SCIENTIFIC STANDARDS FOR CONDUCTING VIABILITY ASSESSMENTS UNDER THE NATIONAL FOREST MANAGEMENT ACT: REPORT AND RECOMMENDATIONS OF THE NCEAS WORKING GROUP

November, 2001

Sandy J. Andelman, Steve Beissinger, Jean Fitts Cochrane, Leah Gerber,
Paola Gomez-Priego, Craig Groves, Jon Haufler, Richard Holthausen, Danny Lee,
Lynn Maguire, Barry Noon, Katherine Ralls, Helen Regan

National Center for Ecological Analysis and Synthesis

Sandy J. Andelman is Deputy Director of the National Center for Ecological Analysis and Synthesis (NCEAS), University of California, Santa Barbara.

Steve Beissinger is Associate Professor, University of California, Berkeley.

Jean Fitts Cochrane is a biologist with the U.S. Fish and Wildlife Service.

Leah Gerber is a postdoctoral associate at the National Center for Ecological Analysis and Synthesis (NCEAS), University of California, Santa Barbara.

Paola Gomez-Priego is a graduate student intern at the National Center for Ecological Analysis and Synthesis (NCEAS), and a student in the Bren School of Environmental Science and Management, University of California, Santa Barbara.

Craig Groves is Director of Conservation Planning for The Nature Conservancy.

Jon Haufler is Executive Director of Ecosystem Management Research Institute in Seely Lake, Montana.

Richard Holthausen is national wildlife ecologist with the U.S.D.A. Forest Service, Flagstaff, Arizona.

Danny Lee is a research ecologist with the U.S.D.A. Forest Service, Pacific Southwest Research Station, Arcata, California.

Lynn Maguire is Associate Professor, Nicholas School of the Environment and Earth Sciences, Duke University.

Barry Noon is Professor, Department of Fishery and Wildlife Biology, Colorado State University.

Katherine Ralls is a senior research biologist at the Smithsonian National Zoological Park's Conservation and Research Center.

Helen Regan is a postdoctoral associate at the National Center for Ecological Analysis and Synthesis (NCEAS), at the University of California, Santa Barbara.

Table of Contents

Chapter 1. Executive Summary

Regulations implementing the National Forest Management Act (NFMA) initially went into effect in 1982. They were replaced by a new set of planning regulations in November, 2000. The 1982 regulations direct the U.S.D.A. Forest Service to provide habitat on National Forests to support viable populations of native and desired non-native vertebrate species, well distributed across National Forest lands. The revision of these regulations issued in November 2000 would extend this requirement to additional species in the plant and animal kingdoms, but provide for significant qualifications in the way the requirement is applied to many species. The Forest Service, however, has deferred implementation of these new regulations, and is attempting to produce a different set of regulations to replace those issued in 1982. Concerns about the viability provision of the regulations have been central to the argument that a new set of regulations is needed.

Since 1982 the Forest Service has used a variety of approaches to meet the NFMA viability requirement, ranging from opinion-based assessments conducted by individual resource specialists, to detailed habitat and demographic modeling. Many Forest Plans have been challenged on either the adequacy of the management guidelines for species, or on the adequacy of the process used to demonstrate that viability requirements were met. The Forest Service now is making a renewed effort to provide consistent and scientifically rigorous input to Forest Plans on species viability. Accordingly, in September, 2000, the Forest Service commissioned an external review of methods used to assess species viability under the NFMA. An independent working group was established by the National Center for Ecological Analysis and Synthesis (NCEAS) in September, 2000. Between December, 2000, and June, 2001, the working group met four times. Over the seven month period, we reviewed the current scientific literature, dozens of Forest Service technical reports and white papers, draft and final environmental impact statements, and consulted with a wide range of Forest Service staff at national, regional and National Forest levels.

Here we summarize the recommendations that have emerged from our review and evaluation. There are two relatively distinct audiences for our report: the Forest Service national leadership, and individual Forest Service biologists and other staff who are responsible for conducting viability assessments. Accordingly, here we highlight a set of nine general recommendations that are intended for those individuals responsible for setting and maintaining policy within the Forest Service. In Chapter 10 we articulate several practical recommendations, intended to provide guidance to National Forest-level and regional-level Forest Service staff.

General Recommendations

1. The entire viability assessment process, and its role in evaluating potential management alternatives, should be thoroughly documented in an accessible format. Current documentation of viability assessments is inadequate. We reviewed thousands of pages of documentation for several of the Forest Plans we examined. However, in spite of this voluminous documentation, we found that personal consultation with individuals who were directly involved with a particular assessment process often was the only way to determine what methods were used to assess viability, why these methods were chosen, and what role the viability assessment played in evaluating management alternatives.

2. The Forest Service should adopt a systematic and consistent approach to identifying which species on National Forest lands require viability assessments. After considering both scientific credibility and practicality of application, we recommend that all species in the planning area of interest with ABI/Heritage G ranks of 1-3 be considered as an initial set of species for assessment of viability considerations. This set should be further refined and supplemented based on species' distributions, relative to the scale of the planning area, and other factors discussed in Chapters 5 and 6.

3. The Forest Service should establish an internal viability working group that would serve three functions: a) develop standards for data, modeling and expert opinion that would constitute an adequate viability assessment; b) develop additional taxon-specific standards for viability assessments; and c) provide service and support to the

forests and regions in implementing scientifically credible viability assessments in conjunction with forest management and planning efforts. In developing standards, the working group should involve knowledgeable experts from other federal resource agencies (e.g., U.S.F.W.S., N.M.F.S., B.L.M., N.P.S., D.O.D., D.O.E., E.P.A.), relevant professional societies (e.g., Society for Conservation Biology, the Wildlife Society, American Forestry Society, etc.) and the academic community.

4. Whenever possible, formal mathematical modeling approaches should be used in viability assessments, rather than approaches that rely solely or primarily on expert opinion. However, we recognize that the choice and appropriateness of any modeling approach will be limited by the specific objectives of the analysis, the availability of sufficient data such that modelers do not have to guess about the estimates for a large number of model parameters, and the model assumptions that are reasonable to make.

5. All measured data used in viability assessments should be identified as such and should be clearly distinguished from expert opinion or inferred data. Measured data should be made available over the Internet so a broad community of scientists has the opportunity to undertake viability analyses and assessments. For species listed as endangered or threatened under the Endangered Species Act, viability assessments should be based on empirical data and formal mathematical modeling approaches, rather than on expert opinion.

6. The Forest Service should utilize structured, credible and repeatable approaches for eliciting, interpreting, and using expert opinion in the viability assessment process. Viability assessments always will involve some degree of expert opinion and judgment. In some cases this might entail expert judgment as to the appropriateness, and/or choice of a quantitative modeling approach, expert interpretation of empirical data to parameterize quantitative models, and/or expert interpretation of model output. In other instances, expert opinion may be the primary or even the sole method used in viability assessments. Forest Service staff responsible for conducting viability assessments should have appropriate training in structured methods for eliciting and assessing expert opinion. When adequately trained staff is unavailable, the Forest Service may need to employ experienced consultants with appropriate training in the academic discipline of decision theory.

7. Regardless of the method used, specific sources and levels of uncertainty associated with viability assessments should be made explicit, as well as the implications of this uncertainty for decisions involving selection of a management alternative. Where measured data are used, an effort should be made to estimate or account for sampling error, so that this source of variability is not wrongly assigned to "environmental variation."

8. Viability assessment should be viewed as an integral part of the ongoing Forest Service land management and decision process, and, in turn, monitoring should be an integral component of the overall process used by the Forest Service to manage for species viability, including selected species whose likelihood of extinction is minimal, to ensure they remain so.

9. Whatever approach is used for a viability assessment make a formal effort to understand the limitations of that approach, via cross-validation studies.

Chapter 2. Introduction

The Forest Service manages “the lands and resources of the National Forest System, which includes 192 million acres of land in 42 states, the Virgin Islands, and Puerto Rico (see Chapter 3). The System comprises 155 National Forests, 20 national grasslands, and various other lands under the jurisdiction of the Secretary of Agriculture” (Federal Register, November 8, 2000). Several laws guide the management of the National Forest System. These include the Multiple Use Sustained Yield Act (MUSYA), the Endangered Species Act (ESA), the National Forest Management Act (NFMA), and the National Environmental Policy Act (NEPA).

This report focuses on methods for assessing species viability, as required by NFMA planning regulations. The NFMA planning regulations (Federal Register, November, 2000, pp. 67580-67581) define species viability as:

“A species consisting of self-sustaining and interacting populations that are well distributed through the species’ range. Self-sustaining populations are those that are sufficiently abundant and have sufficient diversity to display the array of life history strategies and forms to provide for their long-term persistence and adaptability over time.”

The Forest Service commissioned this report as part of a larger process within the agency intended to simplify, clarify and improve the scientific basis for viability assessments. The results presented here will be used by the Forest Service to develop guidance to National Forests for incorporating species viability considerations into land and resource management plans. The objectives of the Forest Service in commissioning this review were to:

- Establish the scientific basis for geographic and temporal scales used in the assessment of viability.
- Identify approaches that could be used to assess species viability within the context of NFMA, recognizing the need to make long-term (50-100 year) projections of

future conditions, the requirement to deal with a broad array of taxa, and the reality of limited information for most species.

- Describe the strengths and limitations of each approach, including an overall comparison of the range of approaches.

Process

The Working Group first met for four days at NCEAS, in Santa Barbara, in December, 2000. At that meeting, we discussed the objectives for the review, as articulated by the Forest Service, and developed a plan for accomplishing our task, as well as an outline for this report.

Subsequently, the Working Group met at NCEAS for two days in February, 2001, for three days in April, 2001, and for one day in Denver, in May, 2001. In between these meetings individuals and small groups reviewed 1) the relevant scientific literature; 2) dozens of Forest Service documents, including regional-scale Forest Service assessments for the Interior Columbia Basin (both Draft and Final and Supplemental), the Sierra Nevada National Forest, the Northwest Forest Plan (FEMAT), and the Tongass National Forest; 3) Forest-scale plan assessments for the Northern Great Plains, the White River National Forest and the National Forests in Wisconsin and Minnesota; 4) numerous Forest Service white papers. In addition, Working Group members consulted with Forest Service staff at national, regional and individual forest levels.

We note here that one individual from among the thirteen working group members did not agree with all of the language used in the report or with every recommendation.

Structure of this Report

We begin our report with a brief, historical perspective on viability assessment in the context of National Forest management (Chapter 3). In Chapter 4, we clarify the management context in which viability assessments are conducted. Chapter 5 discusses the process for identifying species at requiring viability assessments, and Chapter 6 expands this process to identify focal species. In Chapter 7, we review and compare existing methods for viability assessment, including a review of methods that have been used by the Forest Service. Chapter 8 is devoted to discussion of methods for eliciting and assessing expert opinion. In Chapter 9 we review methods for collecting information so that assessments can be updated and improved over time. Finally, in Chapter 10 we present an overview and summary of our findings.

Acknowledgments

The working group meetings that led to this report were supported by the National Center for Ecological Analysis and Synthesis, a Center funded by NSF (Grant #DEB-0072909), the University of California, and the Santa Barbara campus, and by the U.S.D.A. Forest Service. We thank Gay Bradshaw, Peter Kareiva, Bruce Marcot, Fred Samson and Mark Shaffer for their thoughtful reviews of the final draft, which led to numerous improvements and clarifications.

Chapter 3. Viability Analysis in the Context of National Forest Management

Historical Perspective

The conservation of native species and biodiversity has been integral to management of National Forests and grasslands in the United States for over a century. Many excellent narratives on Forest Service history are available, including Hirt (1994), Beesley (1996), Rush (1996), and Steen (1997). The Committee of Scientists (1999) and Floyd et al. (2000) provide reviews supporting their suggestions for improved forest planning. John Fedkiw, a former senior executive with over 30 years experience in U.S.D.A., has written two histories (Fedkiw 1989, 2000), providing unique institutional perspectives[2]. The overview below is paraphrased from these sources.

Many of today's National Forests initially were designated as forest reserves in the late 19th and early 20th centuries. These reserves were created from lands that previously were subjected to a range of uses, from pristine wilderness areas to lands that previously were used for logging, mining, and livestock grazing. Destructive flooding following loss of forest cover was another motivation for limiting exploitation of public lands. Recognizing the interconnectedness of forests and water, the Organic Act of 1897 mandated management of the forest reserves, "for the purpose of securing favorable conditions for water flows and to furnish a continuous supply of timber for the use and necessity of the citizens of the United States." Many of the most productive timberlands and lands better suited for agriculture and mineral development were explicitly excluded from the forest reserves and instead were entrusted to private ownership. The forest reserves were transferred to the newly created Forest Service in 1905, and designated as National Forests in 1907. By 1915, the National Forest system included 162 million acres gleaned from the public domain of the western states.

---

[2] Both documents are available on the internet: http://www.fs.fed.us/pl/rpa/89pdf/Use_Management.pdf and http://www.fs.fed.us/research/publications/managing_multiple_uses_on_nation.htm

The National Forests east of the Mississippi River have a different history. The Weeks Law of 1911 authorized the purchase of private lands for "the protection of the watersheds of navigable streams." This allowed the creation of National Forests in the eastern states where essentially there was no federal land. Much of the purchased land consisted of abandoned homesteads and farms that had been burned or were heavily eroded. Twenty-four million acres were added to the National Forest system from 1911 to 1945. The goal of maintaining navigable flows was an overriding factor in both the location and management of these forests. Protection from wildfire was another major concern.

For most of the period prior to World War II, resource extraction on National Forests was relatively limited. The primary commodity use was widespread livestock grazing in the western forests and grasslands. The Forest Service invested in establishing well-defined boundaries, preventing timber theft, administering special activities, and perhaps most importantly, suppressing fire. As demand for timber increased dramatically during World War II and in the post-war period, the National Forests became an increasingly important supplier of timber. More and more Americans also began looking to the National Forests for recreational opportunities. Timber and recreation thus became the principal drivers of forest management. Rising concerns over the balance among competing resource uses prompted passage of the Multiple Use-Sustained Yield Act of 1960 (MUSYA). The new law restated Congressional intent and widened the set of preferred uses: "the national forests are established and shall be administered for outdoor recreation, range, timber, watershed, and wildlife and fish purposes." Furthermore, the establishment and maintenance of wilderness areas were identified specifically as consistent with the intent of MUSYA. A series of environmental laws with implications for forest management soon followed MUSYA: the Wilderness Act of 1964, the National Environmental Policy Act of 1969 (NEPA), the Endangered Species Act of 1973 (ESA), and others.

While MUSYA specifically acknowledged the inevitable conflict inherent in trying to satisfy multiple objectives within the National Forests, it provided minimal direction as to how such conflicts should be resolved. As Fedkiw (2000) notes, multiple-use management evolved from attempts to accommodate different uses—each championed by a different interest group—without impairing the long-term productivity of the land. It was not an independent objective, per se, with its own supporters.

The National Forest Management Act of 1976 (NFMA) is the principal statute driving National Forest planning today[3]. While not changing the multiple use and sustained yield focus of forest management, NFMA called for extensive planning and public involvement. The intent was to reconcile competing public demands at the scale of the National Forest. Congress recognized that conflicts between resource extraction, amenity values and ecological issues such as biodiversity were an integral part of public land management. Rather than resolve such conflicts legislatively, Congress enacted a procedural planning process. The intent was that thorough and open analysis involving "integrated consideration of physical, biological, economic, or other sciences" could lead to local resolution of conflicts and wider acceptability of decisions. Each National Forest or grassland was required to develop a land and resource management plan (forest plan). The purpose of the forest plan was to guide all resource management activities for a 10-15 year period.

Direction in NFMA for conservation of species and ecosystems is fairly general. Specifically, NFMA directs planners to:

> Provide for diversity of plant and animal communities based on the suitability and capability of the specific land area in order to meet overall multiple-use objectives, and within the multiple-use objectives of a land management plan adopted pursuant to this section, provide, where appropriate, to the degree practicable, for steps to be taken to preserve the diversity of tree species similar to that existing in the

---

[3] Details regarding the Forest Service planning process and the statutes that govern this process are readily available on Forest Service websites. Specific sites are referenced below, but a useful starting point is http://www.fs.fed.us/forum/nepa/.

region controlled by the plan" (16 U.S.C. 1604(g)(3)(B)).

NMFA directed the Secretary of Agriculture to develop a set of planning rules to guide implementation of the Act. The first planning rule was developed in 1979, but never implemented. A second planning rule developed in 1982 has guided National Forest management until the present.

3.1.1 The 1982 Planning Rule

All existing forest plans were developed under the 1982 planning rule. Most forest plans were developed during the mid-1980s, and all were completed by the mid-1990's. The 1982 rule stipulates that revision should occur about every 10 years, but no later than 15 years after approval of the original plan or latest plan revision. In order to comply with the statutory 15-year requirement, plan revision efforts have been completed or are underway on many National Forests. In addition, the plan revision and amendment processes associated with large-scale assessments such as the Northwest Forest Plan, Interior Columbia River Basin Ecosystem Assessment Project, and the Sierra Nevada Forest Plan Amendment have followed the 1982 planning rule.

The 1982 planning rule[4] brought increased visibility and importance to species viability within forest planning. Section 219.19, Fish and Wildlife Resources, stipulates:

> *Fish and wildlife habitat shall be managed to maintain viable populations of existing native and desired non-native vertebrate species in the planning area. For planning purposes, a viable population shall be regarded as one which has the estimated numbers and distribution of reproductive individuals to insure its continued existence is well distributed in the planning area. In order to insure that viable populations will be maintained, habitat must be provided to support, at least, a minimum number of reproductive individuals and that habitat must be well distributed so that those individuals can interact with others in the planning area.*

The rule further defines seven requirements necessary to achieve this objective, including direction regarding management indicator species. As the name implies, "these species shall be selected because their population changes are believed to indicate the effects of management activities." Planning teams are to estimate the effect of proposed alternatives on management indicator species, monitor their population trends through time, and determine habitat relationships.

Non-vertebrates animals and plants are covered more generally through requirements to maintain biodiversity. Section 219.27(g) states:

> *Management prescriptions, where appropriate and to the extent practicable, shall preserve and enhance the diversity of plant and animal communities, including endemic and desirable naturalized plant and animal species, so that it is at least as great as that which would be expected in a natural forest and the diversity of tree species similar to that existing in the planning area. Reductions in diversity of plant and animal communities and tree species from that which would be expected in a natural forest, or from that similar to the existing diversity in the planning area, may be prescribed only where needed to meet overall multiple-use objectives.*

The 2000 Planning Rule

The Forest Service has continually sought to refine and streamline the planning process with the mutual goals of strengthening the scientific basis of management and improving public involvement and agency accountability. The 1982 planning rule was reviewed in 1989 and changes proposed in 1995[5]. In 1997, the Secretary of Agriculture

---

[4] The 1982 Planning Rule is accessible online at http://www.fs.fed.us/forum/nepa/rule/nfmareg.html
[5] See http://www.fs.fed.us/forum/nepa/rule/chronology.doc.

appointed a committee of thirteen scientists to review the Forest Service planning process and offer recommendations for improvements. Based on the scientists' report (COS 1999)[6], an internal Forest Service team developed a new planning rule, and released it for public review in 1999. Following public comment, the new planning rule was adopted in November, 2000[7].

3.1.2 Emphasis on Sustainability

One distinguishing feature of the 2000 planning rule is that it identifies ecological sustainability as the foundation necessary for production of all benefits from National Forests and grasslands. Section 219.2(a) states:

> *The first priority for planning to guide management of the National Forest System is to maintain or restore ecological sustainability of national forests and grasslands to provide for a wide variety of uses, values, products, and services. The benefits sought from these lands depend upon long-term ecological sustainability. Considering increased human uses, it is essential that uses of today do not impair the functioning of ecological processes and the ability of these natural resources to contribute to sustainability in the future.*

Sustainability as an overarching goal of natural resource management has been widely advanced by resource professionals and lay persons alike (COS 1999). The exact definition of the term and its application to management of the public lands is an evolving concept, much debated in the literature and elsewhere (e.g., Kolb et al. 1994, Amaranthus 1997, Floyd et al. 2001). The COS report (1999) and the 2000 planning rule both recognize the interconnectedness of ecological, social, and economic sustainability.

---

[6] The COS report is available at http://www.fs.fed.us/news/science.
[7] Available at http://www.fs.fed.us/forum/nepa/rule/fedreg.pdf.

Ecological sustainability is defined in Section 219.36 (Definitions) as, "the maintenance or restoration of the composition, structure, and processes of ecosystems including the diversity of plant and animal communities and the productive capacity of ecological systems." The rule repeatedly stresses that ecosystem diversity and species diversity are major components of ecological sustainability that must be maintained or restored. Much attention is given to the type of information and analyses that can be used to assess and plan for ecosystem diversity and species diversity (Section 219.20).

The language used in reference to plan decisions (Section 219.20(b)) is perhaps the most relevant to our review:

> 1) *Ecosystem diversity*. Plan decisions affecting ecosystem diversity must provide for maintenance or restoration of the characteristics of ecosystem composition and structure within the range of variability that would be expected to occur under natural disturbance regimes of the current climatic period …
> 2) *Species diversity*. (i) Plan decisions affecting species diversity must provide for ecological conditions that the responsible official determines provide a high likelihood that those conditions are capable of supporting over time the viability of native and desired non-native species well distributed throughout their ranges within the plan area …
> 3) *Federally listed threatened and endangered species*. (i) Plan decisions must provide for implementing actions in conservation agreements with the U.S. Fish and Wildlife Service or the National Marine Fisheries Service that provide a basis for not needing to list a species. … (ii) Plan decisions involving species listed under the Endangered Species Act must include, at the scale determined by the responsible official to be appropriate to

> the plan decision, reasonable and prudent measures and associated terms and conditions contained in final biological opinions issued under 50 CFR part 402.

The 2000 planning rule also clarifies that there are situations in which it is not expected that forests can meet the basic requirement to provide conditions that have a high likelihood of supporting species viability. These include situations:

> 1) When conditions outside the authority of the agency prevent the agency from providing ecological conditions that have a high likelihood of supporting over time the viability of native and desired non-native species…
>
> 2) Where species are inherently rare and not naturally well-distributed….
>
> 3) Where environmental conditions needed to support a species have been so degraded that it is technically infeasible to restore ecological conditions that would provide a high likelihood of supporting viability…

Definition and Interpretation of Viability

The development of the 2000 planning rule involved considerable discussion regarding the definition of viability and the actual wording to be included in the rule.

The 1982 planning rule definition of viability emphasized distribution and persistence (Section 219.19):

> *For planning purposes, a viable population shall be regarded as one which has the estimated numbers and distribution of reproductive individuals to insure its continued existence is well*

> *distributed in the planning area.*

The Committee of Scientists (1999) expanded the definition to include diversity of life histories and forms:

> *A viable species is defined as consisting of self-sustaining populations that are well distributed throughout the species' range. Self-sustaining populations are those that are sufficiently abundant and have sufficient genetic diversity to display the array of life-history strategies and forms that will provide for their persistence and adaptability in the planning area over time.*

The 2000 planning rule definition is a variant of the COS definition, adding a requirement that populations interact and dropping the modifier, "genetic," (Section 219.36):

> *Species viability*: A species consisting of self-sustaining and interacting populations that are well distributed through the species' range. Self-sustaining populations are those that are sufficiently abundant and have sufficient diversity to display the array of life history strategies and forms to provide for their long-term persistence and adaptability over time.

All three definitions are concise statements about the potential or capacity of a species to persist over an indefinite time period, acknowledging the variation in environmental conditions to be experienced over that period.

Methods and Extent of Analysis

Every interdisciplinary team charged with developing or amending a forest plan has to answer 3 basic questions regarding viability assessments:

1) Which species are to be assessed?
2) What type of analysis should be used for each species?
3) How much analysis is enough?

The 2000 planning rule offers limited guidance in answering these questions. Viability assessments for all species listed under the Endangered Species Act are required. For other species, planners can use discretion in selecting surrogate species or species groups to evaluate ecological conditions for all species. We discuss the use of surrogate species in Chapter 6, however, the use of surrogate species is highly speculative, and should be undertaken with considerable caution. Management and monitoring surrogates frequently are selected without adequate testing of their performance ability (Simberloff 1998; Caro and O'Doherty 1999). This, in turn has led to confusion about when a species is being monitored for its own sake, and when a species can appropriately function as an accurate indicator of other organisms or environmental attributes. In Chapter 5, we describe methods for selecting species for viability assessments. With respect to question 2, the planning rule does not mandate a particular type of viability assessment, but it does specify the kind of information to be used (Section 219.20):

> *In analyzing viability, the extent of information available about species, their habitats, the dynamic nature of ecosystems and the ecological conditions needed to support them must be identified. Species assessments may rely on general conservation principles and expert opinion. When detailed information on species habitat relationships, demographics, genetics, and risk factors is available, that information should be considered.*

The methods that can be used to assess the viability of a particular species will be constrained by the kinds of information available. In Chapter 7, we discuss a variety of methods best suited for different kinds of information.

Ultimately, the limits of even the best information bases and analyses are reached and decisions must be made in the face of uncertainty. Thus, monitoring to improve the information base for future decisions and adaptive management are essential elements of effective management. Section 219.11 of the 2000 planning rule addresses monitoring and evaluation for adaptive management. In Chapter 9, we return to the subject of monitoring in some detail.

Chapter 4. Decision Framework for Viability Assessments

Viability Assessment Is a Component of Forest Management

Viability assessment is one component of the larger Forest Service management and decision process. Two phases of this larger process are shown in abbreviated form on the left side of Figure 4.1, intersecting with the steps of viability assessment, shown on the right. “Ongoing management” (upper left) refers to activities guided by existing forest plans and accompanying project plans. The decision process (lower left) is initiated when a forest plan is revised or amended, or when a new project is proposed. Many types of analysis and monitoring (e.g., estimation of fiber yields from timber management alternatives, prediction of water quality impacts from road construction) provide input to the overall management process; and we deal with only one of those here: viability assessment. In addition, our focus is specifically on species viability, which is but one component of ecosystem diversity and sustainability.

Viability Assessment Is a Component of Ongoing Forest Management

Undertaking forest management activities (step 3), monitoring the effects of those activities on forest resources (step 4), and using monitoring data and analyses to assess forest management in the light of social and resource needs (step 1) are part of the ongoing forest management process. Assessing impacts on species viability is part of this ongoing process, too, through monitoring the effects of forest management activities on the status of species of concern and using those data to assess the need for a change in current management.

The 2000 planning regulations provide two approaches for considering ecological sustainability. One approach addresses sustainability at the level of ecosystem diversity. The other approach focuses on species viability. Our group was specifically charged with considering species viability assessments, and while we recognize that ecosystem-level diversity is a consideration in forest management and planning processes, we limit our review to the species viability approach.

Integrating species viability assessment into ongoing forest monitoring and management (connections between upper left and right-hand circles in Figure 4.1 shown by double arrows) will ease the burden of assessing species viability during project or plan level decision processes (lower left) by ensuring that the information necessary to carry out the assessments is already available. Analyzing these data in the periods between active decision processes can provide early warning of threats to viability for species of concern (at step 1), which may in turn suggest that a change in management is needed (step 2).

Viability Assessment Is a Component of Forest Management Decisions

At intervals, assessment of current management (step 1) will suggest the need for decisions (step 2) that require a commitment of resources, triggering a formal analysis of alternatives (steps I-II), followed by the selection of an alternative (step III) to guide the next phase of forest management. Some decision processes, such as revisions of forestland management plans, occur on a predictable schedule. Others may be triggered by social needs (e.g., changes in public opinion about the proper mix of extractive and nonextractive uses of National Forest land) or special resource management needs (e.g., clean-up after natural or human-caused fires). Sometimes the trigger for a management decision process may be the need to take positive action on behalf of species at risk of extinction, as revealed by ongoing monitoring and assessment (steps 4 and 1). Some decision processes are at a broad spatial scale, as in regional assessments or forest plan revisions; some are project-level decisions at a much smaller scale, such as siting a new campground.

Viability assessment plays a role in ongoing management (upper left, Fig. 4.1): the early warning function described above. It plays an even bigger role in active decision processes, when forest plans are being revised or amended, or when project-level decisions to implement forest plans are being made (lower left, Fig. 4.1). Viability is one of many forest management objectives that guide the design of alternative forest management schemes (step I), the analysis of the merits and demerits of those alternatives relative to the multiplicity of management goals (step II), and the weighing of priorities among those goals that leads to a final decision (step III).

Assessment of Management Issues Informs the Viability Assessment

All of the elements of the viability assessment process (right-hand side of Fig. 4.1) are informed by the review of management issues (step 1) that initiates a decision process (lower left): What issues are to be addressed by the upcoming decision? Is species recovery a main target of the decision? What alternatives are likely to be considered? Are these likely to affect habitat for vulnerable species, whether negatively or positively? Will there be direct or indirect impacts on the organisms themselves (e.g., increased hunting due to improved access)? What are the spatial and temporal scales of proposed activities?

Choosing Species for Viability Assessment

The answers to questions arising from an assessment of the management issues will help to identify species (or other taxonomic groupings) likely to be affected by proposed activities and whose viability under various management alternatives must be assessed (step A). Rough lists of species requiring viability assessments can be drawn from existing resources, such as species listed as endangered or threatened under the Endangered Species Act, state Natural Heritage element lists, lists of state endangered, threatened or species of concern, and National Forest lists of sensitive species. Chapters 5 and 6 discuss methods for identifying taxa that should be the subjects of viability assessments.

Although steps A, Choosing Species for Viability Assessment, and B, Selecting an Appropriate Level of Analysis, are presented as separate tasks here, they are really part of a continuum of species viability assessments. The initial step of identifying species at risk is itself an assessment of viability, albeit largely intuitive and unstructured. Many of the species for which assessments must be made will receive only slightly more structured scrutiny. Only a very few high profile and well-studied species will receive the sort of detailed and quantitative analysis usually characterized as population viability assessment.

Selecting An Appropriate Level of Analysis

The next step in the viability assessment (step B) is to determine what type of analysis should be undertaken for each category of species identified in step A. Two factors that enter into that choice include (1) the spatial and temporal scale of proposed activities and (2) the potential for cumulative effects as a result of combinations of activities through time and across the landscape, including actions on lands adjacent to National Forest land. How do the species distributions match up with the spatial scales of proposed actions? It is rarely appropriate to conduct viability analyses at the spatial scale typical of project-level decisions; assessments at broader scales will usually be both more meaningful biologically and more cost effective. For species whose distributions are much larger in scale than the proposed actions, refer to a regional scale viability assessment if one is available; if not, consider undertaking a regional scale assessment. For species with distributions of about the same scale as the proposed actions, plan for a viability assessment for this proposal, unless one is already available for a sufficiently similar situation. For species whose distributions are much smaller than the scale of the proposed activities, as may be the case for narrow endemics or species specializing on small, patchily distributed habitats, plan to do site-specific assessments for any management activities that will affect areas where the species occurs.

When determining an appropriate type of analysis, consider the possibility that existing assessments may do the job adequately. For species already covered by a recovery plan or by agreements with other agencies, and for management proposals that fall within the scope of existing, broader scale viability assessments, no additional analysis may be needed, other than ensuring that proposed alternatives fall within ranges considered acceptable in existing plans and ensuring that the cumulative effects of many actions taken on a smaller scale still fall within the regional guidelines. For the remaining species or groups, some preliminary analysis is needed to identify the level of viability assessment that best matches anticipated changes in management.

