dispersal and movement patterns. The results can be used to guide over what area, in what specific locations, and at what level of resolution to conduct a set of guidelines for species conservation, inventory, survey, and study or to apply a species prediction model.

Combining conservation approaches to achieve management objectives is covered in detail in chapters 8 and 12; chapter 11 provides details on programmatic implementation considerations.

Sampling Considerations

Detecting and sampling rare species entails special statistical considerations (Thompson 2004). It may be possible to estimate the total number of species in a community by using methods of compiling species accumulation curves (Shen et al. 2003) and rarefaction and related analyses (e.g., Haddad et al. 2001; Mao and Colwell 2005). However, although these curves describe total species richness, they may not provide reliable information on individual RLK species.

It may not be critical (or possible) to survey or monitor all RLK species in a community. One reason is that many such species may be very difficult to detect; that is, the probability of their appearing in a survey or inventory sample given that they are actually present may be very low. Once a threat assessment (see chap. 4) helps determine which RLK species to study, one of the more difficult challenges in conserving the species is developing statistically valid sampling schemes to effectively deal with the detectability issues. Such sampling schemes may entail the use of particular statistical distributions to guide the number and dispersion of samples in the field, depending on the expected density and dispersion of the rare species being sampled. Different statistical distributions have different properties, and some are better suited to help ensure that rare species are adequately included in samples at specified levels of confidence.

For example, Venette et al. (2002) suggested use of the hypergeometric, binomial, and beta-binomial distributions for sampling rare invertebrates. Green and Young (1993) suggested that adequate sampling effort to detect rare species could be determined by use of the Poisson and negative binomial distributions, particularly if the organisms are not spatially aggregated. However, with many species, one form of rarity (see table 5.1) is indeed local aggregation with overall total low population numbers. In such cases, other authors have suggested use of various adaptive sampling strategies (papers in Thompson 2004) such as adaptive cluster sampling (Philippi 2005), which begins at known concentration centers or locations of a species and then extends adjacent samples from those points.

Cutler et al. (2001) described sampling RLK species as a problem of representing rare events in space and time. Because of the many unique attributes of RLK species, they stated, "these characteristics provide unique statistical challenges to the design of surveys to provide quantifiable estimates about the species." They prescribed a three-pronged approach to information gathering: (1) conduct coarse-grained distribution surveys to determine the degree of rarity and habitat association of the species and to determine their probability of occurrence and their habitat associations; (2) conduct midscale (e.g., watershed) distribution surveys to model probability of occurrence and ecological association; and (3) gather persistence-related information, at the scale of the population, to determine parameters such as local population density and, as possible, demographic vital rates (survivorship and mortality rates). They described a suite of sampling designs, several of which have been implemented in the Survey and Manage program (see Cutler et al. 2001; Molina et al. 2003; Molina et al. 2006; and chap. 4 for details) and emphasized three primary considerations in design selection: (1) cost effectiveness, (2) ability to quantify the uncertainty in the estimates of population characteristics, and (3) flexibility in making changes to the designs to obtain different types of needed information.

Determining appropriate sampling frames and sample sizes for RLK species means knowing something about two parameters: the probability P that the organism is present at a site, and the probability Q of detecting the organism given that it is present (MacKenzie et al. 2003). That is, if D represents a detection event, then $E(D|\bar{P}) = 0$, $E(D|P) = Q$, $E(\bar{D}|P) = 1 - Q$, and where E = expected value or probability (and the bar represents "not"). Parameter Q is generally referred to as "detectability," and determining this probability is not easy. For example, Royle (2006) noted that estimating the probability of occurrence of a species can be confounded by detectability itself and may vary due to variation in the species abundance (especially when using methods where user experience, learning, and search image are important). Still, estimating detectability is important because false conclusions of species absence may lead to habitat-disturbing activities that would locally reduce or extirpate the organism, adding to overall threat levels. Thus estimates of detectability should be considered when estimating overall abundance, local presence, or viability of RLK organisms (MacKenzie and Kendall 2002; Royle et al. 2005; see also Beavers and Ramsey 1998; Jenouvrier and Boulinier 2006).

For example, in the Pacific Northwest, preliminary results from the Survey and Manage program's regionwide cross-taxonomic surveys of RLK species, using a stratified random sampling grid, have shown that some species are so rare and spotty in distribution that the numbers of encounters have been too low to reliably extrapolate numbers throughout the region (Molina et al. 2006). However, for some of the less rare species, random grid designs at the plan scale have been effective in finding new locations of individuals (Edwards et al. 2004), although a strict analysis of detectability has not been conducted.

