20 Ecoregional Assessments and Conservation Priorities

Time is money. Time, money, and commitment are what make habitat management happen in stands, landscapes, regions, and around the globe, but those resources are limited. Every forest manager has a budget and personnel limitations. Consequently, a manager will need to know where to invest those resources to have the greatest impact on the resources of interest. Getting the "biggest bang for the buck" is the approach that most managers want to take. For instance, consider a forest manager in Alabama with three primary goals: bobwhite quail, white-tailed deer, and timber. Patterns of food patches interfaced with cover are important to deer and quail (albeit at different spatial scales), but making a profit is important as well, so the problem becomes one of optimizing habitat qualities for the two game species, while ensuring profitable timber production. One way to approach this problem is to view habitat for the two species as constraints on the timber production, or alternatively, view timber production as a constraint on habitat for the two species. In either case, the resulting decision is one where one group of resources is given more value than another, and the decision resulting from the analysis can be implemented over space and time to achieve the desired goals (assuming some natural disturbance does not come along and change everything).

Now consider problems likely to occur over much larger areas of space and time. How would you decide where to provide habitat for rare species throughout their geographic range, in order to minimize risk of extinction? Or decide which parcels to buy before they are turned into housing developments? Or decide which nuclei of forests to protect from invasive species before they are overrun? Or decide how to coordinate management actions among landowners over a region to achieve biodiversity goals? Just as landscapes provide the context for stand prescriptions and regions provide the context for landscape management plans, global patterns of biodiversity provide the context for regional conservation strategies (Buchanan et al. 2011). Global patterns of biodiversity will only be conserved if the strategies are implemented among stands over landscapes and among landscapes over regions. Strategies are developed from the top down and implemented from the bottom up. Think globally, act locally. Within this context, it is often difficult to know where to invest the time, money, and commitment to achieve these regional goals (Loyola et al. 2009). Regional assessments can provide the context, and prioritization analyses can provide the guidance for investments.

ECOREGIONAL ASSESSMENTS

Ecoregions are areas of similar climate, topography, soils, and other factors influencing patterns of vegetation and the animals; the processes that support these vegetation and faunal patterns occur and recur predictably (Table 20.1, from Bailey 1980). Ecoregions are often used as the basis for assessments. Habitat conservation and management strategies are developed at the ecoregional scale, which guide landscape management plans, which guide development of stand prescriptions or local management plans.

Ecoregions are displayed as generalized areas of climatically associated patterns of vegetation (Figure 20.1). This map is one of several attempts that have been made at mapping ecoregions, each with differences as influenced by the goals of the organization funding the work. Some systems of delineating ecoregions have greater detail than others (Omernick 1995). Ecoregions are mapped as discrete entities, but in actuality they represent gradients. One will grade into another,

TABLE 20.1
US National Hierarchy of Ecological Units

Planning Scale	Utility
Ecoregion	
Global	Broad applicability for modeling and sampling
Continental	Strategic planning and assessment; international planning
Regional	
Subregion	Strategic, multiforest, statewide, and multiagency analysis and assessment
Landscape	Forest or area-wide planning and watershed analysis
Land unit	Project and management area planning and analysis

Source: From Bailey, R.G. 1980. *Descriptions of the ecoregions of the United States*. Washington DC: U.S. Department of Agriculture, Forest Service. Misc. Pub. 1391.

and no two places within any one ecoregion are the same. The devil is in the details. As you zoom in on any ecoregion, there is variability in patterns of soil, topography, climate, and vegetation, which occurs locally; hence, the need for a hierarchy of ecological units (Nesser et al. 1994, Keys et al. 1995, McMahon et al. 2001, Table 20.1). Hierarchical patterns of ecological units are useful because they do not follow political boundaries and can provide a framework for addressing issues that cross administrative and jurisdictional boundaries (Probst and Crow 1991). Using ecoregional units as the basis for assessments and development of coarse-filter conservation strategies is also intuitively appealing, because disturbance forces and recovery patterns are often more similar within an ecoregion than among regions. Although local modifications are often needed during landscape planning, ecoregional patterns provide a broad context for assigning goals and objectives that are related to the ranges of variability in ecosystem indicators (historic or future) seen in the ecoregion and can provide a logical link to population viability modeling efforts (Polasky et al. 2005; see Chapter 21).

Although ecoregions often have climatically or topographically defined boundaries, ecoregions are not a spatial scale per se. Some are large and some are small. In fact, the interaction of various ecological states and processes can all occur over a range of spatial scales. Allen and Hoekstra

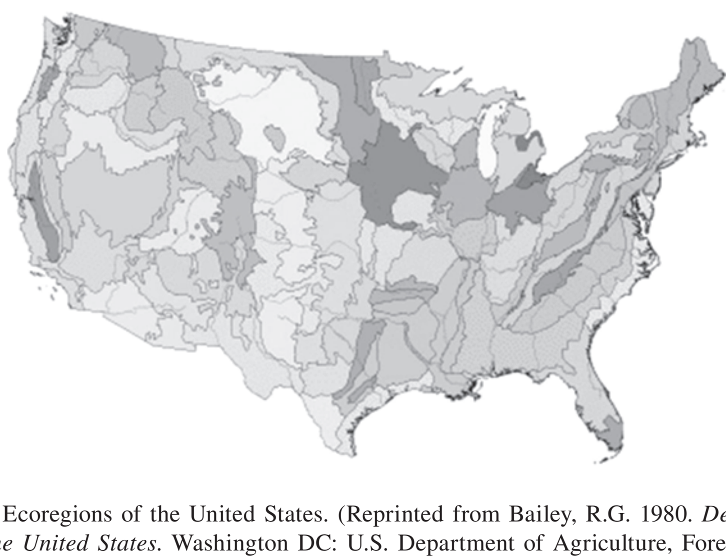

FIGURE 20.1 Ecoregions of the United States. (Reprinted from Bailey, R.G. 1980. *Descriptions of the ecoregions of the United States*. Washington DC: U.S. Department of Agriculture, Forest Service. Misc. Pub. 1391.)