For species with tight connections between habitat characteristics and population dynamics, analyses that focus on changes to habitat may provide the basis for an adequate assessment. When population dynamics may be affected by changes in factors other than habitat (e.g., new roads increase human access, leading to higher mortality), then analyses that incorporate changes in population parameters are needed.

It is very demanding to make a viability assessment that expresses the impact of proposed management activities on a species' viability in terms of likelihood of extinction within a specified period of time within narrow confidence limits. Therefore, identifying how much precision is needed, or even possible to estimate, given available data, is important to selecting an appropriate type of assessment. When the results of viability assessments are used to help analyze management alternatives (step II), decision makers need to know which alternatives are better or worse for species viability. They also need to know how much better or worse an alternative may be, because impacts of an alternative on viability may need to be traded off against impacts on other biological or socioeconomic goals. It also may be necessary to judge which alternatives may push a species below some specified likelihood of persistence that has been deemed unacceptable in the context of legal requirements under the Endangered Species Act or NFMA requirements for maintaining viability. When some management alternatives may push species below this level, the demands for precision in viability analysis are greater than when only a comparative estimate of viability is needed. An example of the latter might be comparisons among activities intended to foster species recovery, where all of the alternatives being considered are expected to improve species viability compared to the status quo.

If some species will have especially high visibility, with attention from outside groups and potential for legal action, it may be desirable to do a more precise viability assessment than would be warranted on the basis of scientific considerations, such as biological characteristics and anticipated management impacts, alone.

All of these considerations will suggest a level of analysis (step B) for each species or group that matches the need for precision in estimating viability levels, appropriate spatial scale, impact on habitat only vs. impact on habitat and also on individuals, tightness of association between population and habitat, and level of public attention. Chapter 7 describes methods for conducting viability assessments that satisfy these different needs and provides guidance for choosing among them.

Evaluating the Adequacy of Available Data

Once the list of species (step A) and the desired level of viability assessment (step B) have been determined, the next question is: Will the available data support the desired level of analysis (step C)? If so, proceed with the analysis (step D) and forward the results to the synthesis step (step G), and then on to the design (step I) and analysis (step II) of management alternatives. In addition to the analysis, recommend a monitoring scheme (step 4) to check the predictions of the assessment as decisions are made (step III) and alternatives implemented (step 3).

When Adequate Data Are Not Available

If, as is often the case, the data are not available to support a viability analysis at the desired level, evaluate whether or not the decision timeline can be delayed in order to collect and analyze such data. If so, gather the information that is needed (step E) and then use it in the desired analysis (step F). If that isn’t feasible, compare the risks of proceeding with a level of assessment that does not match the potential consequences of management action for species viability with the risks of delaying the decision process to gather additional information (step E). If the choice is to proceed with a less suitable assessment than desired, the associated risk of error must be communicated forcefully to those responsible for the larger management decision process (lower left). It is equally important to communicate the necessity of investing in a well-designed scheme for monitoring the effects of management actions (step 4), so that decisions that will appreciably diminish species viability can be detected and corrected before it is too late. Experiencing the risks associated with a viability assessment below the desired level of analysis, or the delays associated with gathering the information needed for the desired level, will help motivate forest managers to look ahead toward the kinds of viability assessments that are likely to arise in future management decision processes, so that they can collect information on vulnerable species as part of ongoing monitoring and data analysis (upper left), rather than waiting until a decision is imminent (lower left).

Synthesizing the Results of Multiple Viability Assessments

Usually, viability assessments for forest management will encompass many species and many kinds of analyses, with results phrased in disparate terms (e.g., some in terms of probability of extinction in a given period of time, and others in terms of a category on a rule-based rating scale, such as that used by the Association for Biodiversity Information/Heritage Programs). When the results of viability assessment are passed forward to the forest management decision process, in order to inform both the design and the analysis of management alternatives (lower left), these disparately phrased results from many species must be integrated with all of the other inputs to forest management decisions (e.g., water quality impacts, socioeconomic impacts) before a decision on a preferred management alternative can be reached. If those responsible for the viability assessment merely pass along the raw results for many species, with no attempt to distill and interpret them, the impact of the viability results in the overall decision process may be severely diminished. Those conducting the viability assessments must make some synthesis of the viability results (step G), but in doing so, they must guard against inadvertently incorporating their own value judgments about the importance of various species, or about the levels of viability that are acceptable or unacceptable, as they combine results across species and species groups.

To some extent, any synthesis will include some value judgments (e.g., an arithmetic average implies equal weights on the items being averaged). Where these can’t be avoided, they should be made as explicit as possible, so their impact on subsequent conclusions will be transparent. One way of guarding against the misuse and misunderstanding of the synthesis of viability results is for those responsible for the viability assessment to communicate ahead of time with those responsible for the overall analysis to negotiate the terms in which viability will be expressed for each species or group of species. Some of the items that should be discussed are: the time frame that should be used, whether a numerical estimate or a qualitative category will be used to express level of viability, and how the level of confidence or level of uncertainty in the results will be expressed. In the absence of input of this sort from the overall analysis, those responsible for viability assessment may be obliged to make a rough categorization of their viability results for each alternative using, perhaps, three categories: (1) species which, under any plausible assignment of value, are likely to remain above an acceptable level of likelihood of viability; (2) species which, under any plausible assignment of value, are likely to fall below this acceptable level; and (3) those species that are in between. In the overall analysis, most attention is likely to be paid to the second category, although most of the scientific debate might be taken up by the third.

Viability Assessments Inform Design and Analysis of Management Alternatives

Information from viability assessment is only one of many factors that go into the choice among alternatives in a multi-objective decision process (step III). For listed species, negative impacts of proposed actions may trump other forest management goals; for many other species, effects of proposed actions on viability will be weighed alongside effects on other important objectives. If some alternatives impose unacceptable burdens on species of concern, but are otherwise desirable, viability assessment may suggest new alternatives or modifications to existing alternatives (step I) that would have less impact. Therefore viability assessment should be viewed as part of the process of designing alternatives, as well as evaluating them. There often may be opportunities to use insights from viability assessment to modify alternatives to enhance the welfare of species of concern, even when that has not been the main impetus for the planning process.

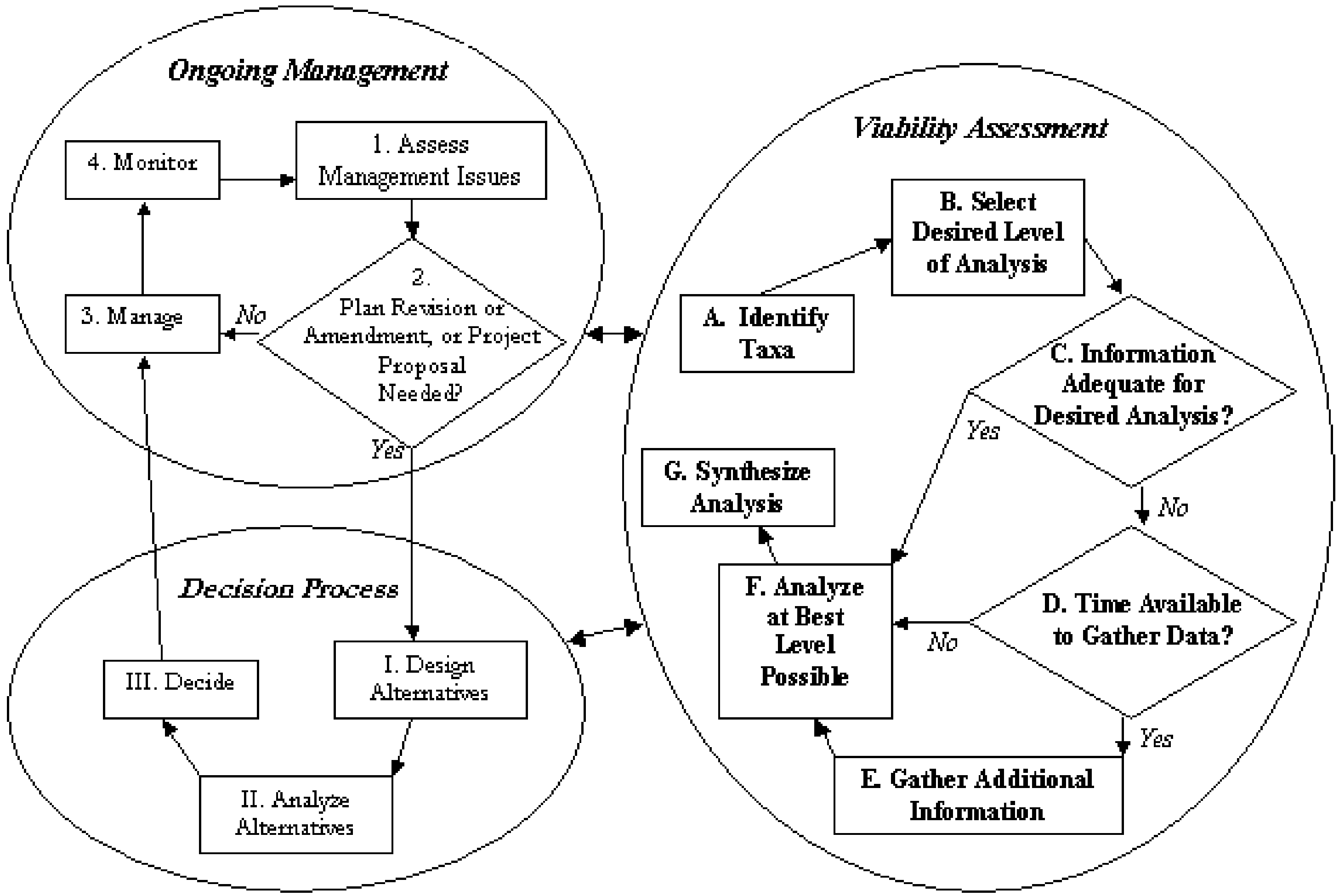
Monitoring Is an Essential Part of Viability Assessment

Once a decision has been made (step III) and the selected alternative implemented (step 3), ongoing monitoring (step 4) must include sampling to check the effects of forest management on species of concern. Even the best viability assessments are laced with uncertainty, due to both limited knowledge and inherent variation in biological systems. Using ongoing monitoring to check population status, estimate population parameters, and verify biological relationships incorporated in viability analyses will incrementally improve both data and models. Chapter 9 recommends monitoring approaches to fulfill these needs. Then, in future decision processes, the information needed to support the desired level of viability assessment (step C) will be immediately at hand. In addition, information gathered on the status of species of concern informs the ongoing assessment of the forest management situation (step 1), alerting forest managers when a new decision process should be initiated (step 2).

Summary

Viability assessment is an essential component of both ongoing forest management and forest management decision processes. Ongoing monitoring of species viability forms part of the management assessment process, which may lead to a need for new forest management decisions. The spatial and temporal scales of proposed activities, the likely impacts of those activities on habitats and individuals for species of concern, and the level of public and legal scrutiny for different species all influence the type of viability assessment that is needed for each species or group. Too often, the data needed to support the desired level of analysis are lacking, and the risks of proceeding with a more limited analysis must be weighed against the risks of delaying a decision while the necessary data are gathered. The more precise the estimates of viability desired, the greater the data demands. Once assessments have been made for species of concern, those results must be synthesized and then passed forward to the forest management decision process. Prior discussion of the terms in which those viability results will be reported will help ensure they are incorporated in both the design and analysis of management alternatives. A viability assessment should also strive to determine the degree to which population size and trends of species of concern are influenced by (1) management activities and (2) other factors. If management activities are the primary influence it is desirable to further elucidate which activities have the greatest impact. If factors other than management activities are most influential, then it should be clarified that alteration of current or proposed management activities will do little to ameliorate adverse viability conditions. Once viability assessments have been weighed, along with all of the other inputs to forest management decisions, and a management alternative has been chosen, well-designed monitoring of effects of management on species of concern will set the stage for improved viability assessments in the future.

Figure 4.1. The interaction of viability assessment processes (on right, A-G) with the Forest Service decision process (lower left) and with ongoing forest management (upper left).



Chapter 5. Selecting Species for Which to Conduct Viability Assessments

Introduction

The NFMA planning regulations define species at risk as "federally listed endangered, threatened, candidate, and proposed species, and other species for which loss of viability, including reduction in distribution or abundance, is a concern within the plan area. Other species-at-risk may include sensitive species and state-listed species. A species-at-risk also may be selected as a focal species" (Federal Register 65:67580, 2000).

Federal agencies, such as the U.S. Forest Service, are legally required to make efforts to protect and recover species listed as endangered or threatened under the Endangered Species Act. However, identifying the full set of species that may be at risk from proposed future management activities is more complex and there is no universally accepted approach (Raphael and Marcot 1994; Holthausen et al. 1999). Numerous methods for classifying species according to their risk of extinction are used by management agencies and governments throughout the world. Some are intended to be applied within a local region or state (e.g., Millsap et al. 1990, Lunney et al. 1996, Breininger et al. 1998), while some have national (e.g., Mexico 2000, Molloy and Davis 1992, Czech and Krausman 1997) or international status (e.g., IUCN 1994, 2000), or are used at multiple political scales (e.g., USFWS 1983, Master 1991). Yet others have been developed to apply to specific taxonomic groups (Carter 2000, Wade 1998). These protocols share many attributes and use similar information, such as population size, extent, number of populations, and trends in at least some of these variables. All result in an assessment of threat, couched in words that reflect the probability of decline or loss of a taxon within some time period.

There also are significant differences among these methods, including the extent to which they consider management variables, taxonomic status, recoverability, and assessments of past or future trends. They also employ different logical systems to interpret data, treat missing data differently, and apply arbitrary weights to the variables.

Typically, a particular protocol has been developed with particular management, regulatory and administrative processes in mind. Thus, the operation of any particular method and the kinds of variables it employs reflect assumptions about management context, although these assumptions usually are implicit, not explicit.

The U.S. Forest Service has used a range of methods to identify sensitive species and/or species at risk. Sensitive species, as defined by the Forest Service, are species known to occur on National Forests, for which population viability is a concern, as evidenced by significant current or predicted declines in population numbers or density, or significant current or predicted declines in habitat capability that would reduce a species' existing distribution. Within the agency, there is no single consistent process for identifying sensitive species at the scale of a National Forest or region. While there have been efforts to establish a consistent approach, these largely have been unsuccessful. Instead, a range of information sources typically is used, including, but not restricted to federally listed species, U.S. Fish and Wildlife Service or National Marine Fisheries Service candidates for federal listing under the Federal Register Notice of Review; state lists of endangered, threatened, rare, endemic, unique, or vanishing species, especially those listed as threatened under state law. Two regions (Regions 1 and 2) that administer the National Grasslands in the Great Plains, and the Intermountain Region (Region 4) have adopted multi-criteria, point scoring schemes to identify potentially species for inclusion on sensitive species lists. The criteria used include: abundance, distribution, level of threat of habitat loss, population impacts, specialized habitat/ecological amplitude and declining population trends. If the total score for a species is greater than a threshold, then the species is nominated for sensitive designation. Other regions base sensitive species lists on U.S. Fish and Wildlife Service candidate listings and Association for Biodiversity Information ranks. The lists of species at risk resulting from these processes often number in the hundreds (Sierra Nevada FEIS 2000) to over a thousand (Thomas et al. 1993a, b) for the National Forests and BLM lands in the Pacific Northwest (Meslow et al. 1994).

In this chapter we focus on systematic approaches for classifying species at risk, with the aim of identifying a scientifically sound, yet practical approach that can be used by the

Forest Service to meet its obligations for identifying species at risk under the NFMA planning regulations. A related issue - the identification and use of focal species to assess species at risk - is addressed in Chapter 6. In addition to methods for ranking species, similar methods have been used to rank environments or habitats for their capacity to support particular species (e.g., Lindenmayer and Possingham 1996, Hanski 1999a). Here we review only existing protocols for classifying species according to their risk of extinction, and evaluate the protocols according to criteria relevant to the Forest Service planning and management process. Finally, we consider additional criteria for identifying species at risk, as a function of the spatial distribution of populations and its relevance to the scale of Forest Service management activities.

Existing Protocols

The classification systems described here are an illustrative sample of the variety of methods available to categorize species according to some direct estimate of the risk of extinction or an indirect estimate using factors associated with the risk of extinction. Classification systems vary significantly, depending on the objectives of the system, the relevant species, the data available, and the scale of consideration (Root, in press). These factors need to be considered explicitly when selecting a protocol for prioritizing species for population viability analysis.

IUCN 1994 and 2000. The IUCN classification system (Mace and Lande 1991; IUCN 1994; 2000) is a rule-based approach to classifying species according to their risk of extinction at a global scale. Four categories of risk are specified: critically endangered, endangered, vulnerable, and lower risk. The criteria are quantitative and based on principles of population biology. Species are classified by population size and number; population growth trends; distribution; degree of fragmentation; and estimates of extinction risk within given time frames. Each parameter has sub-criteria with numerical thresholds or qualitative protocols. Species do not have to satisfy all of the criteria to be placed in a class, just a minimum number of them. The difference between the IUCN 1994 and 2000 protocols is that the latter considers current and future management activities that might be the cause of a population decline. The IUCN protocols have been applied to 2021 species covering a broad range of taxonomic groups in the United States. For a full list of ranked species see http://www.redlist.org/search/search-basic.html.

ABI/Heritage. The Association for Biodiversity Information/Heritage Program Global (G) and State (S) ranking protocols (formerly known as The Nature Conservancy G and S ranks) also were designed to classify species based on their risk of extinction. The classification scheme (modified from Master 1991 and Master et al. 2000) uses the following factors: number of occurrences and number of occurrences with good viability; population size; extent of occurrence; area of occupancy; short- and long-term trends in population size and geographic range; scope, severity and imminence of threats; number of appropriately protected and managed occurrences; and intrinsic vulnerability. Most of the factors involve explicit numerical criteria. There are five classes: critically imperiled; imperiled; rare or uncommon; apparently secure; and secure. Unlike the IUCN classification scheme, all of the criteria are taken into account in the classification of a species. The ABI/Heritage protocols also incorporate information about current and future threats to populations. It is designed to be flexible across many scales. The ABI/Heritage protocols have been applied to 8164 vertebrate and selected invertebrate species (http://www.abi.org/datasets_zoo/overview.htm) and to about 16738 plant species from a broad range of taxonomic groups in the United States and Canada. However, most of these species have been classified according to the scheme in Master (1991) and Master et al. (2000), rather than the modified version of the protocols. It currently is unknown how many species have received ranks under the modified protocols. NatureServe™ (http://www.natureserve.org) provides data, and Heritage conservation status and U.S. ESA ranks for over 46,000 plant and animal species and subspecies in the U.S.

USFWS. The United States Fish and Wildlife Service developed guidelines for classifying species as either endangered or threatened, as defined by the Endangered Species Act of 1973 (ESA) (USFWS 1983). The ESA states that the following factors determine whether or not a species should be listed as endangered or threatened (Nicholopoulos 1999): the present or threatened destruction, modification, or curtailment of the species' habitat or range; over-utilization for commercial, recreational, scientific, or educational purposes; disease or predation; the inadequacy of existing regulatory mechanisms; and other natural or man-made factors affecting the species' continued existence. There are no numerical thresholds or criteria and the classification system is primarily subjective. This system is intended to be quick and straightforward to expedite determination of a species' status. It involves collecting the best available commercial and scientific data, and making a decision about the status of a species based on these data. These protocols take into account current and future threats and management activities that may cause population decline. Approximately 1244 species from a broad range of taxonomic groups have been classified as endangered or threatened under the USFWS guidelines (http://ecos.fws.gov/tess/html/boxscore.html). The National Marine Fisheries Service (NMFS) also lists species under the ESA, however they use a different protocol (Ruckelshaus, personal communication).

Millsap et al. (1990). The Millsap et al. (1990) classification system was designed to categorize vertebrate species in Florida, based on their risk of extinction. The scale is regional rather than global or national. A series of scores is assigned for a range of parameters. These include: population size; population trend; range size; distribution trend; population concentration; reproductive potential; and ecological specialization. Each criterion has a number of subclasses, defined as either numerical thresholds, or as subjective criteria. Scores for each parameter are summed to give a "biological score." A separate set of criteria that includes rankings based on the current state of knowledge of distribution, population trend, limiting factors and the current extent of conservation efforts designates an "action score." The protocols then can be used to rank species to prioritize conservation efforts. The absolute scores also can be used to differentiate between endangered and vulnerable species. Recent and potential threats are not explicitly taken into account. The Millsap et al. scheme has been applied to 668 vertebrate species (up to 1990) for a broad range of taxonomic groups in Florida (excluding marine fish species).

Lunney et al. (1996). This classification scheme (Lunney et al. 1996) is a modification of the Millsap et al. (1990) protocols for the purpose of systematically evaluating the conservation status of all mammals, birds, reptiles and amphibians in New South Wales (NSW), Australia, in accordance with the NSW National Parks and Wildlife Act of 1974. Parameter definitions are modified to address the geography of NSW, which is five times larger than Florida, and the time since European settlement (which is shorter than European settlement in Florida). An additional question on threatening processes was added to satisfy the criteria listed in the Act. The variables are weighted to reflect reliability of available information. Greater weight is given to the ecological attributes of population size and trend, and area of distribution and trend, as these provide estimates of changes in populations. Species are ranked according to their relative summed scores and classified as threatened or vulnerable with one of three knowledge grades: adequate, limited, or inadequate. Modifications to suit NSW geography and the time since European settlement in Australia restrict the applicability of these protocols to NSW, however, further modification can extend their utility to a range of geographic scales and timelines. The Lunney et al. scheme has been applied to 883 vertebrate species (up to 1996) for a broad range of taxonomic groups in NSW, Australia. Although it could be applied to species in the U.S., this has not been done.

Partners in Flight. This classification scheme was originally created to address declining populations of Neotropical migratory songbirds, but it was hoped that it could be applied consistently to any group of species in any geographic region (Carter et al. 2000). Like the Millsap et al. scheme, a series of scores is assigned to parameters, many of which have sub-categories defined by numerical criteria. There are seven parameters: breeding distribution; non-breeding distribution; relative abundance; threats to breeding; threats to non-breeding; population trend; and area importance. Scores are based on quantitative and objective data where possible, however, subjective assessments based on expert opinion also are permissible in the absence of empirical data. The protocols reflect recent and predicted threats that may put a species at risk of decline or extirpation from an area. There is flexibility in the way scores for each parameter can be used to assess risk. The sum of all scores is one indication of priority status, however, parameters also can be weighted if some parameters are considered more important than others. A categorical approach can be applied that sorts species primarily according to thresholds of scores for population trend, relative abundance and threats. The notion of "area responsibility," which highlights an area's contribution to long-term conservation of species, also has been introduced as a way of using the PIF protocols. Between 100 and 300 bird species have been classified according to the PIF protocols in the United States.

California Fish and Game. The California Fish and Game (CFG) classification protocols are working criteria developed by the Birds Species of Special Concern (BSSC) Technical Advisory Committee that are intended to provide a numeric score for the purpose of ranking special concern taxa for assessing conservation priorities in California (BSSC 1998). There are eight parameters: population trend; range trend; population size; range size; percentage of entire range within California; ecological specialization; population concentration; and vulnerability to threats. Each parameter is divided into sub-categories that have numeric or qualitative criteria all with an associated score. Scores are summed to determine the relative rank of conservation status among species. The impacts of realized known threats and potentially irregularly occurring catastrophic events are taken into account. Ranking is not based on speculative threats when there is no past or current evidence of population-level impact. The parameters, as stated, restrict application to California, however, with slight modification they could be extended to other regions. We do not know how many species have been classified using the CFG protocols.

Method for Evaluating the Risk of Extinction of Wildlife in Mexico (MER). This is a point scoring method for assigning categories of threat to species in Mexico (MER 2000). It has four parameters, each of which is assigned sub-categories and criteria. Two of the parameters have numerical thresholds, while two have qualitative criteria. The four parameters are: distribution size of the taxon in Mexico; the condition of the habitat with respect to the natural development of the taxon; intrinsic biological vulnerability of the taxon; and the impact of human activities on the taxon. The impacts and tendencies of threats from general human activities such as urban development, fragmentation of habitat, contamination, commerce, traffic, change in land use, and introduction of exotic species are explicitly considered. There are two categories of threat, based on the final sum of the scores. These are "in danger of extinction" and "threatened." There is an implicit default classification of low risk if a species is not classified in either of the risk categories under the protocols. The MER was published in the Mexican Federal Register in 2000, and we do not know how many species have been classified using this protocol.

Table 5.1 summarizes and compares the parameters used in each of the classification schemes outlined above, whereas Table 5.2 compares each of the methods according to a list of criteria. The criteria are relevant for categorizing species at risk in a forest management context. Since the U.S.D.A. Forest Service uses a range of sources for designating species as sensitive (including IUCN, USFWS, ABI/Heritage and tailored point-scoring schemes), rather than any single approach, the Forest Service is excluded from both Tables 5.1 and 5.2. The criteria used in Table 5.2 are:

Geographic scale: what is the scale at which the protocols can be applied? Are they to be applied at a local, state or regional level (L); at a national level (N); or globally (G)?

Quantity/availability of information: how much data is required to classify a species in a category of threat? Is this information generally available for most species? Low: an assessment can be made with up to two parameter values (or pieces of information) that are generally available for most species. Medium: an assessment can be made with three to seven parameter values (or pieces of information) that are generally available for most species. High: an assessment requires more than seven parameter values.

Current/future management: are current or future management activities explicitly considered in the protocols in the form of projected impacts to population size, geographical distribution or other population-level parameters? Yes/No.

Future population trend: do the protocols incorporate future trends in population size as a result of the underlying population dynamics, impacts of threats, catastrophes or human activities, or response to other ecological processes? Yes/No.

Ecological specialization: are there parameters relating to the degree to which a taxon is dependent on environmental factors (e.g., responses to decreases in availability of preferred food types, breeding sites, or other ecological or behavioral specializations)? Yes/No.

Taxonomic range: can the protocols be applied to a range of taxonomic groups and life histories? Low: the protocols can be applied only to terrestrial vertebrates; Medium: protocols can be applied to terrestrial and aquatic vertebrates; High: protocols can be applied to terrestrial and aquatic vertebrate, invertebrate and plant species.

Geographic distribution: do the protocols include parameters related to the geographic distribution of the species, e.g. the size of the area over which the species is distributed, the total size of the area occupied by the species, and the trend in the geographic distribution? Yes/No.

Threats: are recent or future threats that result from direct or indirect impacts of human activity, or irregularly occurring catastrophes taken into account in the protocols? Yes/No.

Uncertainty: can the uncertainty associated with each of the parameters be dealt with in the protocols, either with existing methods, or by straightforward modification to the protocols? Yes/No.

Reliability/robustness: is the classification method repeatable? Can different assessors achieve identical conservation status when ranking the same species using the same quantitative data? Is ambiguous or vague language used that can lead to vastly different interpretations of criteria? Are definitions provided for each criterion? Yes/No.

Transparency: are the definitions of the criteria clear and easy to apply to any species? Is it clear how a final assessment was achieved? Is it possible to ascertain which parameters contributed significantly to the overall assessment? Yes/No.

Discussion

All of the classification systems outlined here focus on a species' risk of extirpation at some spatial scale. The scale of application among the schemes ranges from global (e.g., Mace and Lande 1991, IUCN 1994 & 2000, USFWS, ABI/Heritage, PIF) to national (ABI/Heritage, MER) to local or regional (Millsap et al. 1990, Lunney et al 1996, CFG, USFS). None of the protocols is designed specifically for application to populations, the most likely unit of management consideration for the Forest Service. Protocols that operate at broad scales (i.e. globally or nationally) will address populations when applied at a national or regional scale if the entire range of the species is within the nation or region of application. Six of the methods, namely Lunney et al., MER, ABI/Heritage, USFWS, PIF, and USFS explicitly address the impact of recent or potential threats. This is important for determining which species are most likely to be adversely affected by management activities. The protocols considered here have a range of data requirements. Rule-based methods, such as the IUCN protocol generally require less data than point-scoring methods. Point-scoring methods require input for every parameter to establish an overall score. All species must have an equivalent amount of data, either empirical or subjectively assigned by an expert. This makes point-scoring methods less useful than rule-based methods for quick assessments when little data are available. Of the methods outlined here, six are point-scoring methods (ABI/Heritage, PIF, CFG, MER, Millsap et al, Lunney et al.), one is rule-based (IUCN), one has little structure (USFWS). Currently, the Forest Service uses a combination of methods.

There are well-known criticisms of multi-attribute point-scoring methods. Some of these are: they are regarded as *ad hoc*, arbitrary and often pay little attention to biological principles; all parameters or criteria are given equal weight, i.e. it is assumed they contribute equally to extinction risk; parameters are treated as if they are independent, even when there are clear correlations and dependencies; the relationship of the aggregated score to extinction risk is unclear. Beissinger et al. (2000) recommend that, rather than aggregating scores, scores for each variable in the PIF prioritization scheme should be used to place species into a category related to conservation priority in much the same way as for the IUCN scheme. They assert that close correlation between variables is not a problem when variables are considered separately in categorizing risk because the variables represent different ways a species might be of conservation concern.

Rule-based classification schemes such as the IUCN categories are not without their own concerns. Even though they provide clear definitions for each of the categories, the true relationship between the probability of extinction and the adopted proxies is not known, but is likely to be taxon dependent. Furthermore, such schemes are usually biased towards certain taxa (see Keith 1998 and Swaay et al. 1999 for amendments of the IUCN categories for application to vascular plants and butterflies, respectively), although this criticism could be made of all classification schemes, both rule-based and point-scoring. Another criticism of rule-based methods is that, while they can be applied with very little data, they do not always use all the available data or differentiate between types of data. Hence, a species with a small population size and distribution could be assigned to the same conservation status category as a species with the same distribution, but with a much larger or unknown population size. Regardless of the various criticisms of point-scoring and rule-based methods, all have merit and utility in identifying species at risk and all are routinely used for different purposes.

All of the methods described here, except the USFWS protocol, provide for the treatment of uncertainty. For the IUCN 1994 and 2000 protocols, a bounding method has been implemented which allows parameters to be entered in terms of lowest, best and highest estimates (Akcakaya et al. 2000). Fuzzy logic is then employed through the rules to give a classification within plausible bounds. This type of treatment can be applied to all rule-based methods that use quantitative and/or qualitative criteria similar to the IUCN categories. Bounding methods also can be applied to all of the point-scoring methods because they numerically grade (or score) species attributes from lowest to highest risk. Best estimates within upper and lower bounds can be given for each score, depending on the uncertainty associated with each input parameter. This type of treatment is implemented in Lunney et al. (1996) to deal with the uncertainty resulting from many expert opinions.

All of the methods described here are useful for classifying species at risk. Those that explicitly include current and future threats and a species' response to those threats are particularly useful in determining which species will be adversely affected by proposed management actions. Despite similarities among the protocols, each was designed for a different purpose. It is important to maintain consistency throughout any viability assessment process. The best method for identifying species at risk will depend on the management scenarios proposed, the amount of data available, the time frame within which the assessment must be completed, and the scale at which the assessment is to be made (Lehmkuhl et al. 2001).