A more broad-brush approach to inventory of RLK species may entail use of rapid survey and ecological assessment methods (Sayre et al. 2000), such as Oliver and Beattie (1996) and Jones and Eggleton (2000) used to inventory invertebrates. Such methods typically entail blanketing a study area with several taxonomic specialists who intensively collect as many specimens or sightings of organisms over a short time period. The samples usually include many (but seldom all) RLK species; the approach will often miss organisms that may be present but that are not detectable during the short time period of the survey. The purpose is to sample and determine at least the presence of individual species, not just to build species accumulation curves and estimate total species richness. The advantage of rapid survey methods is the collection of a relatively large, representative set of organisms from a community in a short time; the disadvantages are that some rare species are still missed in the samples and the statistical properties of the samples may be difficult to assess.

Some species that are secretive or that occur in locations difficult to study may be known from just a few locations, and some of these species might be more abundant than previously suspected. Such seems to be the case with the sharp-tailed snake in the U.S. state of Oregon. The limited inventory previously available on this species suggested that it might be rare, and it is listed as vulnerable by the state of Oregon. Surveys by Hoyer et al. (2006) more recently expanded the known range and number of sightings of this species by almost a factor of 10, suggesting that the secretive nature of this species lent to its being unnecessarily listed as vulnerable. However, because a species is secretive does not necessarily mean that it is more abundant than thought and thus not vulnerable.

Habitat Modeling

One tool that may be useful for assessing and conserving RLK species is that of modeling to predict habitat quality and potential presence of organisms. Species-habitat modeling has a long and rich history in ecology and resource management (e.g., Verner et al. 1986; Scott et al. 2002), but there are special considerations pertaining to RLK species.

Many of the same detectability and sampling issues apply to developing predictive habitat models for RLK species. Failing to account for missing an RLK species during surveys carries similar risks as predicting species absence when the species is actually present. The modeling risks include biasing estimates of site occupancy, colonization, and local extinction probabilities (MacKenzie et al. 2003).

It can be extremely difficult to collect enough habitat information when few sites are available for analysis and when rare species are patchily distributed. Nevertheless, some rare species have been modeled successfully. For instance, Dunk et al. (2004) predicted occurrence of five species of rare mollusks in Northern California, using generalized additive models to estimate each species' distributional range and habitat associations. In the Pacific Northwest, federal land management agencies under the Survey and Manage program (see example later in chapter) have used regionwide potential natural vegetation GIS models (Henderson 2001; Lesher 2005) and Bayesian belief network models (Marcot 2006) to predict Survey and Manage species occurrence and to prioritize landscape and stand-scale surveys.

It is essential to clarify the objectives for modeling RLK species and habitats, including the potential application of the models. One such set of objectives, as used in the Survey and Manage species program, may be to assess quality and spatial distribution of habitat by which to prioritize sites for expensive field surveys for the organisms, as in preproject or predisturbance surveys. In the Survey and Manage example, models of selected RLK species have been developed at two scales: (1) an ecoregional scale using broad topographic and climatic parameters in GIS analyses, by which to map suitability of environments across watersheds (Henderson 2001; Lesher 2005); and (2) a fine scale using within-stand vegetation and substrate characteristics in Bayesian belief network (BBN) models by which to calculate likelihoods of habitat quality classes within the watershed at particular sites (Marcot 2006). These two scales of models can be used in a tiered fashion, first predicting suitable areas broadly at the ecoregional scale, and then within those areas further assessing degree of suitable habitats and substrates within vegetation stands or at particular sites. Alternatively, one could also accrete local model outcomes to model broader-area conditions (Rastetter et al. 1992). Tiering the models may reduce errors of inclusion, that is, avoiding the prediction of suitable environments and potential presence of the species in areas clearly outside the species' range of distribution or range of tolerable conditions.

However, RLK species prediction models may be developed to deliberately err more on the side of commission than omission (presuming that some modeling prediction error is always present), particularly if predicting species absence means that proceeding with ground-disturbing activities would decimate the organisms from the site. The balance between types of errors can be defined according to the real and opportunity costs involved. Predicting high habitat quality or presence of the organism when the habitat is actually low in quality or the organism is absent (a type I error) may unnecessarily trigger expensive and activity-delaying surveys and losing access to valuable resources (e.g., timber). On the other hand, incorrect prediction of low habitat quality or absence of the organism when the habitat is actually high in quality or the organism is present (a type II error) may unnecessarily result in local extirpation of the species.

