CHAPTER

A Decision Framework for Choosing Models in Large-Scale Wildlife Conservation Planning

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This volume presents principles, concepts, methods, and examples of several modeling approaches suitable for planning wildlife conservation in large landscapes. Many approaches are rooted in ecological theory (e.g., Rowland and Wisdom, this volume), with different data needs, assumptions, and methodologies, and have been applied successfully in a diversity of environments. Approaches range from fine filter, single species approaches to multiple species or coarse filter, ecosystem approaches. The diversity of approaches may seem overwhelming to land managers or planners who must select among these options for their project. In this chapter, we present a decision framework (Fig. 24-1) that represents the range of options for planning wildlife conservation considered in this book. The context for this framework, and for the book, is that the reader is interested in the conservation of wildlife species; we do not consider coarse filter approaches focused solely on higher ecological levels without consideration of species. As with any science-based planning effort, careful upfront consideration of objectives and the level of investment you can make will make subsequent methodological questions much easier to address. The topics here are generally arranged from those related to the goals of the project and refining objectives, to model development and application, to model or project evaluation, and conclude with some miscellaneous issues we believe deserve consideration. We use the framework to step through two case studies (Fig. 24-1).

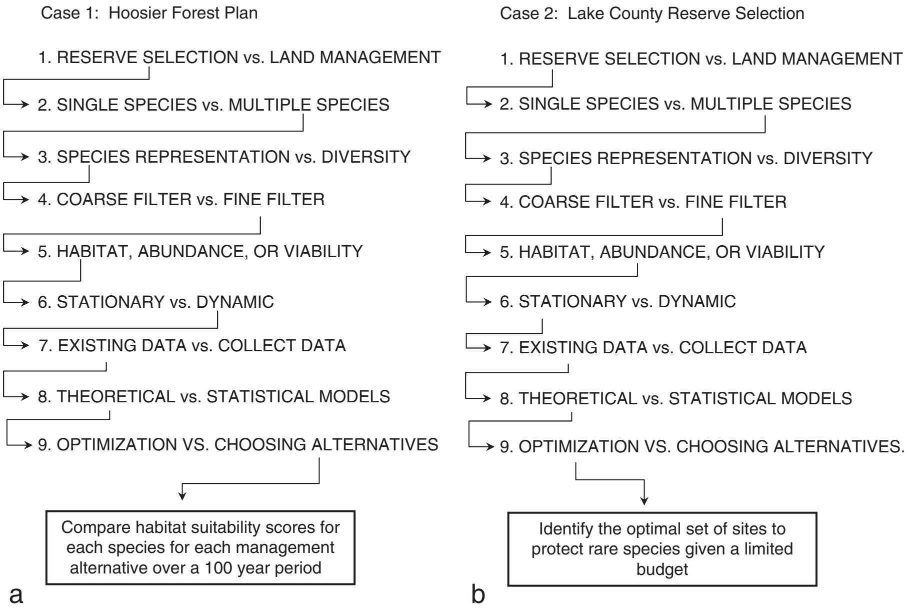

FIG. 24-1

Examples of decision pathways for two different conservation planning problems. Case 1 (A) is part of the Hoosier National Forest Plan that considered habitat suitability for focal species under five plan alternatives. Case 2 (B) is a reserve selection problem in which Lake County Illinois planners identified the optimal set of sites to protect rare plants and animals given a fixed budget. See text for details.

RESERVE SELECTION VERSUS LAND MANAGEMENT PLANNING

The goal of reserve selection is to select a geographic area that best addresses conservation objectives, while the goal of land management planning is typically to select land management practices that best meet conservation objectives for a defined geographic area. The boundary between these methods becomes blurred when the definition of reserves is loosened to include multiple-use lands for which conservation is one of many objectives, and reserve selection methods are used to select lands, perhaps already within public ownership, for different levels of protection. Both reserve selection and land management planning can address single or multiple species. Multiple species problems typically consider habitat needs of as many species as possible, or a subset of high-concern species (Flather et al.; Noon et al., this volume). Single species problems are more likely to focus on abundance and can use gap approaches to see if important areas (i.e., areas of high abundance) are covered (i.e., Fig. 22-6, Fitzgerald et al., this volume). Mathematical optimization procedures can be used to maximize benefits across several objectives (Haight and Gobster, this volume), or simple graphical or map-based assessments can be made, for example, by comparing maps of current protected areas to maps of species abundance.