Viability is defined in the regulations at a local scale. Although the approaches outlined here are useful, they are not completely sufficient for identifying species at risk on Forest Service land because these methods were designed for application at much larger scales. Hence, adoption of any one method in isolation for identifying at-risk species in the context of the NFMA will not appropriately address viability concerns at the requisite scale. Furthermore, existing approaches only will be useful for viability concerns if there are data to support them, and if consideration of future threats and impact of forest management practices is feasible. For these reasons, we recommend the ABI/Heritage protocols as the most suitable of existing protocols to assist in identifying species at risk, because they explicitly deal with the severity, scope, and imminence of threats, and because ABI/Heritage global ranks already exist for almost all species on Forest Service land. This recommendation is made with the caveat that issues of scale need to be handled appropriately. To reconcile the incompatibility of the scale at which classification schemes are intended for application and the scale at which the Forest Service operates, we suggest that the ABI/Heritage protocols be used in conjunction with the Forest Service sensitive species lists to ensure that at-risk species not identified by the ABI/Heritage protocols will not be ignored altogether. An approach that uses both ABI/Heritage ranks and sensitive species lists compiled using a variety of methods already is used by Regions 1 and 2 that administer the National Grasslands in the Great Plains. We see this as a flexible approach that makes the necessary (although not always ideal) compromise between biological justification, data needs and considerations of scale.

5.4 Species Distribution Issues

After generating a list of species potentially at-risk using the protocols described in the previous section, the distribution of those species within the planning region must be considered. The 2000 Forest Service planning regulations state that forest managers have a responsibility to provide ecological conditions which will result in a high likelihood of supporting viable populations of native species and desired non-natives that are well–distributed throughout their ranges within the planning area. The regulations further state that a species is well distributed when individuals can interact with each other in the portion of the species' range that occurs within the planning area. However, to be useful, this criterion necessitates some understanding of the underlying distributional patterns of a species throughout its range. The regulations specifically identify six types of species distributions to which planners must be especially attentive:

(1) Endemic species: Species whose entire distribution is restricted to a single planning area
(2) Disjunct species: Species whose natural distribution has resulted in one or more populations on the planning area that are either isolated from each other and/or from other populations outside the planning area
(3) Peripheral species: Species with a population that only partially occurs within the planning area
(4) Inherently – rare and not naturally well-distributed species: Species that are not naturally well–distributed across the planning area.
(5) Species that presumably occur within the planning area, but for which the conditions necessary to support viable populations are outside the authority of the Forest Service.
(6) Species for which it is not feasible to restore ecological conditions that would provide a high likelihood of viability

In cases 1-3 above, the regulations state that the Forest Service must provide for ecological conditions that are capable of maintaining viable populations. Below, we explore these categories in more detail.

Endemic species (1), as defined by the Forest Service, encompass several patterns of rarity (Rabinowitz et al. 1981). Some have restricted habitat requirements, but can occur in either large or small populations. Others use a broader array of habitats, but again can occur in large or small populations. Because endemic species, by definition, have limited distributions (the only place they can be conserved is within the planning region to which they are endemic), all of them should be considered potentially at-risk. However, not all of these species need to be subjected to viability assessments. Some species will occupy specific sites that may not be affected by any management alternative in the forest planning process (e.g., a cave, cliff, bog), requiring only that site-specific management projects avoid these areas. Whether or not viability assessments should be done for any of these species is determined at the discretion of the planning team, based on interpretation of the policy provided in the planning rule, as well as on the particular alternatives generated and the projects anticipated. All else being equal, the type of species with small populations that is both endemic to a planning region and specific to one or only a few habitats faces the greatest viability risks.

The circumstances under which disjunct (2) populations of a species may be the focus of a viability assessment will be depend, in part, on the status of the species throughout its range. If the species as a whole remains well–distributed, and its populations generally are not declining or threatened, then a viability assessment is not necessary. If, on the other hand, the species is known to be declining in a significant portion of its range, then these populations will require viability assessments. Regardless of whether disjunct (2) populations are subjected to viability assessments, they are likely to have diverged genetically or morphologically from less isolated populations, and their genetic distinctness alone may make them worthy of conservation attention (Lesica and Allendorf 1996). Thus, it is prudent to maintain them on at-risk species lists.

The manner in which the 2000 planning regulations have defined peripheral populations (3) appears insufficient (note the term “peripheral” is not used explicitly in the regulations). The most relevant issue is not whether a population occurs partly on National Forest lands, but whether the natural distribution of the species results in populations on the edge of the range that completely or partly occur on National Forest lands. These populations should be considered important for conservation attention in two sets of circumstances. First, if the population in question occurs at the northern end of the limits of the species distribution, it potentially could become a central or core population in the future under various global climate change scenarios (Halpin 1998). Second, if the peripheral populations belong to a species that has already suffered declines and habitat losses in major portions of the species’ range, recent biogeographic analyses of range contractions suggest that these peripheral populations may represent valuable conservation opportunities in the future, as the species continues to decline (Channel and Lomolino 2000). In these two circumstances, peripheral populations should be considered at-risk and included within viability assessments.

Most species that occur, or potentially occur within the jurisdiction of National Forest lands, but for which the ability to maintain viable populations is outside the control of the Forest Service (5), may already be considered at-risk. For example, this category might include aquatic species for which National Forest lands and waters provide habitat, but for which the major limiting factors for the population occur upstream or downstream. Similarly, species that have been extirpated from National Forest lands, but for which suitable ecological conditions persist might also be in this category. Several wide-ranging carnivores fall in this category. A good example is the grizzly bear in the Selway-Bitteroot Wilderness of Idaho and Montana. Most biologists conclude that grizzly bears no longer occur in this area. However, habitat analyses suggest the area could support a viable population of bears, and reintroductions have been proposed. In most cases, National Forests will want to maintain suitable habitat conditions, to provide future opportunities for Forest Service lands to contribute to species recovery. Viability assessments that analyze the distribution and quality of suitable habitat may be most appropriate for these species.

It is important to distinguish between species that are inherently rare and those that are rare because of population declines and/or habitat loss caused by humans. Nonetheless, as Rabinowitz et al. (1986) demonstrated, even inherent rarity comes in several forms. We already have reviewed those forms of rarity that can be classified as endemic. There are three remaining forms: Type 5) species that are widely distributed but occur in a limited number of habitats in small populations, Type 1) widely distributed species that occur in a limited number of habitats in large populations, and Type 4) widely distributed species that occur in a variety of habitats in small populations. Note that these forms do not correspond precisely to the Forest Service types above. Some cavity-nesting bird species are widely distributed, but within a single planning region may occur in a limited number of habitats (Types 1 and 5), such as recently burned areas (e.g., black-backed and three-toed woodpeckers). How recently the area has burned, the size and number of snags, and other habitat factors will influence whether populations are large or small. Some wide-ranging carnivores, such as the wolverine, use a variety of habitats, but their population numbers are typically small. Type 4 Species should always be candidates for viability assessments because of their small population size and the fact that they occur across many habitats and could therefore be placed at risk by one or more proposed management activities.

Rabinowitz's Types 1 and 4 species also should be considered at-risk, but whether or not they need viability assessments will depend on the proposed management alternatives in the forest planning process and on the nature of proposed projects at the district level. This is why it is important to distinguish between species that are inherently rare and those that are rare because of human activities. In the cavity-nesting bird example, a natural fire may or may not create habitat for particular woodpecker species. Species such as black-backed and three-toed woodpeckers often occur at low densities relative to other woodpecker species. In these situations, viability assessments may be unnecessary. However, if salvage logging operations (human activities) are proposed for these areas, some activities, depending on levels of snag management, could pose a threat to these species. In such situations, a viability assessment for these types of species is advised.

Similarly, some plant species may be inherently rare due to habitat specificity (e.g., restricted to serpentine soils or alpine meadows). Again, whether such species are subject to viability assessments depends upon proposed actions. For rare plants, in most cases their habitat needs can be addressed at the project level and it is not necessary to conduct viability assessments. For the same reasons as noted above, a prudent step is to maintain these species on at-risk lists so that managers are reminded of the specific habitat needs and appropriate conservation measures for these species.

The final category of concern based on species distribution, comprises species for which conditions on National Forest lands are no longer capable of supporting well–distributed populations (6). A variety of species may fall in this category. For example, contiguous forest cover for species with large area requirements may not exist on National Forests that have been highly fragmented by timber harvest. Similarly, some forests may no longer support viable, well–distributed populations of aquatic species whose habitats have been degraded by mining, grazing, or timber harvest. Forest managers and planners will need to consider whether restoration of such lands to the point of supporting viable populations is feasible and realistic from biological and socioeconomic considerations. Species in this category should be classified as at–risk and National Forest operating units should critically evaluate the potential of long-term restoration efforts to support well–distributed viable populations. Species for which restoration of conditions to support viable populations is unlikely in the foreseeable future (6) (e.g., an aquatic species whose habitat has been destroyed by a reservoir) should be eliminated from the at risk category.

5.5 Conclusions and Recommendations

Several of the protocols reviewed here provide credible approaches to identifying species at risk, with the caveat that none was intended to be applied at the specific scale required by the NFMA. After considering both scientific credibility, and practicality of application, we recommend that the ABI/Heritage protocol be applied to species in the planning area as an initial filter for identifying species at risk. ABI/Heritage global and state ranks already exist for most vertebrates and vascular plants, and for some invertebrates on National Forest lands. Additional factors to consider have been outlined by Holthausen et al. (1999), and are summarized here.

1. Identify all species with ABI/Heritage G ranks of 1-3

In addition, consider:

2. Species population size

3. Distribution of the species' population, particularly the extent to which populations are evenly distributed, or relatively patchy and isolated, both within the planning area, and relative to populations on lands outside the planning area

4. Species life history traits, especially body size, reproductive rate, dispersal ability, and migration rate

5. Degree of habitat specialization

6. Known or suspected trends in abundance and distribution

7. Other known risk factors

Table 5.1

Parameters	**Protocols**								
	IUCN 1994	IUCN 2000	ABI/Heritage	USFWS	Millsap et al 1990	Lunney et al 1996	PIF	CAF&G	MER
Population size	YES	YES	YES	NO	YES	YES	YES	YES	NO
Population trend	YES	YES	YES	NO	YES	YES	YES	YES	NO
Number of populations	YES	YES	YES	NO	NO	NO	NO	NO	
Abundance relative to other species	NO	NO	NO	NO	NO	NO	YES	NO	NO
Reproductive potential	NO	NO	NO	NO	YES	YES	NO	NO	NO
Population concentration	NO	NO	NO	NO	YES	YES	NO	YES	NO
Area of occupancy	YES	YES	YES	NO	NO	NO	NO	NO	NO
Extent of occurrence	YES	YES	YES	NO	YES	YES	YES	YES	YES
Range trend	YES	YES	YES	NO	YES	YES	NO	YES	NO
Area importance	NO	NO	NO	NO	NO	NO	YES	NO	NO
Habitat condition	NO	NO	NO	NO	NO	NO	NO	NO	YES
Fragmentation	YES	YES	NO	NO	NO	NO	NO	NO	NO
Vulnerability to threats	NO	NO	YES	NO	NO	YES	YES	YES	YES
Current or future threat occurrence	NO	YES	YES	YES	NO	YES	YES	NO	NO
Ecological specialization	NO	NO	NO	NO	YES	YES	NO	YES	NO
Intrinsic vulnerability	NO	NO	YES	NO	NO	NO	NO	NO	YES
Number of protected Occurrences	YES	YES	YES	NO	NO	NO	NO	NO	NO
Risk of extinction	YES	YES	NO	NO	NO	NO	NO	NO	NO
Rule-based scheme	YES	YES	NO	NO	NO	NO	NO	NO	NO
Point-scoring scheme	NO	NO	YES	NO	YES	YES	YES	YES	YES

Table 5.2

System	Scale	Quantity/ accessibility of information required	Current/ future management	Future population trend	Ecological specialization	Range of taxonomic groups and life histories	Geographic distribution	Threats	Uncertainty	Reliable/ robustness (ambiguous or vague language)	Transparency
Millsap et al 99	L	HIGH	YES	YES	YES	MEDIUM	YES	NO	YES	NO	YES
Lunney et al 99	L	HIGH	YES	YES	YES	MEDIUM	YES	YES	YES	NO	YES
CA Fish & Game 99	L	HIGH	YES	YES	NO	LOW	YES	NO	YES	YES	YES
MER	N	MEDIUM	YES	NO	YES	HIGH	YES	YES	YES	YES	YES
TNC/ABI/Heritage	L/N/G	HIGH	YES	YES	YES	HIGH	YES	YES	YES	NO	YES
USFWS	N	LOW	YES	NO	NO	HIGH	NO	YES	NO	NO	NO
IUCN 94	G	LOW	NO	YES	NO	HIGH	YES	NO	YES	YES	YES
IUCN 00	G	LOW	YES	YES	NO	HIGH	YES	NO	YES	YES	YES
PIF	G	MEDIUM	YES	NO	YES	LOW	YES	YES	YES	YES	YES

Chapter 6. Use of Focal Species To Assess Species At Risk

Introduction

The new planning regulations recognize two broad categories of species of management concern—focal and at-risk species. Focal species are "surrogate measures used in the evaluation of ecological sustainability, including species and ecosystem diversity. The key characteristic of a focal species is that its status and trend provide insights to the integrity of the larger ecological system to which it belongs. Individual species, or groups of species that use habitat in similar ways or which perform similar ecological functions, may be identified as focal species. Focal species serve an umbrella function in terms of encompassing the habitats needed for many other species, play a key role in maintaining community structure or processes, are sensitive to changes likely to occur in an area, or otherwise serve as an indicator of ecological sustainability. Certain focal species may be used as surrogates to represent ecological conditions that provide for viability of some other species, rather than directly representing the dynamics of those other species"(Federal Register 65:67580, 2000). The definition of species-at-risk was discussed in Chapter 5.

Focal and at-risk species are not necessarily mutually exclusive categories. In addition, the focal species concept is defined very broadly in the new regulations and includes several defining attributes:

1) Surrogate measures of ecological sustainability.
2) Representative of larger groups of species with similar habitat requirements or functional roles.
3) Umbrella species whose area requirements include the habitat needs of many other species.
4) Play a significant role in maintaining the structure and processes of ecological systems.
5) Serve as an indicator of ecological sustainability.
6) Surrogate measures of the ecological conditions that provide for the viability

of other species.

There is obvious overlap among these operational definitions, but they differ in subtle ways that have important consequences. Aspects of the definition related to points 1, 4, and 5 more closely reflect the use of the focal species concept as it was used in the Committee of Scientists Report (COS 1999). In that report, focal species were defined as species whose status and temporal trend provide insight to the larger ecological systems to which they belong. The incentive to distinguish this group of species was motivated by the challenges of monitoring complex ecological systems where, for pragmatic reasons, only a few indicators can be measured. The focal species concept was first proposed to the Forest Service during development of the effectiveness monitoring program for the Northwest Forest Plan (Mulder et al., 1999; Noon et al., 1999), but the term focal species has been used by several other authors in various contexts (e.g., Lambeck 1997, Noss et al. 1999) and its meaning is broad and relatively imprecise.

The term focal species also has been used by Lambeck (1997) to describe umbrella species currently at risk from a set of specific processes that threaten their existence (point 3 above). Lambeck's use of the term umbrella includes, not only species with large area requirements, but any species whose functional and compositional requirements are believed to encapsulate the needs of other species. Based on four broad categories of risk, Lambeck recognizes the following types of focal species:

1) Area-limited species*:* Those species for which patches of appropriate habitat simply are too small or too few in the planning area to support a viable population. Wide-ranging carnivores in the northern Rocky Mountains or some species of neotropical migratory birds on National Forest lands in the eastern United States are examples of potentially area-limited species.

2) Process-limited species: Those species dependent upon certain types of ecological processes or disturbance regimes for renewal and persistence.

For example, lodgepole pine forests in western Wyoming need infrequent, but intense stand-replacing fire events to persist on the landscape.

3) Resource-limited species: Those species whose populations are limited by a shortage of one or more critical resources. For example, cavity-nesting birds may be limited by the availability of suitable snags for excavation of nest sites.

4) Dispersal-limited species: Those species for which there are suitable habitat patches to support small populations, but the patches are beyond the distance over which individuals can move or dispersal success is too low to balance local extinction rates. Examples include species with highly fragmented populations, such as the southwestern willow flycatcher, where patch extinction rates often exceed colonization rates.

The new regulations also use focal species to represent groups of species that are similar in terms of their habitat associations or ecological functions (point 2, above). The clustering of species into guilds on the basis of habitat, behavior, taxonomy, response to disturbance, or other dimensions is common practice in the Forest Service land management process (e.g., Wisdom et al. 2000). The motivation is to reduce the complexity of the species viability assessment process by lumping species into groups expected to respond similarly to environmental change (point 6 above). Efficiency is potentially achieved to the extent that the dynamics of a single species within a group is representative of the dynamics of the unmeasured species in the same group. A Forest Service working group recently developed a potential method for grouping species at-risk and identifying one or more potential focal species for each group (Wisdom et al. 2001). Given the charge to our group, we also will focus our discussion in this chapter on the issue of identification and selection of focal species to act as surrogates for species identified as at risk of extinction or significant population decline. We caution at the outset, however, that as desirable as such grouping might be as a “short cut” for viability assessments, as yet there has been little empirical evidence to support the effectiveness of such an approach (e.g., Andelman & Fagan 2000; Rubinoff 2001; Lindenmayer et al. 2002)..

Use of the Focal Species Concept in Viability Assessments

The broadened definition of the focal species concept leads to partial overlap with the at-risk category of species discussed in the NFMA regulations. In this case, a focal species can be selected from among a larger group of at-risk species. Also, it is possible that the ecological conditions of species at risk could be represented by focal species that are not themselves at risk. In either case, the long-term viability of the focal species is assumed to be representative of a group of species with similar ecological requirements and this group is assumed to respond in a similar manner to environmental change. In addition, the focal species is assumed to have more demanding requirements for factors putting other group members at risk of extinction. Subsequent discussion will be limited to the identification of focal species as surrogates for species identified as at risk of extinction or significant population decline.

Identifying Species At-Risk

A review of methods used to identify and rank species at-risk is presented in Chapter 5. In this chapter, we elaborate on procedures developed by Wisdom et al. (2001) to identify focal species. To identify a small number of focal species for measurement, Wisdom et al. (2001) proposed that all species must first be classified into groups based on similarities in degree of risk, risk factors, patterns of habitat use, taxonomy, or some other meaningful set of criteria. Because of the great variability in management situations, it is unlikely that there will be a single “best” method for forming groups—therefore, a diversity of approaches may be most defensible. However, we believe that grouping species by factors putting them at risk of significant population decline will often be most meaningful. In this context, we propose the following step-down process:

1. Identify the species pool within the management area. This will require an

identification of all species whose geographic range overlaps the planning area. In general, for a given area, as the resolution of the analysis increases the number of required focal species will increase. This process will result in an exhaustive list of candidate focal species.

2. Identify those species known or suspected to be at risk of extinction or population decline within the planning area and rank them by degree of risk (see Chapter 5). Two major categories of risk are useful to identify at this point: species at risk due to environmental factors and those at risk due to small population size. In general, we consider environmental factors to be deterministic influences that are under the control of management. In contrast, small populations are subject to stochastic factors that often lie outside of management control. This distinction is useful but often imprecise. For example, most deterministic risk factors will be exacerbated by stochastic events, and many species have small populations because of past deterministic effects.

The environmental factors that most commonly put species at risk include the following:

1) Habitat loss or fragmentation
2) Declines in habitat quality
3) Introduced species, including those that act as predators, competitors, or sources of disease to native species
4) Over-harvest or illegal harvest
5) Environmental contaminants.

Because all of these threats arise as direct or indirect consequences of human behavior, in theory, they are all amenable to management intervention.

Environmental factors are often the deterministic processes that lead to population declines and small population size. Once populations are small, however, in addition to the environmental risk factors, a new set of demographic and genetic risk factors must be considered. Demographic considerations include (Lande 1993):

1) Demographic stochasticity
2) Environmental stochasticity
3) Spatial structure of the population
4) Density-dependence effects including Allee effects and habitat thresholds (Lande 1987)

Genetic considerations include:

1) Migration between local populations
2) Inbreeding depression
3) Genetic drift
4) Effective population sizes of local populations

To assess risks associated with small population size requires the manager to have information on the number and size of local populations, the geographic distribution of these populations and their connectivity, and the degree to which local populations show temporal and spatial covariance in their dynamics. To assess connectivity requires information on movement rates, genetic interchange between local populations, and the degree of local genetic differentiation. The reality is that this information is unknown for most species. As a consequence, viability assessments based on genetic or demographic variables will be restricted to a small number of species that may, or may not, serve as useful focal species.

Criteria for Identifying Focal Species As Surrogates for Species At-Risk

6.1.1 Risk Factors

Within each broad category of risk (i.e., environmental factors or small population size), we recommend that species be grouped according to their primary risk factor (e.g., habitat loss). In addition, it may be valuable to further subdivide species into smaller groups based on a nested hierarchy of classification steps (see Wisdom et al. 2001). For example, species at risk from habitat loss first could be subdivided into major habitat categories (based on classification of vegetation associations and seral stages), and then further subdivided into those at risk from deterministic processes (e.g., succession, fragmentation, climate change, etc.) or from stochastic processes (e.g., storms, fires, floods, volcanic eruptions, etc.) affecting their habitat.

6.1.2 Life History and Ecology

Additional ways to group species may take into account specific aspects of a species' life history and ecology. All species can be categorized along dimensions of average body size, morphology (shape), growth rate, life span, dispersal mode and ability, food web position, response to disturbance, etc. One or more of these dimensions may be useful to identify a focal species for a larger at-risk species group. For example, the following attributes are known to be correlated with extinction risk:

1) Low reproductive potential
2) Delayed age at first reproduction
3) Adult sex ratio favoring males
4) Low pre-adult survival rate
5) Low adult female survival rate
6) Large area requirements
7) Limited dispersal ability

Within a species-group, species with these life history attributes may be most extinction prone, but not necessarily the best candidates for focal species. To serve a focal role, changes in the status of focal species should act as an early warning of declines within the group, and the viability of these species should allow reliable inference to the viability of other species in the group.

Within a group of species at-risk, it may be useful to further partition species by the scale at which they respond to temporal and spatial variation in the environment. A good proxy variable for partitioning species by spatial scale is body size and its relationship to home range size (e.g., Holling 1992). Difference in temporal scale among species is often represented by relative life spans, reproductive potentials, and age at first reproduction.

Wisdom et al. (2001) suggest that a cross-classification of species-group into four or more classes based on combinations of home range size and dispersal ability should prove useful to land managers.

Obviously, there are many possible ways to group species based on a set of nested criteria. However, tradeoffs to achieving an efficient assessment of all species are soon encountered. In theory, a nested classification process could eventually lead to each species being in its own unique group. Recall, the goal is to have a manageable number of focal species to assess and still achieve reliable inference to the status and trend of the unmeasured species.

Practical Focal Species Attributes

In addition to the biological criteria discussed above, to the extent possible, focal species also should have to satisfy a number of practical criteria in order to be reliable surrogates for the unmeasured species. These include:

1) Their dynamics parallel those of the larger environmental component or system of ultimate interest
2) They show a short-term but persistent response to change in the state of the environment
3) Their dynamics can be accurately and precisely estimated (high signal to noise ratio)
4) The likelihood of detecting a change in their value is high, given a change in the state of the system
5) Each demonstrates low natural variability, or additive variation, and changes in their values can be readily distinguished from background variation
6) Differences in the status of the species is known to be able to discriminate among sites along a known condition gradient
7) The species is known/believed to respond to management actions
8) The costs of measurement are not prohibitive
9) Response specificity to known risk factor

The reality is that species which best fulfill the above criteria also will be characterized by the following attributes:

1) Their taxonomy is well-known and unambiguous
2) Their biology and life history are “known”
3) They are “easy” to find and measure (limited mobility?)
4) They show low sampling variability (consistent and high detectability)
5) Their populations show low process variation (demographic and genetic)
6) They have detectable population trends

The possibility exists that only a small number of species meet the above criteria and these will be those species that have been well studied. Unfortunately, previously well-studied species may not necessarily be the best candidate focal species.

A Step-Down Process of Selecting Focal Species

Based on the above discussion, we propose the following process for identifying candidate focal species to assess species at risk (cf. Hilty and Merenlender (2000) and Wisdom et al. (2001)):

1) Determine legal responsibilities of the Forest Service relative to sustaining biological diversity and the viability of individual species. That is, clarify the requirements for species protection within the larger context of Forest Service land planning and management..
2) Define the species pool: make a list of all the species in the affected area for which the Forest Service has a legal responsibility. Pragmatically, this process usually defaults to vertebrates and vascular plants. However, invertebrates and non-vascular plants should be retained to the extent they are at-risk and serve a surrogate role.
3) Apply a risk-based classification process, as outlined above or similar to that proposed by Lambeck (1997). Clearly separate those species at risk due to small population size from those at risk due to environmental factors acting independent

of population size.

4) From each at-risk group, further allocate species to categories based on the specific nature of the risk factor. For habitat-limited species, for example, a classification of species by vegetation associations and structural stage is a logical first step.
5) Within each at-risk category, rank species by their degree of risk (see Chapter 5).
6) Using information on body size or home range, further classify species within each group according to their spatial scale of response to environmental factors.
7) Using information on life span or age at first reproduction, further classify species within each group according to their temporal scale of response to environmental factors.
8) Remove from each group those species whose dynamics are largely independent of activities occurring on Forest Service lands.
9) Further reduce the number of species in each group by retaining only those that best meet the statistical and sampling criteria discussed above (see pragmatic factors).
10) Select at least one or more focal species from each group.
11) Develop a stopping rule. The final list of focal species should be sufficiently comprehensive and complementary that it addresses the legal requirements of the NFMA regulations

The Stopping Rule

How does the manager know when a sufficient number of focal species has been selected? This is a difficult question to address, and probably no definitive answer is possible. To begin with, we might propose that for pragmatic reasons alone 100 focal species is too many and 10 may be too few. Ultimately, funds available for monitoring will set the upper limit. However, between that limit and the minimum number can some guidance be provided?

As outlined above, the process of focal species selection follows a logical sequence:

(1) identify species within the planning area; (2) reduce this list to those at risk of decline; (3) partition this group into categories of risk; (4) identify within each risk category those most at risk; (5) identify within each risk category those that "best" represent the rest of the group. The "best" representative within a group is dependent upon the management questions to be addressed, as well as how well a species fits the biological and practical criteria discussed above. Representative examples of this process are discussed by Lambeck (1997) and McLaren et al. (1998).

Ultimately, the process of partitioning species into more but smaller subgroups needs to stop or the efficiency gained by a focal species approach is compromised. Given the contingent nature of any stopping rule, however, perhaps all that can be offered are a few general rules of thumb. Examples include:

1) Stop when each discrete vegetation community type (or ecosystem) is represented by ≥ 1 focal species.
2) Stop when all specific risk factors have been associated with ≥ 1 focal species.
3) Stop when all species of interest fall within at least 1 focal species grouping.
4) Stop when all discrete taxa (defined at the family level) are represented in 1 or more focal species groups.

Chapter 7. Methods of Assessing Viability

Many methods exist for assessing the viability of a species. They include methods of assessing habitat quantity and quality, rankings of species vulnerability to extinction, demographic models that assess the impact of management on the rate of growth or risk of extinction, genetic models that assess the loss of genetic diversity, analyses of presence-absence data, trend analysis, and panels of expert opinion. All methods have some utility and the range of possible methods offers a trade-off between complexity of analysis and generality of results (Beissinger et al. in press). Alone, none might provide perfect analysis, but when used together, as one of several analyses comprising a species viability assessment, they can provide important insights.

In this chapter, we present information on methods for conducting species viability assessments. We first present an overview of methods, their data needs, advantages, and disadvantages. Next we examine which methods the Forest Service has used in past assessments. We conclude by discussing how the scale at which species viability assessments are conducted and the availability of data affect which approaches should be used in assessing viability.

Approaches for Assessing Viability

Below we briefly review the methods that can be applied to analyze population viability. We concentrate on defining each approach, presenting the data necessary for its use, describing its strengths and weaknesses, and providing a context for use. A summary of these assessment tools is provided in Table 7.1.

7.1.1 Habitat-based Analyses

Most factors that affect the trajectory of a population are either directly or indirectly linked to the environmental conditions in which the species occurs. These environmental influences often are best quantified through analyses of habitat quantity and quality. Thus, habitat is a key factor in addressing species viability. Moreover, jeopardy decisions under the Endangered Species Act frequently focus on habitat. In terrestrial ecosystems, habitat loss is the greatest threat to species persistence (Wilcove et al. 1998). Planning for species viability must first and foremost provide for a sufficient amount and distribution of habitat. It also is important to understand and quantify the relationships between measures of habitat quality and demographic parameters of a species. There are a number of methods for evaluating habitat for species and for planning for the maintenance of future habitat conditions, which we briefly summarize below.

Comparison with Historical Conditions – A comparison of historical and current habitat conditions often is a component of viability assessments. Historical conditions can serve as a baseline for purposes of comparison, but it may not always be simple to determine the appropriate historical reference point. Quantity and quality of habitat historically available frequently are used as a reference point for comparison with existing ecological conditions (Swetnam et al. 1999, Landres et al. 1999, Haufler et al. in press). Historical conditions can be evaluated and described in a number of ways, but care must be taken in the choice of the time frame for the historical benchmark. Two types of description generally have been used. One type strives simply to define the composition of a landscape in some time period in the past. Examples are numerous, and include interpretations of U.S. Public Land Survey information (Bourdo 1956, Stroessner and Habeck 1966, Kapp 1978, Galatowitsch 1990, White and Mladenoff 1994, VanDeelen et al. 1996, Batek et al. 1999), use of old aerial photographs (Quigley et al. 1996), or use of information such as stand reconstruction from dendrochronology (Harrod et al. 1998, Sloan 1998). Another approach to describing historical conditions emphasizes the disturbance regimes that occurred historically, and an understanding of their type, frequency, intensity, and resulting temporal effects on ecological communities (Agee 1993, Morgan et al. 1994, Richter et al. 1996, Cissel et al. 1999, White et al. 1999, Haufler 2000, Richter and Richter 2000).

From a species viability perspective, understanding historical habitat conditions can serve two important purposes. First, knowledge of historical conditions can be used to evaluate the status of a species within a landscape under historical disturbance regimes. Not all species occurring in a landscape historically had viable populations within that landscape. For species occurring along the fringes of their range, trying to maintain species viability in managed landscapes may be challenging. Nonetheless, such populations can be critical for ensuring long-term evolutionary potential of species. Moreover, what may in the short-term appear to be sink or suboptimal conditions may be critical for the maintenance of metapopulation dynamics (Donovon et al. 1995; Murphy 2001). In addition, an historical reference can provide valuable insights about the types of disturbances and conditions that the species routinely responded to within that landscape.

In addition to comparisons among historical and extant habitat conditions, the most frequent comparison between historical and current conditions involves evaluating changes in species ranges. This comparison may be most useful in the context of comparisons among areas of occupancy (i.e., areas within a species range where known populations occur). Information on historical population densities also may be available, and may provide an indicator of change. In contrast, rarely are historical data on species demography available.

Thus, historical information can provide a critical reference for understanding the viability needs of a species. However, because existing landscapes typically differ from historical landscapes in many aspects, methods are needed to evaluate existing capabilities of the landscape to provide for species viability, and to project future probabilities that the landscape can continue to support the species.

