CHAPTER

Geographic Approaches to Biodiversity Conservation: Implications of Scale and Error to Landscape Planning

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Curtis H. Flather, Kenneth R. Wilson,
and Susan A. Shriner

Conservation science is concerned with understanding why distribution and abundance patterns of species vary in time and space. Although these patterns have strong signatures tied to the availability of energy and nutrients, variation in climate, physiographic heterogeneity, and differences in the structural complexity of natural vegetation, it is becoming more difficult to ignore the role that humans play in shaping the composition of species assemblages across landscapes (Gaston 2006). The amount of net primary productivity that goes directly to support humans has become a common, if not contentious, measure of the human footprint on ecosystems. Mean estimates of the proportion of total terrestrial net primary productivity that is appropriated by humans range from 25–40% (Vitousek et al. 1986, Rojstaczer et al. 2001, Imhoff et al. 2004), and the more that is co-opted by humans, the less there is available to support other species (Haberl et al. 2002, Gaston et al. 2003). Although these estimates have low precision (Haberl et al. 2002), there is widespread agreement that human impacts on ecosystems are substantial and growing (Laurance 2001, Wackernagel et al. 2002, Palmer et al. 2004).

The impacts of human activities on biodiversity are projected to have broad global ramifications (Sala et al. 2000), but the spatial extent of those impacts will be uneven (Cincotta et al. 2000, Imhoff et al. 2004, Evans et al. 2006). Similarly, an uneven spatial distribution of biodiversity is among the most conspicuous of patterns in macroecology (Gaston 2000, Myers et al. 2000). It is this dual pattern of regional concentrations of biodiversity and regional concentrations of human impact—areas of concentration that are often spatially correlated (Balmford et al. 2001, Luck et al. 2004, Gaston 2006)—that has encouraged the development of geographic approaches to conservation and fostered an underlying "hope" that a significant portion of species diversity could be conserved in a relatively small fraction of the landscape (Reid 1998).

A geographic perspective has also been triggered by the rapidly growing availability of spatially explicit data on the occurrence and abundance of species (Blackburn and Gaston 1998, Pärtel 2006). In the >15 years since urgent calls for spatially explicit data to assist land managers and policy makers with broad-scale environmental problems (Brown and Roughgarden 1990, Lubchenco et al. 1991), there has been substantial progress on making ecological data sets more readily available in digital format (Graham et al. 2004). The global coverage of these data is far from complete (Hortal et al. 2007, Soberón et al. 2007), but where they do exist the potential uses of geographic data for testing ecological hypotheses and describing nature are numerous (Guisan and Thuiller 2005). A common practical application of these data has been the development of conservation prioritization schemes that ultimately lead to geographically explicit conservation designs.

Establishment of conservation areas, whether they are focused strictly on biodiversity conservation or on conservation that allows for some degree of multiple-use resource management, has become a vital component of most regional strategies to mitigate the oft cited erosion of contemporary biodiversity resources. However, conservation of all resources is impossible due to limited financial resources. Thus, managers are frequently confronted with the problem of "... where should scarce conservation resources be spent?" On the surface this seems a simple question, but unequivocal answers have eluded conservation scientists for a number of reasons. Two important confounding factors in conservation design are scale and error.

Because scale affects the detection of biodiversity patterns across broad geographic areas (Willis and Whittaker 2002), it is also expected to affect our choice of where to focus conservation activities (Shriner et al. 2006). On the other hand, error in our understanding of species occupancy across the landscape, error in our understanding of the environmental attributes that are important to species habitat selection, and error in our measurement of species occurrence or habitat, all contribute to uncertainty in our characterization of biodiversity patterns and the conservation strategies derived from those patterns.

In this chapter we review the implications of scale and error effects on conservation design. Then we provide an overview of geographic-based conservation approaches before examining scale and error effects in detail using data from our work on biodiversity patterns in the southwestern United States. Finally, we present our thoughts on the implications of scale and error effects to conservation planning and some suggestions for future research needs.

GEOGRAPHIC APPROACHES TO BIODIVERSITY CONSERVATION

Setting Context

A geographical perspective in ecology and applied conservation has a long history. Early 19th and 20th century phyto- and zoogeographers focused on documenting the distributional patterns of flora and fauna for the purpose, among others, of delineating realms of biotic similarity according to climatic, physiographic, and evolutionary criteria (Hooker 1859, Wallace 1876, Shelford 1913). It is tempting to classify this early work as merely descriptive—a view promoted by some contemporaries who have characterized this perspective as "the search for patterns of animal and plant life that can be put on a map" (MacArthur 1972*b*:1). However, it is clearly more than simple description. There are numerous examples where explanation for the observed patterns in species occurrence is tied to land mass proximity, dispersal capacity, dispersal agents, allometry, energetics, and evolutionary principles (Spellerberg and Sawyer 1999, Gaston and Blackburn 2000).

How humans may have influenced these geographic patterns has, at times, been ignored (Stott 1984, Spellerberg and Sawyer 1999). For some ecological questions, this is legitimate because an understanding of the factors and processes that affect the natural geography of biodiversity is important (Gaston and Blackburn 2000:295–300). However, accounting for how humans alter this natural geography is equally important if we are to counteract the erosion of biodiversity that is attributed to human activities (Rapport et al. 1985, Cox and Moore 1993, Balmford et al. 1998). So while we do not deny that interesting ecological patterns can be studied in the absence of invoking any human causation, the focus of this chapter is on using geographic-based conservation to stem biodiversity losses attributable to human influences. Thus, we will not discuss floral or faunal realms, range-abundance relationships, latitudinal gradients of species diversity, or species-area relationships, all of which are topics in geographical ecology (MacArthur 1972*b*, Gaston and Blackburn 2000). Rather, we focus on conservation planning issues that are motivated explicitly by human actions, are relevant over broad landscapes, and have a spatially explicit, and therefore geographic, component.

Another context setting issue for our chapter concerns the word "biodiversity." A question that is basic to any conservation plan is: "What exactly are we proposing to conserve?" Answering this question requires the definition of conservation targets—those biodiversity features that we wish to ensure long-term persistence of through conservation plan implementation (Groves 2003). Historically, conservation targets have focused on species, populations, ecosystems, scenery, landscapes, and perhaps the most inclusive target, biodiversity

(Bakker et al. 2000, Redford et al. 2003). In this chapter we restrict our discussion to cases in which species are the primary focus of biodiversity conservation. However, our discussion and examples are not unique to a species focus. Ecosystems (Hoekstra et al. 2005) or genes (Neel and Cummings 2003)—the other mentioned elements that round out the triad of biodiversity features (Dirzo and Raven 2003)—could be subject to the same geographic perspective as reviewed in this chapter.