SINGLE SPECIES VERSUS MULTIPLE SPECIES

The decision regarding single species versus multiple species is primarily a function of project objectives. Some projects may be focused on a single species, such as the recovery plan for an endangered species. Most land management decisions, however, involve multiple species. Reserve selection problems are generally multispecies approaches that focus on species diversity or maximizing coverage of a set of high-priority species. In land management planning, however, multispecies approaches usually involve reapplying single species models. For example, Habitat Suitability Index (HSI) models have been used to evaluate the impact of management alternatives on 10 wildlife species for the Hoosier National Forest (Dijak and Rittenhouse, this volume) or to develop bird conservation plans covering 40 species for multistate bird conservation regions (Fitzgerald et al., this volume). We are not aware of any land management planning exercise that evaluated habitat quality or animal abundance for multiple species in a single, multivariate model. The reason is that predictive statistical models would likely be too complicated, or be unrealistically simplistic, to address the diverse responses of multiple species simultaneously. Some software automates predictions for multiple species by implementing a number of single species models (Beck and Suring, this volume). An alternative to multiple single species models is a coarse filter approach or a focus on species diversity, both of which require additional assumptions and issues that we discuss in the following sections.

COARSE FILTER VERSUS FINE FILTER

Many conservation needs can be met through coarse filter approaches that focus on ecosystem representation and diversity. Coarse filters can be used where ecosystems or other high-level ecological units are the focus of conservation efforts, or where the focus is on species conservation and ecosystems are surrogates that address the needs of single or multiple species (Haufler and

Kernohan, this volume; Hicks et al., this volume; Noon et al., this volume), which is our approach here. For a single species, planners would select ecosystems or habitats as surrogates for a species abundance or viability. This approach is extended to multiple species by repeating the process for groups of species. Most projects, however, will involve the use of fine filters (i.e., species-level modeling), at some stage, whether as the primary planning approach (Fitzgerald et al., this volume), or as a component for selected high-profile species (Hicks et al., this volume), or to validate assumptions about the adequacy of coarse filters to meet species viability needs. At a superficial level, coarse filter approaches may appear like a less complex planning approach, but they rely on many untested assumptions such as the adequacy of vegetation type and structure to represent ecosystems or other ecological units (Noon et al., this volume). Multispecies approaches using a fine filter approach require models for all species of interest or a set focal species.

An alternative to the coarse filter/fine filter dichotomy is a multiscale approach that takes a stepwise approximation approach working from broad scales to successively finer scales (Probst and Gustafson, this volume). This species-based approach begins at the broadest scale in the problem with simple distribution data and then increases resolution by examining habitat gradients and occurrence information, then productivity and survival. Information is synthesized across scales for the problem and key assumptions addressed through monitoring (Probst and Gustafson, this volume).

So whether through a combination of coarse and fine filters, a stepwise multiscale approach, or a fine filter species-level approach, at some point species-level models will be required either as the primary approach or to validate assumptions made in coarser-scale analyses. Hence, much of our focus in this volume and the remaining discussion in this chapter relates to species-level approaches.

SPECIES REPRESENTATION VERSUS DIVERSITY

A focus on species diversity is one potential solution to the trade-offs involved between fine filter/species approaches versus coarse filter/ecosystem approaches. Perhaps the most popular measure of species diversity is species richness, a simple count of the number of species. However, species richness is generally not a suitable metric for conservation planning because it is generally driven by common, widespread species, which are not the species most conservation planning efforts are intended to address (Brooks et al. 2006). Furthermore, maximizing species richness at local scales can reduce species richness at larger scales (Noss 1987). So, simple diversity objectives are generally abandoned in favor of those that address species representation; we generally want all species (or other components of biodiversity) adequately protected in conservation plans (Sarkar et al. 2006). One way to address species representation in reserve design is to ensure that some target set of species pool members, typically species considered at risk or a high conservation concern, are adequately represented in the conservation plan (Flather et al., this volume). Additionally, reserves can consider those that complement, as opposed to those redundant with, species covered in existing reserves or other areas being considered (Flather et al., this volume). In land management planning, species representation is often addressed by focusing on high-concern species or selecting indicator species that serve as surrogates for a larger group of species. For example, bird conservation under the Partner's in Flight plan in North America focuses on a set of priority species (Fitzgerald et al., this volume). Land management planning on national forests in the United States has focused on threatened, endangered, and sensitive species and management indicator species, but is shifting to greater use of coarse filter approaches (Noon et al., this volume). Species representation problems can address presence/absence, abundance, or viability.