Wildlife Habitat Relationship Models. A common approach to predicting species occurrences is to map ecological communities across a planning landscape, and to assign species locations to the mapped communities according to the quality of the habitat represented by each ecological community for each species (Thomas 1979, DeGraaf and Rudis 1983, Hoover and Wills 1984, Hamel 1992). This is the basis for wildlife habitat relationship models, such as those used in California (Dedon et al. 1986, Block et al. 1994). In this approach, the habitat classification system selected will directly determine how species are associated with a predicted location. To implement this method a classification system of ecological communities or ecosystems is needed, that is accurately mapped at an appropriate resolution across the landscape. In addition, each classification category must be rated relative to its quality of habitat for each species. Numerous studies have addressed the issue of how to determine the validity and/or the appropriate geographic scale of application (e.g., Marcot et al., 1983, England & Anderson 1985, Raphael & Marcot 1986, Morrison et al., 1987, Laymon 1990).

This approach provides a mechanism for assessing the potential occurrence or status of many species under different management alternatives by only having to describe the change in the types of ecological communities that is expected from the different management alternatives. It has the disadvantage of not addressing the specific habitat needs of a species because it is only indirectly inferring habitat for species based on their assumed association with the various ecological communities classified for the landscape. Generally, this approach only is applied to species that use a single ecological community or ecosystem, rather than considering various contributions from multiple ecological communities to the habitat needs of a species that is more broadly distributed. This method primarily has been applied to vertebrate species, especially birds and mammals. While this approach can provide an index of potential changes in habitat quality for a species, it is difficult to project habitat in such a coarse manner and to relate it effectively to species demography or viability. Thus, while a useful approach for considering management alternatives at landscape scales, wildlife habitat relationship models typically do not directly address species viability.

Habitat Suitability Models. Another approach is to develop an index of habitat suitability to predict species occurrences based on descriptions of specific habitat variables (Schamberger et al. 1982). A model is built based on known habitat requirements of the species, and then linked to a classification of the landscape. To predict species locations, the habitat variables associated with the habitat model must be mapped across the planning landscape. This requires delineation of polygons of a classification system, and the associated attribute data on each habitat variable in the model to be described for each classified type of polygon. These models can be evaluated at the scale of a single stand or habitat patch, or they can address spatial interspersion and juxtaposition requirements of multiple stands for a species. For habitat suitability models to be accurate, they must have classification systems that allow for accurate description and variability in habitat variables. Furthermore, the variables must be sampled with sufficient rigor to provide an adequate attribute database.

Habitat suitability modeling has the advantage of compiling and relating known information on a species' habitat needs in a systematic and quantifiable manner. It helps identify where good information on the species is known, as well as information gaps on the species habitat requirements. Habitat suitability modeling also allows a quantitative comparison among management alternatives, although many of the model relationships are based on subjective professional opinion. Habitat suitability models require the quantification of relationships between habitat attributes and population fitness, relationships that often are poorly understood. Thus, a disadvantage of habitat suitability models is the lack of quantified information on the habitat requirements of most species. An additional limitation is that the habitat relationships described in the model are used to quantitatively rate alternatives, with the quantification often obscuring the lack of confidence in the model outputs. Furthermore, the models must be applied through use of a classification system of ecological communities or ecosystems, and a set of habitat attributes assigned to each ecological community. Errors can arise from selecting classifications that are too coarse in resolution or that stratify ecological communities in a manner that does not relate to the habitat characteristics needed by the species. Additional sources of error often include a poor description of habitat attributes, inaccurately mapped polygons within the landscape, or mapping ecological communities at a resolution that does not properly reflect habitat use by the species.

Individual Home Range-based Habitat Assessment - An additional type of habitat modeling, termed habitat-based viability analysis by Roloff and Haufler (1997), expands on the habitat suitability approach. For species for which home ranges can be determined, specific home range areas are delineated. Attributes of these areas that are closely linked with species demographic rates are then measured across a planning landscape. Habitat suitability models are designed and applied at the scale of a species' home range, but all species may not have identifiable home ranges. Through a multi-step process, associations between habitat attributes within home ranges and species demographic rates are determined. Then, these habitat attributes are measured within specific home ranges, which can be labor intensive, and requires relatively fine-scale data. Finally, each home range is assigned a habitat quality, where habitat quality is evaluated along a continuum, from low to high, based on relative rates of occupancy, survival, and reproduction (Pullium 1988, 1996, Donovan et al. 1995, Fauth et al. 2000, Roloff and Haufler in press). Outputs are the number of home ranges of specified levels of habitat quality distributed across the landscape

This habitat-based approach to viability assessment is, in many ways, a type of individual-based, spatially explicit model (see subsection in Demographic Models). It assumes specific relationships exist between demographic rates and habitat attributes, at the scale of a home range. Data requirements are the same as for habitat suitability modeling, with the addition of information on home range sizes for each modeled species. Demographic parameters linked to habitat attributes can be used to move this approach to an individual-based demographic model (Ackakaya 2000). The spatial description of habitat quality produced from this approach can be used for a variety of habitat-based population viability assessments (Akcakaya and Atwood 1997, Akcakaya 2000).

An advantage of the individual home range modeling approach is that it quantifies, in a spatially explicit fashion, the expected numbers and attributes of home ranges. It does this in a manner that can provide direct linkages to population sizes and demographic parameters of the species. This approach allows evaluation of management alternatives in terms of expected responses in population size. Further, it may facilitate an understanding of source-sink relationships, and can form the basis for various metapopulation analyses (see below). This type of approach is also used to determine potential umbrella species for conservation planning efforts. Species that occupy large home ranges, are thought to serve as umbrella species for populations of co-occurring species with smaller home ranges.

Such umbrella species have been used to locate boundaries of proposed conservation areas, however there is little empirical data in support of the effectiveness of commonly used umbrella species (Andelman and Fagan 2000). Its disadvantages are the same as for habitat suitability models, in that it requires known habitat relationships as well as home range information and the need for properly classified maps. This method does allow for the aggregation and interpretation of the effects of single or multiple activities at a project level, quantified through the landscape or larger scale (Roloff and Haufler, in press).

7.1.2 Demographic Models of Viability

Demographic models have been used frequently to evaluate population viability. Demographic models, especially many types of population viability analyses (Beissinger and McCullough 2002), attempt to quantify various combinations of decimating factors influencing a population. They differ from other classes of models in this section by explicitly incorporating birth and death rates, and to varying degrees, the processes that affect them. Demographic models vary in complexity, from deterministic matrix models of a single population to stochastic, spatially explicit models that keep track of each individual on specific landscapes. Here we briefly review each model type, including its data requirements, uses and inferences.

Matrix Population Models - Deterministic single-population matrix models (hereafter "matrix population models") are simple descriptions of population dynamics consisting of a set of equations (one for each age or stage class), which often are formulated into a matrix, that predict population size at time $t+1$ from information on the survival, growth and reproduction of individuals at time t. Matrix models are used to make short-term population projections unless it is assumed that long-term averages for the vital rates will experience relatively little change. They can be used to make estimates of the geometric rate of population growth (λ), to indicate population characteristics, such as the distribution of individuals among age classes (stable age distribution) or the reproductive value of age classes, and to evaluate the relative influence of demographic rates on population change (sensitivity or elasticity analysis) (Caswell 2001).

Matrix population models are the least data demanding of demographic models, although years of field study are needed to construct and parameterize them well. They require: (1) determination of ages or stages for analysis; (2) estimation of age or stage of first reproduction; and (3) estimation of reproductive success and survivorship for the different ages or stages. The decision of how many stages to use and their composition should be closely tied to variation in demographic parameters (Sauer and Slade 1987), but often only a few age or stage classes are used because field studies are rarely sufficiently long or detailed to measure rates for many ages or stages. Estimates of fecundity and survivorship are the most important components of matrix population models. Most matrix population models of vertebrates are constructed only for females, since male fecundity often is unknown.

Matrix population models have limited but important uses in assessing viability. Because they are deterministic models, they only indicate whether and how fast a population is increasing or decreasing, and do not give an estimate of the risk of extinction. They complement analyses of trend and distribution. Matrix models also are useful for evaluating impacts of specific management actions. Although the model may not accurately project population size over time, a sensitivity analysis of a matrix population model might indicate how changes in specific demographic rates (e.g., reproductive success versus adult survival) impact rates of population change.

Stochastic Single Population PVA Models - Stochastic PVA models project populations for 50, 100 or more years into the future by allowing demographic rates or λ to change for each year, using the Monte Carlo method, which randomly samples rates from predetermined distributions (Beissinger and McCullough in press; Shaffer 1987, 1981). Since each run of a stochastic model follows a unique trajectory and yields a different ending population size, models must be run 500 to 1000 times to explore the full range of parameter values and portray the distribution of possible ending population sizes (Burgman *et al.* 1993, Harris *et al*. 1987).

Stochastic PVA models can produce the proportion of runs that end at population size zero (“extinction” rate), or at a small size such as ≤ 25 individuals (“quasiextinction” rate). No standard time interval or extinction rate defines a viable population, but intervals of 50-100 years and extinction rates <5% commonly are used. Another result is to compute the mean or median year of extinction for populations that went extinct or the "time to extinction". Perhaps the most complete descriptor of model results is the cumulative probability function for ending population size, or the “quasiextinction function,” which is a basic form of risk analysis (Ginzburg *et al.* 1982, Burgman *et al.* 1993).

Stochastic single population models require estimates of about twice as many rates as matrix population models (Beissinger and Westphal 1998), although many of the additional rates can be derived from the same field data. In addition to estimates of mean age- or stage-specific survival and fecundity, to model the effects of demographic and environmental stochasticity, stochastic models require estimates of variance in fecundity and survival for each age or stage class. This also requires knowledge of the relationship between environmental conditions and vital rates, and a good measure of the variance in vital rates over the full range of environmental conditions encountered. Stochastic single-population models also should include carrying capacity and its variance (Ginzburg *et al.* 1990), and the frequency and effects of catastrophes (Mangel and Tier 1994). Stochastic single population models also can incorporate processes that may occur when populations become small, such as Allee effects or inbreeding depression.

Results from stochastic single-population models should be interpreted carefully. Although PVA models supplied with adequate data have the potential to produce an average trajectory from the hundreds of model runs over short time periods (10-25 years) that may track the trajectory of a real population with some degree of accuracy (Brook et al. 2000), these models are unlikely to produce accurate predictions of the likelihood of extinction over the long-term (Beissinger and Westphal 1998, Groom and Pascual 1998, Ludwig 1999, Fieberg and Ellner 2000, Foley 2000, Meir and Fagan 2000, Coulson et al. 2001). Moreover, point estimates of extinction probabilities will always depend on the time horizon and the threshold size at which a population is considered extinct. The primary prediction from stochastic single-population models, the probability of extinction, is difficult to validate because we cannot know which of the hundreds of simulated population trajectories will most resemble the trajectory of the unreplicated, real-world population. Comparing the model's average population projection to a time series of historical population trends provides a weak way to examine how well the model captures the short-term dynamics, but does not verify the value of stochasticity or other variables used in the model that are responsible for differences among model runs. Stochastic single-population models are best employed in conservation decisions by comparing *relative* differences among management options incorporated into the model rather than basing policies on the *absolute* rates of extinction predicted by the models (Beissinger and Westphal 1998). Such was the basis for comparing conservation strategies based on different sizes and configurations of reserves for the Northern Spotted Owl in the Pacific Northwest (Noon and McKelvey 1996).

Metapopulation Models - Metapopulation models can examine relationships between landscape structure and population dynamics by incorporating site-specific effects on demography. Such models rapidly increase in complexity because additional details of demography, dispersal and landscape dynamics to depict population dynamics accurately (Hanski in press, Harrison and Ray in press). Several types of metapopulation models exist. Here we examine only patch models that are based on demographic rates; incidence function approaches based on presence-absence data are examined in another section.

Patch models track populations in each individual habitat patch. Individuals reside only within these patches, may move or disperse among them, and are not permitted to survive in the surrounding landscape. Population dynamics within each patch are modeled as in stochastic single-population models, with the additional steps of determining the number of individuals that will disperse from and migrate to each patch. Patch metapopulation models provide output on population trajectories, risk of population extinction or decline, and related measures of population size.

Patch metapopulation models require estimates for dozens of parameters (Beissinger and Westphal 1998). Simple patch models may require patch carrying capacities, population growth rates, and immigration and dispersal rates among patches. Complex models may have data requirements similar to stochastic single-population models, but these parameters may vary among patches. Patch models also require parameters related to spatial structure, ranging from patch area and interpatch distances to rates of mortality during migration, which are very difficult to estimate. It also is important to know whether rates of dispersal and immigration are dependent or independent of density within a patch, and most importantly, the degree of correlation among patches for environmental stochasticity (Beissinger and Westphal, 1998). These relationships are extremely difficult to determine from field data, but may have a major effect on model results (Stacey *et al.* 1997).

Spatially Explicit Population Models. Spatially explicit population models (SEPMs) are complex simulation models that incorporate exact spatial locations of individuals, habitats, barriers to dispersal, and other landscape characteristics. They may be built for a single population or a metapopulation. Although their main outputs are population characteristics and processes, SEPMs often are individual-based models (IBMs); individuals are placed in known locations and assigned demographic traits based on the habitat where they are located. Output from SEPMs is comprised of averages over individuals to yield population statistics, such as population size at a specific time, population trajectories, or time to extinction. Spatially explicit models can be used to model movements of individuals across a diverse landscape, or the response of populations to changing landscape structure.

There are two common forms of spatially explicit population models. *Grid or cell-based models* track population sizes in equal-sized cells, which are typically the building blocks of larger habitat patches and are influenced by the inputs and outputs of neighboring cells. Grid-based models often are used for abundant organisms, such as plants, insects or rodents (Bradstock *et al.* 1996, Price and Gilpin 1996), where monitoring the movement or fate of each individual may be intractable. *Individual-based models* track the location and behavior of every individual, and typically have been applied to vertebrates whose biology and demography have been studied extensively, to simulate responses to regional management practices (Boone and Hunter 1996), evaluate translocation options (Akçakaya *et al.* 1995) and simulate the effects of forest management policies (Liu *et al.* 1995, Lamberson *et al.* 1994, McKelvey *et al.* 1993).

An important step in both kinds of models is to link a population model to a landscape map. Maps require designation of habitat quality from the standpoint of the species considered and often incorporate real-world landscapes using extensive GIS databases. The population model often is a life-history simulator that classifies each individual by sex and age, and then follows it through an annual cycle of breeding, survival during the nonbreeding season, and dispersal.

To build SEPMs requires large amounts of data, and as a result, their ability to make accurate predictions is limited. In addition to extensive data on the distributions of ecological communities across landscapes, habitat-specific demography and an accurate quantification of dispersal patterns and movement rules are required. Maps of habitat distribution are useless without ecological knowledge that determines habitat quality for reproduction and survival. Such detail rarely is available for any species. Data often are especially limited for dispersal, which often is a very sensitive model parameter (Wennergren *et al.* 1995, Ruckelhaus *et al.* 1997, Mooij and DeAngelis 1999). Incorporating dispersal using alternative methods that do not depend on modeling step-by-step movement across a landscape may be necessary.

7.1.3 Analyses of Genetically Effective Population Size

Genetic goals related to preservation of genetic variability and avoiding effects of inbreeding have been used to evaluate population viability. The importance of genetic versus demographic goals in population recovery efforts has received much discussion over the past 10-15 years (Schemske *et al.* 1994, Lande 1988, Beissinger et al. in press). The concept of *Effective Population Size or* N_e has been used to incorporate genetic considerations into viability assessments.

N_e measures the effect of genetic drift on a natural population. It can be defined as the number of individuals in a population that contribute genes to the next generation. N_e is a useful way to evaluate the magnitude of genetic loss over time in a small population by comparing how it deviates from an “ideal” population, or by examining the relationship between N_e and the actual or census population size (N) (Waples in press). N_e often is substantially less than N, usually only 10-25% of total population size (Frankham 1995, Waples in press). An “ideal” population has both genetic and demographic characteristics that include: (1) large and stable population size, (2) equal genetic contribution of both sexes, (3) no inbreeding, (4) equal family size, and (5) non-overlapping generations. All organisms violate some or all of these characteristics. However, it is the relative change of N_e over time and comparison of N_e with N that provide a yardstick for assessing viability. Demographic data, genetic data, or some combination of the two may be used in analytical equations to estimate N_e.

N_e was one of the first measures of population viability. Franklin (1980) and Soulé (1980) suggested that small populations should try to attain an N_e of 50 individuals over the short-term and N_e of 500 individuals over the long-term (the "50/500 rule"). The 50/500 rule was quickly incorporated as a guideline for endangered species recovery planning. Although the concept was well intended, the numeric goals of 50 or 500 were sometimes applied to the census population size, without regard to the structure of the population. Lande (1995) later elaborated on the 50/500 rule by proposing that a minimum effective size of 5,000 was needed for long-term viability. More recently, Franklin and Frankham (1998) argued that an N_e of 500-1,000 would be appropriate, but this is a value judgment, rather than a scientific standard.

7.1.4 Incidence Functions

The size of patches or fragments of a given type of environment within a landscape influence the colonization by individuals, persistence of individuals and breeding units, and numbers of species in an area. Incidence functions describe the probability of occurrence of a species as a function of patch or island size. Incidence functions describe the likelihood of a species being found within a patch of habitat of a particular size, and also can be interpreted as the proportion of patches of a given size having species present. Incidence functions also can be used to determine the number of species in a patch of a given size. Measures of species richness are assumed to be area dependent and $S = cA^z$, where S = the number of species present, A = area of patch, c is a scaling constant that varies by taxon and location, and z is the rate at which the number of species increases with increasing area. As a consequence of the z factor, as a general rule of thumb, twice the number of species will require ten times the area.

In some ways incidence function models are simpler than population dynamics models. In simplest form, the state variable is the presence or absence of a particular species in a patch, which is the typical metapopulation model discussed above. Patch metapopulation models quickly can become complex when accounting for spatial geometries in arrangement of patches, multi-species interactions, and in quantifying the transitions between states of presence and absence.

Incidence function models can be developed using two approaches. In the patch-occupancy approach (Hanski 1994*a,b*), simple presence-absence data obtained from a single or multiple surveys is used to determine the probability of a patch being occupied. The patch-turnover approach requires multiple surveys to document extinction and colonization in each patch (Sjogren-Gulve and Ray 1996, Hanski, 1998). Both approaches rely on a combination of analytical and simulation techniques, as detailed in Hanski (1999b). Logistic regression often is used to evaluate the influences of patch area, isolation or other factors on the proportion of occupied patches. Once parameterized, the model then can be applied to other metapopulations or to simulate metapopulation dynamics by substituting new patch area and isolation data (Hanski 1999b). A rapidly growing list of species has been studied using the incidence function approach, including insects (Hanski 1994*a,b*), plants (Quintana-Ascencio and Menges 1996), reptiles (Hokit *et al*. 1999), birds (Cook and Hanski 1995) and small mammals (Hanski 1991).

The incidence function approach is limited by available data on effects of landscape characteristics on colonization and extinction. To apply this approach effectively, we need to better understand which landscape parameters correlate with and directly affect species abundance and distribution. We also need to develop better ways to predict effects of subdividing populations. For example, fragmentation-sensitive species may require different management than other species.

7.1.5 Trend Analysis

Population size and trend (average percent change in population over some specified period, e.g., 20 years) are ingredients in any model that proposes to calculate an extinction risk, and management actions can be evaluated in terms of the change they promote in population trend or size. Thus, evaluating population trend is fundamental to the process of viability assessment. Since point estimates of extinction probabilities will always be highly uncertain, and will always depend on the time horizon and the threshold at which a population is extinct, it often is far better to describe the risk faced by any population with two attributes: current population size and trend (e.g., average change in population size over the last twenty years).

In a typical sequence of counts from a population, estimates do not increase or decrease smoothly over time, but instead show considerable variation around long-term trends. Variability in environmental conditions, which causes birth and death rates to vary from year to year, is likely to be important in such fluctuations. The simplest conceptual model of population growth is $N_{t+1} = \lambda N_t$, where N_t is the number of individuals in the population in year t, and λ is the per capita population growth rate. The discrete growth rate of a population (λ) is a function of many factors, and variation in λ may be produced in several ways. λ may be considered to be either independent of population density or density dependent. When environmental variation causes survival and reproduction to vary from year to year, the population growth rate also must be viewed as varying over some range of values. Further, many environmental fluctuations are unpredictable, so we cannot predict with certainty what the exact sequence of future population growth rates will be. As a general rule, as variability in λ increases, the geometric average growth rate over a set period of time is less than the arithmetic mean average growth rate over the same time period.

Count data, such as the number of individuals in a population surveyed over multiple years, may be relatively easy and cheap to collect, particularly as compared to detailed demographic data. Analyses of extinction risk can be made based on trend data. Dennis et al. (1991) developed an approach that is now widely used in analyses of extinction risk. This approach, based on trend analysis, incorporates variability in growth rates by extracting a maximum likelihood estimator of growth rate and the confidence interval about that growth rate (Dennis et al. 1991) on the assumption that population changes can be approximated by a simple diffusion process with drift. As long as density dependence is not strongly evident in the data, this simple diffusion approximation to population change is thought to work well (Dennis et al. 1991). The method translates the diffusion approximation into analyses of net population change. More recently, these approaches to trend analysis have been extended to allow the use of data with large sampling error, and/or density dependence to estimate risks in declining populations (e.g., Dennis & Otten 2000, Engen & Saether 2000, Saether et al. 2000, Holmes 2001).

Elicitation of Expert Opinion

7.1.6 Types of Expert Opinion

Viability assessments always involve some degree of expert opinion and judgment. For example, even scientists conducting a highly data-driven population viability analysis must make judgments on many questions, such as whether it is appropriate to use an existing PVA model and whether it is permissible to base some model parameters on data from a closely related species (Ralls et al. in press). When data are insufficient to build any type of model, assessments must be based entirely on expert opinion.

The opinion of experts frequently is used to assess the viability of forest management options, and a large variety of techniques are available to elicit expert opinion. Experts can work alone or in groups. They can be provided with multitudes of information on the species and landscapes that they must evaluate. Opinions of experts can be gathered using structured approaches, such as those prescribed in Morgan and Henrion (1990, Ch. 6 and 7), or non-structured techniques such as those used in some Forest Plans, including the Sierra Nevada plan amendment. Experts may be asked to give an overall estimate of species status (a common, but not very sound practice) or make separate estimates of various components of viability (such as population size, juvenile survival, whether or not available habitat is decreasing), which are subsequently combined to produce an overall estimate of viability (a much better approach).

Although it sounds simple to ask experts for their opinions, the proper use of expert opinion actually is an academic field belonging to the larger area of decision analysis (Morgan and Henrion 1990, Goodwin and Wright 1991, Meyer and Booker 1991). A naive approach to using expert opinion can fall prey to well-known potential biases and errors in the human judgment of uncertainty (Goodwin and Wright 1991, chapter 8; Anderson 1998). When decisions must be based largely or entirely on expert opinion, it is important to use well-tested, established techniques. Because these techniques are unfamiliar to most biologists, collaboration with people trained in their proper use is highly advisable.

In brief, it is important to (1) identify the variables for which expert opinion is needed; (2) choose unbiased experts, or at least a balance of opposing biases among the experts consulted, to improve credibility of the final results; (3) motivate the experts; (4) create a "mental model" outlining the chain of inference from the information provided by the experts to the final viability assessments; (5) provide the experts with a common pool of information; (6) use proper procedures for eliciting and verifying probability estimates; (7) decide how to combine individual judgments if more than one expert is used; and (8) thoroughly document the entire process used so that it can be evaluated by others. Chapter 8 provides a more detailed explanation of each of the above steps.

7.1.7 Natural Heritage Program Method for Assessing Viability of Occurrences of At-Risk Species

Many National Forests utilize a specialized type of expert opinion-derived information on Threatened and Endangered Species, USFS Sensitive Species, and state species of special concern that they obtain from state Natural Heritage Programs (NHPs) and the Association for Biodiversity Information (ABI). Typically, NHPs provide the Forest Service with information on the species taxonomy, specific locations of occurrences or populations on the National Forest of concern, dates of observations, names of observers, relevant population or habitat information if it exists, and a rank of the estimated viability of an occurrence or population. The basic unit that NHPs use for tracking and maintaining on-the-ground information on occurrences or populations of at-risk species is referred to as the *element occurrence* or *EO.* An element occurrence is defined as an area of land and/or water in which a species or natural community is, or was, present. For species-level elements, the EO often corresponds with a local population, but when appropriate, may be a portion of a population or a group of nearby populations (e.g., metapopulations).

Element occurrence ranks or EO ranks provide a structured assessment, based on expert opinion, of the estimated viability or probability of persistence of individual occurrences of a given element (http://www.natureserve.org). Another way of stating this is that EO ranks provide an assessment of the likelihood that an occurrence will persist for a specified period of time if the current conditions at the site persist. An EO rank represents a Heritage Program biologist's assessment of the relative value of an occurrence with respect to other occurrences for that same species. The basic EO ranks that are used in ABI/Heritage methods are: A = excellent estimated viability; B = good estimated viability; C = fair estimated viability; and D = poor estimated viability.

In essence, EO ranks provide Forest Service biologists with information in a standardized format on the quality of a given occurrence or population of an at-risk species relative to other occurrences or populations for that species. As noted below, this information is derived from primary scientific literature and from expert opinion. It is necessary and helpful, but not sufficient information to judge whether the species is viable on a particular National Forest. By itself, the EO rank information does not provide a biologist or planner with information on whether the species is well distributed across a forest or even how that species should be distributed. Nor does it usually provide specific information concerning viability analyses (e.g., probability of persistence or extinction) unless such work has been published and is available for a particular species. It does, however, provide forest biologists and planners with an indication of the condition and quality of individual populations or occurrences, which may provide some insights into the viability of the species across the planning area.

EO ranks are based on factors that reflect the present status or quality of an element occurrence. In general, three factors provide the basis for estimating the viability of an EO: size, condition, and landscape context. *Size* is a quantitative measure of the area and/or abundance of an occurrence. Major components of the size criterion are area of occupancy, population abundance, population density, and population fluctuation. *Condition* is an integrated measure of the quality of biotic and abiotic factors, structures, and processes within the occurrence. Components of this factor include reproductive status, population structure, ecological processes, and abiotic physical/chemical factors. *Landscape context* is an integrated measure of the quality of biotic and abiotic factors, structures, and processes surrounding the occurrence including the condition of the landscape and connectivity from the occurrence to adjacent habitats.

In general, current stress and trends on an occurrence should be reflected in the criteria of size, condition, and landscape context. Therefore, trend *per se* is not used as a factor in ranking an EO. Other factors that, in the past have been considered important for ranking an EO are future stresses, defensibility of the EO, manageability, and restoration potential. Because these factors do not represent the relative conservation value of an EO in its current condition and relate more to the uncertain impacts of future events, they generally should not be considered in the ranking of an EO.

Size is the primary factor influencing EO rank for many species with condition and landscape context used secondarily (especially vertebrates). The assumption in this case is that large size would generally not occur without a favorable rating of the condition and landscape context criteria. For species where limited information is available on size (e.g., many plants and invertebrates), condition and landscape context factors may be relied upon more heavily.

In practice, species experts or teams of experts develop element occurrence specifications that define what constitutes an occurrence for a given plant or animal species. Then these same experts also develop specifications for what should be considered an “A” element occurrence rank, a “B” element occurrence rank, a “C” rank, and so on. These specifications are based both on expert opinion and the primary literature on these species. The information on EO definitions and EO ranks is maintained centrally by ABI and is made available to all NHPs in the United States. Individual Heritage Programs then use this information to assign EO ranks to individual populations and occurrences of at – risk species in their respective states.

Element occurrence specifications or definitions and the accompanying EO rank specifications have been developed for some, but not all, vertebrate species and vascular plant species in North America. In addition, only some Heritage programs have assessed the viability and condition of individual occurrences in the field and assigned these ranks to those occurrences. As a consequence, only a portion of the Heritage information on at–risk species provided to National Forests by their local Natural Heritage Programs will actually contain specific information related to viability (i.e., the EO rank information).

In the end, expert opinion is exactly that – the best judgment of a group of experts. It can be no better than the information they have to work with, combined with their intuition about how that information should be interpreted. However, the quality of assessments based on expert opinion can be easily degraded by using haphazard procedures rather than following established protocols described in Chapter 8.

Use of Species Viability Assessment Tools in Past Forest Service Planning

We reviewed viability assessment methods used in eight Forest Plans. Five of these were broad-scale assessments: Interior Columbia Basin (Draft), Interior Columbia Basin (Final and Supplemental), Sierra Nevada National Forest Plan, Northwest Forest Plan and Tongass Land and Resource Management Plan. We divided the Interior Columbia Basin into two plans ("Draft" and "Final & Supplemental"), because the viability assessment was approached in a different way in each. The other three were forest-scale assessments: Northern Great Plains, White River National Forest and National Forests in Wisconsin and Minnesota. The broad-scale assessments were complete at the time of our review, however, the forest-scale assessments were in draft form and, therefore, some revisions or changes could occur before the final Environmental Impact Statements (EIS) for those plans are released.

For each management plan, information was obtained on 1) plan attributes, including spatial scale (i.e., broad-scale vs. individual forest-scale); 2) taxonomic groups considered in the viability assessment (i.e., non-vascular plants, vascular plants, invertebrates, fish, amphibians, reptiles, birds, mammals); 3) the total number of species and number of federally listed species considered in the viability assessment; 4) the format for supporting analyses (e.g., appendices, separate technical reports, etc.); 5) the number of management alternatives evaluated. In addition, for each plan, information was obtained on the type of analysis (i.e., habitat-based analysis, demographic models, or other). For habitat-based analyses, four types of analyses were distinguished: 1) comparison with a historical reference state; 2) wildlife habitat relationship models; 3) habitat suitability models; and 4) individual home range-based habitat assessments. Demographic models were characterized as 1) matrix population models; 2) stochastic single population PVA models; 3) metapopulation models; or 4) spatially explicit population models. Other analytical methods included: 1) analyses of genetically effective population size; 2) incidence function approaches; or 3) trend analysis. Where species viability was assessed using expert opinion, the following information was obtained: 1) use of structured vs. non-structure approaches; 2) use of panel of experts vs. individual experts; 3) number of panels; 4) number of panelists; 5) use of decomposition methods (overall assessment vs. components); 6) process of reconciliation; 7) method used to combine information; 8) peer review process, including content review and process review.

We extracted the information from final or draft EIS, technical reports, and from personal communication with Forest Service personnel. Tables 7.2, 7.3, and 7.4 summarize the results of the management plans we reviewed.

Plan Attributes are the basic information about the species assessed and the characteristics of the management plans reviewed. The number of alternatives ranged from 3 to 11 and the number of species assessed from 6 to 623 (Table 7.2).

Analysis Attributes are the quantitative methods described in this chapter that were used in the species assessment for each management plan. These methods can be used in combination with expert interpretation to complete a viability assessment. However, Table 7.2 shows that few quantitative approaches were used in broad-scale assessments or forest planning scale assessments. Spatially explicit demographic analysis and trend analysis was used for only a single species, the northern spotted owl, in the Northwest Forest Plan and metapopulation analysis was used for the western prairie fringed orchid in the Northern Great Plains (FEMAT 1993, Appendix N, NGP-PRMP). Matrix population models, stochastic single population models, analyses of genetically effective population size and incidence functions were not used in any of the plans we reviewed. In six of eight cases, expert opinion was the final and most important approach in the viability assessment process (Table 7.4). The Sierra Nevada Plan was exceptional because it used habitat-based models and NHPs to assess the viability of all 623 species of concern (Volume 3, Chapter 3, part 4 SNFPA-FEIS). Finally, habitat suitability models were developed in two plans: for one species in the Tongass and for two species in the Sierra Nevada (Table 7.3; Appendix N, TNF-FEIS).