A final context setting issue concerns the notion of "reserves" in geographic conservation planning. Reserves are often defined as lands strictly managed for the conservation of biological resources and permanently protected from human development (Noss et al. 1999). Although reserves are most certainly a critical component of large-scale conservation planning efforts (Pimm and Lawton 1998, Noss et al. 1999), sole reliance on reserves will be insufficient to meet conservation objectives because of biological, economic, social, and political constraints (Langholz and Lassoie 2001, Liu et al. 2001). Consequently, it is important to accommodate a continuum of human uses within units of land comprising the conservation plan. Thus, we use the term "conservation area" (*sensu* Groves 2003) to define geographic units that are to be managed in a way that maintains the biodiversity features (in our case, species) associated with those units. We will restrict our usage of the term "reserve" when referring to that subset of conservation areas where land, by some formal designation, is managed strictly for biodiversity conservation.

Species Criteria Used in Biodiversity Conservation Planning

One of the early attempts to relate species geography to applied conservation biology is attributed to Wilson and Willis (1975). Focusing on the number of species that could be conserved in a network of habitat patches, they proposed a set of geographic rules for conservation design (e.g., contiguous habitat patches will conserve more species than fragmented habitat patches of the same area; when fragmentation is unavoidable, minimize separation distances). Although these rules quickly gained the status of conventional wisdom, subsequent work exposed their failure to apply generally (Simberloff 1988, Hof and Flather 1996). Our intent here is not to rehash past criticisms of these geographic conservation rules in particular, but to use this work as a stepping-off point to review two basic species criteria used in geographic-based biodiversity conservation planning.

One is based on a simple *species count* where the composition of the species pool being counted is immaterial except to the extent that the pool is often constrained by some common taxonomic (e.g., birds, arthropods) or ecological (e.g., habitat specialist, endemic, rare) attribute. Species count, also called *species richness* (we use count and richness interchangeably), is certainly the simplest and most easily understood criterion (Purvis and Hector 2000). For this reason, focusing on areas where species richness is concentrated has received extensive consideration in conservation planning (Brooks et al. 2006, Ceballos and Ehrlich 2006). There are important ecological reasons why a particular area supports more species than surrounding areas and species richness should be the criterion of choice when mean site diversity is an important conservation goal (Williams et al. 1996, Shriner et al. 2006).

However, species counts are nothing more than an integer representation of a nameless list of species. This failure to consider species identity forms the basis for this criterion's main criticism. In particular, within biogeographic regions, areas of high species counts have been found to have many species in common (Lennon et al. 2001). Moreover, there is evidence to suggest that spatial variation in species counts appears to be driven largely by the landscape occupancy pattern among common and widespread species—species that are unlikely to be most deserving of our conservation efforts (Brooks et al. 2006). In the absence of composition information, the degree to which comprehensive conservation of the species pool is being achieved is impossible to evaluate (Flather et al. 1997).

To address this weakness, systematic conservation strategies began to rely on a second criterion in conservation planning—*species representation*. The criterion is focused on ensuring some target set of species pool members are adequately represented in the conservation plan. The use of species representation in conservation planning traces back to Australian ecologists and geographers of the early 1980s (see Margules and Usher 1981, Kirkpatrick 1983) who noted that if the goal of conservation is to conserve biodiversity broadly, then we should be focusing on adding units of land to conservation networks that contribute the greatest marginal increase in species coverage (Sarkar et al. 2006). Conservation planning under this criterion becomes a search for units of land whose occupying species complement those, as opposed to being redundant with, species covered either in extant conservation areas or among a set of potential units of land that are being considered for conservation status (Vane-Wright et al. 1991). As such, species representation as a conservation planning criterion is linked inextricably to the concept of *complementarity*.

Because species are not distributed randomly, but occur on the landscape with varying degrees of spatial structure, species composition tends to be more similar among sites that are near to one another (Nekola and White 1999). This underlying structure in compositional similarity results in a well-known problem with conservation designs based on the species representation criterion—namely, sites selected to maximize complementarity are often well dispersed throughout a region, making management of these areas more costly, logistically difficult, and subject to elevated edge effects and dispersal constraints (Bedward et al. 1992, Possingham et al. 2000, Williams et al. 2005). Furthermore, there is evidence that strict application of a species representation criterion could result in the selection of sites that disproportionately represent areas that are at the periphery of species' geographic ranges, which may predispose these so-called marginal populations to future extinction events (Araújo and Williams 2001). These weaknesses can be addressed explicitly by including a variety of spatial constraints (e.g., adjacency requirements, minimization of boundary lengths, focus on core distributions) that effectively broaden the species representation criterion to include additional ecological considerations (Sarkar and Margules 2002).

A commonly expressed weakness of geographically explicit conservation planning, regardless of the species criterion used, is that it often fails to address the persistence of species (Lambeck and Hobbs 2002, Wiersma and Urban 2005). In their simplest forms, richness and representation criteria are based on the presence-absence pattern of species across the landscape. However, the likelihood of species persistence increases as the population size increases. Unfortunately, abundance data are not generally available for many species, making the oft noted data constraints associated with conservation planning (Lamoreux et al. 2006) even more severe.

Addressing the persistence issue has resulted in the rapid expansion of species criteria that get used in conservation planning. A few of these criteria are *redundancy*—a measure of species incidence across conservation networks such that representation occurs at least *k* times (ReVelle et al. 2002); *irreplaceability*—a measure that reflects the importance of a potential unit of land to the overall conservation design and is sensitive to unique or rare targets like local endemics (Pressey et al. 1994, Cabeza and Moilanen 2001); *vulnerability*—a measure of threat, either to species or habitat persistence or habitat conversion (Redford et al. 2003, Ricketts et al. 2005); and *robustness*—a measure that merges notions of redundancy and vulnerability by quantifying the degree to which conservation goals are maintained in the face of anthropogenic or natural disturbance (O'Hanley et al. 2007). Although these additional criteria complicate the subject of conservation planning, we find it reassuring that, for the most part, these emerging criteria can be thought of as variations on the fundamental criteria that we began this section with—namely, species counts or representation. For example, many of these new criteria simply invoke weighting schemes that permit the conservation practitioner to emphasize species or areas differentially.