(1990) and Hoekstra et al. (1991) made an excellent case that ecological functional units, such as genotypes, organisms, populations, communities, landscapes, ecoregions, and biomes, all interact over a range of spatial scales, and that there are more potential interactions among these units at small spatial scales than at large spatial scales (because the planet is only so big) (Figure 20.2). So although we can use ecoregional units as a basis for planning, ecoregional units may have, and often do have, genotypes, organisms, populations, communities, and landscapes, that occur within a limited portion of an ecoregion, or which extend beyond ecoregional boundaries. Because ecoregions are developed in a hierarchical manner, conservation strategies for various species often cross ecoregional boundaries and do not always align with those boundaries. Consider a community of large carnivores in the Rocky Mountains containing wolves, lynx, wolverines, and cougars. This community represents a collection of interacting species that clearly transcend multiple ecoregional units in their genotypes, individual organisms, and populations. Using ecoregional units may be useful, but the appropriate level in the classification hierarchy must be interfaced with the spatial domains represented within each of these species, if species conservation is a goal. Nonetheless, ecoregions continue to be used as the basis for prioritizing conservation efforts for a wide range of species (Loyola et al. 2009).

Indeed, it is the spatial scaling properties of the region and the spatial requirements of the species using the region, that interface to provide information about the potential risk of losing species due to changes in patch areas, edges, and other factors that describe the spatial complexity of a region. Wiens (1989) defined "domains of scale" or spatial patterns of patches that emerge as you perceive increasingly large areas of a region. For instance, given two regions, each with different land-use or disturbance histories, patch sizes and configuration might differ (Figure 20.3). Taking a transect from any point on the region and estimating an ecologically important metric over increasingly large areas produce a trend that should increase to an asymptote in a landscape in which all patches were uniformly distributed. But in many regions, patches are not uniformly distributed, and the trend in landscape metrics is not asymptotic (Wheatley 2010). Instead, there

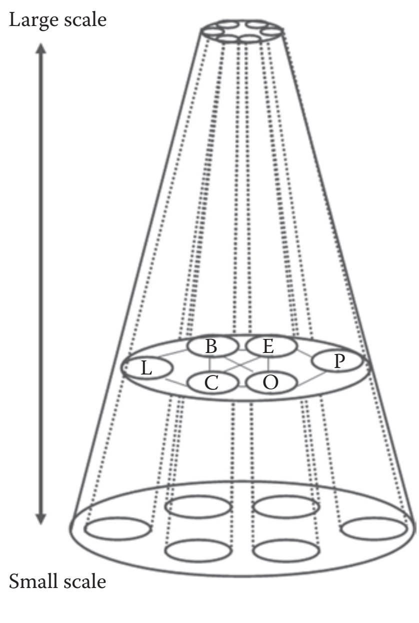

FIGURE 20.2 Organisms (O), populations (P), communities (C), landscapes (L), ecosystems (E), and biomes (B) all interact over a range of spatial scales, with many levels of interaction possible at small spatial scales than at large scales. (Redrafted and adapted from Allen, T.F.H., and T.W. Hoekstra. 1990. *Journal of Vegetation Science* 1:5–12.)

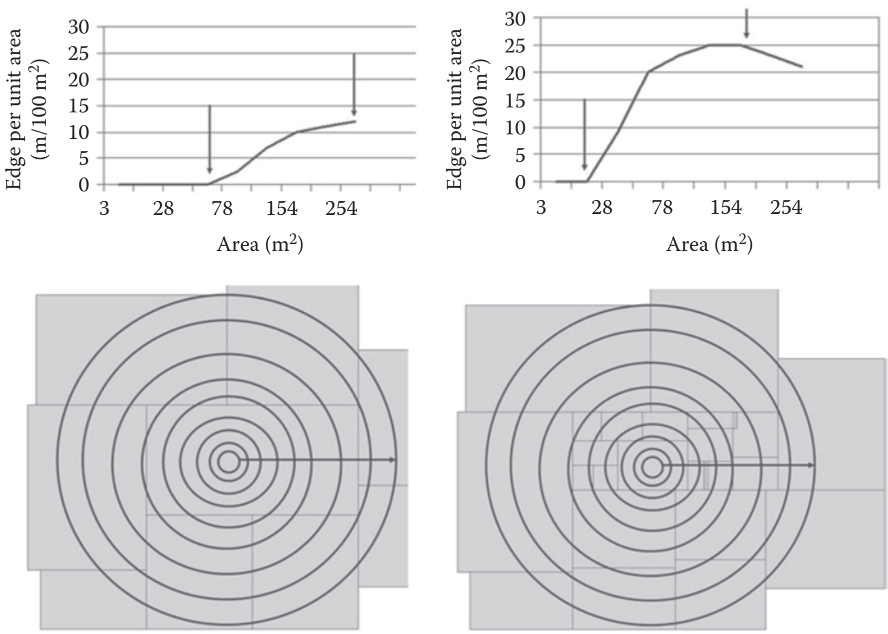

FIGURE 20.3 Simplified example of identifying domains of scales in portions of two landscapes. In a landscape where patches are uniform and evenly distributed (left), a threshold in edge density emerges at 50 m^2 and approaches an asymptote at 254 m^2. In a more heterogeneous landscape (right), two thresholds emerge at 12 m^2 and at 202 m^2 where the patch heterogeneity is expressed in two domains of scale. Depending on the domains of scale associated with a species life history, one landscape may provide higher quality habitat than another.

are thresholds that emerge from these analyses that define domains of scale. These domains can be used as information in regional assessments in two ways. First, the domains identified in current landscapes can be compared to a reference condition (or conditions), such as the conditions that might be seen under the historical range of variability or a desired future condition. An implicit assumption is that, as the domains depart from the reference condition, then there is increasing risk of losing species that are not well adapted to using new domains of scale that fall outside of the historic ranges. In a more explicit analysis, the home range, territory, or metapopulation sizes for a species can be compared to the domains of scale associated with the species' habitat over a region. The greater the disparity between the domains of scale needed by a species and those found in a region, the greater the risk to the species of not having adequate patch sizes, configuration, or connectivity.

In addition, it is often the edges between ecoregional units that can be of considerable interest during conservation planning and management prioritization. On a map, the edges between ecoregions are sharp lines due to the need to classify units, but in reality they are blurred boundaries (Bailey 1980), oftentimes with rich plant and animal communities occurring at climatic or topographic ecotones. Simply viewing the ecoregional unit as the basis for planning, without recognizing the potential importance of the ecoregional hierarchy and ecoregional ecotones, may miss important drivers of species richness. Climate change is likely to change ecoregional boundaries. Beaumont et al. (2011) indicated that of the 238 ecoregions with exceptional biodiversity, 82% are likely to be under significant stress by the year 2100. As ecoregions rearrange in response to climate, it is likely that the domains of scale will also shift, and that species shifts will first be noticed at ecoregion boundaries (Wiens and Blanchet 2010). Cross-region connectivity will be critical to allow species to migrate, if they can, as climates change.