HABITAT, ABUNDANCE, OR VIABILITY

We and others in this volume have discussed the need to focus on viability (Akçakaya and Brooks, this volume; Beissinger et al., this volume; Bekessy et al., this volume; Millspaugh et al., this volume) or setting population goals to meet desired levels of viability (Fitzgerald et al., this volume; Johnson et al., this volume). Indeed, persistence is one of the key tenets to biodiversity conservation in addition to representation (Sarkar et al. 2006). However, there will be many applications where habitat will serve as a surrogate for populations and the amount of habitat or population size as a surrogate for viability. Projects dependent on existing knowledge such as the scientific literature and expert opinion will be more amenable to simpler models (i.e., habitat matrix or habitat suitability models) that predict habitat quality and not abundance. However, because of their simplicity and availability of data, habit models can be used for more species than population or viability models. Predicting relative or absolute abundance usually involves fitting a statistical model to a suitable data set and using it to predict the response variable of interest, often in the form of continuous surface maps (Fitzgerald et al., this volume; Niemuth et al., this volume). These approaches require all the usual assumptions of statistical models, and though often ignored, should consider issues of detection probabilities and potential biases if ignored. See related discussion later on conceptual versus statistical models.

The majority of projects have an ultimate goal of ensuring species viability, but default to modeling approaches that consider only habitat or abundance because of real or perceived project limitations. A lack of data about population vital rates and the impact of environmental factors on those rates often precipitates the use of habitat or abundance models. Most wildlife research and monitoring activities have focused on habitat use and abundance, in part because data to estimate productivity and survival can be more difficult to collect. We suggest projects that have an ultimate goal of ensuring species viability, but that take a habitat or abundance approach because of the preceding constraints, should attempt to develop population viability models based on their best understanding of population parameters to validate their approach for select species. Available software greatly facilitates the development of such models (Akçakaya and Brook, this volume; He, this volume; Roloff et al., this volume) and the experience of parameterizing a model and conducting simple sensitivity analyses can identify important knowledge gaps and areas of uncertainty related to the assumptions and data used in habitat or abundance-based models.

PROCEED WITH EXISTING DATA OR COLLECT NEW DATA

Often, it is tempting to postpone modeling until more and better data are available. However, we encourage moving forward with modeling efforts despite a lack of complete information (Millspaugh et al., this volume). Simple models without a full complement of data can be developed, which still allow for important evaluations, such as sensitivity analyses that can be used to guide future data collection. When data collection and needs are considered within an adaptive management framework, alternative models and key assumptions can be evaluated.

For many species, existing literature about habitat relationships can be derived from the literature. Such information can form the basis for preliminary model development and evaluation. Thus, one can often proceed with model development and application despite a lack of site-specific data. Subsequent investigation can evaluate the validity of those existing data and assumed relationships. At the very least, literature-derived estimates and relationships offer some insight into possible factors of importance and can help guide study objectives and experimental design (e.g., determination of sample size requirements). Without model validation, though, literature-derived models should be used with some suspicion.

Whereas general habitat requirements might be derived from the literature, models requiring vital rates might be more difficult to parameterize. Because habitat studies are more common and generally applicable across a species range, there might be less danger in applying simple habitat models versus viability models, which are more data hungry. Also, vital rates are often more likely to differ across a species range when one considers population processes that are site and population specific (e.g., density dependence, density-independent factors such as weather, predation, habitat fragmentation). Regardless of the approach, we believe it is prudent to make full use of existing data while acknowledging limitations and uncertainty and identifying ways of reducing both. One will never have all the data he wants or desires; however, it is necessary to move forward and make management decisions in a timely manner.