The emergence of these additional criteria offers extreme flexibility in tailoring a conservation plan to the idiosyncrasies of a locale or species. With this flexibility comes the burden of choice—and the set from which to make that choice is growing. Since a choice must be made, it is tempting to evaluate the effectiveness of this growing list of species criteria in terms of which is "the best." MacArthur (1972*a*:259) once made the observation that "[a]nyone familiar with the history of science knows it [science] is done in the most astonishing ways. . .", as a commentary on ecology's search for *the* scientific method. The same observation holds for geographic approaches to conservation. The land management landscape is too complicated by land ownership patterns and conflicting resource values to allow a single "best" approach to geographically based conservation planning to emerge. Therefore, the preference for a particular species criterion, or set of criteria, will be dictated more by underlying value structures and circumstances specific to the conservation problem being addressed rather than any simple prescription of how geographically based conservation planning should be done (Redford et al. 2003).

Data Types for Conservation Planning

The fundamental datum for any geographic approach is a spatial location for a given species. Although there may be other attributes of species or areas brought along in the datum record, "knowing" where each species occurs on the landscape is the minimum prerequisite needed to characterize biodiversity across some area of interest (Ferrier 2002). How does the conservation practitioner go about obtaining information on species identity and location? There are four broad classes of data types or sources for such information.

First, and most obvious, are survey-based data that provide spatially explicit *empirical observations* of species occupancy. Much of these data take the form of simple locational records where a species was observed or collected. Data of this form that are collected on a probabilistic sample, where both presence and absence can be discerned, are extremely valuable to landscape-level conservation planning. However, such data tend to be restricted taxonomically and geographically—being available for only certain species in a particular locale. For this reason, when the planning area is large, conservation planners must make use of existing data from museum, herbarium, academic, or private collections (Bender et al. 2005). These data often represent an accumulated set of observations because they have been collected by a number of individuals, over varying periods of time, using a number of field collection techniques, and visiting sites in an opportunistic fashion (Funk and Richardson 2002). Consequently, the data are characterized by a number of inherent biases (Williams et al. 2002). Furthermore, the data often only record the presence of an individual, making it difficult to distinguish true absence from areas that have not been surveyed (Ferrier 2002).

Short of designing new surveys to eliminate the noted constraints of using existing data—which under most circumstances is infeasible given the time and financial constraints associated with most planning efforts—the conservationist must look for supplemental data. A second data type that is often used is that generated by *expert judgment* (Groves 2003). When data gaps are severe, expert panels may be the only source for species occurrence information. However, expert-generated data are difficult to evaluate, difficult to replicate, and often highly variable among comparable experts (Ferrier 2002, Neel and Cummings 2003). These well-known shortcomings of expert-generated data have motivated some to call for a shift away from expert judgments to evidence-based conservation (Sutherland et al. 2004) for biodiversity planning efforts.

This brings us to our third data type—*predicted occurrence* of species. Spatial interpolation that is based only on the geometry of location records is perhaps the simplest form of filling in species occupancy gaps (see Rapoport 1982). Such approaches are ecologically neutral and have given way to methodologies that quantify the covariation between species locations and some set of environmental attributes. Data of this type start with empirical observations of species that are then linked geographically with environmental predictors that allow the planner to extrapolate occupancy across a region of interest. Such approaches are often discussed under the rubric of species distribution modeling and have become commonplace in ecology over the past decade (Fortin et al. 2005). In addition to standard linear modeling approaches such as logistic and autologistic models (Augustin et al. 1996), sophisticated methods such as maximum entropy (Phillips et al. 2006), ecological niche factor analysis (Hirzel et al. 2002), Bayesian modeling (Gelfand et al. 2003), and genetic algorithm for rule-set production (Stockwell and Noble 1991) have emerged as tools for predicting species distributions. These models can be classified according to the type of data required. Some require only those locations where a species is known to occur and are classified as presence-only models (e.g., Zaniewski et al. 2002). Others also require information about where a species is known not to occur and are classified as presence-absence models (e.g., Engler et al. 2004). Although the rapidly growing number of approaches to predict species occupancy across a landscape (see Elith et al. [2006] for a review) has the potential to overwhelm conservation planners, these approaches do provide repeatable results that can be evaluated quantitatively.

In the best of all worlds planners would have comprehensive biodiversity data across all taxonomic groups with adequate spatial coverage. This is far from the case, and many have commented that our understanding of biodiversity is woefully incomplete (Brown and Roughgarden 1990, Pimm and Gittleman 1992, Flather and Sieg 2000). One approach for overcoming this data constraint is to assume that the biodiversity pattern of well-studied taxa can be used as a surrogate for other, less well-known taxa (Caro and O'Doherty 1999, Marcot and Flather 2007). In this sense, *surrogates* represent a fourth data type. Like predicted occurrence, this data type is inferential rather than measured *per se*. Although there has been recent evidence in support of surrogacy (Lennon et al. 2004, Lamoreux et al. 2006), the support is certainly not general (Ceballos and Ehrlich 2006). This latter finding is consistent with a growing number of papers that have cautioned conservation planners against blindly using surrogacy in geographic conservation approaches (Flather et al. 1997, Ricketts et al. 1999, Hess et al. 2006*b*).

Overview of Geographic Approaches

Once species criteria are selected and the data are in hand, the conservation planner must decide where, geographically, biodiversity conservation efforts will be put into practice. This involves formalizing the conservation problem and implementing some algorithm to solve it (Sarkar et al. 2006). There are two broad classes of approaches for solving the conservation network delineation problem: (1) those that focus on some quantile of a frequency distribution, and (2) those that focus on efficiently meeting an explicitly stated conservation objective.