Models can and should be tested to determine their frequency (and thus probabilities) of type I and type II errors. The land and natural resource manager (hereafter, resource manager) would then need to weigh these tradeoffs associated with each type of error (Marcot 1986; Mapstone 1995) and may also consider opportunity costs and likelihoods of legal appeals and of not meeting other resource management objectives. In this way, habitat models for RLK species could be part of a broader decision-modeling framework in which alternative management actions (e.g., to survey or not; to proceed with ground-disturbing activities or not), their costs and their effects on habitat conditions, and the utility values of resulting habitat states or resources produced, can be integrated into decision trees or other tradeoff analyses.

Another use of habitat modeling of RLK species is to expand our ecological understanding of the species. This can be done using traditional multivariate statistical analyses of inventory and survey data to identify major environmental correlates. It should be emphasized that nothing replaces sound empirical field research for expanding ecological understanding. Models play a subservient role in such ventures, to help identify major correlates and causes.

Habitat modeling of RLK species can also be used to identify and prioritize vegetation or other habitat attributes for conservation or restoration. Once an RLK species habitat model is built (and hopefully calibrated and perhaps further validated with field data), sensitivity analysis (e.g., Keitt et al. 1997) can reveal which prediction variables (vegetation or habitat attributes) most account for variation in response variables (habitat quality or RLK species presence). For instance, rank-ordering prediction variables according to their explanatory power (i.e., the degree to which variation in the species response is accounted for by variation in each prediction variable) tells which vegetation or habitat characteristics have the most influence on RLK species habitat quality or species presence. Of course, the reliability of using this approach depends in part on the set of habitat variables included in the model and their correlations. The importance of any variable is always measured in the context of the other variables.

The explanatory power of prediction variables can be gauged in various ways. A popular approach is the use of Akaike's information criterion (AIC). This index denotes which model—that is, which combination of predictor variables—best fits the observed data without overfitting the data with too many parameters (Burnham and Anderson 2002). The modeler will still need to decide if the best-fit model makes ecological sense. Other approaches to determining the explanatory power of prediction variables and the best model to use include information theoretic modeling (Burnham and Anderson 2002), which entails depicting the relation among variables as likelihoods and comparing the relative values of those likelihoods. This approach too can suggest the best suite of variables that have the best explanatory power for a given response.

Another way to gauge explanatory power of prediction variables uses variance reduction in analysis of model sensitivity. Variance reduction is the degree to which incremental changes in the value of a predictor variable accounts for variation in the response variable. High variance reduction of a predictor variable means much of the variation in the response variable is accounted for by that predictor. Variance reduction is used with continuous variables; an analogous "entropy reduction" is used with categorical variables. The resource manager could then use the rank-order list of site variables to prioritize variables for restoration or conservation. This presumes that the validated model truly represents the set of prediction (site) variables that affect the species response variables. If validation is not done with such models, then the models represent little more than the belief structures of the modelers, so that sensitivity analyses cannot be used to truly determine the main influential parameters. It may still be useful to pose testable hypotheses.

For example, sensitivity analysis of a site-scale habitat model of Townsend's big-eared bat (*Corynorhinus townsendii*) revealed the following site attributes listed in decreasing influence (decreasing values of entropy reduction) on prediction of habitat quality for this species: presence of caves or mines, large snags or live trees, forest edges, cliffs, bridges or buildings, and boulder piles (Marcot et al. 2001). The influence of each of these habitat attributes was positive on habitat quality. Most of the influence came from the first four attributes, so in this case, the resource manager could get the greatest return by focusing initial funding and conservation efforts for this species on gating caves and mines, conserving large snags or live trees, retaining forest edges, and not disrupting cliff environments. However, in this particular example, the bat model was developed from knowledge of bat experts, not from empirical data, so the interpretation of the sensitivity test results should be treated as testable working hypotheses, not as definitive knowledge of which site factors more influence presence of this bat species.

One should realize that testing, validating, updating, and testing sensitivity of such models using correlation analysis does not necessarily reveal the true causal variables. The truism is that correlation is not necessarily causation. However, if the results of testing fundamentally disagree with the conceptual models of experienced biologists, then the basic model structure should probably be reexamined.