EXAMPLES OF ECOREGIONAL ASSESSMENTS

Federal, state and NGO groups have been involved in ecoregional assessments across the United States, Canada, Australia, and other countries around the world. The scope of the assessments conducted represents a broad spectrum of spatial scales, processes, and political entities. Some, such as the Forest Ecosystem Management Assessment Team (FEMAT), limited their assessment to federal lands in the Pacific Northwest and focused largely on late successional species (FEMAT 1993). Others, such as Coastal Landscape Analysis and Modeling Systems (CLAMS) project, considered all landowners, long timeframes, and a multitude of processes and species (Spies et al. 2007). In the CLAMS approach, both past and likely future conditions are considered in the face of current and alternative future policies. The Interior Columbia Basin Ecosystem Management Planning (ICBEMP) assessment considered a huge multistate area and resulted in an assessment of forest-related ecosystem process and species over the region (Wisdom et al. 2000, Figure 20.4). States also have conducted much smaller ecoregional analyses, such as the Berkshire Ecoregion Assessment in Massachusetts (Fleming 2006). Oftentimes, when states or federal agencies are involved in assessment, the assessment stops at political boundaries although the ecoregion extends across boundaries. This was the case in the Berkshire assessment. Indeed, the scale of analyses is at times aligned with ecoregional boundaries and at times with political boundaries, but often does not consider the domains of scale of the system being assessed (Wheatley and Johnson 2009). The issue is further confounded when ecoregional analyses include international borders. FEMAT and ICBEMP largely stopped at the Canadian border, although the contributions of resources from Canada were considered as part of the context for the assessment. Nongovernment organizations (NGOs), especially The Nature Conservancy (TNC), also have used ecoregional assessments in their planning

FIGURE 20.4 U.S. Forest Service and BLM lands covered by the Interior Columbia Basin Ecosystem Management Plan. (Reprinted from the USDA Forest Service Interior Columbia Basin Ecosystem Management Plan. See Wisdom M.J. et al. 2000. Source habitats for terrestrial vertebrates of focus in the interior Columbia Basin: Broad-scale trends and management implications. USDA Forest Service General Technical Report PNW-GTR-485 for more detail.)

FIGURE 20.5 Example of a cross-border ecoregional assessment coordinated by The Nature Conservancy. Six ecoregions are included in this assessment. (From Marshall, R. et al. 2004. An ecological analysis of conservation priorities in the Apache Highlands Ecoregion. Prepared by The Nature Conservancy of Arizona, Instituto del Medio Ambiente y el Desarrollo Sustentable del Estado de Sonora, agency and institutional partners. With permission from the Arizona Chapter of The Nature Conservancy.)

and prioritization work. NGOs are not restricted to political boundaries to the degree that states, provinces, and countries might be. For instance, TNC (2001) completed a multistate assessment for the Appalachian forests of Maryland, Virginia, Pennsylvania, and West Virginia and facilitated a cross-border assessment in the southwestern United States and Mexico that included six ecoregions (Marshall et al. 2004, Figure 20.5).

Conducting an Ecoregional Analysis

Will you ever be involved in conducting an ecoregional assessment? Well, maybe, but you quite likely will be developing landscape management plans or working within management plans that are tiered to an ecoregional assessment. In order to provide a useful a context for landscape plans,

TABLE 20.2
Primary Steps in an Ecoregional Assessment

1. Identify the ecoregion and spatial extent to be included in the analysis.
2. Identify the species of conservation concern.
3. Determine the habitat associations of species.
4. Delineate the boundaries of the species range and map distribution within the range.
5. Identify the natural disturbances and human activities.
6. Identify the potential risks to species or its habitat.
7. Map the extent of individual and cumulative risk factors.
8. Identify and develop the management actions.

Source: From Wisdom, M.J., M.M. Rowland, and L.H. Suring (eds.). 2005. *Habitat Threats in the Sagebrush Ecosystem: Methods of Regional Assessment and Applications in the Great Basin*. Alliance Communications Group, Allen Press, Lawrence, KS.

ecoregional assessments should be effective in identifying likely risks to species, their habitats and plant communities over large areas. Recent systematic ecoregional assessments and associated conservation planning approaches probably have been more effective at conserving biological diversity than approaches of the past (Margules and Pressey 2000), but expectations of scientists and stakeholders are not always realized (Bottrill et al. 2012). Past approaches often resulted in a biased distribution of lands identified for protection or management specifically for biodiversity goals, with many areas occurring on lands not useful for other purposes such as high elevations and steep slopes (Scott et al. 2001).

Wisdom et al. (2005) outlined the steps for conducting an ecoregional assessment (Table 20.2), and Groves et al. (2002) proposed a process to identify the conservation areas (which may or may not require management) across ecoregions (Table 20.3). Whereas large ecoregional assessments of the past have cost millions of dollars and taken 5 or more years to complete, Groves et al.'s (2002) process has a median cost of $234,000 per plan (in 2002 U.S. dollars) and an average completion time of just less than 2 years. Cork and Tait (2009) conducted a review of known information, including an identification of high priority conservation areas, and a workshop to develop strategic priorities for biodiversity conservation across Australia. These latter two examples indicate that a regional or national assessment may take many forms and still provide the basis for development of landscape plans and stand prescriptions.

ASSESSING PATTERNS OF HABITAT AVAILABILITY AND QUALITY

Data used to set goals, develop assessments, and establish priorities often come from remotely sensed sources such as satellite imagery (e.g., LANDSAT), Light Detection and Ranging (LIDAR) data, aerial photography (orthophotos), and the resulting Geographic Information Systems data that are derived from these techniques. Consequently, the data are usually restricted to some minimum grain (perhaps a 30×30 m pixel for LANDSAT) and logical extent (simply from the standpoint of managing too many pixels of data). Using satellite data, reflectance values in various spectral bands are usually classified into conditions on the ground that are recognizable to humans as land cover classes. These classes become the basis for further associations as habitat for various species. But each species has its own habitat requirements, so one classification system is unlikely to work well for all species; indeed, Cushman et al. (2010) discourage the use of cover classes at all, recognizing that landscapes are not composed of discrete patches but rather gradients of habitat quality or resource values. Nonetheless, past efforts have classified data into land cover classes that then are related to the likely occurrence of various species and processes across the area of interest. Wildlife Habitat Relationships (WHR) models have traditionally been used to relate classified land types

TABLE 20.3
A Seven-Step Conservation Planning Framework

Step 1: Identify the conservation targets:
- Abiotic (physically or environmentally derived targets)
- Communities and ecosystems
- Species: imperiled or endangered, endemic, focal, keystone

Step 2: Collect information and identify the information gaps:
- Use a variety of sources
- Rapid ecological assessments and rapid assessment programs
- Biological inventories
- Workshops with species' experts

Step 3: Establish the conservation goals:
- Address both representation and quality
- Distribute the targets across environmental gradients
- Set a range of realistic goals

Step 4: Assess existing conservation areas:
- Gap analysis

Step 5: Evaluate an ability of conservation targets to persist:
- Use criteria of size, condition, and landscape context
- Use GIS-based "suitability indices" to assess current and future conditions

Step 6: Assemble a portfolio of conservation areas:
- Use site or area selection methods and algorithms as a tool
- Design networks of conservation areas employing biogeographic principles

Step 7: Identify the priority conservation areas:
- Use the criteria of existing protection, conservation value, threat, feasibility, and leverage to prioritize areas

Source: From Groves, C.R. et al. 2002. *Bio Science* 52:499–512.