QUALITATIVE OR CONCEPTUAL VERSUS STATISTICAL MODELS

Most planners, managers, and scientists would rather have a model based on good empirical data (i.e., a statistically fit model) than a more qualitative or conceptual model based on existing knowledge in the form of literature and expert opinion. The reality is many more of the latter type models exist, and will be built, than the former. Qualitative or conceptual-based models can be based on a variety of data types (expert opinion, literature, and empirical) and built with whatever knowledge currently exists, which is a strength. So, for example, habitat matrix models have been built for literally thousands of vertebrate species across the northeastern (DeGraaf and Rudis 1986), southern (Hamel 1992), and western (Airola 1988, Fitzgerald et al., this volume; Hepinstall et al., this volume; Hicks et al., this volume) United States. GIS-based habitat suitability models, which add important landscape components, making them spatially explicit, are now available for many species (Larson et al. 2003; Rittenhouse et al. 2007; Tirpak et al. 2008).

The shortcoming of qualitative or conceptual models is that without some type of data-based validation, there is no way to assess model validity. With a statistical model we should at least know how well the model describes the data on which it was built. This assessment of the fit of a statistical model, however, can create a false sense of security and lead to the application of the model outside its true scope of inference, which could result in large prediction errors. Thus, appropriate assessment of a statistical model should consider standard metrics of model fit (e.g., deviance), but also whether the model is appropriate for the site.

Good modeling practices (see Millspaugh et al., this volume) should be used regardless of which approach is taken. We encourage hypothesis-based or mechanistic models because most models in landscape planning will be used for prediction with new data and simple correlative relationships from a single data set may not work well for this purpose. We see convergence between conceptual versus statistical models in current information-theoretic frameworks. For example, suitability indices in HSI models can serve as the basis for candidate models in an information theoretic framework evaluating model support when empirical data become available (Rittenhouse 2008). Also, this approach can guide data collection when applied in an adaptive management framework that includes model evaluation.

STATIONARY VERSUS DYNAMIC

Stationary approaches typically are either focused on current conditions; assume habitats, landscapes, or populations are not going to change significantly over the planning horizon; or assume current landscape conditions are representative of future conditions, just not in a spatially exact way. Dynamic approaches directly address landscape or population change over time. In a statistical model, this usually means fitting a model to a time series and making assumptions about the applicability of current trends to forecasting the future. In simulation modeling it involves parameterizing vital rates of populations or landscape processes to project current conditions into the future.

Others in this volume (Akçakaya and Brook, Bekessy et al., He, Hepinstall et al., Oliver et al., McKenzie et al.) and elsewhere (Akçakaya et al. 2004, Wintle et al. 2005, Pichancourt et al. 2006, Shifley et al. 2006) have demonstrated the benefits of modeling landscape change as part of wildlife conservation planning or viability assessments. A failure to account for succession in understory and overstory, natural disturbances, changes in land use, climate change, or planned management activities can result in inaccurate or biased estimates of habitat suitability, abundance, or viability. We believe dynamic approaches have great utility and will increase in use; however, we suggest users consider the following issues in their application. Dynamic modeling approaches can be a large undertaking and may not fit the time frame of the project or the objectives of the planning process. While some simple approaches undoubtedly exist, problems that consider multiple management alternatives in large landscapes can take years to assemble data, parameterize models, run the models, and compile and interpret output (Shifley et al. 2006). Second, consider how the time series generated by dynamic approaches will be used. The addition of time as an axis to analyses that already consider multiple species and multiple management alternatives may provide too much information. An alternative is to consider a single point in time, for example, at the end of the planning horizon. Dynamic landscape modeling approaches are constrained in spatial extent or require trade-offs between resolution and extent. For very large-scale planning efforts, the coarser resolution required to address large spatial extents may be too coarse to capture important spatial processes affecting the species of interest. These problems require rethinking the important dynamic process at larger spatial scales because given current computing limitations, small-scale processes simply cannot be replicated over larger spatial extents, although features such as understory vegetation growth might be important (McKenzie et al., this volume). Lastly, approaches that directly incorporate dynamic landscape modeling will almost certainly involve heuristic approaches to optimization or more likely "choosing from alternatives" approaches to decision making simply because they are too complex for true optimization approaches to landscape design.