The former are commonly discussed as a *hotspot* analysis, and this approach is linked directly with the species count criterion discussed earlier. Reducing the conflict between human land use intensification and areas possessing globally significant counts of species remains an important challenge to biodiversity conservation (Burgess et al. 2007). As noted by Ceballos and Ehrlich (2006), few topics in conservation planning have received more attention than species diversity hotspots. Originally, this approach involved the enumeration of species that were most threatened or vulnerable to human activities within some geographic unit (Myers 1988). Those units could be countries (e.g., Sisk et al. 1994), a systematic grid (e.g., Balmford et al. 2001), or a habitat patch (e.g., Wilson and Willis 1975). The patch-based definition is appropriate for local conservation problems, but is intractable at regional, continental, or global scales. Geographic units that are defined by administrative or political boundaries lend themselves to macroecologic investigation but suffer from unequal areas with little ecological basis that can skew evaluations of conservation importance. For this reason, species occupancy patterns across some systematic grid have become the more common empirical basis for hotspot analysis. The flexibility of this approach has resulted in usage that extends well beyond its species richness roots with the term “hotspot” being invoked any time the analysis seeks to identify geographic areas that rank particularly high (i.e., some upper quantile) on one or more axes of species (genus, family) richness, levels of endemism, numbers of rare or threatened species, intensity of threat, or indicator of ecosystem degradation (Prendergast et al. 1993, Flather et al. 1998, Reid 1998, Hof et al. 1999, Margules and Pressey 2000). The designation of the upper quantile (i.e., the frequency distribution threshold that identifies those geographic units as “hot”) varies in the literature but is usually $\leq$10%. However, because there is no ecological justification for the choice of quantile, the conservation planner is left to make this subjective decision.

Although the hotspot approach has played a central role in conservation planning, it has been criticized for the same reason that the species count criterion has been criticized; namely, it ignores species composition (Possingham and Wilson 2005, Fleishman et al. 2006). At the heart of this criticism is the principle of efficiency. Given limited conservation resources, an efficient strategy is one that concentrates on the fewest high-quality sites that meet the conservation objective (Redford et al. 2003). Because hotspot approaches ignore composition, they are generally thought to be inefficient unless the number of conservation units is constrained to be very small (Reid 1998, but see Shriner et al. 2006).

Efforts to develop algorithms that identify those conservation areas that give the biggest biodiversity bang for the conservation buck have resulted in an extremely rich set of tools that have their origins in the operations research literature—namely, *optimization* analysis (Haight and Gobster, this volume). The use of optimization models in biological conservation is increasing as evidenced by Rodrigues and Gaston's (2002) list of >30 optimal conservation design studies in the previous decade. Although there is an impressive variety of algorithms, they tend to fall into two broad classes: (1) those that define conservation networks based on iterative or stepwise algorithms, and (2) those that seek exact optimal solutions. Although they share the goal of designing efficient conservation strategies, they differ in that iterative algorithms are often referred to as inexact heuristics, since they can only approximate an efficient design (Cabeza and Moilanen 2001); while those based on a closed-form optimization formulation offer a globally optimal prescription (Hof and Flather 2007). So, why would conservation planners choose to use an inexact heuristic? There are a number of reasons that procedures not offering true optimal solutions get used. Many realistic conservation problems are unsolvable in closed form, while heuristic approaches tend to be intuitive, simple, and appear to provide reasonably good solutions when compared to exact solutions (Pressey et al. 1997).

Factors Affecting Our Ability to Describe Biodiversity

To this point we have implicitly ignored a number of factors that are known to affect our ability to characterize biodiversity and to develop tenable conservation plans. In this chapter we wish to highlight two: scale and error.

There is a substantial literature on the subject of scale and its effects on ecological study and the conservation recommendations derived from that research (Wiens 1989, Hoekstra et al. 1991, Schneider 2001, Willis and Whittaker 2002). There is growing evidence that patterns of covariation can shift when the analysis scale is changed (Lennon et al. 2001, Hess et al. 2006*a*, Pautasso 2007), and this explains, to a large degree, why unequivocal conservation recommendations have been so difficult to make.

Similarly, geographic conservation planning requires accurate data on the identity and location of species and the adequacy of biodiversity surveys. Data limitations have long been recognized as an important constraint associated with geographic-based conservation efforts (Prendergast et al. 1999), and there is a concern that data quality is not keeping pace with the growth in algorithmic sophistication (Possingham et al. 2000). Moreover, the ease with which large quantities of biodiversity data are now being made available raises additional doubts concerning their reliability (Cherrill and McClean 1995). Like scale, the impact of error in geographic-based conservation plans is rarely considered outside scientific journals, and even then, the sensitivity of geographic-based analyses to varying levels of error is seldom explored.

How do scale and error potentially affect geographically based conservation planning efforts? In order to explore some of the answers to this question, we now turn to a set of case studies that are motivated by our biodiversity assessment work in the southwestern United States. We present these case studies not as definitive works on the issues of scale or error, but rather offer them as examples of the kinds of issues that can emerge when practitioners consider their potential impacts on biodiversity conservation planning.

CONSIDERATION OF SCALE IN CONSERVATION PLANNING: DO BROAD BRUSHES COVER A GNAT'S ASSETS?

Before reviewing the specifics of our work on scale effects in conservation planning, we first need to define what we mean by scale. Probably the most frequent definition of scale in an ecological context (as opposed to a cartographic context) refers to the relationship between grain and extent of a particular investigation (Wiens 1989, Schneider 2001). *Grain* refers to the physical size or time period of the observation unit, whereas *extent* refers to the overall area or time period of the study or the geographic or temporal dimension to which inferences are drawn. It is the combined characterization of grain and extent that defines the scale of any investigation or conservation planning effort.

Given the biodiversity data deficiencies discussed previously (see "Data Types for Conservation Planning"), the input data into either hotspot or optimization analyses often stem from relatively coarse-grained observation units (Shriner et al. 2006); e.g., commonly analysis units are $\geq$10,000 km^2 (see Andelman and Willig 2003, Larsen and Rahbek 2003, Orme et al. 2005) for global, continental, and regional extents. How does this analysis scale compare with conservation implementation scale? We examined data from the World Database on Protected Areas (WDPA; WDPA Consortium 2004) to characterize the size distribution of currently implemented conservation areas. We restricted our examination of these data to those conservation areas that are terrestrial and classified by the World Conservation Union (IUCN) to category I and II (i.e., chief purpose is biodiversity conservation), qualifying them as biodiversity reserves. Furthermore, we purged all reserves with areas $\leq$1 ha because a large proportion of these very small reserves protect historic monuments or unique geologic formations (see Shriner et al. 2006). The median was $\sim$5 km^2 ($n = 8{,}967$) and nearly 75% of the reserves were $<$62 km^2 (Fig. 4-1).