For any kind of habitat modeling of RLK species, it may be valuable to record field data when species are not detected as well as when they are detected (e.g., Zielinski and Stauffer 1993; Carroll et al. 1999; Kery 2002). With such data, type I and II errors can both be analyzed, as with so-called confusion matrices, which present error rates of false prediction of presence and absence. Too often, however, empirical data on RLK species are only on presence, although modeling species–habitat relations based on use-availability data can also be of value (e.g., Hirzel et al. 2002). Museum and collection records, and most field inventory and survey records, generally provide only presence data.

With no way to gauge how well models predict absence (i.e., statistical power; Steidl et al. 1997; Thomas 1997), the resource manager would be unsure if prediction of low habitat quality or "lack of presence" of an RLK organism in an area truly implies a high confidence that it is absent. Thus there may be uncertainties associated with gauging risk of inadvertent local extirpation. However, the methods of MacKenzie (2006) provide for ways to adjust species responses for the probability of their detection given a level of sampling effort, and this might help reduce some of the uncertainty associated with lack of data on species absence.

For plants and allies (fungi, lichens, bryophytes), lack of detection in well-surveyed vegetation plots may be taken as a reasonable "absence" for modeling purposes, although some cryptic life forms may still be difficult to detect, and people conducting field surveys may require special training. For any taxa, it may be worth establishing and testing field survey protocols for ensuring that both presence and absence can be determined at acceptable levels of confidence. Also, it is important to denote in species databases whether lack of presence means confirmed absence (searched for, according to a survey protocol, and not found) or unknown absence (a survey protocol was not used to determine absence, and the species was not reported present).

Habitat modeling of RLK species could also distinguish between prediction of habitat quality and prediction of actual presence of the organism (Church et al. 2000). With rare species, habitat may seem entirely suitable but the organism may often still be absent because of many other factors affecting rarity (see chap. 3). Such effects are particularly confounding with RLK species. Even with fully suitable habitat, to the best of expert knowledge and as far as can be identified and predicted from field data, only a small percentage of such sites might actually harbor an RLK organism. The resource manager may find that habitat models predicting habitat quality instead of species presence are entirely sufficient for most purposes, such as to help prioritize sites for field surveys.

As an example, Imm et al. (2001) developed models of rare plants based on resource and vegetation characteristics in southeastern U.S. hardwood forests. They found that the models accurately predicted habitat but, not surprisingly, only when the plants were strongly associated with these variables and the scale of modeling coincided with habitat size. They concluded that habitat prediction models can be useful tools for managing rare plants, especially when combined with information from research and monitoring.

As new information is gathered, it can be integrated back into the models to further calibrate, validate, and amend them to be more accurate. Some modeling approaches, such as BBN models and many "data mining" tools (e.g., associations and pattern discovery methods, Bayesian hierarchical modeling, and probabilistic relational modeling), can handle missing data and can integrate new data to update underlying probability tables or functions (Dominici et al. 1997). In some approaches, databases, even those having some missing data, can be used to generate a set of rules that optimally predict some outcome, such as habitat quality, based on a set of environmental or habitat predictor variables. These are "rule induction" algorithms and they include classification and regression trees and fuzzy-logic-based inductive modeling methods (Stockwell et al. 1990; Jeffers 1991; Uhrmacher et al. 1997). Other updating techniques include sequential and empirical Bayes methods (Johnson 1985, 1989; Ver Hoef 1996) and expectation maximization and other methods (Dempster et al. 1977). The value of such approaches is to create a predictive model that can at least be used to generate hypotheses about causal influences of species presence. The risk, though, with rule induction approaches is in overfitting the data so that the resulting model may work well with the (often limited) data at hand but is not general enough to apply anywhere else or under any other conditions.

These very powerful tools extend the toolkit of traditional ("frequentist") statistical analyses and can be used to construct and update habitat models of RLK species. In particular, using methods that account for missing data—a common situation with RLK species data sets—may be quite useful. The initial modeling, in fact, can be done based on quizzing a species expert and building a prediction model based on the expert's best judgment of what constitutes suitable habitat (building such models has long been called knowledge engineering in the artificial intelligence computer programming literature; e.g., Fox 1984). The reliability of models based solely on expert knowledge—"expert systems" in the broadest sense—may be unknown or difficult to judge without subsequently testing it with field data. However, a rigorous peer-review process (or a sequential interview or rigorous expert panel process) can still be used to build the initial expert-based model *if* there are such other peer experts available who have knowledge of the species.