(seral stage and plant communities) to the occurrence of each of the vertebrate species that occur in an area (DeGraaf and Yamasaki 2001, Johnson and O'Neil 2001). For many species the WHR models produce reasonable estimates of the availability of habitat for many species across a region (Block et al. 1994), but not all. Ideally classification would have to be tailored to each species to more accurately reflect the collection of habitat elements important to the species (Betts et al. 2006, Cushman et al. 2010). Further, the grain of the assessment (say 30×30 m) may be too large to reliably capture the information needed to assess habitat for some species, both because one classification system does not apply to all species and some species use habitat at scales smaller than the grain size. Consequently, WHR models may be a useful guide to patterns and changes in habitat area and configuration for species over an area, but the reliability of the information varies considerably from species to species. These approaches only provide generalized estimates of habitat availability indicating where the species could occur and not necessarily where it would find habitat of better or worse quality (as it might influence animal fitness). Indeed, knowledge of site-specific sizes, abundance, and distribution of habitat elements would be needed to understand specifically how habitat quality might change from place to place or over time in a region (McComb et al. 2002, Betts et al. 2006, Spies et al. 2007, Cushman et al. 2010). Some of this information could be extracted from air photos and from LIDAR data. These techniques can provide some information on individual trees and even smaller structures that can be seen from the air. Additional information can be derived from generally available GIS layers which includes topography, soils, hydrography, and climate. These classified images and supplementary remotely sensed data can provide the information needed to assess habitat availability for many species. Habitat element information for some species would better be assessed from the ground. For instance, the amount of cobble in a stream important to torrent salamanders or the presence of hollow trees for swifts will not be reasonably reflected in remotely sensed data. Ground plot information is needed.

Ground plot data are systematically collected from multi-resource inventories such as Forest Inventory and Analysis data on private (and many public) forest lands in the United States and these efforts can be used to infer patterns of habitat availability for species over space and time (Ohmann et al. 1994). In addition, many industrial land managers have continuous forest inventory plots distributed across their properties to monitor tree growth and death, and these plots can be adapted to allow collection of site-specific data on habitat elements as well. Of course these ground inventories are samples and not inventories, and so they have been of limited value when representing habitat availability in situations where both fine-scale habitat elements as well as landscape composition and structure might be important to a species.

Ground plot data have been interfaced with remotely sensed data to allow representation of habitat elements across complex regions (Ohmann and Gregory 2002, Spies et al. 2007, Ohmann et al. 2011). Using this approach, the ground plot data are georeferenced to the physical location, topography, climate, reflectance values, and many other features on GIS layers to create a subset of "informed" pixels (pixels with a ground plot within them). For all pixels that do not have associated ground data ("uninformed pixels"), the same descriptive characteristics are also estimated, but of course there are no corresponding ground plot data associated with the uninformed pixels. To provide a "seamless" representation of ground plot information, characteristics of informed pixels are used to "inform" those pixels without ground plot data that are most similar in these descriptive characteristics. Hence, fine-scaled ground plot data can be imputed to all pixels in the extent of the assessment (Ohmann and Gregory 2002, Ohmann et al. 2011). Once ground plot data have been assigned to the uninformed pixels, then the pixels can be reclassified based on the imputed ground plot data to create species-specific habitat quality maps across the planning area (McComb et al. 2002, Spies et al. 2007, Cushman et al. 2010). These maps can then be used as the basis for assessing net gains and losses of habitat over space and time as well as population viability analysis (see Chapter 21) for species of high risk of being lost from the area in the future.

Prioritizing Management and Assessing Policies

How would you decide which tools to use and approaches to follow to ensure that your biodiversity conservation goals will be effective? The tools available to assist in decision making for biodiversity protection have exploded in number and complexity over the past decade. Gordon et al. (2004) identified over 50 decision support tools that could be used to assist in biodiversity conservation and that number is increasing annually. Choosing which to use, if any, is an overwhelming task and is highly dependent on the specific questions, goals and objectives of the assessment (Johnson et al. 2006). Below are examples of a few commonly used and powerful approaches to assessments and prioritization.

Coarse-Filter Approach

There are numerous examples of how estimates of habitat patterns, availability and quality have been used to provide a means of prioritizing management decisions. These same techniques often can be used to assess alternative management plans or policies across the area of assessment. Since time and money are usually limited when making decisions regarding management to conserve biodiversity, prioritization of the areas to manage or protect becomes paramount.

One such approach is the Conservation Assessment and Prioritization System (CAPS) which uses a coarse filter approach to parcel prioritization (Gordon et al. 2004). The CAPS approach was developed by Dr. Kevin McGarigal and uses potential biodiversity valuation that applies "biodiversity screens" to each patch in the landscape. These screens are applied to a map of predicted natural communities modeled from remotely sensed and GIS data. Biodiversity screens are models

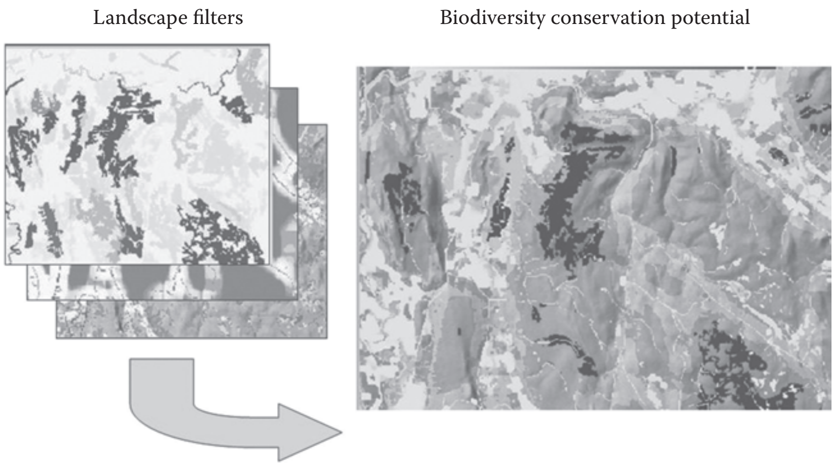

FIGURE 20.6 Example of the application of biodiversity screens or filters to a landscape resulting in the identification of high priority blocks for management or protection. (Provided by Dr. Kevin McGarigal. With permission.)