This high variability in conservation planning scales does raise a question of whether there is a "right" scale to analyze biodiversity patterns. Although Wiens (1989) made a plea, >15 years ago, for objective approaches by which ecologists can define appropriate investigative scales, we still tend to treat scale with an exploration of system behavior resulting from varying grain or extents in an arbitrary manner. The danger with such an approach is that it becomes difficult

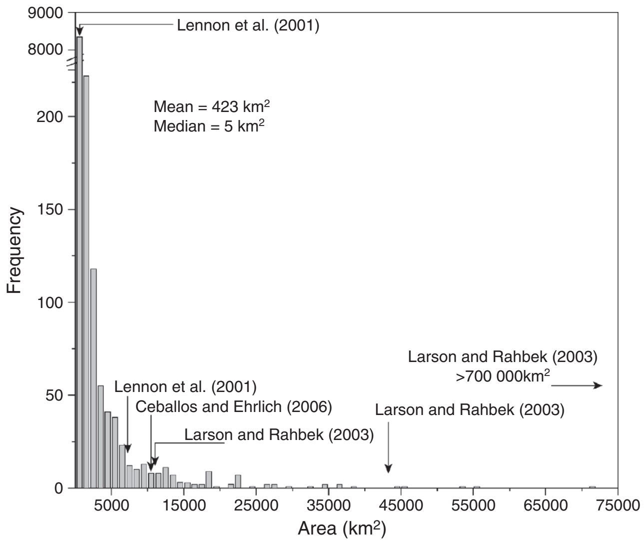

FIG. 4-1

Frequency distribution of the size of terrestrial conservation reserves dedicated to strict biodiversity conservation (IUCN categories I and II) from the World Database on Protected Areas.

to distinguish scale artifacts from ecologically meaningful patterns (Wiens 1989). This concern is particularly relevant to geographically based conservation efforts given the discrepancy between conservation analysis scales and the scales of conservation implementation (Fig. 4-1).

A common argument to address the scale mismatch between conservation data/analyses and implementation is to couch conservation planning as a hierarchical process whereby coarse-grained analyses provide a broad brush depiction of biodiversity that serves to focus conservation attention on regions that warrant attention (Ferrier 2002; Probst and Gustafson, this volume). The actual identification of land units that will comprise a particular conservation network can then be identified with finer scale data within these priority regions (Pressey et al. 1993, Harris et al. 2005, Fjeldså 2007). This approach implicitly assumes that conservation priorities are nested geographically, which is to say that conservation analyses at coarse (regional) scales are consistent with conservation actions that are implemented at fine (local) scales (Larsen and Rahbek 2003). Do we have evidence that conservation designs based on fine-scale data are generally nested within coarse-scaled designs?

The Approach and Database

We examined conservation designs with species count and species representation criteria using richness hotspot and optimization approaches. We based our analyses on predicted range maps for mammals and birds developed by the Arizona and New Mexico Gap Analysis Programs (Thompson et al. 1996, Halvorson et al. 2001) available at 90 m and 100 m grid resolution, respectively. For the purposes of this analysis we assumed that these range maps reflected the "true" occupancy pattern of species across these two states. We used these "known" distributions to derive 1, 100, 625, 2500, and 10,000 km^2 grid cell representations of each species' distribution based on a binary rule that classified a cell as occupied if any portion of the species' true geographic range intersected a particular grid cell. We had suitable range map data for four state-species groups: Arizona birds (279 species), New Mexico birds (324 species), Arizona mammals (129 species), and New Mexico mammals (138 species).

For the richness hotspot analysis we first generated the total species count within each grid cell, at each of the grain sizes, by simply summing occupancies across all species. We defined hotspots as grid cells exceeding the 95th quantile as in Prendergast et al. (1993); that is, we identified those 5% of grid cells with the highest richness estimate. For the optimization analysis we used the occurrence data for each grid cell, at each grain size, to select that set of cells such that each species was represented at least once in the set. We used the MARXAN conservation design software, in particular simulated annealing (Ball and Possingham 2000), to identify the most efficient (minimum area) set of cells meeting the representation objective. Simulated annealing is an inexact heuristic that has been shown to perform well in conservation design applications (Possingham et al. 2000).

Total area of the conservation network designed under our representative criteria varied greatly with grain size (Shriner et al. 2006:1665); networks based on small grain units reached solutions after affecting <1% of the total extent, whereas large grain units affected nearly 20% of the extent. These size differences confound interpretation of scale effects because networks based on smaller grain units are more likely to overlap networks based on large grain units given the larger total area included in the large-grain solution. For this reason, we further constrained the conservation network solutions from MARXAN based on the notion of irreplaceability. An irreplaceability score was generated by MARXAN that reflected the number of times any one grid cell was selected as a member of the "best" network in 1000 realizations of the design solution with the score ranging from 0 (never selected) to 1000 (always selected). We rank-ordered grid cells based on these irreplaceability scores and selected those cells that exceeded the 95th quantile as in the hotspot analysis. Because of the numerical intensity of this analysis, we only explored scale effects for optimally designed conservation areas for Arizona birds. Furthermore, we did not develop a conservation network based on an irreplaceability ranking at the 1 km^2 grain because too few cells were selected to meet the 5% area goal (i.e., we met our representativeness criterion with <5% of the extent's area).

Results

The amount of overlap observed for species hotspots varied from 0.0% to 63.1% with an overall mean of 28.1% (Table 4-1). The degree of overlap in conservation networks was different between taxa (grand mean across birds in both states = 25.6%; grand mean across mammals in both states = 31.6%), but was very similar between states (28.7% for Arizona and 28.6% for New Mexico).

Conservation areas designed under a species representation criterion showed similarly low degrees of overlap across grain sizes. Percentage overlap in pair-wise grain comparisons for Arizona birds varied from a low of 15.3% to a high of 44.2% (mean = 23.9%). Although network overlap was generally low, it is noteworthy that the selected conservation units tended to cluster in certain geographic locales (Fig. 4-2). So, while conservation areas showed low overlap, there was a high degree of adjacency, suggesting that there may be some underlying ecological mechanism causing the proximity of network units at different scales. This pattern of proximity notwithstanding, it is also notable that there are some portions of the state that were selected uniquely for conservation focus at individual scales (e.g., the south central portion of Arizona at the 100 km^2 grain size; Fig. 4-2).