One dynamic process that we have largely ignored in this volume is global change. The reason for this omission is that most operational land-management planning at landscape scales ignores global change, operating under the assumption that impacts resulting from land-use change and resource management practices are of greater concern under typical planning horizons. This assumption, however, may be challenged as new information on the magnitude or time frame of global change is discovered. Global change modeling generally focuses on larger scales than land management planning. Traditionally, global change modeling addressed vegetation and wildlife through modeling the impact of climate change on species distribution through characterizations of species bioclimatic envelopes. In response to critiques that factors other than climate change affect species distribution, however, models have become more complex and include factors such as biotic interactions, dispersal, and disturbance (Pearson et al. 2003, Beaumont et al. 2007). For example, a frame-based spatially explicit model (ALFRESCO) was developed to simulate landscape-level response of vegetation to interactions between fire, climate, and vegetation in the boreal forest of interior Alaska (Rupp et al. 2000, 2002). As climate change models become more realistic and step down to the landscape level, and concerns for global change impacts grow, we expect to see global change addressed by more large-scale wildlife conservation and land management planning. For example, it is likely that the incorporation of such spatio-temporal changes will become routine in future PVA modeling (Akçakaya and Brook, this volume).

OPTIMIZATION VERSUS CHOOSING FROM ALTERNATIVES

All but the simplest projects will ultimately have to address how to maximize benefits from competing objectives. Projects focused exclusively on wildlife conservation will need to address conflicts among species needs and maximizing viability given financial constraints (Haight and Gobster, this volume). Multiple-use projects will need to maximize benefits associated with species viability and other benefits such as recreation or wood or mineral production. Selecting a course of action to meet objectives can involve an optimization approach or choosing among defined alternatives. Optimization methods are generally empirical, and methods to choose among alternatives range from qualitative to empirical (Haight and Gobster, this volume).

Choosing among defined alternatives is a common approach used for national forest land management plans in the United States. A number of management alternatives are considered that span the interests of stakeholders and balance competing objectives in different ways. Information is gathered on the effects or outcomes of each alternative and can range from expert opinion to predictions from empirical models (Fig. 24-2; Dijak and Rittenhouse, this volume). A planning team considers which alternative best meets the public interests and legal mandates based on information gathered from the stakeholders and resources specialists. If objectives can be clearly defined in some measurable form, and some form of decision framework or weighting agreed upon for competing objectives, a model could be developed to choose the optimal alternative. We are not aware of any examples of this approach; one of the benefits of choosing from alternative models is the simplicity of relying on a consensus after reviewing the evidence for the alternatives. A potential important shortcoming of this approach is that the optimal solution is not likely among the