In some cases, the species may be just so rare, so difficult to detect, or so unknown, that any field surveys are not practical. In such cases, expert-based habitat models should be used with caution and as working hypotheses. Very RLK species may be impossible to model individually. In those cases, developing models of species habitat groups (e.g., old-forest canopy-dwelling arthropods, or mesoscopic soil organisms) or species functional groups (e.g., rock-adhering stream bryophytes that serve to filter water) may be a useful approach (chap. 6), although prediction of individual species within such species groups becomes problematic.

Dealing with Uncertainty

This section characterizes the kinds and sources of uncertainty and discusses ways to address uncertainty to help reduce management risk.

Assessing, modeling, and managing any species carries various kinds of uncertainty, or, more properly, sources of error. Sources of error traditionally include both measurement error and random error. Measurement error can be reduced by using statistically rigorous research or survey designs, by attending to adequate training of individuals conducting the surveys and research, and by calibrating measurement tools, be they calipers, densiometers, eyeballs, identification keys, or models. Random errors can stem from inherent variation in species' distributions and environmental conditions affecting species' presence, and can be reduced (but usually not eliminated) with adequate sample sizes and sampling designs.

Random error introduces unexplainable noise in biological studies. Moller and Jennions (2002) conducted a meta-analysis of a variety of 43 biological studies and found that the mean amount of variance (R^2) by which the response variables were explained by the predictor variables was a rather dismal 2.51 to 5.42%. One implication was that the average sample sizes needed to conclude that a particular relationship was absent with a power of 80%, and alpha = 0.05 (two-tailed) was considerably larger than that usually recorded in studies of evolution and ecology. This has dire implications for studies of RLK species where sample sizes are often far lower than needed for meeting widely accepted levels of statistical confidence and power. Of course, if a survey includes all members of some rare biological population, then it constitutes a complete census of the entire statistical population and there are no errors of estimation involved. This is seldom the case, or at least, it can seldom be demonstrated that an entire biological population has been located or that environmental conditions encountered during surveys would be invariant. Thus some errors of estimation or prediction are nearly always involved.

Beyond these traditional types of uncertainty, assessment and conservation of RLK species also carry some other special considerations. Some of these were mentioned earlier, particularly problems of detectability, taxonomy, and field sampling methods and expertise. When species are inconspicuous or rare, research studies and surveys are expensive, detections tend to be few, and data are sparse by which to project population estimates and determine habitat correlates. Estimators of relative abundance of species can account for low detectability of RLK species (e.g., MacKenzie and Kendall 2002), but with rare species it takes a lot of field effort to reliably develop such correction factors. Further, experts who can effectively locate the species in the field tend to be few, and training of survey technicians needs to be done rigorously to ensure reliable outcomes (e.g., Scott and Hallam 2003; Wilkie et al. 2003), particularly with data on species absence.

When modeling RLK species, many of these sources of uncertainty combine to create greater error in prediction of habitat quality and species presence (and absence). There is no one practical method for calculating such *propagation of error* from all these sources of uncertainty. The traditional analytic method of calculating propagation of error entails expansion of the Taylor series to include terms for covariance among affector variables (Meyer 1986). Practically, however, such an approach is nearly impossible with RLK species. Clark (2003) has proposed a hierarchical approach to dealing with propagation of error in estimating population growth rates. In general, various sources of error can be characterized in a variety of ways.

One of the simplest ways to characterize uncertainty of scientific understanding of RLK species is to have species experts denote a general level of confidence, such as depicting a level of confidence on an ordinal scale (e.g., low, moderate, high) representing the overall level of scientific understanding of the species. Such an approach has been used in several regional species–environment relations databases that include information on RLK species, and has been useful to sort species needing further study (Marcot et al. 1997; O'Neil et al. 2001).

A more detailed listing of species information in addition to identifying information gaps and research needs on each species can be useful for guiding studies. For example, a database on research needs was compiled on terrestrial vertebrate species for the Interior Columbia Basin Ecosystem Management Project (Marcot 1997). The database represents significant gaps in scientific knowledge and includes 482 potential research study topics on 232 individual species and 18 groups of species. The main keyword subjects include basic ecology, distribution, inventory and monitoring, environmental disturbance, effects of land management activities, and other topics. The database can be searched a variety of ways, such as to determine which species share specific information gaps, or the full array of research needs on a specified species or group of species. A research needs assessment can also address a broader array of biodiversity attributes related to RLK species and can be directed at prioritizing studies to meet the needs of resource managers (e.g., Smythe et al. 1996).