Table 4-1 Pairwise Comparisons of Percent Overlap for Richness Hotspot Reserves Developed at Five Grain Sizes: 1 km^2, 100 km^2, 625 km^2, 2500 km^2 and 10,000 km^2. Percent Overlap is Calculated by Dividing the Area of Overlap by the Area of the Smaller Reserve

	Arizona	**New Mexico**	**Arizona**	**New Mexico**
Map Comparison (km^2)	**Birds**		**Mammals**	
10,000, 2500	16.7	37.5	17.8	50.0
10,000, 625	21.9	18.8	0.0	25.3
10,000, 100	26.5	11.2	0.0	12.4
10,000, 1	33.4	13.0	5.1	8.7
2500, 625	17.6	34.1	54.5	56.9
2500, 100	27.8	21.4	43.4	36.1
2500, 1	14.6	19.6	27.7	21.8
625, 100	47.1	42.7	63.1	50.2
625, 1	25.1	18.4	40.5	23.9
100, 1	32.7	29.4	57.8	36.2
Mean	26.3	24.6	31.0	32.1

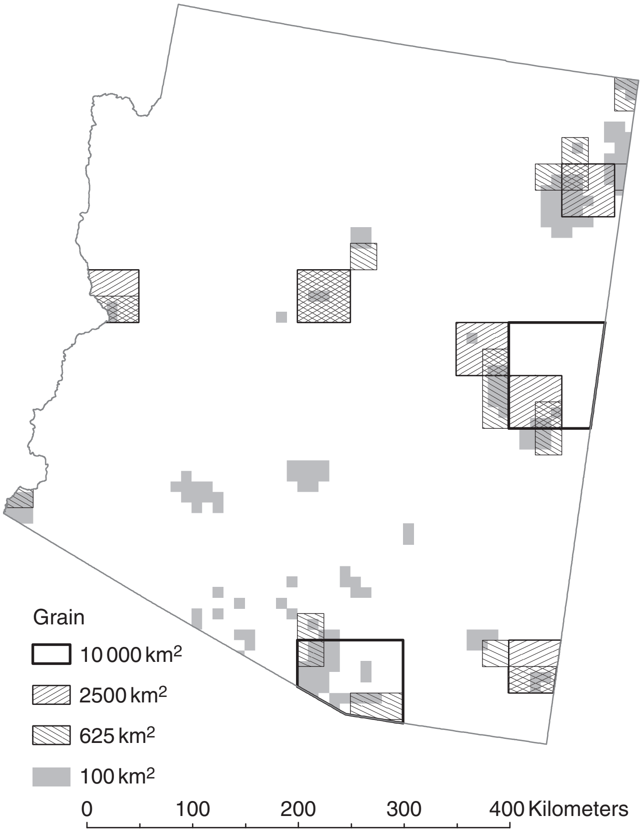

FIG. 4-2

The spatial overlap of conservation areas selected at four conservation unit grain sizes (100 km^2, 625 km^2, 2500 km^2, and 10,000 km^2). The conservation networks selected at each grain size were based on an irreplaceability criterion and an inexact heuristic optimization (i.e., simulated annealing using MARXAN).

Although tangential to an evaluation of scale effects, a comparison of conservation networks selected under species counts and representation criteria is also interesting. Like the other overlap comparisons, richness hotspots and representation reserves showed low overlap across all grain sizes (Fig. 4-3). In fact, the two criteria appear to be focusing on very different aspects of the state's geography. Overlap was minimal at the coarsest grain (0.0% at 10,000 km^2 grain) and reached a maximum (17.5%) at the 2500 km^2 grain. Reserves based on the representation criterion also had higher degrees of species coverage than reserves based on species counts (as expected), but it was surprising that hotspot reserves at the finest scale (1 km^2) did cover most species used in the analysis.

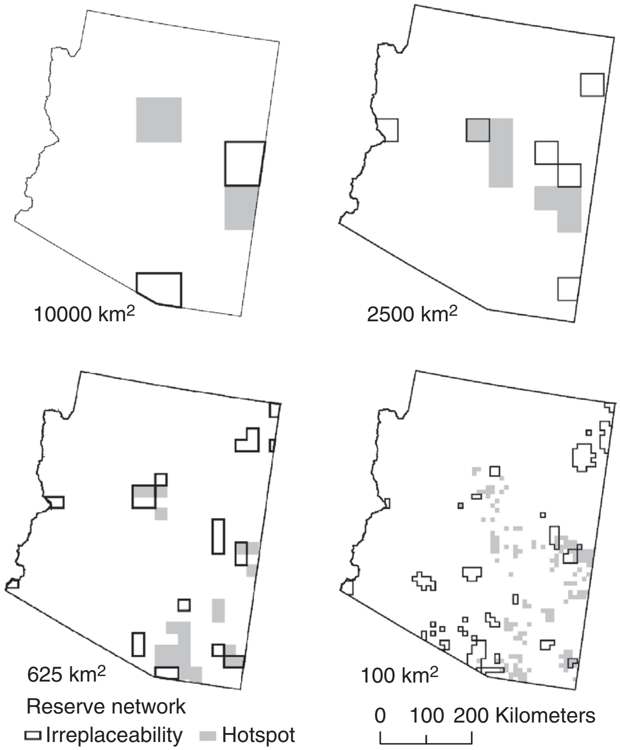

FIG. 4-3

Spatial congruity between reserve networks selected using hotspot and optimization analysis at four reserve unit grain sizes (100 km^2, 625 km^2, 2500 km^2, and 10,000 km^2).

Implications

Because it is rarely feasible politically to establish large conservation networks (Margules and Usher 1981:99), it is legitimate to ask: "If the observation units in a geographically based conservation analysis can't be implemented (i.e., too large), then what is the point of coarse grain assessments for biodiversity conservation?" Justifications for retaining coarse grain biodiversity assessments tend to proceed according to the following logic: (1) Conservation practitioners do not hold detailed knowledge of species occurrence because comprehensive biodiversity data are limited spatially; (2) coarse-grain assessments allow some of those spatial limitations to be relaxed because the heterogeneity in sampling effort becomes less detectable at large grain sizes; and (3) therefore, coarse grain assessments provide a means of identifying broad areas that should be the focus of more detailed conservation study. This logic is based on the assumption that conservation designs form a spatial hierarchy where coarse-grain designs subsume fine-grain designs. That is, fine-grain designs would be nested within coarse-grain designs.