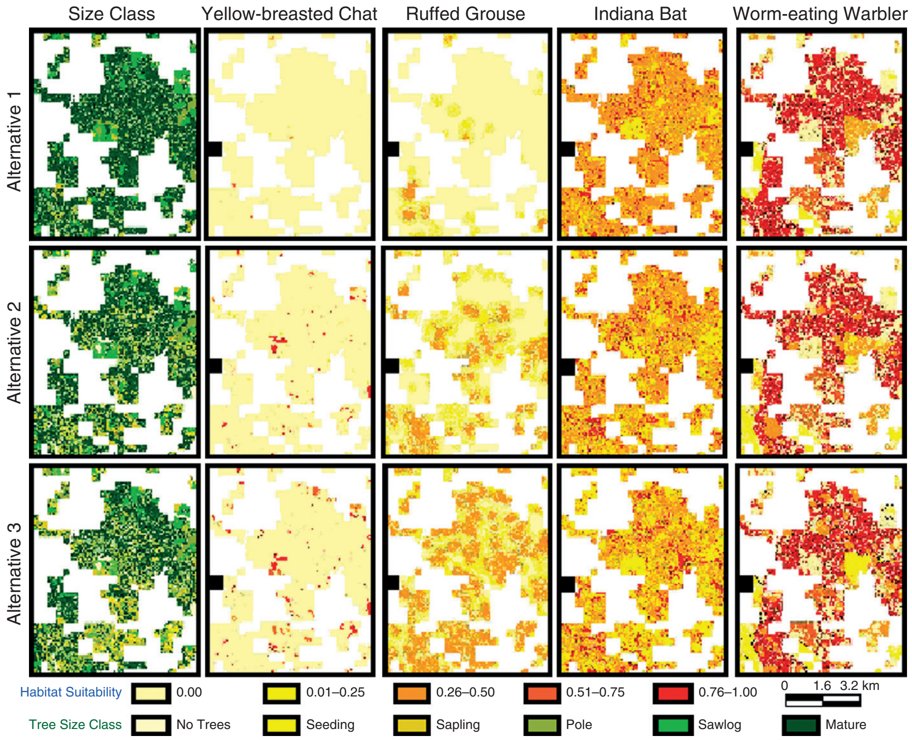

FIG. 24-2

An example of the "choosing from alternatives" approach. Maps of predicted changes in forest size-class and habitat suitability for four wildlife species under three management alternatives considered in the Hoosier National Forest Land Management Plan. Maps and graphs such as these can be used to inform selection of a preferred alternative.

few alternatives typically considered; however, we assume that one of the alternatives is close enough.

Optimization approaches seek the optimal solution based on well-defined objectives or evaluation criteria (Flather et al., this volume; Haight and Gobster, this volume). The use of optimization analysis in conservation is not common but is increasing (Rodrigues and Gaston 2002). There are two broad classes of optimization approaches: those that seek exact optimal solutions (Hof and Flather 2007) and more heuristic approaches that use iterative or stepwise algorithms that approximate an efficient design (Cabeza and Moilanen 2001). Some conservation problems can be reduced to a few important variables and solved through an exact optimal solution (i.e., Haight and Gobster, this volume). However, many conservation problems are too complex and intractable in a closed-form exact optimization model; in these cases, more heuristic approaches offer a reasonable alternative (Pressey et al. 1997). Often, more species in the analysis results in consideration of more management options and objectives, which makes the problem more difficult to solve. Somewhat ironically, these more difficult cases become too complex for any reasonable effort using mathematical optimization, and planners often fall back to simpler approaches such as choosing among a defined set of alternatives that have been characterized by an evaluation of their impact on a limited set of resources and species; models may or may not be used. However, in some cases spatial optimization has been used to maximize persistence or habitat suitability for multiple species (Noon et al., this volume).

Careful articulation of objectives is important in any decision, but especially so for empirical-based decisions. For example, maximizing (1) total habitat suitability across all species, (2) average total habitat suitability per species, or (3) average total habitat suitability per species, with the constraint that total habitat suitability must be greater than some minimum for each species, will likely result in different solutions.

CASE STUDIES

We present two case studies based on examples presented in this volume to illustrate the use of this decision framework. For illustrative purposes we present plausible reasons for the decisions made in these examples; these may or may not represent the thought process of the parties involved.

Case 1: Hoosier National Forest Plan

In Case 1 we consider one element of the Hoosier National Forest Plan: planning for the viability of a set of focal wildlife species (Rittenhouse 2008). This case study involves a land management problem where a plan will be developed to manage lands encompassing the Hoosier National Forest to meet stakeholder interests and legal mandates (Fig. 24-1A, step 1). By legal mandate, the agency must consider viability of native species, so it is a multiple species problem that addresses species representation (Fig. 24-1A, step 2 and 3). The planning team chose a fine filter approach by considering the habitat needs of a set of focal species representing a mix of management indicator species and species of concern (Fig. 24-1A, step 4). The plan considered the amount and quality of habitat, as opposed to species abundance or viability directly, because habitat could most easily be linked to the forecasted changes in forest composition and structure under the plan alternatives (Fig. 24-1A, step 5). The plan used a dynamic approach so it could address short- and long-term affects of management by estimating habitat suitability from the outputs of a dynamic landscape change model (Fig. 24-1A, step 6; Dijak and Rittenhouse, this volume). The plan had to be completed within a short time frame, so it relied on existing data and utilized habitat suitability models that could be developed from expert opinion and published studies (Fig. 24-1A, step 7 and 8). The plan considered management alternatives (Fig. 24-2) and relied on the development of a consensus based on input from resource specialists, stakeholder meetings, and deliberations of the planning team (Fig. 24-1A, step 9).