If at least some field information is available on RLK species, such as from surveys or collection and voucher records, statistical analyses can aid further understanding of the ecology of the species, and initial models can be built to predict habitat quality and perhaps species presence. With few sample points, however, traditional statistical estimation procedures may be fraught with high levels of error. For example, extrapolating a regional population from a small sample of a rare species may be risky if the sampling design fails to block on, or account for, some major environmental or habitat correlates of species presence, such as presence of a key pollinator or dispersal agent. The estimate may be precise (such as having a low variance or low odds ratio) but it may be biased if the sample does not accurately represent the full range of conditions present throughout the region. Statistical methods of bootstrapping, rarefaction analysis, or cumulative error analysis may help reveal such biases but cannot eliminate such errors. Further, nothing short of solid field autecology studies can truly reveal the mechanisms underlying distribution and abundance patterns of RLK species.

Other kinds of uncertainty in assessing and managing for RLK species pertain to metapopulation dynamics (Thomas 1994). If RLK species are patchily distributed with at least some interchange among patches (populations), two factors can have a major (positive) effect on long-term persistence of the overall metapopulation: (1) if populations tend to vary asynchronously between one another, and (2) if organisms can successfully disperse among habitat patches and recolonize empty patches. These two factors likely vary greatly among many taxonomic groups, rendering it impossible to generalize among all RLK species.

Other factors pertaining to population viability—including demographic vital rates, and genetic isolation and simplification of gene pools—also likely vary greatly among taxa and species. It is not useful to predict viability of an RLK species based solely on general life history characteristics of a taxonomic group or even of a closely allied congeneric (i.e., belonging to the same genus) species (Lehmkuhl et al. 2001; Wilcox and Possingham 2002); there is simply too much variation among species (see chap. 6).

Sampling and analysis schemes can be devised to estimate richness of species, including rare species. Rosenzweig et al. (2003) tested six methods of estimating total species richness of butterflies in Canada and the United States, at the broad scale of biogeographical provinces. According to their findings, methods that rely on extrapolating from frequency of sampled species did not work well to estimate the total number of species. Other methods rely on estimating the asymptote of the species-accumulation curve, which is a graph of the number of species in a set of samples by the number of species occurrences in those samples, where the number of species at the asymptote is the estimated total number of species in the community. Asymptote methods served as reliable estimators of total number of species even with relatively small samples (10% of the ecoregions sampled) when samples were well dispersed spatially but not when samples were clustered. In this way, at least the total number of species in a community can be estimated, although such approaches do not reveal which species, particularly the rare ones, are present.

In summary, despite best efforts and use of analytic and evaluation methods to describe and reduce errors of estimation and to improve understanding and prediction, some uncertainty in scientific knowledge and management of RLK species will remain.

Management Implications

By definition, managing for RLK species carries great uncertainty. How might the resource manager use the information on uncertainty as discussed in the previous section?

This is the case with surveys of northern spotted owls (*Strix occidentalis caurina*), a relatively rare species in conifer forests of the Pacific Northwest, where a strict and rather complicated survey protocol (USDA Forest Service 1993) has been established to determine nesting status (box 5.1). The complexity of this example is not unusual, given that spotted owls are scarce everywhere (rarity condition [a] in fig. 5.1), occupy large home ranges, and are not necessarily highly detectable during any given survey outing, either by direct visual observation or by soliciting responses from playing taped calls and songs. Also, should nesting *not* be confirmed after all the survey visits stipulated in the protocol are conducted, only then can one state with a moderate degree of confidence that nesting is indeed not occurring at that site. Similar, complex survey protocols to determine presence and absence have been established for a wide variety of species and taxa under the Survey and Manage program, including for RLK fungi, lichens, bryophytes, and mollusks, and other species of concern within the Northwest Forest Plan, such as marbled murrelets (*Brachyramphus marmoratus*). For example, the marbled murrelet protocol uses detection probabilities to set the number of site visits to provide a high confidence (95%) of observing the birds if they are present.

Box 5.1. Survey protocol standards for determining nesting status of northern spotted owls (*Strix occidentalis caurina*) in the Pacific Northwest (USDA Forest Service 1993).