that reflect the content, context, spatial character, or condition of a patch to arrive at an index of potential biodiversity value. Stakeholders are involved in deciding how various parameters of the screens guide the identification of high priority patches in the region. Parameters such as the size of a natural community patch, edge contrast, edge density, its proximity to water, the soil type, or road density (among many others) can be identified and weighted to help identify priority patches for management or protection. The result of applying a set of screens is a biodiversity value ranging from 0 (low value for biodiversity conservation) to 1 (high value) for each patch on the landscape that then can be used to highlight those patches of highest value (Figure 20.6). The resulting high priority patches represent areas that may receive special management practices, or could be placed in reserves, conservation easements, or purchased from private landowners to protect species associated with the priority patch characteristics. The species receiving protection include species that are known to be associated with the priority patches as well as those represented in the "hidden diversity," or those species assumed to be associated with these patch conditions but which have not yet been identified. Further, the approach has been used not only in ecoregional assessments (e.g., the Berkshires), but also in mitigation to replace areas gobbled up by roads and development with patches of appropriate sizes and conditions.

Integrated Coarse-, Meso-, and Fine-Filter Approaches

Many assessments use a combination of coarse-, meso-, and fine-filter approaches to understand the current conditions across complex ecoregions. Some, such as the Willamette Alternative Futures approach (Hulse et al. 2002) used the likely changes in abundance and distribution of vertebrates across the ecoregion as a primary assessment of current and future effects of alternative future landscapes. They also considered the areal extent and distribution of various plant communities across the planning area, but did not assess the landscape metrics associated with the patches in a patch prioritization manner such as used in CAPS. Nonetheless the results of this effort have been widely used to inform land use planning decisions in the region so that planners can consider the effects on forest land and potential impacts on biodiversity of land use decisions. Another assessment, the Coastal Landscape Analysis and Modeling Systems Project (Spies et al. 2007) was designed to analyze the ecological and socio-economic consequences of various forest policies across multiple ownerships. The process includes a complex set of interacting models that consider the disturbance and regrowth of forests as guided by forest management policies and

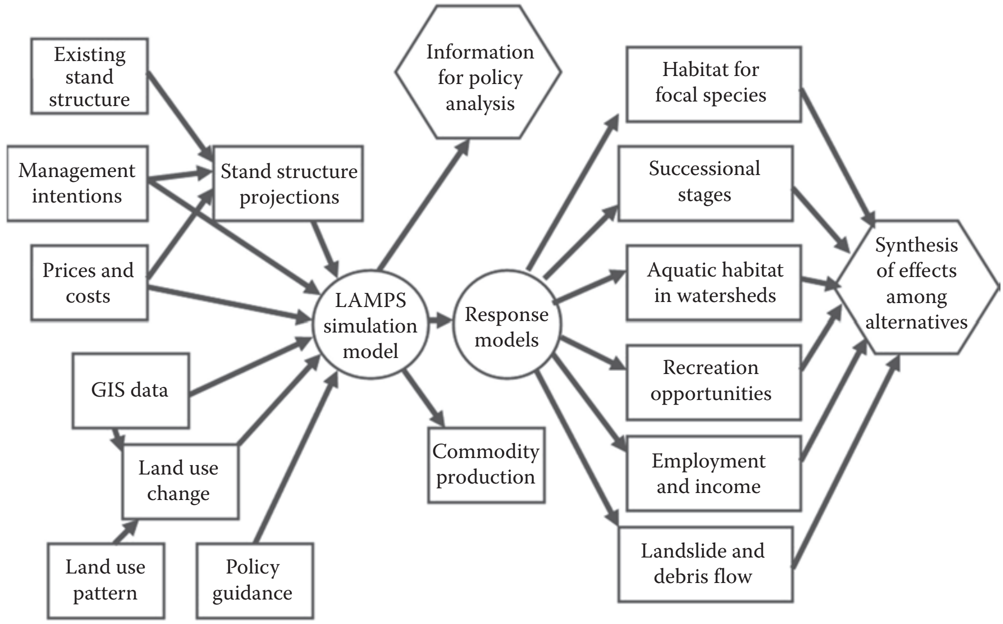

FIGURE 20.7 Complex interacting models produce current and likely future estimates of plant communities, habitat for focal species, and economic goods and services for the Oregon Coast Range under alternative forest policies. (From Spies et al. 2002. *Congruent Management of Multiple Resources: Proceedings from the Wood Compatibility Initiative workshop.* USDA Forest Service Gen. Tech. Rep. PNW-GTR-563.)

the resulting patterns of plant communities and habitat quality for focal species across the region (Figure 20.7). The results can not only be used to assess alternative forest policies but also identify locations in the region which might be particularly important as core patches or linkages across complex multi-ownership landscapes. The projected changes in plant communities (as a coarse filter index to protecting hidden diversity) as well as changes in habitat quality and distribution for focal species (those selected to represent certain ecological associations) are used to compare policy alternatives (Figure 20.8).

These approaches can be influential with stakeholders because the ability to map plant communities, habitat for various species, other resources, and land ownership can more directly engage stakeholders with scientists around visual portrayals of these resources over space and time (Wright et al. 2009).

Fine-Filter Approaches

Other approaches to ecoregion assessments take a species by species approach to identifying areas for particular management or protection. Because most of the species assessments rely on WHR models to develop maps of occurrence of species, the underlying maps can also be used as a coarse filter assessment as well. One such approach is a nationwide effort called Gap Analysis (Scott et al. 1993). The goal of Gap Analysis is to "keep common species common" by identifying those species and plant communities that are not adequately represented on existing conservation lands. By identifying habitat for all vertebrates in a region, Gap Analysis provides information that can be used to make decisions regarding vertebrate species conservation and management.

Gap Analysis consists of three main data layers, a landcover layer, a layer showing the predicted distributions of vertebrate species, and a stewardship layer (Figure 20.9). These layers are used in a Gap Analysis consisting of three primary steps. The first step is to map plant communities to develop a landcover layer. Landcover is mapped using satellite data as well as other supporting information from existing GIS layers, air photos, and ground plot data.