Case 2: Lake County Reserve Selection

In Case 2 we use the example presented by Haight and Gobster (this volume) where Lake County Illinois planners wanted to identify a cost-effective set of sites to be acquired to protect rare plants and animals. This is a reserve selection problem that addresses multiple species (Fig. 24-1B, step 1 and 2). The planners wanted to select sites for protection that optimized the number of species protected within a defined budget. While the quantity being optimized is a count of species, they considered only a pool of rare plants, so this is a species representation problem (Fig. 24-1A, step 3). They took a fine filter approach because they wanted to explicitly account for the occurrence of rare plants and animals (Fig. 24-1A, step 4). The approach was based on occurrence, the simplest metric of abundance, presumably because these data were available and they did not see enough additional benefit for the additional effort required to collect or analyze additional data on density or viability (Fig. 24-1A, step 5 and 7). This represents a stationary approach because the planners did not model how sites or populations might change over time (Fig. 24-1A, step 6). Instead, they relied on existing data about occurrence, which is essentially a qualitative model of occurrence, because they did not model occurrence from survey data (Fig. 24-1A, step 8). They used a true closed form optimization model to select the optimal set of sites given their budget (Haight and Gobster, this volume).

OTHER ISSUES

Many of the procedures discussed in this book require combined skills of GIS applications, vegetation modeling, wildlife-habitat modeling, and social and economic considerations. Given the data and technical expertise required, the availability of large spatial data sets and concepts being addressed, model sophistication is quickly outpacing the ability of agencies to apply them. Model and data complexity require teams of scientists, planners, and managers to work collaboratively to address planning requirements. Furthermore, the availability of high-quality data is not keeping pace with sophistication of analytical methods. As pressures increase for large-scale conservation planning, agencies will need to continually retool to meet mandates and planning needs.

SUMMARY

We outlined a decision framework for choosing among the many modeling approaches presented in this book. Elements in the framework include choosing models that address reserve selection versus land management; single species versus multiple species; species representation versus diversity; coarse filter versus fine filter; habitat, abundance, or viability; stationary versus dynamic; existing data versus new data; theoretical versus statistical; and optimization versus choosing from alternatives. After reviewing elements in the decision framework discussed in detail in other parts of the book, we worked through the decision process of two case studies. In doing so, we identified several pitfalls, such as lack of data, and offered guidance about the modeling process. Careful consideration of objectives is necessary to select the appropriate procedures, metrics, and tools in any conservation planning activity. Although reminding one to revisit objectives might seem like an overly generic recommendation, it is a fundamental consideration that drives all subsequent decisions. Without clearly articulated and followed objectives, one could easily become lost in the myriad of methods, available data, future data needs, and other decisions within the framework we have provided. One must avoid the temptation to use metrics that are simply easy to measure or readily available. Instead, it is far better to move forward with a lack of complete knowledge using the most appropriate metrics and concepts, while acknowledging the need to collect additional data to test assumptions and reduce uncertainty.

LITERATURE CITED

Airola, D. A. 1988. *A guide to the California Wildlife-Habitat Relationship system.* California Department of Fish and Game, Rancho Cordova, CA. 74p.

Akçakaya, H. R., V. C. Radeloff, D. J. Mladenoff, and H. S. He. 2004. Integrating landscape and metapopulation modeling approaches: Viability of the sharp-tailed grouse in a dynamic landscape. *Conservation Biology* 18:526–537.

Beaumont, L. J., A. J. Pitman, M. Poulsen, and L. Hughes. 2007. Where will species go? Incorporating new advances in climate modelling into projections of species distributions. *Global Change Biology* 13:1368–1385.

Brooks, J. M., R. A. Mittermeier, G. A. B. da Faonseca, J. Gerlach, M. Hoffman, J. F. Lamoreux, C. G. M. Mittermeier, J. D. Pilgrim, and A. S. L. Rodrigues. 2006. Global biodiversity conservation priorities. *Science* 313:58–61.