The results from our case study do not support the assumption of nested designs for conservation analyses across spatial scales. Our results clearly indicate that conservation planning outcomes can be scale dependent (but see Fjeldså 2007) and that conservation planners should proceed cautiously when conservation priorities are based on analysis scales that are disparate from implementation scales. The low spatial overlap of conservation networks based on different scales (observation grain in our case) suggests that a simple comparison of biodiversity analysis scales with conservation implementation scales would be a useful attribute for conservation planners to consider in judging whether their plan is likely to be an efficient mechanism for conserving biodiversity.

This does present a quandary for conservation planners. In the absence of fine-scale biodiversity data, there may be no opportunity for avoiding inefficient conservation plans. Obviously, a solution to this predicament is to develop fine-scale species occupancy data for biodiversity conservation planning. Alternatively, conservation practitioners could implement conservation plans at the coarser scales that correspond to current knowledge. Ultimately, the best approaches will likely blend management at coarse scales for large spatial extents while simultaneously integrating fine-scale management, potentially resulting in more effective conservation of the species in question.

Given the prohibitive costs associated with collecting primary species occupancy data, it would seem important to also invest in detailed distributional modeling efforts that can accurately predict species occupancy. Efforts to assess our ability to predict species distribution are often overlooked (Wilson et al. 2005), even though the uncertainty in our predictions can be substantial (Flather et al. 1997, Elith et al. 2002). Moreover, it is important to understand how robust our conservation plans may be to species distribution errors—which is the subject of the next case study.

ERROR AND UNCERTAINTY: THE DIVINER'S LAMENT

> *As in life, sins of commission and omission are often fatal. I can write with feeling on this subject, having dowsed…things that I ought not to have dowsed and left undowsed those things I ought to have dowsed. (Terry Ross, founder and president of the American Society of Dowsers, Danville Vermont. First published in* Rod & Pendulum*, No. 68, September 1992).*

The objective of the dowser is not unlike that of the modeler who is attempting to predict the occupancy pattern of species across the landscape. Whether using the L-Rod, Y-Rod, or the pendulum to predict the location of water, or GLMs, GAMs, or maximum entropy to predict species occurrence, the underlying

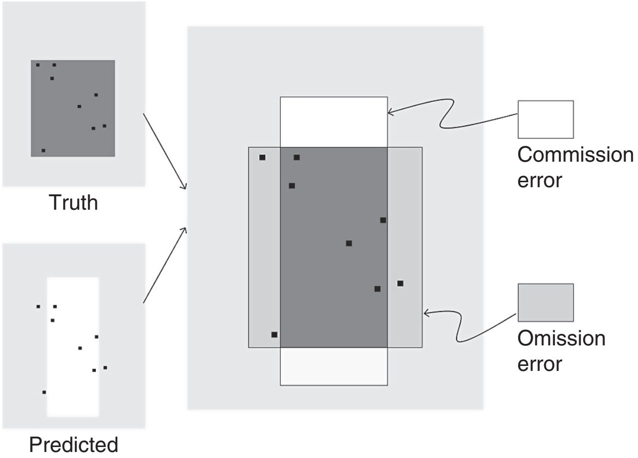

FIG. 4-4

Conceptual model illustrating commission (false presence) and omission (false absence) errors associated with predicting species distributions.

goal is the same—assign presence or absence in a way that minimizes error and is subject to validation by the well driller or species survey.

The representation of landscape occupancy by a species can be wrong in two ways (Fig. 4-4). A species can mistakenly be predicted to occur in an area (false presence or commission error), or a species can mistakenly be predicted to be absent in an area (false absence or omission error). The sources of uncertainty that lead to such errors are many. Sampling errors occur with species surveys due to the sample units selected, identification errors, incomplete or biased sampling, and imperfect detectability; models used to predict occupancy may introduce error because of imperfect habitat relationships or model misspecification; and cartographic errors can manifest due to inaccuracies (thematic or locational) in species point observations or map layers used in predicting occupancy—not to mention data transcription errors or errors associated with summarizing information across multiple observation grains. As is the case with all geographically based conservation planning endeavors, the outcome is only as reliable as the underlying data (Burgess et al. 2007:174). There are numerous examples in the literature demonstrating the sensitivity of geographically based conservation plans to errors (Dean et al. 1997, Gaston and Rodrigues 2003, Wilson et al. 2005), yet explicit consideration of error in conservation prioritization schemes is still wanting (Rondinini et al. 2006).

In this case study, we review the findings of our efforts in the southwestern United States to examine the effects of omission and commission errors on analyses to support systematic conservation designs. In particular, we were interested in the following questions: (1) How robust are spatially explicit conservation recommendations to errors in the underlying biodiversity data?; (2) does the type of error differentially affect design sensitivity?; (3) does the manner that errors manifest on the landscape (random versus spatially constrained) affect conservation plans?; and (4) does the number of species of conservation interest affect the conservation plan's sensitivity to error?

The Approach and Database

Our general approach and data are similar to the previous case study, and we refer the reader to that section for the details. The underlying data were again based on the Arizona and New Mexico Gap Analysis Programs, and the distribution maps available were treated as the "known" landscape occupancy pattern for each species. As before, *true* hotspots were generated for birds and mammals based on the known landscape occupancy maps. Species richness hotspots were again defined as the set of map cells that exceeded the 95th quantile for species richness.

Errors of omission or commission were imposed on the known occupancy maps for each species in two fundamentally different ways: (1) *spatially random* in which error occurred in a simple, spatially random fashion, and (2) *boundary correlated* in which errors were spatially correlated and more likely to occur at the boundary of the known species distribution. In the spatially random error case, for omission, all cells where the species was known to be present were equally likely to be selected and changed to an absence; and for commission, all cells where the species was known to be absent were equally likely to be selected and changed to a presence. We simulated the boundary correlated error case by weighting the probability that a given cell would be selected as an error cell based on its proximity to a range boundary using the following distance decay function:

$$\text{Prob}(i) = 1 - (1 - \theta^{D_i})^{\beta},$$

where Prob(i) is the probability that cell i is selected for error imposition, D_i is the distance that cell i is from a range boundary, and θ and β are parameters that affect the maximum distance from the range boundary and the rate of decay that a cell could be considered for misclassification, respectively. We used a combination of θ and β such that little error occurred beyond 30 km of a known range boundary with most errors occurring within 7–10 km.