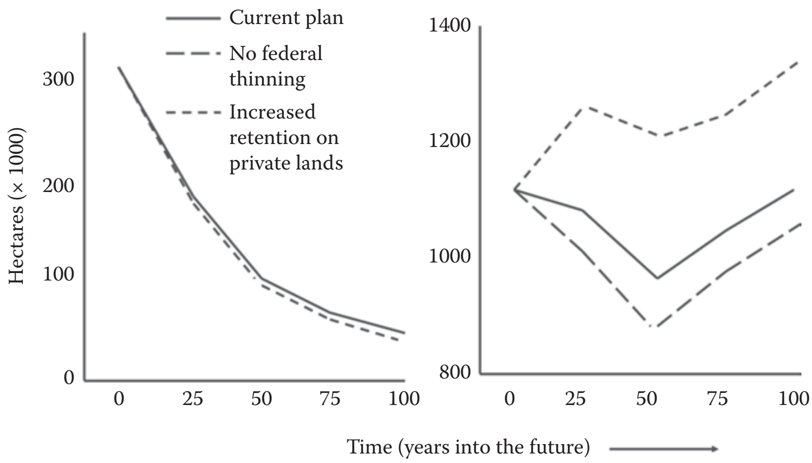

FIGURE 20.8 Projections of habitat availability for an example focal species (olive-sided flycatchers, right) and a plant community type (hardwoods, left) in the Oregon Coast Range, under three policy alternatives: current policies (solid), no thinning allowed on federal lands (large dashes), and green tree retention on private lands (small dashes). (From Spies, T.A. et al. 2007. *Ecological Applications* 17: 48–65.)

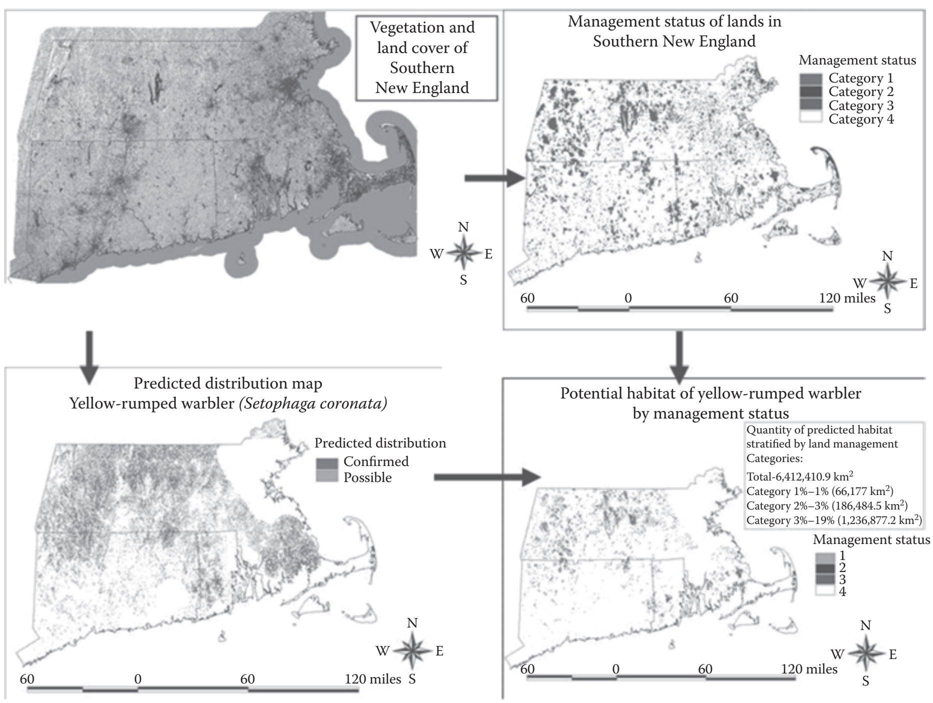

FIGURE 20.9 Steps in the process of Gap Analysis. Maps of land cover using remotely-sensed and ground-plot data (top left), species distribution based on geographic ranges, and habitat relationships (top right), and management status (bottom left) are overlain to identify gaps in the protection network for each species (bottom right). (From Zuckerberg B., C.R. Griffin, and J.T. Finn. 2004. A Gap Analysis of southern New England: An analysis of biodiversity for Massachusetts, Connecticut, and Rhode Island. Final contract report to USGS Biol. Resour. Div., Gap Analysis Program, Moscow, Idaho. With permission.)

The second step is to map predicted distributions of vertebrate species known to breed or use habitat in the region. Known, probable, and possible occurrences are used to define the geographic range of each species. Then a WHR model is developed for each species that relates the land cover data to the likely occurrence of the species across the region. The process does not usually include any assessment of habitat quality or viability.

The third step of a gap analysis is to assign a land stewardship rank between one and four to each patch on the assessment area. Status one lands have the highest degree of management for conservation, status four lands have the lowest. Stewardship ranks are based on the long-term intent of the managing entity (owner or steward). Ranks are based on (Scott et al. 1993)

- Permanence of protection from conversion of “natural” land cover to “unnatural” (human-induced barren, arrested succession, cultivated exotic-dominated).
- Amount of the tract protected, with 5% allowance for intensive human use.
- Inclusiveness of the protection, that is, single feature such as wetland versus all biota and habitat
- Type of management program and degree that it is mandated or institutionalized.

The fourth step is to analyze the representation of each species (or plant community) in areas managed for the long-term maintenance of biodiversity. To accomplish this, maps showing animal and plant community distributions are intersected with stewardship maps to identify areas where the species that are not receiving protection based on management status or appropriate management could occur. Identification of high priority areas for protection or management can be based on individual species (Figure 20.9) or on species richness patterns (Figure 20.10). Gap analysis has been completed for every state in the United States and composite assessments across state lines now allow ecoregional analyses. The gap analysis approach has been combined with other biological assessments to allow national mapping of biodiversity indicators (Boykin et al. 2012).

Utility and Effectiveness of Ecoregional Assessments

Of course the ultimate test of effectiveness is in conserving species and preventing listing of species in the future through hierarchical planning, implementation, and monitoring. Bottrill et al.