Shown here is only part of a much longer protocol that provides guidelines for determining pair status, resident single status, unknown (single owl) status, absence (verified unoccupied site), and reproductive status (nesting status and reproductive success).

1. **Nesting is confirmed** if any of the following conditions are observed. Two observations, at least 1 week apart, are required to determine nesting status if the first observation occurs before 1 May. This is necessary because the owls may show signs of initiating nesting early in the season without actually laying eggs, and their behavior could easily be mistaken for nesting behavior. After 1 May, a single observation is sufficient. Nesting is confirmed if, on two visits before 1 May, or one visit after 1 May:
   a. the female is detected (seen) on the nest; or
   b. either member of a pair carries natural or observer-provided prey to the nest; or
   c. a female possesses a brood patch when examined in hand during mid-April to mid-June. Only one observation is required. Dates may vary with the particular areas. Be careful not to confuse the normal small areas of bare skin (apteria) on the abdomen with the much larger brood patch. A fully developed brood patch covers most of the lower abdomen, extending to the base of the wings. Describe the brood patch on the field form, including length, width, color, and texture of the skin, and any evidence of regenerating feathers around the edge (NOTE: while a scientific research permit from the U.S. Fish and Wildlife Service is not required for calling spotted owls, any capture or handling of spotted owls does require such a permit); or
   d. young are detected in the presence of one or both adults. Because young barred owls look like young spotted owls until late in the summer, the presence of young alone is not sufficient to confirm nesting.
2. **Nonnesting (and nonreproduction) is inferred** if any of the following are observed. Two observations are required during the nest survey period, with at least 3 weeks separating these observations to ensure that late nesting attempts are not missed. The second observation should occur after 15 April. Because nesting attempts may fail before surveys are conducted, the nonnesting status includes owls that did not attempt to nest as well as those that have failed. Nonnesting is inferred if:
   a. the female is observed roosting for 60 minutes, particularly early in the season (1 April to 1 May). (Be aware that nesting females with large nestlings often roost outside the nest during warm weather. If in doubt, be sure to schedule one or more visits in mid-June to check for fledglings); or

(*continues*)

Box 5.1. *Continued*

b. the female does not possess a brood patch when examined in-hand between mid-April and mid-June. Only one observation is required; or
c. the pair is not located after a 4-hour search on two separate visits; or
d. you offer prey to one or both members of the pair and they cache the prey, sit with prey for an extended period of time (30 to 60 minutes), or refuse to take additional prey beyond the minimum of two prey items. To be considered a valid nesting survey, an owl must take at least two prey items.

Surveys where the bird(s) leave the area with prey and you are unable to determine the fate of the prey cannot be classified as to nesting status and do not count toward the required two visits. Banded or radio-marked birds may be reluctant to take prey at all; therefore, nesting status should be inferred from other means (e.g., checking for fledglings later in the season).

3. If nesting is not determined before the latest date (by province) listed for nesting status visits, you cannot classify the owls as nonnesting using the criteria listed above. **Nesting is unknown** if:
   a. owls are found after these dates, without young; or
   b. no owls are found after these dates at those sites where owls were present prior to these dates.

A related and more general question is, how far should assumptions of species' presence and habitat associations be relied upon? Another way of asking this is, what are acceptable levels of type I and II errors when modeling or predicting occurrence of an RLK species? As discussed earlier, a type I error is a false prediction of species presence or adverse effect, and a type II error is false prediction of species absence or no effect. Here, the answer is less clear than simply demanding proof of presence or of adverse effects. A resource manager may wish to determine opportunity costs and values of outcomes ("utilities" in decision modeling), as well as possible harm or benefit to the species, under each possible type of error, and then balance them according to their management risk attitude.

To do this, resource managers would first have to characterize their risk attitude. Risk attitude refers to the degree to which one is risk seeking, risk neutral, or risk averse for particular kinds of decisions. For example, a resource manager may be averse to moderate or high probabilities that an RLK species would become imperiled because it may mean that the species would then become petitioned for listing as a threatened or endangered species, which could then entail expensive and time-consuming revisions of natural resource and land management plans. Or the resource manager may be tolerant (seeking) of such risks if social, administrative, or political pressures were high for maintaining or increasing rates of extraction of other natural resources. In some cases, risk may even be judged according to potential for changes in one's career path, potential promotion or demotion, possibilities of lawsuits and legal actions, and so forth. These are legitimate causes for concern and consideration by decision makers and line officers, and they do figure into their risk attitudes, albeit usually tacitly and not as part of explicit decision criteria. In other cases, one's risk attitude may be dictated, to a degree, by policy or law, such as leaning toward risk aversion when dealing with potential harm to a listed threatened or endangered species because national law and resource and land management agency policy and regulations dictate so.