Half of the plans we reviewed used expert panels to estimate the level of risk to a species or a group of species from alternative management scenarios (Table 7.4). The number of experts on a panel ranged from 3 to 7, and the number of panels from 8 to 13 (Table 7.4). Where expert panels were used, discussion among panel members with no requirement for consensus was the primary process used to reconcile differing points of view among experts. The Tongass Plan used a modified Delphi approach, but did not require consensus among experts (Shaw 1999). Scores from individual experts typically were averaged to obtain a single outcome per species (Shaw 1999, FEMAT 1993, ICB-Draft 1997).

The remaining four plans, with the exception of the Interior Columbia Basin (Final and Supplemental), used an unstructured panel of experts or an individual expert to assess species viability. The experts were not necessarily experts on the species, but were members of the EIS team. The Sierra Nevada Plan consulted a single individual for each species. The individuals consulted were considered by the EIS interdisciplinary team to be the most knowledgeable expert for that particular species or taxonomic group. In total, fourteen experts worked individually on the viability assessments for the Sierra Nevada Plan.

The Sierra Nevada, Interior Columbia Basin (Draft, Final and Supplemental), and the National Forests in Wisconsin and Minnesota plans decomposed the expert opinion into environmental and population components (Table 7.4). The Interior Columbia Basin (Final and Supplemental) was unusual in that it used Bayesian Belief Network Models (BBN) to structure, decompose and combine available information, including both empirical data and expert judgment to assess species viability under a range of management alternatives (Marcot et al. in press).

Considerations Affecting Which Approach(es) Should Be Used to Assess Viability

Deciding which approach or approaches should be used to assess viability and how to implement them provides a challenge for the Forest Service. The choice of methods depends in part on the number of species to be assessed and the quantity and quality of data available to support an analysis. As the scale of analysis increases, more species will need to be considered and more landscapes will need to be analyzed. This increases the demand for data and affects which analyses are appropriate. More complex models require greater technical skills and more time to create. Below we discuss the choice of scale, data requirements, and technical constraints before finally returning to the question of which methods to select.

7.1.8 Spatial and Temporal Scale Considerations

A significant consideration for species viability assessment is the scale at which the viability analysis is applied. Viability analysis requires that a sufficiently large area be considered so that a reasonably sized population is assessed. Project level activities cannot address viability of species, but need to tier to analyses of larger areas. Forest plans may address large enough areas to address viability analyses for a majority of the species in the landscape, but even at this scale, viability assessments for some species may not be possible. Broad scale assessments should allow for viability assessment of all but a handful of species with very low population densities or very large home range requirements. Obviously, migratory species present additional challenges with respect to viability assessments.

For most species, viability assessments should be conducted at the scale of the planning landscape for one or more National Forests, or with broad scale assessments. If viability is assessed at the forest planning scale, it will help assure the continued persistence of the species through redundant analyses of planning landscapes, as well as partially address the distributional objectives for each species. Even within these landscapes, mixed ownership of lands can make viability assessments problematic.

For species that require viability to be assessed at larger scales than included in a forest plan, other options may be possible. If broad-scale assessments have or are being conducted, this should address viability needs for nearly all species. If these assessments are not available, then Forest Service planners may wish to coordinate with other agencies to conduct a larger scale assessment, or alternatively, the viability assessment may identify the contribution of the planning landscape towards the overall viability of the species. In this later case, comparisons to the estimated historical role of the landscape in maintaining the viability of the species may be useful information.

Obviously, no one scale can be recommended as appropriate for all species. Forest planners will need to evaluate the appropriate scale based on the needs of each species, and the availability of data for different areas.

Temporal scales for viability assessments also are difficult to define clearly. Forest planning typically considers the next 10-15 years. However, viability analyses generally address time spans of 50-100 years or longer. Viability goals in forest planning should strive to assure that activities to be conducted over the planning period will not compromise the long-term persistence of a species. Thus, assessments should use 50 or 100 years when analyzing the impacts of land use change on viability, especially for long-lived organisms with long generation times (Lande in press). There is a trade-off, however, in the duration of the time span analyzed and the accuracy of predictions from models or other assessment methods. Errors are propagated with each time step (often one year) into the future that the model or assessment evaluates. Thus, as the time span is lengthened to assess important characteristics of viability, confidence in model outcomes decreases and it becomes important to carefully evaluate the objectives of the modeling exercise and the way the model output will be interpreted, as discussed in the next section.

7.1.9 Quality and Quantity of Data Available Affect Model Objectives, Choice and Testing

Various types of data may be available for conducting species viability assessments and the data requirements for each type of model are summarized in Table 7.1. As models become more complex and realistic, the amount of data required increases rapidly (Beissinger and Westphal 1998). Furthermore, as the size of the area for analysis increases and the number of species to be analyzed grows, the demand for data increases greatly. Whenever the data permit, it is useful to employ several models that increase in complexity to analyze viability.

When there is sufficient knowledge of demography, dispersal, habitat use, and threats, there is potential to use some of the kinds of mathematical models discussed above. Statistical models (e.g., incidence function models) may yield precise model outputs for some situations, but their generality may extend only to a specific set of data and they may have limited applicability due to the assumption of a stationary landscape configuration required by the model (Hanski in press). The most realistic models often are stochastic PVA, metapopulation or spatially explicit models that can be developed for specific species, management scenarios, or landscapes. These models, however, often yield less precise results because they are stochastic, so that model outputs are in the form of probability distributions rather than a single result, and because all model parameters can rarely be estimated from data specific to the system of interest. *Thus, the choice of a model often will be limited by the objectives of the analysis, the data that are available and the assumptions that are realistic to make* (Ralls et al. in press).

The typical objective of a viability assessment is to inform managers about the relative benefits and risks of different alternatives to a set of species, and to determine which alternatives lie above some minimum acceptable level of risk, recognizing that the determination of what levels of risk are acceptable is a matter of policy, not science. This objective often is concerned with comparisons of the risk (e.g., of extinction) to a species posed by different management options relative to one another (i.e., *relative risk*), as well as estimating an accurate risk or impact from a particular management action (i.e., *absolute risk*). A model that attempts to estimate absolute risk should strive to develop an accurate, comprehensive representation of reality and would include all factors influencing the probability of extinction (Ralls et al. in press). A model that is intended to be an accurate representation of reality should be judged by its accuracy and its ability to predict the future should be extensively validated. Such a model would incorporate a wealth of data that are not available for most, if any, endangered species. Building models that can accurately estimate absolute extinction risk and the data required for these models are, for most species, still beyond our capabilities (Ludwig, 1996, 1999; Fieberg and Ellner 2000). Models designed to compare differences among management options (i.e., estimate relative risks of extinction), however, may be less comprehensive than a model intended to estimate absolute risk of extinction and can be useful for assisting in decision making.

Such a model should be judged by how well it incorporates the most important processes thought to affect the species and by whether it provided insights helpful to the decision-making process.

Inherent in effective use of any of these analyses is the need for quality data. Data of questionable accuracy or resolution cannot be enhanced through application of analysis tools that may mask high variability or inaccuracies in the measurement of input data. Even for the simplest models of habitat associations, little attention has been paid to factors that can influence accuracy: the resolution of the mapped information (Hollander et al. 1994, Haufler et al. 1999), the quality of the attribute data used to quantify relationships (Bender et al. 1996), and the accuracy of the mapped information (Roloff and Kernohan 1999, Karl et al. 2000). At a recent symposium addressing scale and accuracy issues in modeling species occurrences, the importance of habitat classification systems was hardly mentioned in the 150+ presentations. Yet, the classification system selected to define habitats potentially can cause major errors in model performance, and has contributed to skepticism about the ability to accurately model species occurrences or abundances. Similar examples of problems associated with population or demographic data and models also have been described (Beissinger and Westphal 1998, Groom and Pascual 1998).

Models that can and have been tested with independently gathered data, and those that explore the range of uncertainty in parameter estimates, are likely to be more useful for decision-making than untested, unexplored models. Models can be validated by testing primary (e.g., population projections) or secondary (e.g., dispersal distributions) predictions (Beissinger and Westphal 1998), or by evaluating the ability of the model to replicate past system behavior (Bart, 1995). Habitat models can be validated to some extent with presence-absence data collected in field surveys. Validation of population models and genetic models is more difficult. Population projections resulting from stochastic single-population, metapopulation, and spatially explicit Individual Behavior Models are difficult to validate directly, but secondary predictions of the models can be tested to evaluate their general performance. For example, a SEPM might predict the proportion of successful nests of a songbird to be a function of patch size and isolation, which could readily be tested. If the prediction proved false, it might suggest that a simpler, perhaps not spatially explicit, model would suffice to represent the dynamics of the population. Sensitivity analysis is useful to explore the effects of uncertainty in model input values and to direct attention to the parameters that matter most to model performance.

7.1.10 Technical Skills

Considerable technical skill and experience are required to create and parameterize many of the models discussed in this chapter. Although some models, like stochastic single population PVA models, can easily be built with user-friendly software (Lindenmayer et al. 1995), most models require analyses programmed from scratch to evaluate management options that are particular to each viability assessment. Even in the case of canned software programs, sophisticated statistical analyses are required to develop robust and defendable estimates of the vital rates and other variables that are input into the models (White et al. in press). Past experience and professional expertise also can be important to properly facilitate panels of experts in viability assessments (see next chapter).

Lack of technical skills and experience are likely to inhibit viability assessments because it would be unusual to find the appropriate skills represented by the biological staff at the individual forest level. Thus, the Forest Service will need to provide a source of technical assistance to support forest level staff responsible for viability assessments. One model for providing such support would involve establishing an internal Forest Service working group on viability assessment, as well as an inter-agency working group on viability.

The interagency working group could consist of individuals with appropriate technical expertise from federal resource management agencies (e.g., U.S.F.S., U.S.F.W.S., N.M.F.S., B.L.M., N.P.S., D.O.D., D.O.E., E.P.A.), relevant professional societies (e.g., Society for Conservation Biology, the Wildlife Society, American Forestry Society) and the academic community. The group might also be organized into taxon-specific subgroups. The goal of this umbrella group would be to establish and maintain taxon-specific standards for viability assessments. Such a structure might be modeled along the lines of the World Conservation Union (IUCN) Species Survival Commission and its specialist groups. The viability working group could be charged with developing and maintaining standards for data, modeling and expert opinion that would constitute an adequate viability assessment, as well as with developing data collection and research priorities to advance these goals. The sub-groups would provide a mechanism for both reviewing and potentially conducting species-specific viability assessments. Such a group would be a cost effective approach to providing the technical support to Forest Service (and other agency) field personnel responsible for conducting viability assessments.

Although constituting an interagency working group would represent significant coordination and implementation challenges, the issue of how best to conduct credible viability assessments is important enough to merit such an effort. Population viability s still an emerging field without clearly identified standards or broadly accepted protocols. This argues for a broad-based and inclusive approach to assure some level of consistency, avoid duplication of effort, leverage the efforts of many participants and agencies to advance the field, and to develop a consensus on standards that is testable and capable of being refined over time.

The Forest Service should also consider establishing an internal viability working group. This group would serve two functions: a) representation of the Forest Service within the larger inter-agency working group described above; and b) providing service and support to the forests and regions in implementing scientifically credible viability assessments as part of ongoing forest management and planning efforts.

7.1.11 Suggestions for Selection of Viability Assessment Methods

Conservation biology and ecology are still young sciences that offer few general theories and empirical laws to guide policy decisions (Shrader-Frechette and McCoy 1993). Given this situation, conflicts over natural resource management decisions are inevitable and forging consensus is difficult. Tools that allow us to anticipate the impacts of potential management actions, evaluate responses of future landscape changes, and make other kinds of projections to evaluate the potential future consequences of current decisions are especially valuable.

Habitat-based approaches will be a frequently used tool for species assessments in forest planning. Projections of the future status of species must be linked to spatially explicit estimates of the amount and quality of future habitat under each management alternative considered. Where additional levels of analysis are required, and where data permit, these habitat-based approaches can be linked with demographic parameters to provide predictions of future species status. The quality of habitat-based models will depend critically on the quality of the information used for parameterization, including the accuracy of habitat classifications and the resolution and accuracy of habitat mapping.

If they are well grounded in scientific knowledge of the system depicted, formal mathematical and simulation models can be extremely useful for evaluating such impacts and for assessing viability. Models offer a repeatable, transparent method of assembling and evaluating information. Model projections are especially useful when there are many management options and the system is sufficiently complex that the potential impacts of management actions are not obvious. Thus, whenever possible, it is preferable to employ a modeling technique in Table 7.1 to assist in decision-making, rather than use a less repeatable or less transparent method, like expert opinion. Sometimes, however, models foster an illusion of precision, which can make models especially appealing to decision-makers, even when the biology behind them is not sound.

When data are missing or highly uncertain, scientists often differ on how best to use formal, mathematical models in decision-making. Some scientists believe that it is *always* beneficial to build a formal mathematical model because it makes explicit the relationships among variables or factors of concern (Burgman and Possingham 2000, Caswell 2001). Others suggest that models are useful for assessing viability only when there are data of sufficient quantity and quality to support the objectives of the modeling exercise (Beissinger and Westphal 1998, Groom and Pascual 1998, Ralls et al. in press). Models can be and often are built with incomplete or missing data, and models can be used to evaluate how much the missing data affect predicted outcomes (Starfield et al. 1995, Starfield 1997). Nonetheless, there often are so few data about the vast majority of species in a regional viability assessment that modeling, particularly with complicated models, may be misleading (Noss et al. 1997, Ralls and Taylor 1997, Ruckelshaus et al. 1997, Beissinger and Westphal 1998, Groom and Pascual 1998). For example, Green and Hirons (1991) found that data suitable for population modeling were available for only two percent of threatened bird species. If there are few basic data on the distribution, abundance, demography, dispersal, habitat use, natural history, and threats to a species (Table 7.1), perhaps a more desirable approach would be to use some other form of structured decision making, such as expert panels, providing the panels with as much detail as is feasible (see next chapter).

Unfortunately, there often is too little known about the system of interest – species' habitat use and population dynamics, or the impacts of specific landscape changes due to management options – to justify creating a formal mathematical model. To determine whether to use a model and which modeling approach to use, first be sure that the objectives of the modeling exercise are clearly stated. Specific objectives, such as "estimate the risk of extinction during the next 100 years," will be more helpful than general ones. Then determine whether or not any analysis will be adequate, depending upon the availability of data. Although some unknowns always are to be expected, and are acceptable, modelers should not have to guess about the estimates for a large number of parameters, especially parameters that may critically affect the outcome of the model. The complexity of the model should be appropriate for the amount and quality of data available. Beissinger and Westphal (1998) and Groom and Pascual (1998) discuss these issues in more detail. As a general rule, the model should be simple if there are few data and more complex models should not be used if there are not reasonable datasets available. Finally, determine if the assumptions of the model are appropriate to the situation being evaluated. More complex models often make fewer simplifying assumptions, but they require more data. The validity of assumptions of equilibrium conditions is especially important for deterministic matrix population models and incidence function models, whereas assumptions about dispersal patterns and mortality associated with movement are critical to metapopulation and spatially explicit models (Beissinger et al. in press).

In summary, a variety of methods can and should be brought to bear to assess the viability of species under forest management plans. When possible, a method of formal mathematical modeling often is preferable to the use of less quantitative methods. However, the potential for formal models to be built is likely to be limited to only a few or a dozen of the 50-300 species that are assessed in management planning. Thus, it seems likely that the Forest Service will depend on expert opinion in some form to assess viability for the majority of affected species. How expert opinion should be used in this context is discussed in detail in the next chapter.

Table 7.1

	General Data Needs				
Analysis	**Habitat Maps**	**Occurrence**	**Demography**	**Dispersal**	**Pop. Estimate**
Habitat based					
Historical Reference	Past, Current	Past, Present	No	No	No
Habitat Relationships	Current	Present	No	No	No
Habitat Suitability	Current	Present	Population	No	No
Home Range Assessment	Current, Future	Present	Individual	Individual	No
Demographic					
Matrix	No	No	Population	No	No
Stochastic PVA	No	No	Population	No	Population
Metapopulation	Current, Future	Population, Patches	Patch	Population	Patch
Spatially Explicit IBM	Current, Future	Individuals, Patches	Individual	Individual	Patch
Other					
Genetic Effective Size	No	No	Population	No	Population
Incidence Function	Current	Population, Patches	No	No	No
Population Trend Analysis	No	No	No	No	Population, Index

Table 7.2 Plan Attributes. # of alternatives= number of management alternatives assessed; INV= invertebrates; NV=Non-vascular plants; V=Vascular plants; F=Fish; A=Amphibians; R=Reptiles; B=Birds; M=Mammals; # Species=Total number of species assessed for viability; #*= some species were grouped.

		Taxonomic Groups Considered								
	# Alternatives	INV	NV	V	F	A	R	B	M	# Species
Broad-scale Assessments										
Interior Columbia Basin Draft	7 +historical & current			Y	Y	Y	Y	Y	Y	219*
Iinterior Columbia Basin Final and Supplemental	3				Y		Y	Y	Y	34
Sierra Nevada National Forest	9			Y	Y	Y	Y	Y	Y	601
Northwest Forest Plan	10	Y	Y	Y	Y	Y		Y	Y	1120
Tongass National Forest	11				Y			Y	Y	16*
Forest Planning Scale Assessments										
Northern Great Plains	5	Y	Y	Y	Y	Y	Y	Y	Y	47
White River National Forest	6							Y	Y	6
National Forests in Wisconsin and Minnnesota	8/6		Y	Y	Y	Y	Y	Y	Y	176

Table 7.3 Methods used to assess species viability. Some species were assessed using more than one method.

	Habitat Based				Demographic				Other			
	Comparison with historical reference	Wildlife habitat relationship models	Habitat suitability models	Individual home-range based models	Matrix population models	Stochastic single population models	Metapopulation models	Spatially population explicit	Genetically Effective Population Size	Incidence Function	Trend analysis	NHPS
Broad-scale Assessments												
Interior Columbia Basin Draft	0	0	0	0	0	0	0	0	0	0	0	0
Iinterior Columbia Basin Final and Supplemental	28	28	0	0	0	0	0	0	0	0	0	0
Sierra Nevada National Forest	0	270	2	3	0	0	0	0	0	0	0	601
Northwest Forest Plan	0	0	0	0	0	0	0	1	0	0	1	0
Tongass National Forest	1	0	1	0	0	0	0	0	0	0	0	0
Forest Planning Scale Assessments												
Northern Great Plains	0	0	0	0	0	0	1	0	0	0	0	0

Table 7.4 Methods for eliciting expert opinion in forest plans. Panel=Panel of experts; Individual = Individual expert or unstructured panel; OA=no decomposition (Overall assessment); BBN=Bayesian Belief Network Models; #C=Number of components; DISC=Panel discussion; Delphi*=modified Delphi; AVE=non weighted average of outcomes from experts; ND =No data available; Empty spaces indicate the information is not applicable in that record

	Structure	Number of panelists per panel	Number of panels	Decomposition	Process of reconciliation	Combination of information	Process and Content reviewed
Broad-scale Assessments							
Interior Columbia Basin Draft	Panel	3 to 5	9	OA	Delphi*	AVE	Y
Iinterior Columbia Basin Final and Supplemental				BBN			Y
Sierra Nevada National Forest	Individual			2 C			Y
Northwest Forest Plan	Panel	3 to 7	13	OA	Delphi*	AVE	Y
Tongass National Forest	Panel	4	8	OA	Delphi*	AVE	Y
Forest Planning Scale Assessments							
Northern Great Plains	Individual			OA			ND
White River National Forest	Individual			OA			ND
National Forests in Wisconsin and Minnnesota	Panel	107	10	7 C	DISC	ND	Y

Chapter 8. Expert Opinion

Expert opinion sometimes will be the only source of information for assessing the viability of many species of concern; and it often will be an important input, along with more objective data, to many of the analytical methods used for viability assessment. For example, in the Tongass forest plan viability assessments, data were sufficient to build a computer model for Sitka black-tailed deer, while expert panels were used to evaluate viability for 15 other species or species groups (Shaw 1999). A structured approach for eliciting expert opinion and incorporating it into a viability assessment is essential so that the results will be trustworthy and defensible. Appropriate structures for using expert opinion are available from the field of decision analysis, but these approaches are not widely known to resource biologists and managers and, consequently, some previous uses of expert opinion for viability assessment have been haphazard.

In this chapter, we review the steps for using expert opinion and illustrate them with some examples from viability assessment. We hope this review will encourage Forest Service users to follow a more structured approach. However, eliciting and using expert opinion is a complex assessment task in itself, and we urge those who are inexperienced with structured approaches to eliciting expert opinion to make use of experienced consultants whenever possible, both within and outside of the Forest Service. In addition, some useful references describing methods of eliciting expert opinion include: chapters 6 and 7 from Morgan and Henrion (1990); a book-length, very practically oriented treatment by Meyer and Booker (1991); chapters 10 and 11 in Goodwin and Wright's decision analysis book (1991); and an on-line paper on some psychological aspects of elicitation using biological examples (Anderson 1998).

Steps in Elicitation of Expert Opinion

The outline we give here follows that in Clemen's decision analysis text (2001, pp.321-6), supplemented by our own experience and opinions:

8.1.1 Background: Identify the variables for which expert opinions should be assessed

A review of information available from ongoing monitoring (Fig. 4.1, step 4) in relation to the information needed to carry out a viability assessment at the desired level of analysis (Fig. 4.1, step C) will show where objective data are lacking entirely, or where objective data should be supplemented with expert opinion (e.g., to update old information, or to modify information collected on another species or in another region for the species or region of interest). This step interacts with step 4 of the expert opinion process (below), creating a "mental model" of how the pieces of information that need to be collected should fit together to allow inferences about species viability.

8.1.2 Identification and Recruitment of Experts

This is a critical step, on which the credibility of the remainder of the process rests. The experts chosen must be perceived to be well-qualified by reason of both education and experience to make judgments about the variables being elicited by those who might be inclined to criticize the analysis. Mighton et al. (2000) defined an "expert" for their panels as someone who is recognized by her peers, whether academic or research scientists or other resource professionals, as having experience and knowledge of biology, habitat needs and ecological processes for a selected species or group.

Equally important, or perhaps even more important, the experts should have a reputation for objectivity and must be perceived by critics to be unbiased, in the sense of having no personal or professional stake in the outcome of the viability assessment. Sometimes this latter criterion is impossible to meet. Those who are knowledgeable enough to be experts may all have some sort of stake in the outcome of the analysis, including defending past research findings or positions taken. Often the best that can be done is to balance the biases that are represented among several experts from whom opinions are elicited. Structuring the elicitation process carefully according to the principles described below is one of the best protections against unknown effects of bias on the assessed variables. There are several potential types of biases. For example, motivational bias, occurs when an expert is more inclined to respond a particular way because of tacit, subjacent distrust of the agency, etc. Another is positional bias, where different experts or panelists might tend to want to represent a particular attitude, such as resource managers being inclined to think that environments or habitats "should" be managed.

As described in the following sections, methods for managing against bias include (1) breaking down the assessment problem into discrete and measurable components that can be easily compared with other data sources and (2) using Delphi or similar processes (Linstone and Turoff 1975). These processes reduce individual experts' dominance within groups, and provide unbiased methods for aggregating opinions. Observers also may by employed to evaluate the objectivity of expert panelists during the process, identifying instances where an expert's personal or organizational values appear to be overriding the technical basis of his opinions (Shaw 1999).

Identifying experts usually begins with a literature review and proceeds through a process of successive referrals (see Mighton et al. 2000). It is natural to start close to home, but analysts should weigh the benefits of consulting experts who are well acquainted with the local situation against potential charges that these local experts may be biased. It is customary to ask experts to suggest other experts. This generally is a good tactic, but be aware that experts are apt to recommend others who think similarly. To diversify the opinion that is being sampled, and possibly gain some important new insights, it may be prudent to ask experts explicitly to recommend someone who is likely to have a different opinion or experience from their own. Experts' perspectives are likely to have been influenced strongly by the circumstances and location of their own work, thus the process can benefit from tapping into these diverse experiences. Geographical variation may not matter for highly localized viability assessment, but can be critical to regional or species-wide evaluations. For more complex issues it may be valuable to seek experts from varied disciplines. For example, in the viability assessment for the Tongass forest plan revision, separate panels were convened for fishery biologists and for geomorphologists and hydrologists to consider sustainability of fish habitats and river channel conditions, respectively (Shaw 1999).

How many experts are needed for a particular assessment? There is no single answer to this question. Some things to consider are how many candidate experts there are, the level of scrutiny the assessment is likely to face, and the budget available for eliciting expert opinion. More experts are not automatically better, if they are all espousing the same opinions. If the experts are to work in a group, effective group size must be considered. Smaller groups (4-8 experts) can be less cumbersome to manage, while still representing key perspectives. Previous Forest Service panels have averaged five to six experts per taxonomic group, both for complex assessments such as the Northwest Forest Plan (FEMAT 1993) and for individual forest plans (Mighton et al. 2000). Larger groups may be subdivided around different tasks. Because not all experts will be available or willing to participate, the initial list of potential participants needs to be larger than the desired final group size. Mighton et al. (2000) recommend inviting two to three times more experts than needed and contacting them two to three months before convening a group meeting (longer for agency staff).

8.1.3 Motivating experts

Candidate experts may not be enthusiastic about providing their opinions in a structured and quantitative format. It is common for biologically trained scientists, and particularly those without management experience, to be very skeptical of the concept of expert opinion, particularly when it is expressed quantitatively. They may be reluctant to commit themselves to expressing any opinion at all on scientific parameters that they don't feel quite certain about.[8] It is up to the analyst to cajole them into participating (1) by explaining that decisions are likely to be made with or without the benefit of their input and (2) by expressing the hope that the expert will participate in improving the quality of

---

[8] Experts may, however, be more open to offering opinions on management programs or impacts that are presented in general terms. These opinions are not necessarily appropriate to the viability assessment task, as discussed below under Proper Use of Experts.

the information on which those decisions are based. Also, assure them structured procedures will be followed to guard against undue biases in the elicitation process and that uncertainty will be documented, carried through the analysis, and communicated to decision makers.

Both experts and the decision makers who use expert-based assessments should clearly understand that expert opinion provides informed risk judgments, which are not the same as rigorous probability estimates. Marcot et al. (1997) described the results from expert opinion viability assessment as “tentative working hypotheses” about species-habitat relationships and possible trends. Understanding this distinction should help experts overcome their reluctance to provide quantitative risk estimates.

Although experts may be glad to share their knowledge, fair compensation for the time and effort they invest is a helpful incentive. When the experts will be convened in a small group setting, they typically are motivated by the opportunity to interact with their peers. So when inviting experts to attend, explain who else will be involved (or invited) and how the format will allow for useful information sharing.

8.1.4 Creating a “mental model” for the chain of inference from elicited information to viability assessment

This is a vital, but much neglected, step. In the decision analysis literature, the process of creating a mental model for the chain of inference from elicited information to viability assessment is usually called “structuring and decomposition.” It is possible to ask an expert to render a judgment about the likelihood of extinction of a species or group of species for each of several forest management scenarios, but it is bad practice to do so. In many of the Forest Service’s attempts at large-scale viability assessment covering hundreds of species—the Northwest Forest Plan, Interior Columbia Basin Plan, and Tongass Forest Plan—experts have provided this kind of summary judgment in addition to other information (FEMAT 1993, Lehmkuhl 1997, Shaw 1999). Subsequent work for the Interior Columbia Basin and the Sierra Nevada regional plan has moved away from experts' summary judgments to explicitly decomposed assessments (Marcot et al. 1997, Marcot et al. *in press*, Raphael et al. *in press*).

Asking the expert to synthesize a whole chain of inferences intuitively, from forestry activity to species population effect, makes the assessment unnecessarily difficult, unavailable for scrutiny, and prone to error and bias. Instead, ask the expert to participate in breaking the overall judgment into its component parts, thus describing a "mental model" of the connection between actions taken and effects on species populations (e.g., effect of forest management on habitat vertical structure, effect of habitat structure on foraging behavior, effect of foraging behavior on vulnerability to predation, effect of different predation rates on total mortality). Such a model can be described verbally, mathematically or graphically, perhaps by a box and arrow diagram of population dynamics, perhaps by an influence diagram (Clemen 2001) or conceptual diagrams (Goodwin and Wright 1991). Such diagrams can represent the chain of inference an expert wants to follow and a framework for structuring the elicitation of the component influences.

Influence diagrams also can be quantified and used directly for viability analysis by creating decision trees (Goodwin and Wright 1991:Chapter 5) or Bayesian belief networks (Lee and Rieman 1997, Shepard et al. 1997). For example, in a recent assessment of terrestrial vertebrates in the Interior Columbia Basin, Marcot et al. (*in press*) and Raphael et al. (*in press*) developed Bayesian belief network models based, in part, on the judgment of experts. These models were used to rank the likelihood of different population outcomes resulting from projected habitat changes (through an environmental index subset of the model) and non-habitat influences (Figure 8.1). The initial model structures or causal webs were developed using information on species' key environmental correlates (factors) developed by expert panels (Marcot et al. 1997, Wisdom et al. 2000). Additional expert opinion, gathered through one-on-one conversation and unstructured group meetings, was used to establish conditional probabilities linking model nodes where empirical information was unavailable (Marcot et al. *in press*, R. Holthausen, pers. comm.). These models had distinct advantages compared with previous Forest Service use of expert panels to make judgments about the consequences of management alternatives: (1) they clearly disclosed linkages between habitat change and hypothesized response of species; (2) they could be easily rerun to test sensitivity to different assumptions, or to project effects of new alternatives; (3) model results included measures of uncertainty; and (4) results were spatially explicit (Raphael et al. *in press*). Creating more detailed viability models requires much more time than does asking experts for simple summary judgments, thus fewer species are likely to be evaluated and criteria for selection of species to evaluate (Chapter 5) become even more important.

Experts also can be asked to offer judgments about components of formal and informal models that others have developed. For example, expert opinion could be used to parameterize functions for density-dependence in vital rates in matrix projection models, to provide qualitative or quantitative input to rule-based protocols, or to provide functions relating habitat conditions to population consequences for habitat-based protocols.

An important part of the mental modeling step is ensuring that the items to be assessed have been defined in a clear and operational way, so that any expert could understand what s/he was being asked to make a judgment about, and so that all the experts will be responding to the same question. For example, instead of asking an expert to estimate the “viability” of species X if action Y is taken, ask the expert to estimate the likelihood that the population of species X in management area Z will decline below a specified number of individuals within the next 50 years due to a species management prescription. Do not ask an expert to assign categories such as “high,” “medium” and “low” risk of extinction without first defining unambiguously what these terms mean for the particular situation; otherwise, the assessment becomes some opaque combination of the expert’s judgment about species’ prospects and the expert’s own intuitive definitions of “high,” “medium” and “low,” rendering any analysis using these judgments difficult to interpret.

Unambiguous definitions may be difficult to achieve, especially when applied across diverse taxa and situations. For example, the original outcome definitions derived for the

Northwest Forest Plan assessments were revised between the first and second round of panels to increase their reliability through greater clarity and reduced scope (applied to more narrow circumstances) (FEMAT 1993). The same scale of five discrete outcomes developed by FEMAT (1993) later was used for the Interior Columbia Basin (Lehmkuhl et al. 1997, Marcot et al. 1997) and Tongass (Shaw 1999) assessments—again with revisions adapted between sequential panels and for different species groups—while Mighton et al. (2000) condensed the scale to three levels. These examples illustrate that outcome definitions can be clear and explicit for most cases, but "borderline" or atypical cases will continue to challenge participating experts and may require additional assessment.