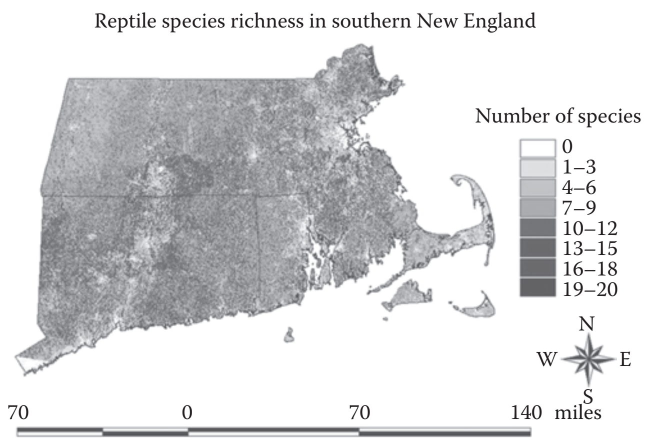

FIGURE 20.10 Example of a composite map of species-richness for reptiles in southern New England. Protection or management of darker areas is more likely to capture more species than lighter areas. (From Zuckerberg B., C.R. Griffin, and J.T. Finn. 2004. A gap analysis of southern New England: An analysis of biodiversity for Massachusetts, Connecticut, and Rhode Island. Final contract report to USGS Biol. Resour. Div., Gap Analysis Program, Moscow, Idaho. With permission.)

(2012) found that the expectations that species would be conserved were not always realized but that additional benefits accrued from ecoregional assessments including improvements in social interactions, attitudes toward conservation, and institutional knowledge. Few regional assessments have monitored effectiveness sufficient to provide evidence for having “saved” species that otherwise would have been regionally or globally eliminated. Measuring effectiveness requires that measurable goals are set to quantify success or failure (Tear et al. 2005). Only through monitoring over long time periods can effectiveness be assessed, and even then how can we know what we did not lose? Monitoring and evaluation provides an opportunity for accountability, as well as an opportunity to learn from past efforts (Botrill and Pressey 2012). Much of the effectiveness of assessment and planning is based on assumption and faith that to plan objectively and monitor the quantitative objectives is less risky than following unstructured approaches to species conservation.

SUMMARY

Ecoregional assessments are often used as a context for development and implementation of landscape management plans which provide direction for development and implementation of stand prescriptions. Ecoregions are hierarchically defined classes of vegetation based largely on climatic and other physical features that seem to drive patterns of plant communities. The utility of ecoregional assessments in providing useful contexts for other efforts are highly dependent on matching the appropriate spatial hierarchy with the spatial and temporal scales associated with the species and communities of concern. Assessments can use coarse-, fine-, and meso-filter approaches but most use some combination of these approaches. Usually these approaches are designed to prioritize areas for protection or management. The degree to which these efforts have been effective in protecting species is questionable, but the knowledge gained from the efforts has been significant. The effectiveness of such approaches in conserving species remains largely unknown and will likely only come after years of monitoring over large areas.

REFERENCES

Allen, T.F.H., and T.W. Hoekstra. 1990. The confusion between scale-defined levels and conventional levels of organization in ecology. *Journal of Vegetation Science* 1:5–12.

Bailey, R.G. 1980. *Descriptions of the Ecoregions of the United States*. U.S. Department of Agriculture, Forest Service. Misc. Pub. 1391, Washington, DC.

Beaumont, L.J., A. Pitman, S. Perkins, N. E. Zimmermann, N.G. Yoccoz, and W. Thuiller. 2011. Impacts of climate change on the world’s most exceptional ecoregions. *Proceedings of the National Academy of Sciences, USA* 108:2306–2311.

Betts, M.G., G.J. Forbes, A.W. Diamond, and P.D. Taylor. 2006. Independent effects of fragmentation on forest songbirds: An organism-based approach. *Ecological Applications* 16:1076–1089.

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.

Bottrill, M.C., M. Mills, R.L. Pressey, E.T. Game, and C. Groves. 2012. Evaluating perceived benefits of ecoregional assessments. *Conservation Biology* 26:851–861.

Bottrill, M.C., and R.L. Pressey. 2012. The effectiveness and evaluation of conservation planning. *Conservation Letters* 5:407–420.

Boykin, K.G., W.G. Kepner, D.F. Bradford, R.K. Guy, D.A. Kopp, A.K. Leimer, E.A. Samson, N.F. East, A.C. Neale, and K.J. Gergely. 2012. A national approach for mapping and quantifying habitat-based biodiversity metrics across multiple spatial scales. *Ecological Indicators* 33:139–147.

Buchanan, G.M., P.F. Donald, and S.H.M. Butchart. 2011. Identifying priority areas for conservation: A global assessment for forest-dependent birds. *PLoS ONE* 6(12):e29080.

Cork, S., and J. Tait. 2009. Rapid desktop assessment of Australian ecosystem and biodiversity conservation opportunities. Consultancy Report to the Australian Conservation Foundation. Carlton, Victoria, Australia.

Cushman, S., J.S. Evans, K. McGarigal, J.M. Kiesecker, and M. Joseph. 2010. *Toward Gleasonian Landscape Ecology: From Communities to Species, from Patches to Pixels*. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, 2010.

DeGraaf, R.M., and M. Yamasaki. 2001. *New England Wildlife: Habitat, Natural History and Distribution*. University Press of New England, Hanover, NH.

Fleming, M. 2006. *Landscape Assessment and Forest Management Framework: Berkshire Ecoregions in Massachusetts*. Executive Office of Environmental Affairs, Boston, MA.

Forest Ecosystem Management Assessment Team (FEMAT). 1993. *Forest Ecosystem Management: An Ecological, Economic, and Social Assessment*. U.S. Government Printing Office: 1993-793-071.

Gordon, S.N., K.N. Johnson, K.M. Reynolds, P. Crist, and N. Brown. 2004. Decision support systems for forest biodiversity: Evaluation of current systems and future needs. Final Report Project A 10. National Commission on Science and Sustainable Forestry.

Groves, C.R., D.B. Jensen, L.L. Valutis, K.H. Redford, M.L. Shaffer, J.M. Scott, J.V. Baumgartner, J.V. Higgins, M.W. Beck, and M.G. Anderson. 2002. Planning for biodiversity conservation: Putting conservation science into practice. *Bio Science* 52:499–512.

Hoekstra, T.W., T.F.H. Allen, and C.H. Flather. 1991. Implicit scaling in ecological research. *Bio Science* 41:148–154.

Hulse, D.W., S.V. Gregory, and J.P. Baker. (ed.). 2002. *Willamette River Basin Planning Atlas: Trajectories of Environmental and Ecological Change*. Oregon State University Press, Corvallis, OR, USA.

Johnson, D.H., and T.A. O'Neil. (managing directors), 2001. *Wildlife-Habitat Relationships in Oregon and Washington*. Oregon State University Press, Corvallis.