A risk analysis would then help outline alternative or sequential management decisions and probabilities of various outcomes such as response by an RLK species. In risk analysis parlance, a "policy" becomes a set of alternative management decision options along with the probabilities of outcomes and the values or utilities of each possible outcome. Risk attitudes can determine which decision or sets of sequential decisions are "best" in terms of a stated decision objective, such as minimizing harm to an RLK species.

Uncertainty over the occurrence or response of little-known species can be represented as a range of possible outcomes of decisions and can be depicted as a range of probabilities of species presence or as a general trend of populations or habitats. Depending on their risk attitude, resource managers could then look at the spread of possible outcomes and not just the median or mean outcome, or just the extreme "best" or "worst case" outcome. With RLK species, the spread is likely to be far wider than with more common and better-known species.

Outcomes could pertain to possible effects on species persistence but also to secondary effects on ecological processes resulting from changes to RLK species. Such processes may include the ecological roles that RLK species might play as key pollinators, dispersal agents, contributors to ecosystem resistance to exotic species invasion, decomposers and other roles in nutrient cycling, and so on. The aim would be to view RLK species not just as independent entities but also as dependent players in the broader ecological tapestry.

The implications of managing for RLK species are further discussed in subsequent chapters. Chapters 6 and 7 evaluate how various species- and system-level management approaches might suffice for conservation of RLK species. Chapter 8 then discusses effectiveness of the various management approaches; chapters 9 and 10 address social and economic considerations; chapter 11 reviews implementation considerations; and chapter 12 presents a process for determining appropriate and useful approaches for conservation of RLK species.

Conclusion

RLK species comprise a diverse lot. They span a wide range of taxonomic classes, life histories, ecologies, levels of abundance, and distributional patterns. Some are cryptic, secretive, and difficult to find, and thus there is little to no information available on their abundance, distribution, and autecology. Some are truly rare, and others might be more common than their apparent rarity suggests.

One way to help make sense of this confusing array of conditions is to denote the type of rarity and the reason for poor knowledge levels of a given RLK species (Table 5.1). These categories of rare and little known can help clarify why a species might be rare, explain why it might be little known, and describe what could be done to aid in its conservation and to increase our knowledge.

For example, a species that is rare because it is generally scarce everywhere could benefit from dispersed conservation activities, be they multiple reserves or guidelines to conserve or restore the species' key habitat attributes in selected locations dispersed across the species' range. On the other hand, a species that is rare because it has an overall small total population but that might be locally more abundant, or because it consists of a distinctive or peripheral population, could benefit by concentrating reserves or activities for conserving its key habitat attributes at its population centers.

Likewise, a species that is little known because of incomplete taxonomic descriptions could benefit from systematic studies to better determine its taxonomic status. A species that is little known because limited inventories have been conducted could benefit from expanded survey efforts or surveys with statistically sound protocols to provide more reliable information on its presence and absence. A species that is little known because there is poor understanding about its ecology could benefit from surveys and studies designed to provide information on life history and habitat associations.

In many cases of RLK species, knowledge is power. Knowledge from ecological studies and general surveys can greatly reduce uncertainty over whether a species is truly rare, should really garner status as threatened or endangered, or requires special consideration for conservation reserves or management guidelines. Data from well-designed surveys can be used to develop models that predict the habitat and perhaps occurrence of RLK species, for use in land management and for prioritizing species for conservation focus. Examples we have cited suggest that with additional information, conservation of some species may become less stringent than initially suspected and may need to be reaffirmed or increased for other species. The next two chapters explore potential approaches to species conservation and their effectiveness for providing for RLK species.

RLK species should not be discounted in terms of the potential ecological importance they may play in ecosystem functions. Certainly, many poorly known micro- and mesoarthropods play crucial ecological roles in soils and forest canopies. Many such species are uncharistmatic and may seldom be chosen for specific conservation focus. But they may be contributing important functions to help ensure productivity of ecosystems and provision of services to people.

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