Although conceptual models help break down the viability assessment into distinct component parts for which experts can make reasonable judgments, these models still should be limited to important parameters. Very large models, designed to represent every possible scenario, are counterproductive because they are too complex for experts to work with. Models that are too complex are more likely to obscure than facilitate understanding. Similarly, Shaw et al. (1999) recommend limiting the number of management alternatives and the scope of impacts considered at one time. In the Northwest Forest Plan assessment, the scope was narrowed to habitat after broader evaluation of population viability became too cumbersome (FEMAT 1993), and this focus on federally controlled habitats has carried into later assessments (e.g., Shaw 1999). The Interior Columbia Basin experts' assessment was divided into separate critiques focused first on federal habitats, and then on a more comprehensive cumulative effects analysis (including non-federal lands and non-habitat factors) (Lehmkuhl et al. 1997).

8.1.5 Providing a common pool of information

Providing the experts with thorough background information is one means of avoiding bias and error in expert opinion assessments. Providing pertinent information helps ensure that the experts work with common knowledge of existing conditions and proposed management actions. This reduces the number of assumptions that must be made by the experts, and allows them to focus on the portion of the assessment for which they have actual expertise. The following types of information would be helpful to experts assessing species implications of forest management activities (see Appendix 4 in Shaw 1999 for the full list of information given to experts on the Tongass forest panels). Both maps and tabular information should be provided where possible. Mighton et al. (2000) recommend giving this information to experts at least four weeks before meeting:

1) Current vegetation conditions
2) Past changes in vegetation
3) Complete descriptions of proposed management
4) Projections of future vegetation conditions, accompanied by the assumptions used in making the projections
5) Quantification of past, current, and projected future habitat, accompanied by the assumptions used in making the projection
6) Information on species range and occurrence, including past and present trends.
7) Information on species life history, particularly traits that indicate relative degree of vulnerability to environmental change and management actions.

Convening experts in a group setting, where feasible, also helps ensure that a common information base is used. In addition to sharing new information, face-to-face discussion helps clarify differences in knowledge as well as in interpretation or perspective. The point is not necessarily to reach consensus, but to assure that differences result truly from different opinions, not just knowledge gaps.

8.1.6 Probability assessment protocols and training.

To avoid introducing biases and error in assessing expert opinion, specific protocols for framing and administering the assessment questions must be used. These techniques are described in Morgan and Henrion (1990) and in Meyer and Booker (1991), as well as in decision analysis texts such as Clemen (2001, Ch. 8) and Goodwin and Wright (1991). These techniques often use “reference lotteries,” where experts are asked to consider hypothetical gambles as a means of articulating their judgments about uncertain events. Since these assessment techniques are unfamiliar to most biologically trained experts, some training of the expert is needed to be sure they understand the assessment task. It often is helpful to use everyday examples for the training (e.g., the probability it will rain tomorrow, given that it was cloudy today) so that the expert can focus on learning the assessment method rather than on the subject matter being assessed. This training process also gives the analyst an opportunity to demonstrate to the expert how potential biases and errors can be counteracted during the assessment, thus increasing the expert's confidence in the assessment task.

Although the structured questioning methods that are used in elicitation of expert opinion may seem tedious and convoluted to those unfamiliar with the process, it's very important to use these established protocols in order to avoid known, systematic errors in the way people (even experts) assess uncertain information intuitively (von Winterfeldt and Edwards 1986, Freudenburg 1992, Cleaves 1994). Anderson (1999) discusses elicitation from biological experts, noting that biological scientists are often inexperienced in making probability estimates, falling prey to mental shortcuts that lead to systematic errors. She suggests that analysts reframe questions of probability in terms of frequencies instead. For example, instead of eliciting a single event probability for a particular bird population (e.g., a decimal likelihood between 0 and 1.0 that the specific population will become extinct), ask the experts to consider the proportion of 100 equivalent populations (e.g., groups of <500 spectacled eiders nesting in eastern arctic Russia) that will become extinct within 10 years. Anderson's paper contains good explanations of the behavioral psychology of assessment of expert opinion, showing where faulty reasoning can compromise assessment results and, alternatively, how to structure the process to take advantage of human cognitive strengths.

Beginning with the Northwest Forest Plan, the Forest Service has elicited probability assessments by having experts allocate 100 "likelihood points" among possible future habitat or population outcomes (FEMAT 1993, Lehmkuhl et al. 1997, Shaw 1999, Mighton et al. 2000). The points represent the experts' degree of belief that the outcome will happen, and thus provide a measure of the experts' level of uncertainty. Spreading the points among alternative outcomes indicates the experts' personal and scientific uncertainty about the outcome.

8.1.7 Probability elicitation and verification.

In this step, the analyst uses the assessment techniques with which the expert is now familiar to elicit the expert's judgments about the items that were defined in step 4. This elicitation process is necessarily iterative, with the analyst asking some questions, and then often doing some analysis, and then asking some more questions, to ensure that the elicited information is as consistent and free of errors and biases as possible. Allowing too little time for this step is a common pitfall, especially when the expert or the analyst must travel some distance to interact. Attempting to carry out the training and elicitation steps remotely also is risky, although, with very careful advance preparation, it can be done (e.g., acid rain assessments, De Steiguer et al. 1990). Usually, it will be much better for the analyst and the expert to meet face to face. Another option is to conduct the elicitation during a group process, either with open discussion among experts or a structure that keeps expert opinions independent, but allows for feedback and information sharing (e.g., Delphi technique or modifications that do not retain anonymity for experts).

The results of this step are not merely the expert's estimates of the quantities being elicited, but also the chain of reasoning the expert used to arrive at these estimates. This chain of reasoning may be illuminated during group discussions, or illustrated by drawing or reviewing and editing an influence diagram. The analyst should probe the expert's thought processes, since these reasoning steps may be as informative as the resulting estimates (in a group process, experts similarly probe each other's thinking while the analyst directs the process and records the information). A record of the reasoning the experts used is very useful in the next step, when information from several experts must be combined.

Another elicitation step that may be important, depending on the situation, is eliciting the expert's degree of confidence in his own judgments or having him estimate the underlying uncertainty in the parameters at hand (e.g., fecundity tied to habitat quality). Confidence or uncertainty estimates can be as simple as a numerical range around an estimate (e.g., a low to high range or a triangular range with minimum, maximum, and most likely estimates) or the spread of likelihood votes as described earlier (FEMAT 1993). More complicated descriptions of uncertainty include "fuzzy sets" that allow for relatively straightforward, linear gradations of likelihood within a low to high range of outcomes (Todd and Burgman 1998), and probability distributions (expressed as a non-linear mathematical function or graph of likelihood across the full range of possible outcomes). Uncertainty in viability assessments may also be estimated indirectly by measuring variation among experts' judgments (Cleaves 1994).

8.1.8 Combining the judgments of several experts

The first option at this stage is to keep the judgments of different experts separate and repeat the analyses with each set of estimates to see if the resulting conclusions change. If the final conclusion is much the same over the range of estimates supplied by different experts, then managers can proceed with a higher degree of confidence in their actions.

If this approach isn't satisfactory and the judgments of different experts must be combined into a single analysis, there are several ways to proceed. One is to use the range of estimates for each component of the analysis (e.g., range of mortality rates for a given age class of a particular species) to conduct a sensitivity analysis, to see if variation within this range makes a difference in the conclusion about impacts of management actions. This is different from the first option because, instead of making a complete analysis using just the opinions of a single expert, the range of opinions of all experts consulted about a single variable are the input to the sensitivity analysis. The range of variation among expert opinions may also be treated as a rough estimate for uncertainty in the projections.

If the desire is to come up with single estimates for each component of the analysis by combining the judgments of different experts, the analyst must choose between some sort of averaging process versus an interactive process designed to help the experts reach a consensus among themselves. Straight averaging among estimates from different experts can be appropriate, as can a weighted average, where the judgments of different experts might receive more of less weight according to their perceived levels of expertise in particular areas. The experts themselves might provide self-ratings of expertise, or they might rate each other (treat this option with caution), or the analyst or decision maker might supply the weights on different experts. Of course, the assignment of weights as an expression of degree of confidence in the judgment of one expert relative to another is fraught with its own potential for bias and abuse, if the differences in expertise are not commonly acknowledged. In Forest Service panels that did not strive for consensus, summary outcome ratings have been averaged across panelists (FEMAT 1993, Lehmkuhl et al. 1997, Shaw 1999). FEMAT (1993) and Lehmkuhl et al. (1997) reviewed their panel results and subjectively revised the few cases that appeared to be internally inconsistent or unsupportable, while for the Tongass panels, results were not altered (Shaw 1999). Lehmkuhl et al. (1997) completed simple statistical summaries of the experts' estimates (means and standard deviations of categorical levels).

If an interactive, consensus-seeking process is desired, the experts may either interact face to face or remotely. Face to face consensus processes are expensive and time-consuming, but the opportunity for experts to discuss their reasoning personally has both advantages and disadvantages. Face to face conversation is easier and more fluent, and different avenues of thought can be followed as they come up. On the other hand, dominant personalities and personal animosities can inhibit free discussion and can stand in the way of consensus. Alternatively, groups may unwittingly fall into agreement ("group think") that implies scientific certitude, where in fact uncertainty is still substantial.

The experts can interact during any parts of step 4 through 7 of the elicitation process, jointly structuring the model of items to be elicited, receiving a common pool of background information, training to use the elicitation procedures, and providing estimates.

Such interaction provides an opportunity for experts to exchange information and develop a common basis from which to make their individual, or joint, judgments. In developing the Bayesian belief networks for the interior Columbia Basin assessment, experts were allowed to interact informally in working groups to provide probability estimates (Marcot et al. *in press*, Raphael et al. *in press*). While this unstructured group process runs the risks of biased or unbalanced results as described above, at least the product is a fully transparent model where the builders' assumptions can be seen and challenged.

Delphi processes, where experts interact anonymously without knowing whose estimates or lines of reasoning are whose, are one way to counteract difficulties in reaching consensus face to face (such as undue influence by dominant personalities) (Linsone and Turoff 1975). In Delphi processes, experts are asked to provide initial estimates of the quantities being elicited separately (either on paper or verbally in face to face interaction with the analyst or via phone or written surveys), and they are also asked to provide some reasoning to support their answers. The answers and accompanying reasoning are then provided to all the experts, without attributing answers and reasoning to individual experts. The experts then review all the answers and have an opportunity to revise their own estimates and provide additional reasoning. This process can then be repeated as many times as desired to approach a consensus.

Modified Delphi processes may not retain full anonymity for the experts, but do attempt to elicit the full range of individual knowledge and opinions. For example, experts may be convened in a room together, provided background information and instructions, and asked to answer questions independently, but then allowed discussion as a group with subsequent opportunity to revise their answers. The Forest Service used this approach for the Northwest forest plan (FEMAT 1993), interior Columbia Basin study (Lehmkuhl et al. 1997, Marcot et al. 1997), and Tongass forest plan (Shaw 1999) expert panels. If the group knows that consensus is not required, then it may be less important to maintain full anonymity in the discussion following the initial probability estimates. Not asking a group to reach consensus may also avoid violating the Federal Advisory Committee Act, which regulates how federal agencies convene advisory groups (also see panel processes in

FEMAT 1993, Marcot et al. 1997).[9]

8.1.9 Document the process

Each step of the elicitation process should be documented carefully, both in order to show that a credible analysis has been used and to enable others to scrutinize and modify the resulting estimates. Poorly documented assessments are nearly as useless as poorly executed assessments. It is inadequate to simply provide a list of the experts who were consulted. Documentation of the process should show how good practice was observed at each step. That's essential to establishing the credibility of the resulting information.

When properly elicited and documented, the information gained through expert opinion provides more than just the quantitative likelihood estimates or ratings. Forest Service examples include experts recording the factors influencing their judgments (via a prepared list; FEMAT 1993), listing the assumptions used in the analysis and its limitations (Lehmkuhl 1997), providing information for species-environment databases and detailed models (Marcot et al. 1997, *in press*, Raphael et al. *in press*), recommending management guidelines and conservation measures (FEMAT 1993, Shaw 1999), and estimating the reliability of information sources used in the analysis (Mighton et al. 2000).

Computers are helpful for recording the detailed steps in the viability analysis process, including preserving a record of individual assessments as in the modified Delphi process of Marcot et al. (1997). Software is available or can be created in a standard spreadsheet program for designing and quantifying decision trees (e.g., Supertree; McNamee and Celona 1989) and Bayesian belief networks (e.g., BayVAM; Lee and Rieman 1997, and Netica [Norsys, Inc.]). Electronic formats are especially useful for sensitivity analysis or revisions based on new information. During group processes it is also helpful to use computers to rapidly compile and distribute or project results for

[9] FACA considerations limited participation on expert panels for the Tongass forest plan to federal and state employees (Shaw 1999); otherwise, the Forest Service has convened expert panels without FACA constraints because the panelists were not asked or allowed to reach consensus (FEMAT 1993, Marcot et al. 1997).

discussion.

Proper Use of Experts

In addition to following a sound protocol for assessing expert opinion, such as the steps described above, making proper use of experts means asking them only for the kinds of information they are uniquely well-qualified to give, namely, their judgments about how management actions are likely to affect species well-being via a chain of interconnected dependencies. Do not ask experts questions like "Will species X be viable under management scenario Y?" unless the criteria for viability have been clearly defined. Without a prior definition for viability—which implies a social standard for species security—such questions require the expert to express opinions on both the effects of management scenario Y on species X and the level of security of species X that should be considered acceptable.

Assessing management effects is an appropriate function for experts, drawing on their superior knowledge of ecology and management. But asking scientific experts to define standards for species protection is wholly inappropriate; the values experts place on different levels of security for different species should not carry any greater weight than those of other participants in forest management in determining how effects on species will be integrated with effects on other resources in choosing a management alternative (Fig. 4.1, steps G, II and III). Therefore, do not ask questions of experts that inadvertently mix "facts" about effects on species with "values" about what levels of effects are acceptable to society (von Winterfeldt 1992, Shaw 1999). That said, the value judgments of experts, and others, will inevitably color the elicitation of expert opinion to some extent through the choice of species to analyze with different levels of scrutiny. Structuring the solicitation of expert opinion helps to keep the focus on topics in which the "experts" are truly expert; for example, biologists estimating bird population effects from specific tree harvesting regimes.

In structuring the viability assessment it may be useful to separate the likelihood that different environmental or management events will occur from the biological consequences of such events. Then different experts can provide opinions only on the parts of the problem—or influence diagram—where they have expertise. For example, forest economists could project land use or harvesting trends on private lands surrounding a National Forest while biologists assess the probable responses of species within the forest to such trends when combined with forest management plans. In forest planning, plan alternatives prescribe many future events, but other events such as bioregional land uses and environmental trends will be beyond Forest Service control. It should not be assumed that biologists or species experts are the best available source of information on those trends. Bayesian belief networks provide good examples of combining different information sources, including empirical data as well as expert opinion, into a viability assessment (see Lee and Rieman 1997, Marcot et al. 1997).

Repeatability of Expert Assessments

Should expert assessments be repeatable, both across experts and through time? Yes, in some respects; perhaps not in others. In general, experts with the same training and access to the same information will probably reach roughly similar conclusions. But, part of the point of consulting several experts is to hear a variety of opinions and learn something different from each, rather than just hearing the same story over and over again. Analysts and managers may be tempted to see such diversity of opinion as an annoying impediment to smooth decision-making, but they should be encouraged to see it as an opportunity to learn more about the systems they must manage, and as guidance for gathering additional information (Fig. 4.1, step E). Differences of opinion may reflect true stochasticity in biological systems as observed by scientists working in different locations and times, instead of simply differences of interpretation or bias. Thus, diversity of opinion may reflect rangewide variation in species ecology.

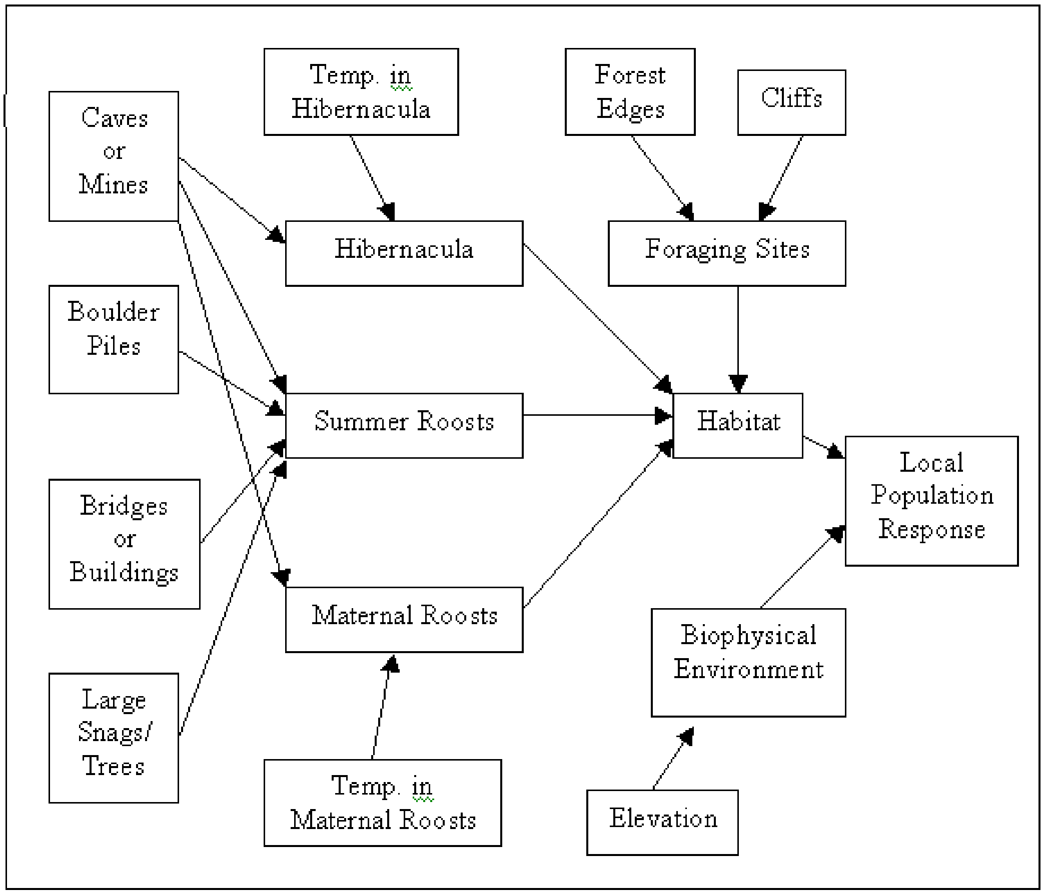
Over the long run, dissecting the source of disagreements among experts will improve the knowledge base for sound decision making far more surely than seeking out a false congruence of opinion. And, experts should be expected to change their opinions as they learn more about the species and ecosystems they are examining, perhaps through interactions with other experts during an elicitation process. If experts did not respond to new information in an open-minded way, we would be disinclined to acknowledge them as experts.

Training and Use of Outside Assistance

Few Forest Service biologists or managers at any level have received training in structured methods of assessing expert opinion. Because a failure to use properly structured methods will compromise both the quality and the credibility of the assessed information, it’s important to remedy this gap both through training within the Forest Service and through use of outside assistance. Although it’s a very good idea to train forest-level biologists and managers in the rudiments of methods for assessing expert opinion, it’s probably not reasonable to expect them to acquire sufficient expertise to carry out such assessments without assistance. The Forest Service could consider training existing personnel, or hiring additional personnel who are already well-trained in expert opinion assessment, to provide the analytical expertise needed for credible assessments. Another option would be to hire outside consultants from the field of decision analysis, being careful to choose people who are able to communicate effectively with biologically trained scientists and managers. Recent population viability assessments for National Forests in Wisconsin and Minnesota, for example, were facilitated by biologists from the Conservation Breeding Specialist Group who have extensive experience with expert panels, although they are not formal decision analysts.

Computer software is available for structuring, compiling, and evaluating expert opinion (e.g., Clemen 2001). These tools can greatly facilitate the expert opinion process. Again, however, we caution against using a tool just because it is available and easy to learn without first understanding the proper use of structured methods to elicit expert opinion.

Figure 8.1 Influence diagram (causal web) for the Townsend big-eared bat (*Corynorhinus townsendii*) corresponding to the Bayesian Belief Network in Figure 2 of Marcot et al. *in press.*



Chapter 9. Collecting Information to Update Assessments Over Time

Collecting data to update assessments

The process of assessing and managing for species viability is fraught with uncertainty. Sources of uncertainty include the inherent stochasticity of ecological processes, incomplete understanding of species ecological requirements, incomplete understanding of interactions between species, inability to accurately forecast future human activities, and incomplete understanding of the effects of human activities on ecosystems. Because of the high level of uncertainty, an ongoing monitoring program is a critical component of the overall management-decision framework (Figure 4.1). In fact, once a management plan is in place, monitoring could be viewed as the central activity upon which all other management actions are based. Continuing to manage without an ongoing monitoring program is analogous to flying a plane with no instruments. There is no way to know your speed, how much fuel you have remaining, the altitude at which you are flying, or the condition of your engine. You might, through good fortune, reach your destination without mishap. More likely, you will run out of fuel, or be forced to turn back, or suffer a calamity. Even if you are lucky enough to reach your destination safely, it is likely that you will have sacrificed efficiency by making unnecessary fuel stops or flying at inappropriate speeds or altitudes.

Despite general agreement on the need for monitoring in the face of uncertainty, there are few examples of successful monitoring of U.S. Forest Service management for species viability (Morrison and Marcot 1995; Noon et al. 1999; Noss and Cooperrider 1994). Where monitoring programs are in place, they generally focus on individual high profile species (e.g., northern spotted owls, red-cockaded woodpeckers, salmon), and do not provide the information needed to assess overall success in managing for viability. Recently, however, some effort has been made to establish guidelines for monitoring particular taxonomic groups (e.g., Elzinga et al. 1998). Monitoring must be considered as an integral component of the overall process used by the Forest Service to manage for species viability. The importance of monitoring is heightened in situations where little existing information is available on which to base management decisions. Such decisions may of necessity be based on expert judgment and general conservation principles, but should be accompanied by commitments to gather better information through monitoring.

While managing for species viability under NFMA is a Forest Service responsibility, monitoring should be coordinated across boundaries and jurisdictions. Monitoring will be more effective and more feasible if it encompasses the entire landscape, and if other agencies and cooperators share in the planning, execution, and budgetary support of monitoring programs.

9.1.1 Goals

In the NFMA regulations guiding planning on National Forests, management for native species viability is one of the tools used to maintain the diversity of plant and animal species. The objectives for monitoring should reflect both the specific goal of species viability and the larger goal of providing for diversity. The following are overall goals of monitoring species viability.

Goal 1. Reduce uncertainty surrounding the effects of ongoing management activities on species and the ecological conditions that support them. A primary goal of monitoring by the Forest Service must be to determine how activities conducted by the agency affect species and ecological conditions. This improved understanding will result in the ability to more reliably forecast management effects. This goal may be most effectively achieved by investigating assumptions made about the effects of management on both species populations and the ecological conditions that support them. The construction of ecological models is a useful way to make the statement of assumptions explicit, and to determine the sensitivity of overall projections to underlying assumptions. Monitoring selected indicators on areas where management treatments are applied and untreated control areas provides for rigorous investigation of assumptions (Walters and Holling 1990).

Goal 2. Provide a more complete understanding on which to base future management actions. Understanding the effects of ongoing management actions is a necessary component of a monitoring program, but alone may be insufficient to allow significant improvement in management. Gathering information that can lead to improved management requires active experimentation to determine effects of possible management actions that differ from the ongoing actions. This is termed active adaptive management (Bormann et al. 1999; Committee of Scientists 1999). Active adaptive management may also focus on the effects of management on sites with ecological conditions that are not currently prevalent on the landscape, but might become more prevalent in the future. For example, active adaptive management might be used to determine effects of possible management actions on areas that are heavily invaded by exotic species.

Goal 3. Determine the overall status of species and the ecological conditions that support them over time. The status of species may be affected by many factors in addition to Forest Service actions. Determination of species status provides additional understanding of overall species diversity, which species should be considered at-risk, the level of risk associated with individual species, and the contribution of Forest Service lands and activities to overall species status. As part of this goal, we should determine whether or not all species that ought to be of concern were identified in the planning process. Ecological modeling suggests that some common, competitively dominant species may be at risk from even moderate habitat loss in patchy landscapes (Kareiva and Wennergren 1995; Tilman et al. 1994), but that there is a time lag between habitat loss and population decline of these species. Addressing this area of uncertainty requires implementation of a broad based monitoring effort that extends beyond the species-at-risk identified in the Forest Plan.

9.1.2 Monitoring based on ecological theory

Monitoring programs grounded in ecological theory should produce defensible and useable information. Noon et al. (1999) propose 6 steps for development of a scientifically-based monitoring program: specify goals, identify stressors, develop conceptual model, select indicators, establish sampling design, and define response criteria. Use of this conceptual framework should result in monitoring programs that are more cost-effective, more informative about ecological sustainability, and clearly linked to the decision-making process. Information derived from such monitoring programs can be used to continually update and improve the assessments (Figure 4.1) that are used in making and refining decisions.

9.1.3 Priority setting for effective use of monitoring resources

There is one unavoidable truth associated with monitoring – we will never have adequate resources to conduct all monitoring that might be appropriate. Inevitably, we must set priorities and implement monitoring for only a subset of the ecosystem components and processes that are being manipulated by management. The following concepts should be helpful in priority setting.

(1) Use focal species

A process for selecting focal species for the purposes of viability assessment is described in Chapter 6. Within the context of managing for viability, the purpose of selecting focal species is to simplify the task of planning and managing for the viability of hundreds or thousands of plant and animal species on a planning area. In this context, focal species are selected because they represent ecological conditions needed to support some larger set of species. Because they are intended to simplify the task of managing for viability, consideration of focal species should play a prominent role in setting priorities for monitoring. A large portion of an overall monitoring effort should be devoted to the focal species that are selected for a planning area.

(2) Identify key areas of uncertainty

As noted earlier, uncertainty about the projected future status of a species may stem from several sources: the inherent stochasticity of ecological processes, incomplete understanding of species ecological requirements, incomplete understanding of interactions between species, inability to accurately forecast future human activities, and incomplete understanding of the effects of human activities on ecosystems. Specific sources of uncertainty should be identified for both the systems being managed and the species affected by that management. Levels of uncertainty should be estimated as part of the assessments of management context and viability. Uncertainty can then be used in prioritizing monitoring activities. For example, those species deemed to be at risk, and for which future outcomes are highly uncertain, would have high priority for monitoring. For many species, such high levels of uncertainty may stem simply from lack of information, so increasing knowledge about species would become a high priority. High levels of uncertainty may also be associated with entire ecosystems. For example, the future status of high elevation ecosystems may be uncertain because of global warming, so monitoring of species associated with those systems could be high priority.

(3) Identify key stressors

Monitoring of stressors simultaneously with monitoring of response parameters allows investigation of correlation. For this type of correlative investigation to be most useful, key stressors must be identified. In some cases, this is quite straightforward. For example, the key stressor for species associated with late-successional forests in the Pacific Northwest was loss of those forests due to harvest (Noon et al. 1999). Stressors to be monitored in this situation could be limited to harvest activities and the resulting amount, distribution, and quality of late-successional forests. In other cases, the relative importance of various stressors may be less clear. Again in the Pacific Northwest, an array of stressors negatively impacts populations of salmon. These include water impoundment, mortality associated with turbines, reduced water quality, and competition with non-native stocks. Here, all potential stressors may have to be monitored in order to begin determining the relative contribution of each to the population status of salmon. Monitoring of several population response variables may be necessary as there may not be a single population response variable associated with all of these stressors.

(4) Determine the degree of risk to species

The degree of risk may be one of the criteria used to prioritize species for monitoring. Criteria for determining degree of risk may include federal listing as threatened or endangered, or ranking based on criteria such as those established by the IUCN-The World Conservation Union (formerly International Union for the Conservation of Nature) or the state Heritage Programs. We recommend the use of ABI/Heritage Global Ranks as common approach to identifying species at risk (see Chapter 5 for additional discussion). These risk rankings should be supplemented locally through the application of expert opinion (Todd and Burgman 1998; U.S. Forest Service 2000) as the established protocols do not necessarily reflect risk at the scale of a National Forest.

(5) Identify species contribution to hypothesis testing

Monitoring that investigates assumptions about the relationships among management activities, species, and the environmental conditions that support species may be designed around specific hypotheses. The characteristics of some species may lend themselves to testing these hypotheses, making monitoring of those species a priority (McLaren et al. 1998). For example, Aberts squirrels, which inhabit closed canopy forests, may be useful for testing hypotheses about the need for retaining high levels of canopy coverage in fire treatments of ponderosa pine forests in the southwest. In a different application of this criterion, focal species may be purported to represent the ecological requirements of larger species groups. Such hypotheses should be tested through sampling of both the focal species and a selection of species, or ecological conditions, that it is being used to represent.

(6) Determine feasibility, ease, and cost effectiveness of sampling

The following characteristics contribute to the ability to reliably sample a species: 1) the species is relatively easy to detect, 2) sampling techniques exist, and 3) the species is distributed in a way that allows statistically valid sampling. Cost effectiveness of sampling is influenced by detectability, weather conditions during the recommended sampling period, remoteness of species location, operational difficulty of sampling technique, and stochasticity of population levels. Where other considerations are equal, priority for monitoring should be given to species for which reliable sampling is both feasible and cost effective.

9.1.4 Data collection designs and techniques

Because monitoring will always be constrained by limited resources, programs must focus on efficient data collection for species at risk with results directly linked to decision-making. To be useful, data must be appropriate for the specific viability assessment techniques used and directly applicable for updating assessments and deciding whether management actions need to change. A monitoring framework that supports management decision-making will anticipate information needs and continually or periodically gather relevant information. The following issues need to be considered carefully in designing an efficient program for monitoring species at risk and focal species and their habitats.

(1) Data quality

In a management-monitoring program the first premise of data quality is that data must be appropriate for their intended use in the decision process. For valid inference about management effects or environmental trends, data collection and analysis must also adhere to fundamental statistical principles of sampling and estimation. Standards for data reliability, precision and statistical power must be addressed in the data collection design (Marcot 1998).

Reliable data discriminate true effects or “signals” of environmental change from background “noise” - either small, unimportant changes or random environmental variability that does not represent a permanent effect or change. Data reliability is typically measured in classical statistics with *p* values estimating the likelihood of obtaining the data if in fact no effect or change had occurred—low *p* values indicate that the result is likely quite real or “significant.” All ecological field data include variation between samples, due to natural variation in nature and sampling problems. Precise data are comparatively less variable or more tightly defined; for example, an estimate of 100-120 birds is far more precise than an estimate of 10-1,000 birds in a planning area. Finally, statistical power (1 - β) is the likelihood of detecting effects or trends of concern (e.g., a decline below 100 nesting pairs) at a given significance level (e.g., $p = 0.05$). It can only be estimated in relation to a desired minimum level of significance or the effect size selected for biological and management significance (e.g., Reed and Blaustein 1997). Power and significance are reciprocal. Power is the probability of rejecting the null hypothesis when it is false. Power goes up as the significance level (or *p* value classifying an effect as significant) goes down. Power and significance can only be raised simultaneously by increasing sample size and hence, estimate precision. Power is especially critical in effective monitoring programs for at-risk species because failing to detect true declines in time to take corrective action reduces viability and in the extreme leads to extinction (Taylor et al. 1993).