Cabeza, M., and A. Moilanen. 2001. Design of reserve networks and the persistence of biodiversity. *Trends in Ecology and Evolution* 16:242–248.

DeGraaf, R. M., and D. D. Rudis. 1986. New England wildlife: Habitat, natural history, and distribution. U.S. Forest Service, Northeastern Forest Experimental Station, General Technical Report NE-108. Broomall, Pennsylvania, USA.

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

Hof, J., and C. H. Flather. 2007. Optimization of landscape pattern. Pages 143–160 *in* J. Wu, and R. Hobbs, editors. *Key topics and perspectives in landscape ecology.* Cambridge University Press, Cambridge, United Kingdom.

Larson, M. A., W. D. Dijak, F. R. Thompson, III, and J. J. Millspaugh. 2003. Landscape-level habitat suitability models for 12 wildlife species in southern Missouri. U.S. Forest Service, North Central Research Station, General Technical Report NC-233, St. Paul, Minnesota, USA.

Noss, R. F. 1987. Do we really want diversity? *Whole Earth Review* 1987 (Summer):126–128.

Pearson, R. G., and T. P. Dawson. 2003. Predicting the impacts of climate change on the distribution of species: Are bioclimate envelope models useful? *Global Ecology and Biogeography* 12:361–371.

Pichancourt, J.-B., F. Burel, and P. Auger. 2006. Assessing the effect of habitat fragmentation on population dynamics: An implicit modelling approach. *Ecological Modelling* 192:543–556.

Pressey, R. L., H. P. Possingham, and J. R. Day. 1997. Effectiveness of alternative heuristic algorithms for identifying indicative minimum requirements for conservation reserves. *Biological Conservation* 80:207–219.

Rittenhouse, C. D. 2008. Wildlife response to spatial and tempered changes in forest habitat. PhD Dissertation. University of Missouri, Columbia, Missouri, USA.

Rittenhouse, C. D., W. D. Dijak, F. R. Thompson, III, and J. J. Millspaugh. 2007. Development of landscape-level habitat suitability models for ten wildlife species in the Central Hardwoods Region. U.S. Forest Service, Northern Research Station, General Technical Report NRS-4, Newtown Square, Pennsylvania, USA.

Rodrigues, A. S. L., and K. J. Gaston. 2002. Optimisation in reserve selection procedures—Why not? *Biological Conservation* 107:123–129.

Rupp, T. S., A. M. Starfield, and F. S. Chapin, III. 2000. A frame-based spatially explicit model of subarctic vegetation response to climatic change: Comparison with a point model. *Landscape Ecology* 15:383–400.

Rupp, T. S., A. M. Starfield, F. S. Chapin, III, and P. Duffy. 2002. Modeling the impact of black spruce on the fire regime of Alaskan boreal forest. *Climatic Change* 55:213–233.

Sarkar, S., R. L. Pressey, D. P. Faith, C. R. Margules, T. Fuller, D. M. Stoms, A. Moffett, K. A. Wilson, K. J. Williams, P. H. Williams, and S. Andelman. 2006. Biodiversity conservation planning tools: Present status and challenges for the future. *Annual Review of Environment and Resources* 31:123–159.

Shifley, S. R., F. R. Thompson, III, W. D. Dijak, M. A. Larson, and J. J. Millspaugh. 2006. Simulated effects of forest management alternatives on landscape structure and habitat suitability in the Midwestern United States. *Forest Ecology and Management* 229:361–377.

Tirpak, J. M., D. T. Jones-Farrand, F. R. Thompson, III, D. J. Twedt, M. D. Nelson, and W. B. Uihlein, III. 2008. Predicting bird habitat quality from a geospatial analysis of FIA data. *In* R. E., McRoberts, G. A. Reams, P. C. Van Duesen, W. H. McWilliams, eds. Proceedings of the 8th annual forest inventory and analysis symposium. U.S. Forest Service, Northern Research Station, General Technical Report WO-xx, Newtown Square, Pennsylvania, USA. *In press*.

Wintle, B. A., S. A. Bekessy, L. A. Venier, J. L. Pearce, and R. A. Chisholm. 2005. Utility of dynamic-landscape metapopulation models for sustainable forest management. *Conservation Biology* 19:1930–1943.