Johnson, K.N., S. Gordon, S. Duncan, D. Lach, B. McComb, and K. Reynolds. 2006. Conserving creatures of the forest: A guide to decision making ad decision models for forest biodiversity. National Commission on Science for Sustainable Forestry Final report. NCSSF, Washington, DC.

Keys, J.E., Jr., C.A. Carpenter, S.L. Hooks, F.G. Koenig, W.H. McNab, W.E. Russell, and M.L. Smith. 1995. *Ecological Units of the Eastern United States: First Approximation*. Geometronics, Southern Region, USDA Forest Service, Atlanta, Georgia.

Loyola, R.D., L.G.R. Oliveira-Santos, M. Almeida-Neto, D.M. Nogueira, U. Kubota, Diniz-Filho, J.A.F., and T.M. Lewinsohn. 2009. Integrating economic costs and biological traits into global conservation priorities for carnivores. *PLoS ONE* 4(8):e6807.

Margules C.R., and R.L. Pressey. 2000. Systematic conservation planning. *Nature* 405:243–253.

Marshall, R. M., D. Turner, A. Gondor, D. Gori, C. Enquist, G. Luna, R. P. Aguilar et al. 2004. An ecological analysis of conservation priorities in the Apache Highlands Ecoregion. Prepared by The Nature Conservancy of Arizona, Instituto del Medio Ambiente y el Desarrollo Sustentable del Estado de Sonora, agency and institutional partners.

McComb, W.C., M. McGrath, T.A. Spies, and D. Vesely. 2002. Models for mapping potential habitat at landscape scales: An example using northern spotted owls. *Forest Science* 48:203–216.

McMahon, G., S.M. Gregonis, S.W. Waltman, J.M. Omernik, T.D. Thorson, J.A. Freeouf, A.H. Rorick, and J.E. Keys. 2001. Developing a spatial framework of common ecological regions for the conterminous United States. *Environmental Management* 28(3):293–316.

Nesser, J., L. Freehouf, W. Robbie, T.M. Collins, C.B. Goudey, R. Meurisse, W.H. McNab et al. 1994. Ecoregions and subregions of the United States. in R.G. Bailey, P.E. Avers, T. King, and W.H. McNab (eds.). *USDA Forest Service*. Map published by U.S. Geological Survey.

Ohmann, J.L., and M.J. Gregory. 2002. Predictive mapping of forest composition and structure with direct gradient analysis and nearest neighbor imputation in coastal Oregon, USA. *Canadian Journal of Forestry Research* 32:725–741.

Ohmann, J.L., M.J. Gregory, E.B. Henderson, and H.M. Roberts. 2011. Mapping gradients of community composition with nearest-neighbour imputation: Extending plot data for landscape analysis. *Journal of Vegetation Science* 22:660–676.

Ohmann, J.L., W.C. McComb, and A.A. Zumrawi. 1994. Snag abundance for primary cavity-nesting birds on nonfederal forest lands in Oregon and Washington. *Wildlife Society Bulletin* 22:607–619.

Omernick, J.M. 1995. Ecoregions: A framework for managing ecosystems. *George Wright Forum* 12(1):35–51.

Polasky, S., E. Nelson, E. Lonsdorf, P. Fackler, and A. Starfield. 2005. Conserving species in a working landscape: Land use with biological and economic objectives. *Ecological Applications* 15:1387–1401.

Probst, J.R. and T.R. Crow. 1991. Integrating biological diversity and resource management. *Journal of Forestry* 89:12–17.

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

Scott, J.M., F.W. Davis, R.C. McGhie, R.G. Wright, C. Groves, and J. Estes. 2001. Nature reserves: Do they capture the full range of America's biological diversity? *Ecological Applications* 11:999–1007.

Spies, T.A., K.N. Johnson, P. Bettinger, W.C. McComb, J.L. Ohmann, and G.H. Reeves. 2002. Challenges to integrating wood production biodiversity, and other socioeconomic values at broad scales: An example from Coastal Oregon. Pages 13–26 in A.C. Johnson, R.W. Haynes, and R.A. Monserud, (eds.). 2002. *Congruent Management of Multiple Resources: Proceedings from the Wood Compatibility Initiative workshop*. USDA Forest Service Gen. Tech. Rep. PNW-GTR-563.

Spies, T.A., B.C. McComb, R. Kennedy, M. McGrath, K. Olsen, and R.J. Pabst. 2007. Potential effects of forest policies on terrestrial biodiversity in a multi-ownership province. *Ecological Applications* 17: 48–65.

Tear, T.H., P. Kareiva, P.L. Angermeier, P. Comer, B. Czech, R. Kautz, L. Landon et al. 2005. How much is enough? The recurrent problem of setting measurable objectives in conservation. *Bioscience* 55:835–849.

TNC [The Nature Conservancy]. 2001. *Ecoregions of North America*. The Nature Conservancy, Western Conservation Science Center, Boulder, CO.

Wiens, J. 1989. Spatial scaling in ecology. *Functional Ecology* 3:385–397.

Wiens, J.A., and D. Bachelet. 2010. Matching the multiple scales of conservation with the multiple scales of climate change. *Conservation Biology* 24:51–62.

Wheatley, M. 2010. Domains of scale in forest-landscape metrics: Implications for species-habitat modeling. *Acta Oecologica* 36:259–267.

Wheatley, M., and C. Johnson. 2009. Factors limiting our understanding of ecological scale. *Ecological Complexity* 6:150–159.

Wisdom M.J., R.S. Holthausen, B.C. Wales, C.D. Hargis, V.A. Saab, D.C. Lee, W.J. Hann et al. 2000. Source habitats for terrestrial vertebrates of focus in the interior Columbia Basin: Broad-scale trends and management implications. USDA Forest Service General Technical Report PNW-GTR-485.

Wisdom, M.J., M.M. Rowland, and L.H. Suring (eds.). 2005. *Habitat Threats in the Sagebrush Ecosystem: Methods of Regional Assessment and Applications in the Great Basin*. Alliance Communications Group, Allen Press, Lawrence, KS.

Wright, D.J., S.L. Duncan, and D. Lach. 2009. Social power and GIS technology: A review and assessment of approaches for natural resource management. *Annals of the Association of American Geographers* 99:254–272.

Zuckerberg B., C.R. Griffin, and J.T. Finn. 2004. A gap analysis of southern New England: An analysis of biodiversity for Massachusetts, Connecticut, and Rhode Island. Final contract report to USGS Biol. Resour. Div., Gap Analysis Program, Moscow, Idaho.