Data quality objectives are not neutral scientific criteria; desired data standards depend upon decision-making criteria, such as the extent of reduction in a focal species' range that would trigger proposals for revised management. Data quality standards must also be developed with knowledge of data collection limitations and trade-offs in potential estimation errors (e.g., a Type I or $\propto$-error projecting a false trend or Type II or β-error failing to detect a true trend). Through monitoring protocols, managers must set standards for minimum acceptable data reliability, precision and power and how these objectives will be balanced. A manager adhering to the "precautionary principle" (Raffensperger and Tichner 1999)—that is, erring toward protecting species despite uncertainty about effects or trends—will have a high standard for data power. For example, managers could specify that sampling be intensive enough to provide at least an 80 percent likelihood (*power with* $\beta = .20$) of detecting five percent or larger (*precision*) declining trends in an index to nesting songbird density with at least 90 percent reliability (*statistical significance with* $\propto = .10$). Desired sampling intensity, or frequency, distribution and duration, should be derived from these objectives as well as logistical and budgetary constraints.

In most cases, it will be difficult (costly, logistically demanding) if not impossible to gather data of sufficiently high standards to meet expectations derived from more tractable (less uncertain and complex) realms of science and public policy. Managers and monitoring program scientists must jointly address the challenges of setting high but operational data collection standards. These standards need not be static—regular review and evaluation of monitoring results should lead to adjustments in sampling design, improving the adequacy and suitability of monitoring data.

For an example of how a Federal agency has established standards for data reliability and power in species monitoring programs, see Taylor et al.'s (2000) description of the 1994 amendments to the Marine Mammal Protection Act. Under these amendments, the National Marine Fisheries Service limits killing of marine mammals to conservative (protective), default thresholds unless statistically valid, population-specific data demonstrate that greater take will not cause declines below management targets. This approach provides great motivation for effective monitoring and data collection—because new data are likely to increase manager's flexibility to allow take. Further, the scheme fosters greater monitoring effectiveness because decision criteria are based on population parameters that agency biologists knew they could estimate fairly well (Taylor et al. 2000).

(2) Scale

A critical concern in data collection design is determining the appropriate geographical scales and scope to match both species' distributions and management actions (Marcot 1998). A range of geographical scales will be represented across different assessments, reflecting the variety of habitats and species groups that need to be monitored. For example, assessment scales can range from local sites for a rare fen plant community, to landscapes for breeding goshawks, to bioregions for a grizzly bear population. In many cases the appropriate monitoring scale will cross administrative boundaries. It will be crucial to coordinate both across these boundaries and between inventory and monitoring programs (forest inventory and analysis, breeding bird surveys, etc). Except for narrowly distributed species and endemics, data collection and evaluation will be more efficient and reliable at bioregional scales. Thus, an effective monitoring program will require careful coordination across programs and agencies, including with non-agency stakeholders.

Attention must also be given to coordinating the time scales or monitoring duration required to gather adequate data and the time frames mandated for management decision making. For many desired data, it may not be possible to match these time scales, especially for project-level or short time frame management decisions (compared with long-range forest plan cycles). Selection of species for monitoring should favor those for which timely data are more likely to be obtained, while monitoring programs must also be maintained for adequately long time scales to provide requisite data.

(3) Repeated measures versus snapshot data

Monitoring must be designed to continue over time because plant and animal populations and communities are always changing. To detect persistent trends, sampling must be repeated over time with sufficient frequency (sample size) to demonstrate changes in abundance or distribution (Caughley and Gunn 1996). The minimal sampling frequency depends upon the desired minimum statistical power or, as described previously, the likelihood of detecting a trend that is truly occurring. To test cause and effect relationships, monitoring must follow an experimental design with repeated measures (replicate samples of the same environment) under controlled experimental treatments or conditions with parallel control or no-treatment measures. These samples may be repeated across space or time, depending on the specific case. Without controlled experiments, monitoring can only suggest correlations between management actions, or environmental stressors, and population effects (Marcot 1998). In many cases controlled replicates may not be feasible, especially for very rare species or where effects develop over long periods. In these cases, less rigorous inference can still be sought with alternative approaches including retrospective and observational analysis (Marcot 1998).

(4) Inference from indirect measures

In many cases it will not be feasible to measure a population directly, however indirect measures may provide adequate indices to population status. For example, species with tight links to a vegetation type or features (e.g., snags) may be monitored by sampling trends in ecological conditions. Preferably, the link between populations and the indirect measures has been established through field-testing. The links may be either cause and effect relationships or a strong history of correlation that gives a reliable index to population trend. In other cases, the best information available to select indirect measures may be conceptual models describing the logical associations between specific environmental variables and population dynamics, based on professional judgment and selective data. An important part of monitoring programs that use indirect measures is testing to validate such models.

Most forest plan effectiveness monitoring focuses on the status and trends of habitat types (Noon et al. 1999). By definition, focal species' trends are indicative of the functioning of the larger ecological system they inhabit (Committee of Scientists 1999) and hence, may provide an indirect measure for the status of a group of associated species. Other indirect measures may include observation of related species or situations, or even expert opinion (Marcot 1998). However, both to confirm the validity of species-habitat models and to assess the status of species that are not clearly tied to habitats or sensitive species, viability monitoring will need to include direct population data in some cases.

(5) Balancing and integrating habitat and population data collection

Categories of data relevant to viability monitoring include population abundance, distribution, and demography, as well as habitat status and trends, and other measures of environmental stressors or threats. Because forest management predominantly affects vegetation and the Forest Service is not directly responsible for managing wildlife populations, habitat data are a primary component of Forest Service viability assessment. For focal species, it is also important to monitor the reliability of landscape-process-species relationship models to confirm that the species truly represents trends in an ecosystem.

To determine whether population trends in sensitive species are due to management actions, data must be collected simultaneously on the status and trends of managed habitats as well as other environmental conditions or anthropogenic factors (weather, disease, harvest, etc.) outside forest management control. The integration of habitat, environment and population information is summarized in conceptual models of cause-effect relationships. In a conceptual model, key environmental variables and species life history traits or population variables are drawn and linked with arrows showing how they influence each other. These models should guide the choice of elements to be monitored, with emphasis on the variables known to be highly influential or on the least certain links and relationships in the causal web. For example, see James et al.'s (1997) "envirogram" for effects of forest fire history on Red-Cockaded Woodpecker populations. A quantified conceptual model or influence diagram (Howard and Matheson 1981) is a critical component of a decision analysis framework for monitoring (Lee and Bradshaw 1998) as well as adaptive management approaches (Walters and Holling 1990) (for more information on conceptual models see the section on problem structuring and decomposition in Chapter 8).

9.1.5 Overcoming institutional barriers

Historically, monitoring programs on federal lands have suffered from inadequate funding, resources and administrative commitment (Mulder et al. 1999). Chronic under-commitment to monitoring is due partly to institutional barriers such as a bias toward immediate actions over the uncertain, future benefits of long-term data collection, or concern that new information is likely to raise new "problems" rather than help managers. For sensitive or listed species, in particular, the risk to the status quo posed by monitoring is that it will reveal more species occurrence sites or a greater threat to the species than previously assumed, requiring special protection measures. Thus monitoring may be seen as reducing rather than enhancing manager's choices or options for management. An effective monitoring program needs to recognize and overcome these inherent barriers.

9.1.5 Institutional Motivation

Inducements begin with ensuring that the entire monitoring program is forthrightly geared to solving management questions. Managers are a monitoring program's primary stakeholders. The links between information gathered in monitoring and management objectives should be direct, through an explicit decision process. Gathering more data should lead to progressively more defensible decisions over time. Lee and Bradshaw (1998) promote a rigorous framework tying monitoring to decision analysis in the Interior Columbia Basin Ecosystem Management Project.

By emphasizing ongoing or repetitive data gathering, with results "always ready" to support changes in management direction, managers should feel that today's investment in monitoring is not a gamble on the distant future, but pays practical, useful returns in the near future. For example, the Northwest Forest Plan called for site surveys and mitigation measures for more than 400 species, including invertebrates, nonvascular plants, fungi, and vertebrates, that were considered to be at risk. Implementation of the surveys revealed that roughly 70 of these species were more common than initially estimated, so requirements for those species were eliminated (USDA and USDI 2000).

Procedurally, managers should be involved in monitoring program development and ongoing reviews so they fully understand the potential, purposes and limitations of the data collection program (Lee and Bradshaw 1998, Sit and Taylor 1999). Engaged participation produces "buy in" to both the need for monitoring and the particular approaches used. Monitoring should be seen as fully integrated with management decision-making, not a separate domain strictly for non-manager scientists or technicians. For example, make sure managers understand and support the process for selecting focal and at-risk species and related indicators. Conceptual models, developed early in the monitoring planning process, clearly demonstrate current understanding of the links between management actions and effects. Through their involvement in the monitoring program—from developing conceptual models to sampling intensity schemes—managers can help assure that the results from monitoring will be truly relevant in a decision context. They need to concur with the monitoring focus and the frequency and method for transferring new, data-based information. For example, monitoring can cover a blend of time and spatial scales, so that at least some benefits are apparent in the manager's "here and now."

Managers, based on agency mandates and public and scientific input, decide how much change in a population or habitat is needed to trigger a change in management actions. Clearly established decision standards are key to developing cost- and labor-efficient and valid sampling protocols. Most likely, an iterative process will be needed back and forth between managers' needs or desired standards and the budgetary and logistical realities of data collection. So it is important both during planning and throughout implementation to address with managers the level of data quality that will be needed to identify whether management decision thresholds or triggers have been crossed—and to help them develop decision criteria for which adequate, timely data can be collected. Managers should see that the monitoring program is cost effective compared with alternatives, including failing to collect adequate data.

Another inducement to securing support for monitoring programs is to make clear the benefits of early detection of adverse management effects or environmental trends to providing flexibility in management choices. The monitoring program should be designed to provide sufficient advance warning so that management can respond before a crisis in species viability pre-empts or fully upsets other management priorities. If trends are not detected until they are severe, management options are going to be highly restricted (Caughley and Gunn 1996). In addition, the cost of remediation will certainly outweigh the direct costs of preventing severe species declines. It may be useful for managers to consider the trade-offs between the expense of monitoring and the potential future costs of restoring species viability if a negative trend were not detected until well into the future because we failed to monitor adequately.

Through their participation in the monitoring program, managers should see the appeal of routinely gathering good information—timely information should provide more choices in how to respond to potential population declines or threats. A caveat, however, is that early detection is always constrained by the statistical difficulty of discriminating true, deterministic declines or threats from non-threatening stochastic variation in species' status. Again, managers need to understand the inter-relationship between their decision-making needs and scientists' ability to gather reliable data or make valid inferences from data.

A final suggestion for inducing institutional commitment to monitoring is to link Forest Service programs collaboratively with other agency programs and across boundaries to activities outside forests. Especially for long-term monitoring efforts, external commitments may help bind forest administrators to a monitoring program. Collaboration should also provide more data and more cost-effective information to the forest. In general, conducting viability analysis and attendant monitoring at bioregional rather than individual forest scales will be most cost effective and robust.

9.2 Basic elements of a monitoring program

At the beginning of this chapter, the overall goals for monitoring species viability are presented. The following four elements would accomplish those goals, and we propose that they be considered standard components of a monitoring program focused on species viability. These elements need to be integrated and prioritized into a unified conceptual monitoring strategy (Lee and Bradshaw 1998), and that strategy should be coordinated across National Forests and with other agencies and cooperators. Such a coordinated strategy should provide the accurate assessments of resource status and management consequences necessary to inform and improve decision-making and motivate managers to support the program.

1. Monitoring the status and trends of ecological conditions needed to support focal species and selected species-at-risk. Priority for monitoring ecological conditions should be given to A) focal species that are chosen because their ecological requirements are thought to reasonably represent ecological requirements of some

larger set of species, and B) additional species-at-risk for which there is or appears to be reasonable correlation between ecological conditions and population.

2. Monitoring population status for selected focal species and selected species-at-risk. For some species, monitoring of population status may be necessary. Population monitoring may be mandated for some species with special status, such as those that are listed under the Endangered Species Act. For other species, monitoring of population status may be key to the effectiveness of adaptive management. Such species would include those for which population changes are not strongly linked to habitat. For example, rare plant species may often be candidates for actual population monitoring as their habitat relationships are not well understood, so it is not possible to make reliable population inferences from habitat. Other candidates for actual population monitoring would include species for which identified changes in abundance, spatial distribution, or any demographic parameter would trigger a review of management. For example, monitoring of fish survival at dams may be necessary to allow appropriate adjustment of dam operations.

3. Investigating assumptions about the effect of management on ecological conditions needed to support focal species and species-at-risk, and on populations of those species. Monitoring experiments, targeted at specific ecological conditions, can be used to investigate these assumptions. For example, Saab and Dudley (1998) investigated changes in snag levels and cavity-nesting bird response on burned areas treated with different levels of salvage logging in southwestern Idaho. Priority for these investigations should be given to focal species that are chosen because their ecological requirements are thought to reasonably represent ecological requirements of some larger set of species

4. Monitoring the presence/absence of a suite of species in order to 1) obtain better baseline information on the distribution of species, 2) improve knowledge of habitat relationships (Carroll et al. 1999), and 3) determine whether there are unexpected changes in habitats or populations for species that were not identified to be of concern

during the planning process. Using a grid sampling design may be an effective way to accomplish this objective (USDA Forest Service 2000). A grid design with a starting point that varies randomly from year to year may reduce problems associated with impacts to permanent plots (Guerrant 1998).

Chapter 10. Summary and Conclusions

Viability assessment is an essential component of ongoing forest management and forest planning processes. A variety of methods can and should be incorporated into viability assessments. Deciding which species require individual assessments, and which approach or approaches are most appropriate represent significant challenges for the Forest Service. Below we summarize five general recommendations based on the material provided in this report.

Adopt a Systematic and Consistent Approach to Identifying Species at Risk

Currently, the processes used to identify the set of species at risk that require viability assessments vary from one planning region to the next. To enhance the efficiency of the forest planning process, and avoid unnecessary duplication of effort among forests and regions, the Forest Service should adopt a systematic and consistent approach to identifying species at risk on National Forest lands. After considering both scientific credibility and practicality of application, we recommend that all species in the planning area of interest with ABI G ranks of 1-3 be considered as an initial set of species at risk. This set should be further refined based on species' distributions, relative to the scale of the planning area, and other factors discussed in Chapters 5 and 6.

Utilize Broad-Scale and Quantitative Analyses Where Possible

Where possible, broad-scale viability assessments, at the scale of several National Forests should be encouraged. In general, a method of formal mathematical modeling is often preferable to the use of more subjective methods. The potential for developing formal models is likely to be limited, however, to only 10-30 of the 50-300 species at risk that are assessed as part of the management planning process. It therefore seems inevitable that the Forest Service will rely on some form of expert opinion to assess viability for the majority of affected species.

Document the Assessment Process

Regardless of whether viability assessments rely on formal mathematical models, expert opinion, or a combination of the two, we cannot emphasize strongly enough the importance of carefully documenting each step of the assessment process. This is critical, both to demonstrate that a credible analysis has been carried out, and to enable others to scrutinize and modify the assessment results. We reviewed thousands of pages of Forest Service documents for our assessment, and in most cases, we found it impossible to reconstruct the viability assessment process that was employed on the basis of the documents alone. For both the regional-scale and individual forest-scale assessments we examined, personal consultation with individuals who were directly involved with the assessment process often was the only way we were able to ascertain what methods were used in the viability assessment, why they were chosen, and/or what role any or all of these methods played in the process of evaluating management alternatives.

Use Structured, Credible and Repeatable Approaches for Eliciting, Interpreting, and Using Expert Opinion

Viability assessments always will involve some degree of expert opinion and judgment. The extent of expert opinion ranges from expert judgment regarding the appropriateness, and/or choice of a quantitative modeling approach, expert interpretation of empirical data to parameterize quantitative models, to expert interpretation of model output, or at the other extreme, expert opinion as the primary or even the sole method for viability assessments. Because available data typically are limited, most viability assessments will depend primarily on expert opinion. It therefore is essential that Forest Service staff responsible for conducting viability assessments have adequate training in structured methods for eliciting and assessing expert opinion. When suitable personnel are unavailable, the Forest Service may need to employ experienced consultants with appropriate training in the academic discipline of decision theory.

Make Uncertainty and its Implications Explicit

Regardless of the method used, all viability assessments are laced with uncertainty, stemming from the inherent stochasticity of ecological processes, incomplete understanding of species ecological requirements and species interactions, limited ability to accurately forecast future human activities, and incomplete understanding of the effects of human activities on ecological systems. Often, sufficient data will not be available to support the desired level of analysis for a viability assessment. This requires a choice as to whether to delay management decisions until suitable data can be gathered, or proceeding with a less suitable assessment than desired. Where the choice is to proceed, the associated risk of error must be estimated and communicated clearly to appropriate decision-makers.

Practical Recommendations for Conducting Viability Assessments

We firmly believe that meaningful improvement in viability assessment processes must be coordinated at the Regional and National levels of the Forest Service organization. We also recognize, however, that individual Forest biologists have an ongoing responsibility to complete assessments as part of Forest Plan revisions and project analyses. We offer the following suggestions to individual biologists who must undertake such assessments with little guidance or assistance.

- **Determine the geographic and biological or taxonomic scope for your assessment and management.** Combining several National Forests in a single effort to assess and manage for viability may be very helpful, particularly where coordinated, large-scale assessments have not been completed.

- **Enlist help from local experts**, including Forest Service scientists, biologists from adjacent National Forests and other federal agencies (e.g., BLM), state agency biologists, university personnel, and other local biologists.

- **Develop a complete species list for the taxonomic groups for which the Forest**

Service has legal responsibility within the area being assessed. If an accurate species list does not currently exist, use existing sources (e.g., automated Forest Service data bases, Natural Heritage data bases) to develop a draft list, and then enlist species experts to help refine it.

- **Use the protocols and systematic screening processes described in Chapters 5 and 6 of this report to identify species-at-risk and select focal species.** The objective should be to identify the set of species that adequately represents viability concerns for the planning area. A structured process, based on solid information, is necessary to make this a credible step.

- **Do a thorough search for information on the species identified in the above step.** First start with any broader scale assessments that have been done for the species. If good, recent literature reviews are not available for the species, do them now. Coordinate this activity with the Regional Office, and consider using contracts to ease the workload. Remember, if your plan is legally challenged, it is critical that you be able to demonstrate that you have used the available scientific information on species.

- **Work with planners to ensure that plan alternatives address the ecological conditions needed to support species at risk.** The process of assessing viability is much simpler if there is appropriate management direction to consider in the assessment.

- **Using the information in Chapter 7, determine the appropriate type of assessment to be done for each species at risk.** Recognize that you often will have to use expert opinion as an assessment tool simply because information needed for other types of analyses is lacking. Also, remember that the techniques described in Chapters 7 and 8 are not mutually exclusive, i.e., expert opinion based on habitat analysis is more robust than expert opinion used alone. As far as possible, integrate quantitative data on at least habitat amount, distribution and trends in both amount and distribution with expert judgment on its implications for providing for persistent populations.

- **Enlist help from the Regional Office, Washington Office, and other experts in designing your assessment process.** The design of any assessment process, even one relying on expert opinion, requires expertise that often is not available at the level of individual forests. Getting assistance from those trained in the appropriate assessment and analysis techniques is critical.

- **Get necessary support from planners to generate needed summary information on plan alternatives, including summaries of management direction and projections of habitat and other critical environmental factors through time.** It is very important that the viability assessment and other assessments being carried out for the forest plan be based on the same understanding of management proposals and likely future conditions.

- **Thoroughly document the viability assessment process, starting with early decisions on the geographic scope of the assessment and going through the final results of the assessment process.** When you are challenged, the quality of your documentation is much more critical than the sophistication of your process. You must be able to demonstrate that you considered the appropriate information, used a logical process to conduct your assessment, and reached reasonable conclusions.

- **Work with your planners to design or refine a monitoring program that will ensure that you have better information to work with next time!** We recognize it is not feasible to monitor all factors for all species. Determining which species require monitoring is discussed in Chapter 9.

Literature Cited

Agee, J. K. 1993. *Fire ecology of Pacific Northwest forests*. Island Press, Washington, D.C

Akçakaya, H. R. 2000. Viability analyses with habitat-based metapopulation models. *Population Ecology* **42**:45-53.

Akcakaya, H. R., and J. L. Atwood. 1997. A habitat-based metapopulation model of the California Gnatcatcher. *Conservation Biology* **11**:990-999.

Akçakaya, H. R., McCarthy, M. A., and Pearce, J. L., 1995, Linking landscape data with population viability analysis: management options for the helmeted honeyeater *Lichenostomus melanops cassidix*, *Biological Conservation* **73**:169-176.

Akcakaya, H.R., S. Ferson, M.A. Burgman, D.A. Keith, G.M. Mace, and C.R. Todd. 2000. Making consistent IUCN classifications under uncertainty. *Conservation Biology*. **14**:1001-1013.

Amaranthus, M. P. 1997. Forest Sustainability: An Approach to Definition and Assessment at the Landscape Level. General Technical Report PNW-GTR-416, U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR.

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**:5954-5959.

Anderson, J.L. 1998. Embracing uncertainty. *Conservation Ecology* **2**:2. Available from the Internet URL: http://www.consecol.org/vol2/iss1/art2

Bart, J. 1995. Acceptance criteria for using individual-based models to make management decisions. *Ecological Applications* **5**:411-420.

Batek, M. J., A. J. Rebertus, and W. A. Schroeder. 1999. Reconstruction of early nineteenth-century vegetation and fire regimes in the Missouri Ozarks. *Journal of Biogeography* **26**:397-412.

Beesley, D. 1996. Reconstructing the landscape: An environmental history, 1820-1960. Pages 3-24 in *Sierra Nevada Ecosystem Project, Final Report to Congress, vol. II, Assessments and Scientific Basis for Management Options*. University of

California, Centers for Water and Wildland Resources, Davis, CA.

Beissinger, S.R., J.R. Walters, D.G. Catanzaro, K.G. Smith, J.B. Dunning, Jr., S.M. Haig and B.R. Noon. In press. Modeling approaches in avian conservation and the role of field biologists. *Current Ornithology*.

Beissinger, S. R., and D. R. McCullough, editors. 2002. *Population Viability Analysis*. Chicago: University of Chicago Press.

Beissinger, S. R., and Westphal, M. I. 1998. On the use of demographic models of population viability in endangered species management. *Journal of Wildlife Management* **62**:821-841.

Beissinger, S.R., J.M. Reed, J.M. Wunderle Jr., S.K. Robinson, D.M. Finch. 2000. Report of the AOU Conservation Committee on the Partners in Flight Species Prioritization Plan. *The Auk*. **117**:549-561.

Bender, L. C., G. J. Roloff, and J. B. Haufler. 1996. Evaluating confidence intervals for habitat suitability models. *Wildlife Society Bulletin* **25**:163-167.

Block, W. M., M. L. Morrison, J. Verner, and P. N. Manley. 1994. Assessing wildlife - habitat relationships models: a case study with California oak woodlands. *Wildlife Society Bulletin* **22**:549-561.

Boone, R. B., and M.L. Hunter. 1996. Using diffusion models to simulate the effects of land use on grizzly bear dispersal in the Rocky Mountains, *Landscape Ecology* **11**:51-64.

Bormann, B.T., J.R. Martin, F.H. Wagner, G.W. Wood, J. Alegria, P.G. Cunningham, M.H. Brookes, P. Friesma, J. Berg, and J.R. Henshaw. 1999. Adaptive Management. Pages 505-534 in N.C. Johnson, A.J. Malk, W.T. Sexton and R.Szaro (eds.), *Ecological Stewardship: A Common Reference for Ecosystem Management*. Oxford: Elsevier Science Ltd.

Bourdo, E. A. 1956. A review of the General Land Office survey and its use in quantitative studies of former forests. *Ecology* **37**:754-768.

Bradstock, R. A., Bedward, M., Scott, J., and Keith, D. A., 1996, Simulation of the effect of spatial and temporal variation in fire regimes on the population viability of a *Banksia* species, *Conservation Biology* **10**:776-784.

Breininger, D. R., M. J. Barkaszi, R. B. Smith, D. M. Oddy and J. A. Provancha. 1998.

Prioritizing wildlife taxa for biological diversity conservation at the local scale. *Environmental Management* **22**:315-321.

Brook, B. W., J. J. O'Grady, A. P. Chapman, M. A. Burgman, J. R. Akcacaya, and R. Frankham. 2000. Predictive accuracy of population viability analysis in conservation biology. *Nature* **404**:385-387.

Burgman, M. A. and H. P. Possingham. 2000. Population viability analysis for conservation: the good, the bad, and the undescribed. Pp in A. G. Young and G. M. Clark, editors. Genetics, Demography, and Viability Of Fragmented Populations. London: Cambridge University Press.

Burgman, M. A., Ferson, S., and Akçakaya H. R., 1993, *Risk Assessment in Conservation Biology*, London: Chapman and Hall.

Caro, T.M. and G. O'Doherty. 1999. On the use of surrogate species in conservation biology. *Conservation Biology* **13**:805-814.

Carroll, C., W.J. Zelinski, and R.F. Noss. 1999. Using presence-absence data to build and test spatial habitat models for the fisher in the Klamath Region, U.S.A. Conservation Biology **13**:1344-1359.

Carter, M.F., W.C. Hunter, D.N. Pashley, and K.V. Rosenberg. 2000. Setting conservation priorities for landbirds in the United States: the Partners in Flight approach. *The Auk*. **117**:541-548.

Caswell, H., 2001, *Matrix Population Models*, second edition, Sinauer and Associates, Sunderland, Massachusetts.

Caughley, G. and A. Gunn. 1996. Conservation Biology in Theory and Practice. Blackwell Science, Cambridge, MA. 459pp.

Channel, R. and M. V. Lomolino. 2000. Dynamic biogeography and conservation of endangered species. *Nature* **403**:84-86.

Cissel, J. H., F. J. Swanson, and P. J. Weisberg. 1999. Landscape management using historical fire regimes: Blue River, Oregon. *Ecological Applications* **9**:1217-1231.

Cleaves, D.L. 1994. Assessing uncertainty in expert judgments about natural resources. GRR SO-110. USDA Forest Service, Southern Forest Research Station, Asheville, NC. 17pp.

Clemen, R.T. (with T. Reilly). 2001. Making Hard Decisions. 2nd rev. ed. with

Decision Tools. Duxbury, Belmont, CA.

Committee of Scientists. 1999. Sustaining the people's lands: recommendations for stewardship of the National Forests and Grasslands into the next century. U.S. Dept. of Agriculture, Washington, D.C.

Conaughton, K. 2001. Sustainability: The Key Forest Policy Issue of the New Millennium? *Journal of Forestry* **99**:7.

Cook, R. R., and I. Hanski. 1995, On expected lifetimes of small-bodied and large-bodied species of birds on islands, *American Naturalist* **145**:307-315.

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**:219-221.

Czech, B. and P. R. Krausman. 1997. Distribution and causation of species endangerment in the United States. *Science* **277**:1116-1117.

Dedon , M. F., S. A. Laymon, and R. H. Barrett. 1986. Evaluating models of wildlife - habitat relationships of birds in black oak and mixed conifer habitats. Pages 115-119 in J. Verner, M. L. Morrison, and C. J. Ralph, editors. *Wildlife 2000: modeling habitat relationships of terrestrial vertebrates.* University of Wisconsin Press, Madison, WS, USA.

DeGraaf, R. M., and D. D. Rudis. 1983. New England wildlife: Habitat, natural history, and distribution. USDA Forest Service GTR- NE- 108. Radnor, PA, USA.

Dennis, B., P. L. Munholland, and J. M. Scott. 1991. Estimation of growth and extinction parameters for endangered species. *Ecological Monographs* **61**:115-143.

Dennis, B. and M.R.M. Otten. 2000. Joint effects of density dependence and rainfall on abundance of San Joaquin kit fox. *Journal of Wildlife Management* **64**: 388-400.

DeSteiguer, J.E., J.M. Pye and C.S. Love. 1990. Air pollution damage to U.S. forests. *Journal of Forestry* **88**:17-22.

Donovan, T. M., F. R. Thompson, III, J. Faaborg, and J. R. Probst. 1995. Reproductive success of migratory birds in habitat sources and sinks. *Conservation Biology* **9**:1380-1395.

Elzinga, C.L., D.W. Salzer and J.W. Willoughby. 1998. Measuring and monitoring plant populations. Bureau of Land Management Technical Reference 1730-1, Denver, CO.

Engen, S. and B.-E. Saether. 2000. Predicting the time to quasi-extinction for populations far below their carrying capacity. *Journal of Theoretical Biology* **205**: 649-658.

England, A.S. and D.W. Anderson. 1985. Avian community ecology in northern California chaparral. Evaluation of wildlife-habitat relationship matrix models for chamise-redshank and mixed chaparral. For: USDA Forest Service, PSW, Berkeley, Ca under Agreement No. PSW-83-0022CA. University of California, Davis, CA.

Evaluation of EIS Alternatives by the Science Integration Team (ICB-Draft), 1997. PNW -GTR-406, Volumes I & II. USDA Forest Service, Pacific Northwest Research Station.

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**:621-631.

Fedkiw, J. 1989. The Evolving Use and Management of the Nation's Forests, Grasslands, Croplands, and Related Resources: A Technical Document Supporting the 1989 USDA Forest Service RPA Assessment. General Technical Report GTR-RM-175, U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Fedkiw, J. 2000. Managing Multiple Uses on National Forests, 1905-1995: A 90-year Learning Experience and It Isn't Finished Yet. U.S. Department of Agriculture, Forest Service, Washington, DC.

Fieberg, J., and Ellner, S. P., 2000, When is it meaningful to estimate an extinction probability? *Ecology* **81**:2040-2047.

Floyd, D. W. (editor), K. Alexander, C. Burley, A. W. Cooper, A. DuFault, R. W. Gorte, S. G. Haines, B. B. Hronek, C. D. Oliver, and E. W. Shepard. 1999. *Forest of Discord: Options for Governing Our National Forests and Federal Public Lands*. The Society of American Foresters, Bethesda, MD.

Floyd, D. W., S. L. Vonhof, and H. E. Seyfang. 2001. Forest Sustainability: A Discussion Guide for Professional Resource Managers. *Journal of Forestry* **99**:8-16.

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

Forest Ecosystem Management Assessment Team (FEMAT). 1993. Forest ecosystem management: an ecological, economic, and social assessment. U.S.D.A. Forest Service and five other federal agencies.

Frankham, R., 1995, Effective population size-adult population size ratios in wildlife: a review, *Genetic Research* **66**:95-107.

Franklin, I. R., 1980, Evolutionary change in small populations, in: *Conservation Biology: An Evolutionary-Ecological Approach* (M. E. Soulé and B. A. Wilcox, eds.), Sinauer and Associates, Sunderland, Massachusetts, pp. 135-149.

Franklin, I. R., and R. Frankham. 1998. How large must populations be to retain evolutionary potential? *Animal Conservation* **1**:69-73

Freudenburg, W.R. 1992. Heuristics, biases, and the not-so-general publics: expertise and error in the assessment of risks. Pp. 229-249 in S. Krimsky and D. Golding (eds.). *Social theories of risk*. Praeger Publishing, Westport, CT. 412pp.

Galatowitsch, S. M. 1990. Using the original land survey notes to reconstruct presettlement landscapes in the American west. *Great Basin Naturalist* **50**:181-191.

General Accounting Office (GAO). 1999. Forest Service Priorities: Evolving Mission Favors Resource Protection Over Production. GAO/RCED-99-166, U.S. General Accounting Office, Washington, DC.

Ginzburg, L. R, Slobodkin, B., Johnson, K., and A.G. Bindman. 1982, Quasiextinction probabilities as a measure of impact on population growth, *Risk Analysis* **2**:171-181.

Ginzburg, L. R., Ferson, S., and Akçakaya, H. R., 1990, Reconstructability of density dependence and the conservative assessment of extinction risks. *Conservation Biology* **4**:63-70.

Goodwin, P. and G. Wright. 1991. Decision analysis for management judgment. Chichester, U.K.: John Wiley & Sons.

Green, R. E. and G. J. M. Hirons. 1991. The relevance of population studies to the conservation of threatened birds. Pp. 594-633 in C. M. Perrins, J. D. Lebreton, and G. J. M. Hirons, editors. *Bird Population Studies: Relevance to conservation and management*. N.Y.: Oxford University Press.

Groom, M. J., and Pascual, M. A. 1998. The analysis of population persistence: an outlook on the practice of viability analysis. Pp. , in: *Conservation Biology*, 2nd Ed. Pp. 4-

27 in P.L. Fiedler and P.M Kareiva, eds., N.Y.: Chapman and Hall.

Guerrant, E.O., Jr. 1998. Monitoring Nelson's checker-mallow (*Sidalcea nelsoniana*) at Walker Flat, Neverstill, and South McGuire: 1997 and 1998. USDI Bureau of Land Management. Salem, OR.

Halpin, P. N. 1998. Global climate change and natural area protection: management responses and research direction. Ecological Applications **7**:828-843.

Hamel, P. B. 1992. Land manager's guide to the birds of the South. The Nature Conservancy, Southeastern Region, Chapel Hill, NC, USA.

Hanski, I., 1991. Inferences from ecological incidence functions. *American Naturalist* **139**:657-662.

Hanski, I., 1994*a*. A practical model of metapopulation dynamics. J*ournal of Animal Ecology* **63**:151-162.

Hanski, I., 1994*b*. Patch-occupancy dynamics in fragmented landscapes. *Trends in Ecology and Evolution* **9**:131-135.

Hanski, I., 1998. Metapopulation dynamics. *Nature* **396**:41-49.

Hanski, I. 1999a. Habitat connectivity, habitat continuity and metapopulations in dynamic landscapes. *Oikos* **87**:209-219.

Hanski, I., 1999b. *Metapopulation Ecology*, N.Y.: Oxford University Press.

Hanski, I. In press. Animal metapopulations in highly fragmented landscapes and PVA. in: *Population Viability Analysis* (S. R. Beissinger and D. R. McCullough, eds.), University of Chicago Press, Chicago, Illinois, in press.

Harris, R. B., Maguire, L. A., and. Shaffer, M. L. 1987. Sample sizes for minimum viable population estimation *Conservation Biology*. **1**:72-76.

Harrison, S., and Ray, C. Plant Population Viability and Metapopulation-level Processes, in: *Population Viability Analysis* (S. R. Beissinger and D. R. McCullough, eds.), University of Chicago Press, Chicago, Illinois, in press.

Harrod, R. J., W. L. Gaines, W. E. Hartl, and A. Camp. 1998. Estimating historical snag density in dry forests east of the Cascade Range. United States Forest Service General Technical Report PNW-428.

Haufler, J. B. 1999. Strategies for conserving terrestrial biological diversity. Pages 17-30 in R. K. Baydack, H. Campa III, and J. B. Haufler, editors. Practical approaches to the conservation of biological diversity. Washington, D.C.: Island Press.

Haufler, J. B. 2000. Ecosystem management: from rhetoric to reality. *Transactions of the North American Wildlife and Natural Resources Conference* **65**:11-33.

Haufler, J. B., R. K. Baydack, H. Campa, III, B. J. Kernohan, L. J. O'Neil, L. Waits and C. Miller. (in press). Performance measures for ecosystem management. Report of The Wildlife Society Technical Committee. The Wildlife Society, Bethesda, Maryland.

Haufler, J. B., T. Crow, and D. Wilcove. 1999. Scale considerations for ecosystem management. Pp. 331-342 in: N. C. Johnson, A. J. Malk, W. T. Sexton, and R. Szaro, editors. Ecological stewardship: a common reference for ecosystem management. Oxford: Elsevier.

Hirt, P. W. 1994. A Conspiracy of Optimism: Management of the National Forests Since World War Two. University of Nebraska Press, Lincoln.

Hokit, D. G., Stith, B. M., and L.C. Branch. 1999. Effects of landscape structure in Florida scrub: a population perspective. *Ecological Applications* **9**:124-134.

Hollander, A. D., F. W. Davis, and D. M. Stoms. 1994. Hierarchical representations of species distributions using maps, images and sighting data. Pages 71-88 *in* R. I. Miller, editor. Mapping the diversity of nature. Chapman and Hall, London, England.

Holmes, E.E. 2001. Estimating risks in declining populations with poor data. *Proceedings of the National Academy of Sciences (USA)* **98**:5072-5077.

Holthausen, R.S., M.G. Raphael, F.B. Samson, D. Ebert, R. Hiebert and K. Menasco. 1999. Population viability in ecosystem management. Pp. 135-156 in Szaro, R.C. , N.C. Johnson, W.T. Sexton and A.J. Malk, editors. *Ecological stewardship: a common reference for ecosystem management.* Oxford: Elsevier Science.

Hoover, R. L., and D. L. Wills, editors. 1984. Managing forested lands for wildlife. Colorado Division of Wildlife, Denver, CO, USA.

Howard, R. and J. Matheson. 1981. Influence diagrams. Pp. 721-762 *in* Howard, R. and J. Matheson (eds). Ecological Scale: Theory and applications. N.Y.: Columbia University Press.

IUCN (Species Survival Commission). 1994. International Union for the Conservation of

Nature Red List Categories. Gland, Switzerland.

IUCN (Species Survival Commission). 2000. International Union for the Conservation of Nature Red List Categories. Gland, Switzerland.

James, F.C., C.A. Hess, and D. Kufrin. 1997. Species-centered environmental analysis: indirect effects of fire history on Red-Cockaded Woodpeckers. *Ecological Applications* **7**:118-129.

Kapp, R. O. 1978. Presettlement forests of the Pine River watershed (central Michigan) based on original land survey records. *The Michigan Botanist* 17:3-15.

Kareiva, P. and U. Wennergren. 1995. Connecting landscape patterns to ecosystem and population processes. *Nature* **373**:299-302.

Karl, J. W., P. J. Heglund, E. O. Garton, J. M. Scott, N. M. Wright, and R. L. Hutto. 2000. Sensitivity of species habitat-relationship model performance to factors of scale. *Ecological Applications* **10**:1690-1705.

Keith, D.A. 1998. An evaluation and modification of World Conservation Union Red List Criteria for classification of extinction risk in vascular plants. *Conservation Biology* **12**:1076-1090.

Kolb, T. E., M. R. Wagner, and W. W. Covington. 1994. Concepts of Forest Health. *Journal of Forestry* **92**:10-15.

Lamberson, R. H., Noon, B. R., Voss, C., and McKelvey, K. S. 1994. Reserve design for territorial species: the effects of patch size and spacing on the viability of the Northern Spotted Owl. *Conservation Biology* **8**:185-195.

Lande, R., 1988. Genetics and demography in biological conservation. *Science* 241:1455 -1460.

Lande, R., 1995. Mutation and conservation. *Conservation Biology* 9:782-791.

Lande, R. In press. Incorporating stochasticity in population viability analysis. In: *Population Viability Analysis* (S. R. Beissinger and D. R. McCullough, eds.), University of Chicago Press, Chicago, Illinois, in press.

Laymon, S.A. 1990. A test of the California wildlife-habitat relationship system for breeding birds in valley-foothill riparian habitat. Pp. 307-313 *in* D.L. Abell, editor, Proceedings of the California riparian systems conference: Protection, management and restoration for the 1990s. USDA Forest Service Gen. Tech.

Report PSW-110. Berkeley, CA.

Lee, D.C. and B.E. Rieman. 1997. Population viability assessment of salmonids using probabilistic networks. *North American Journal of Fisheries Management* **17**:1144-1157.

Lee, D.C. and G.A. Bradshaw. 1998. Making monitoring work for managers: thoughts on a conceptual framework for improved monitoring within broad-scale ecosystem management efforts. Paper online at http://www.icbemp.gov/spatial/lee_monitor (October 26, 1998 draft version).

Lehmkuhl, J.F., M.G. Raphael, R.S. Holthausen, J.R. Hickenbottom, R.H. Nancy, and J.S. Shelly. 1997. Historical and current status of terrestrial species and the effects of proposed alternatives. Chapter 4 *in* Quigley, T.M., A.M. Lee, and S.J. Arbelbide. Evaluation of EIS alternatives by the Science Integration Team. Volume II. General Technical Report PNW-GTR-406. U.S.D.A. Forest Service, Pacific Northwest Research Station, Portland, OR.

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

Lesica, P. and F. W. Allendorf. 1995. When are peripheral populations valuable for conservation? *Conservation Biology* **9**:753-760.

Lindenmayer, D. B., M. A. Burgman, H. R. Akçakaya, R. Lacy, and H. P. Possingham. 1995. A review of three models for metapopulation viability analysis – ALEX, RAMAS/Space and VORTEX. *Ecological Modeling* **82**:161-174.

Lindenmayer, D.B. and H.P. Possingham. 1996. Ranking conservation and timber management options for Leadbetter's possum in southeastern Australia using population viability analysis. *Conservation Biology* **10**:235-251.

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*, in press.

Linstone, H. and Turoff, M. 1975. The Delphi Method: techniques and applications. Addison-Wesley, Wokingham, England.

Liu, J., Dunning J. B., and Pulliam, H. R. 1995. Potential impacts of a forest management plan on Bachman's Sparrows: linking a spatially explicit model with GIS. *Conservation Biology* **9**:62-75.

Ludwig, D. 1996. Uncertainty and the assessment of extinction probabilities. *Ecological Applications* **6**:1067-1076.

Ludwig, D. 1999. Is it meaningful to estimate a probability of extinction? *Ecology* **80**:298 -310.

Lunney, D., A. Curtin, D. Ayers, H.G. Cogger, and C.R. Dickman. 1996. An ecological approach to identifying the endangered fauna of New South Wales. *Pacific Conservation Biology* **2**:212-231.

Mace, G.M. and R. Lande. 1991. Assessing extinction threats: Toward a reevaluation of IUCN threatened species categories. *Conservation Biology* **5**:148-157.

Mangel, M. and Tier, C., 1994, Four facts every conservation biologist should know about persistence, *Ecology* **75**:607-614.

Marcot, B.G., M.G. Raphael and K.H. Berry. 1983. Monitoring wildlife habitat and validation of wildlife-habitat relationships models. *Transactions of the North American Wildlife and Natural Resources Conference* **48**:315-329.

Marcot, B.G., M.A. Castellano, J.A. Christy, L.K. Croft, J.F. Lehmkuhl, R.H. Naney, R.E. Rosentreter, R.E. Sandquist, and E. Zieroth. 1997. Terrestrial ecology assessment. Chapter 5 *in* Quigley, T.M., A.M. Lee, and S.J. Arbelbide. Evaluation of EIS alternatives by the Science Integration Team. Volume II. General Technical Report PNW-GTR-406. U.S.D.A. Forest Service, Pacific Northwest Research Station, Portland, OR.

Marcot, B.G. 1998. Selecting appropriate statistical procedures and asking the right questions: a synthesis. Pp. 129-143 *in* Sit, V. and B. Taylor (eds). Statistical Methods for Adaptive Management Studies. British Columbia Ministry of Forests Research Branch, Land Management Handbook No. 42, Victoria, BC.

Marcot, B.C., Holthausen, R.S., Raphael, M.G., Rowland, M.M., Wisdom, M.J. (In press) Using Bayesian Belief Networks to Evaluate Fish and Wildlife Population Viability Under Land Management Alternatives from an Environmental Impact Statement. USDA Forest Service.

Master, L. L. 1991. Assessing threats and setting priorities for conservation. *Conservation*

Biology **5**: 559-563.

Master, L.L., B.A. Stein, L.S. Kutner and G.A. Hammerson. 2000. Vanishing Assets: Conservation status of U.S. Species. Pages (??) in B.A. Stein, L.S. Kutner and J.S. Adams, editors. *Precious Heritage: The status of biodiversity in the United States.* Oxford University Press, New York.

McKelvey, K., Noon, B. R., and Lamberson, R. H., 1993, Conservation planning for species occupying fragmented landscapes: the case of the northern spotted owl, in: *Biotic Interactions and Global Change* (P. M. Kareiva, J. G. Kingsolver and R. B. Huey, eds.), Sinauer and Associates, Sunderland, Massachusetts, pp. 424-450.

McLaren, M.S., I.D. Thompson, and J.A. Baker. 1998. Selection of vertebrate wildlife indicators for monitoring sustainable forest management in Ontario. *The Forestry Chronicle* **74**:241-248.

McNamee, P. and J. Celona. 1989. Decision Analysis with Supertree. Scientific Press, San Francisco. 324pp.

Meir, E. and W.F. Fagan. 2000. Will observation error and biases ruin the use of simple extinction models? *Conservation Biology* **14**:148-154.

MER. 2000. Methodo de Evaluacion del Riesgo de Extincion de las Especies Silvestres en Mexico. Draft paper.

Meslow, E.C., R.S. Holthausen, and D.A. Cleaves. 1994. Assessment of terrestrial species and ecosystems. *Journal of Forestry* **92**:24-27.

Meyer, M. and J. Booker. 1991. Eliciting and Analyzing Expert Judgment, a Practical Guide. Academic Press, London

Mighton, S.R., R. Meier, E. Padley, T. Doyle, M. Shedd, and J. Gallagher. 2000. Population viability assessment in forest plan revision: National Forests of Wisconsin and Minnesota. Unpublished report, U.S.D.A. Forest Service, Region 9, Milwaukee, WI. May 22, 2000. 154pp.

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

Molloy, J. and Davis, A. 1992. Setting Priorities for Conservation of New Zealand’s Threatened Plants and Animals. Department of Conservation, Te Papa Atawhai, Wellington.

Mooij, W. M., and DeAngelis, D. L., 1999, Error propagation in spatially explicit population models: a reassessment, *Conservation Biology* **13**:930-933

Morgan, M. G. and M. Henrion. 1990. Uncertainty. A Guide to Dealing with Uncertainty in Qualitative Risk and Policy Analysis. N.Y.: Cambridge University Press.

Morgan, P., G. H. Aplet, J. B. Haufler, H. C. Humphries, M. M. Moore, and W. D. Wilson. 1994. Historical range of variability: A useful tool for evaluating ecosystem change. *Journal of Sustainable Forestry* **2**:87-112.

Morrison, M.L., I.C. Timossi and K.A. With. 1987. Development and testing of linear regression models predicting bird-habitat relationships. *Journal of Wildlife Management* **51**:247-253.

Mulder, B.S., B.R. Noon, T.A Spies, M.G. Raphael, A.R. Olsen, C.J. Palmer, G.H. Reeves, and H.H. Hart. 1998. The strategy and design of the effectiveness monitoring program for the Northwest Forest Plant. Northwest Forest Plan, Intergovernmental Advisory Committee.

Murphy, M.T. 2001. Source-sink dynamics of a declining eastern kingbird population and the value of sink habitats. Conservation Biology 15(3):737-748.

Nicholopoulos, J. 1999. The endangered species listing program. *Endangered Species Bulletin*. **14**:6-9.

Nineteenth-century vegetation and fire regimes in the Missouri Ozarks. *Journal of Biogeography* **26**:397-412.

Noon, B. R., and McKelvey, K. S., 1996*a*. Management of the spotted owl: a case history in conservation biology. *Annual Review of Ecology and Systematics* **27**:135-162.

Noon, B.R., T.A. Spies, and M G. Raphael. 1999b. Conceptual basis for designing an effectiveness monitoring program. Pages 21-48 *in* B. S. Mulder, B.R. Noon, T.A. Spies, M.G. Raphael, C.J. Palmer, A.R. Olsen, G.H. Reeves, and H.H. Welsh, technical coordinators. The strategy and design of the effectiveness monitoring program for the Northwest Forest Plan. USDA Forest Service General Technical Report PNW-GTR-437. Portland, OR.

Northern Great Plains- Proposed Revised Management Plans (NGP-PRMP), 2001. USDA Forest Service.

Noss, R. F., M. A. O'Connell, and D. D. Murphy. 1997. The Science of conservation planning: habitat conservation under the Endangered Species Act. Island Press, Washington, D.C.

Noss, R.F., and A.Y. Cooperrider. 1994. Saving nature's legacy. Washington, D.C.: Island Press.

Price, M. V., and Gilpin, M. 1996. Modelers, mammalogists, and metapopulations: designing Stephens' kangaroo rat reserves. Pp. 217-240 in D. R. McCullough, ed., *Metapopulations and Wildlife Conservation.* Washington, D.C.: Island Press.

Pullium, H. R. 1988. Sources, sinks, and population regulation. *American Naturalist* **132**:652-661.

Quigley, T. M., R. W. Hayes, and R. T. Graham. 1996. Integrated scientific assessment for ecosystem management in the interior Columbia Basin. USDA Forest Service, General Technical Report PNW-382. Portland, OR.

Quintana-Ascencio, P. F., and Menges, E. S. 1996. Inferring metapopulation dynamics from patch-level incidence of Florida scrub plants. *Conservation Biology* **10**:1210–1219.

Rabinowitz et al. 1981. Seven forms of rarity. Pp. 205-217 in H. Synge, editor, *The biological aspects of rare plant conservation*. New York: Wiley.

Raffensperger, C. and J. Tichner (eds.). 1999. Protecting Public Health and the Environment. Island Press, Washington, DC.

Ralls, K. and B. L. Taylor. 1997. How viable is population viability analysis? Pages 228 -235 *in* S.T.A. Pickett, R. S. Ostfield, M. Shachak and G. E. Likens, editors. The ecological basis of conservation. N.Y.: Chapman & Hall.

Ralls, K., S. R. Beissinger and J. Fitz Cochrane. In press. Guidelines for using PVA in endangered species management. In: *Population Viability Analysis* (S. R. Beissinger and D. R. McCullough, eds.). Chicago: University of Chicago Press.

Raphael, M.G. and B.G. Marcot. 1986. Validation of a wildlife-habitat-relationships model: vertebrates in a Douglas-fir sere. Pp. 129-138 in J. Verner, M.L. Morrison and C.J. Ralph, editors, Wildlife 2000: Modeling habitat relationships of terrestrial vertebrates. University of Wisconsin Press, Madison, WI.

Raphael, M.G. and B.G. Marcot. 1994. Key questions and issues – species and ecosystem viability. *Journal of Forestry* **92**:45-47.

Raphael, M.G., J.J. Wisdom, M.M. Rowland, R.S. Holthausen, B.C. Wales, B.G. Marcot, and T.D. Rich. *In press*. Projected effects on selected terrestrial vertebrates from actions proposed in the interior Columbia Basin. Forest Ecology and Management.

Reed, J.M. and A.R. Blaustein. 1997. Biologically significant population declines and statistical power. *Conservation Biology* **11**:281-282.

Richter, B. D., and H. E. Richter. 2000. Prescribing flood regimes to sustain riparian ecosystem along meandering rivers. *Conservation Biology* **14**:1467-1478.

Richter, B. D., J. V. Baumgartner, J. Powell, and D. P. Braun. 1996. A method for assessing hydrologic alteration within ecosystems. *Conservation Biology* **10**:1163-1174.

Roloff, G. J., and B. J. Kernohan. 1999. Habitat suitability models. *Wildlife Society Bulletin* **27**:973-985.

Roloff, G. J., and J. B. Haufler. In Press. Modeling habitat-based viability from organism to population. Pages *in* Scott et al. editors. Predicting species occurrence. Washington, D.C.: Island Press.

Roloff, G.J., and J.B. Haufler. 1997. Establishing population viability planning objectives based on habitat potentials. *Wildlife Society Bulletin* **25**:895-904.

Root, K.V. in press. Classification models. In R. Pastorok, S.M. Bartell, S. Ferson and L.R. Ginzburg, editors. *Ecological modeling in risk assessment.* Lewis Publishers, Boca Raton.

Ruckelshaus, M., Hartway, C., and Kareiva, P. 1997. Assessing the data requirements of spatially explicit dispersal models. *Conservation Biology* **11**:1298-1306.

Ruth, L. 1996. Conservation and Controversy: National Forest Management, 1960-95. Pp. 145-162 *in* Sierra Nevada Ecosystem Project, Final Report to Congress, vol. II, Assessments and Scientific Basis for Management Options. University of California, Centers for Water and Wildland Resources, Davis, CA.

Saab, V.A. and J.G. Dudley. 1998. Responses of cavity-nesting birds to stand-replacement fire and salvage logging in ponderosa pine/Douglas-fir forests of southwestern Idaho. USDA Forest Service Research Paper RMRS-RP-11. Ogden, UT.

Saether, B.-E., S. Engen, R. Lande, P. Arcese and J.N.M. Smith. 2000. Estimating the time to extinction in an island population of song sparrows. *Proceedings of the Royal Society Biological Sciences Series B* **267**: 621-626.

Sauer, J. R., and Slade, N. A. 1987. Size-based demography of vertebrates. *Annual Review of Ecology and Systematics* **18**:71-90.

Schamberger, M., A. H. Farmer, and J. W. Terrell. 1982. Habitat suitability index models. U. S. Fish and Wildlife Service. FWS/OBS-82/10. Fort Collins, CO, USA.

Schemske, D. W., Husband, B. C., Ruckelhaus, M. H., Goodwillie, C., Parker, I. M., and J.G. Bishop. 1994. Evaluating approaches to the conservation of rare and endangered plants, *Ecology* 75:584-606.

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

Shaffer, M. L. 1987. Minimum viable populations: coping with uncertainty. Pp. 69-86 in M.E. Soule, ed., *Viable Populations for Conservation.* Cambridge: Cambridge University Press.

Shaw, C.G. III. 1999. Use of risk assessment panels during revision of the Tongass Land and Resource Management Plan. General Technical Report PNW-GTR-460. U.S.D.A. Forest Service, Pacific Northwest Research Station, Portland, OR. 43pp.

Shepard, B.B., B. Sanborn, L. Ulmer, and D.C. Lee. 1997. Status and risk of extinction for westslope cutthroat trout in the Upper Missouri River basin, Montana. North *American Journal of Fisheries Management* 17:1158-1172.

Shrader-Frechette, K. S, and McCoy, E. D., 1993, *Method In Ecology: Strategies For Conservation*, Cambridge University Press, Cambridge.

Sierra Nevada Forest Plan Amendment-Final Environmental Impact Statement (SNFPA -FEIS), 2001. USDA Forest Service, Pacific Southwest & Intermountain Regions.

Simberloff, D. 1998. Flagships, umbrellas and keystones: Is single species management passé in the landscape era? *Biological Conservation* **83**:247-257.

Sit, V. and B. Taylor (eds). 1998. Statistical Methods for Adaptive Management Studies. British Columbia Ministry of Forests Research Branch, Land Management Handbook No. 42, Victoria, BC.

Sjorgen-Gulve, P., and Ray, C. 1996. Using logistic regression to model metapopulation dynamics: large-scale forestry extirpates the pool frog. Pp. 111-137 in D.R.

McCullough, ed. *Metapopulations and Wildlife Conservation.* Washington, D.C.: Island Press.

Sloan, J. P. 1998. Historical density and stand structure of an old-growth forest in the Boise Basin in central Idaho. Pages 258-266 *in* L. A. Brennan, and T. L. Pruden, editors. Fire in ecosystem management: shifting the paradigm from suppression to prescription. Proceedings of the 20th Tall Timbers Fire Ecology Center Conference. Tall Timbers Research Station, Tallahassee, Florida, USA.

Soulé, M. E., 1980, Thresholds for survival: maintaining fitness and evolutionary potential, in: *Conservation Biology: An Evolutionary-Ecological Approach* (M. E. Soule and B. A. Wilcox, eds.), Sinauer and Associates, Sunderland, Massachusetts, pp. 151-169.

Spanos, A. 1999. Probability Theory and Statistical Inference: Econometric Modeling with Observational Data. Cambridge University Press, Cambridge.

Stacey, P. B., Johnson, V. A., and Taper, M. L., 1997, Migration within metapopulations: the impact upon local population dynamics. Pp. 267-292 in I. A. Hanski and M. E. Gilpin, eds., *Metapopulation Biology: Ecology, Genetics, and Evolution.* San Diego: Academic Press.

Starfield, A. M. 1997. A pragmatic approach to modeling for wildlife management. *Journal of Wildlife Management* **61**:261-270.

Starfield, A. M., J. D. Roth, and K. Ralls. 1995. “Mobbing” in Hawaiian monk seals (*Monachus schauinslani*): the value of simulation modeling in the absence of apparently crucial data. *Conservation Biology* **9**:166-174.

Steen, H. K. 1997. The Beginning of the National Forest System. *in* C. Miller, editor. American Forests: Nature, Culture, and Politics. University of Kansas Press, Lawrence.

Stroessner, W. J., and J. R. Habeck. 1966. The presettlement vegetation of Iowa County, Wisconsin. Wisconsin Academy of Sciences, Arts and Letters 55:167-180.

Swaay, C.A.M., M.S. van Warren. 1999. Red Data Book of European Butterflies (Rhopalcera). *Nature and Environment*, No. 99. Council of Europe Publishing, Strasbourg.

Swetnam, T. W., C. D. Allen, and J. L. Betancourt. 1999. Applied historical ecology: using the past to manage the future. *Ecological Applications* **9**:1189-1206.

Taylor, B.L. and T. Gerrodette. 1993. The uses of statistical power in conservation biology: the vaquita and the northern spotted owl. *Conservation Biology* **7**:489-500.

Taylor, B.L., P.R. Wade, D.P. DeMaster, and J. Barlow. 2000. Incorporating uncertainty into management models for marine mammals. *Conservation Biology* **14**:1243-1252.

Thomas, J. W., ed. 1979. Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. USDA Forest Service, Agricultural Handbook #533. Washington, D.C., USA

Thomas, J.W., M.G. Raphael, R.G. Anthony, E.D. Forsman, A.G. Gunderson, R.S. Holthausen, B.G. Marcot, G.H. Reeves, J.R. Sedell and D.M. Solis. 1993a. Viability assessments and management considerations for species associated with late-successional and old-growth forests of the Pacific Northwest: the Report of the Scientific Analysis Team. U.S.D.A. Forest Service, National Forest System, Forest Service Research. Portland, OR.

Thomas, J.W., M.G. Raphael, R.G. Anthony, E.D. Forsman, A.G. Gunderson, R.S. Holthausen, B.G. Marcot, G.H. Reeves, J.R. Sedell and D.M. Solis. 1993b. Forest ecosystem management: an ecological, economic, and social assessment. Report of the Forest Ecosystem Management Assessment Team. U.S.D.A. Forest Service, U.S.D.I. Bureau of Land Management, and U.S.D.I. Fish and Wildlife Service, Portland, OR.

Tilman, D., R.M. May, C.L. Lehman, and M.A. Nowak. 1994. Habitat destruction and the extinction debt. *Nature* **371**:65-66.

Todd, C.R. and M.A. Burgman. 1998. Assessment of threat and conservation priorities under realistic levels of uncertainty and reliability. *Conservation Biology* **12**:966-974.

Tongass National Forest, Land & Resource Management Plan- Final Environmental Impact Assessment (TNF-FEIS), 1997. USDA Forest Service, Alaska Region.

U.S.D.A. Forest Service 2000. Sierra Nevada forest plan amendment draft environmental impact statement. Appendix E: Monitoring and adaptive management plan. Pacific Southwest Region, Vallejo, CA.

U.S.D.A. Forest Service and U.S. Bureau of Land Management. 2000. Final Supplemental Environmental Impact Statement for Amendment to the Survey & Manage, Protection Buffer, and other Mitigation Measures. Portland, OR

USFWS. 1983. United States Fish and Wildlife Service. Endangered and threatened species listing and recovery priority guidelines. Federal Register Vol. 48, No. 184. Wednesday, September 21, 1983. Notices. Pp. 43098-43105.

Van Deelen, T. R., K. S. Pregitzer, and J. B. Haufler. 1996. A comparison of presettlement and present-day forests in two northern Michigan deer yards. *American Midland Naturalist* **135**:181-194.

Von Winterfeldt, D. 1992. Expert knowledge and public values in risk management: the role of decision analysis. Pp. 321-342 *in* Krimsky, S. and D. Golding (eds). *Social Theories of Risk*. Praeger Publishing, Westport, CT. 412pp.

Wade, P.A. 1998. Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. *Marine Mammal Science* **14**:1-37.

Walters, C.J. and C.S. Holling. 1990. Large-scale management experiments and learning by doing. *Ecology* **61**:2060-2068.

Waples, R. S., Definition and Estimation of Effective Population Size in the Conservation of Endangered Species, in: *Population Viability Analysis* (S. R. Beissinger and D. R. McCullough, eds.), University of Chicago Press, Chicago, Illinois, in press.

Wennergren, U., Ruckelhaus, M., and Kareiva, P., 1995, The promise and limitations of spatial models in conservation biology, *Oikos* **74**:349-356.

White, G. C., A. B. Franklin, and T. M. Shenk. In press. Estimating parameters of demographic PVA models from data on marked animals. In S. R. Beissinger and D. R. McCullough. Population viability analysis. Univ. of Chicago Press, Chicago, Ill.

White, M. S., and D. J. Mladenoff. 1994. Old growth forest landscape transitions from pre -European settlement to present. *Landscape Ecology* **9**:191-205.

White, P. S., J. Harrod, W. H. Romme, and J. Betancourt. 1999. Disturbance and temporal dynamics. Pages 281-312 *in* R. C. Szaro, N. C. Johnson, W. T. Sexton, and A. J. Malk, editors. Ecological stewardship: a common reference for ecosystem management, Volume II. Elsevier Science, Oxford, UK.

Wiersum, K. F. 1995. 200 Years of Sustainability in Forestry: Lessons from History.

Environmental Management **19**:321-329.

Wisdom, M.J., R.S. Holthausen, B.C Wales, C.D. Hargis, V.A. Saab, D.C. Lee, W.J. Hann, T.D. Rich, M.M. Rowland, W.J. Murphy, and M.R. Eames. 2000. Source habitats for terrestrial vertebrates of focus in the Interior Columbia Basin: broad-scale trends and management implications. Volume 1—Overview. General Technical Report PNW-GTR-485. U.S.D.A. Forest Service, Pacific Northwest Research Station, Portland, OR.

Wisdom, M.J., G. Hayward, S. Shelly, C.D. Hargis, R.S. Holthausen, J. Epifanio, L. Parker and J. Kershner. 2001. Using species groups and focal species for assessment of species at risk in forest planning. Unpublished manuscript. U.S.D.A. Forest Service, Rocky Mountain Research Station, Flagstaff, Arizona.