A Monte Carlo simulation experiment was developed that compared species richness hotspots that included overlays of $n = 10, 25, 50, 75, 100$, or 125 species, and again the overlay of these species distributions without error was considered truth. To examine error, as each species was overlaid, either spatially random or boundary correlated error was generated with each species having either (1) all omission error, (2) all commission error, (3) balanced omission and commission error, or (4) a random combination of omission and commission error. The resulting landscape occupancy maps with error were used to develop new species richness hotspots. Our response variable in this analysis was the percent overlap between hotspots defined using the known (truth) occupancy maps and the error imposed maps. Because the rules used to generate the original landscape occupancy maps varied by state and by taxon (birds versus mammals), we also considered state and taxa as factors that may explain observed variations in hotspot overlap across the simulation experiment. A total of 30 replicates for each combination of error factors was run for the experiment, each with a fixed error rate (20%). The numerical intensity of the MARXAN optimization analyses prevented us from examining how error would affect conservation designs based on a species representation criterion.

Results

Mean percent overlap between error and true hotspots varied from 62–93%. Spatially random error (mean overlap = 86%) had less of an impact on hotspot coincidence than did boundary error (mean overlap = 77%) (Fig. 4-5). Commission error (mean overlap = 85%) had less of an impact than omission error (mean overlap 75%). In general, the percent overlap increased as the number of species increased when the error was random, whereas it remained essentially stable when the error was spatially constrained to occur near range boundaries. A case that deviates notably from this pattern is boundary omission errors where we observed a monotonic decline in error as the number of species increased (Fig. 4-5b).

Looking at all possible factors that could affect the degree of overlap between true and error imposed hotspots revealed that taxa (bird versus mammal) had the

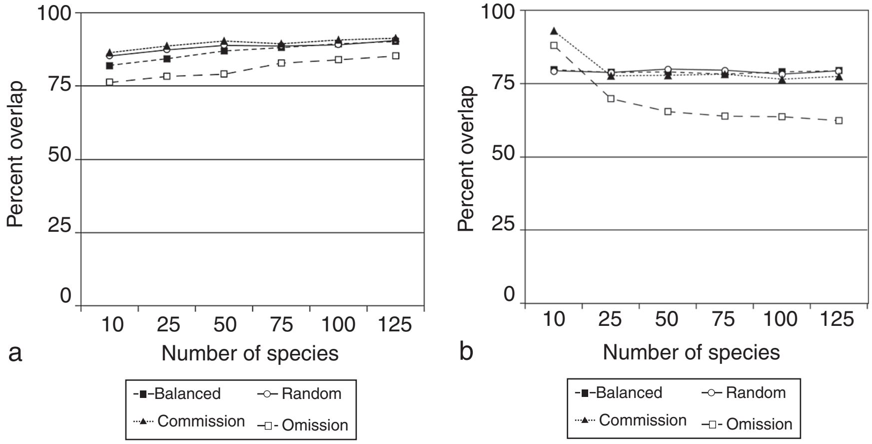

FIG. 4-5

Percent overlap between species richness hotspots for birds and mammals in Arizona and New Mexico for (a) spatially random error and (b) boundary error. Each species had a fixed error rate (20%) that was imposed as all omission, all commission, balanced omission and commission, and random mix of omission and commission.

Table 4-2 Analysis of Variance Results for 5 Factors That May Affect the Degree of Overlap Between True Species Richness Hotspots and Error Imposed Hotspots in Arizona and New Mexico. The Factors are Taxa (Birds, Mammals), Error Type (All Omission, All Commission, Balanced Omission and Commission, Random Omission and Commission), Number of Species (10, 25, 50, 100, 125), Location (Spatially Random, Constrained to Range Boundary), and State (Arizona, New Mexico).

Source	DF	Type III SS	MS
Taxa	1	22 863	22 863
Error Type	3	25 412	8471
No. of Species	5	29 006	5801
Location	1	1617	1617
State	1	1527	1527

Note: Statistical significance is not reported, since rejection of the null can be guaranteed by simply increasing the number of realizations in the simulation experiment.

greatest impact on observed variation in hotspot overlap (Table 4-2). This was followed by error type (all omission, all commission, balance, and random) and the total number of species in the analysis. The spatial location of the error (i.e., whether it was distributed randomly or associated with range boundaries) had relative low explanatory power relative to the other factors.

Implications

The results from this simulation experiment were surprising for a number of reasons. First, the dominating effect of taxa was not anticipated *a priori*. In hindsight, this factor is likely related to the differences in mean landscape occupancy between birds and mammals. Birds, being a more vagile taxon, are able to occupy more of the landscape than mammals. This part of the United States is characterized by isolated montane habitats set in a hostile (at least to endothermic vertebrates) arid matrix. This particular abiotic context likely resulted in a higher proportion of mammalian species with smaller, and therefore restricted, distributions when compared to birds. This fundamental difference in range size and shape is a likely explanation for the pattern we observed. Such speculation could be refuted or confirmed by repeating our analysis on species with widespread versus restricted landscape occupancy patterns. We suspect that this is not a taxon *per se* effect, but rather points to landscape occupancy as a key attribute affecting the robustness of conservation designs based on hotspot criteria.

Another surprising finding was the moderate effect attributed to the number of species overlaid in the analysis. Our expectation was linked to work that has quantified error propagation and the observed decline in overall map accuracy as the number of overlay operations increased (see Veregin 1989). We actually observed an increase in the robustness of hotspot selection under random error as we increased the number of species, and essentially no decline in robustness for error that was spatially constrained to range boundaries as we increased the number of species of conservation concern from 10 to 125. This is likely related to our use of the hotspot criterion in this case study. It is probable that the actual estimate of species richness in any one cell showed much greater degrees of error than the geographic location of the 95th quantile. So while the magnitude of the richness estimate may be sensitive to the number of species overlays, the relative ranking of cells for hotspot identification appears to be somewhat immune to both omission and commission errors.

Finally, the generally high degree of overlap between true hotspots and error imposed hotspots was not anticipated. We intentionally chose a high fixed error rate (20% of the true distribution of the species) to ensure that an effect would be observed. We were surprised that the degree of overlap exceeded 75% in most of the error simulations conducted (Fig. 4-5). This pattern is likely related to the fact that richness is compositionally neutral. We suspect that if optimization based on a species representation criterion had been used to prioritize conservation units, the design solutions would have shown more sensitivity, and therefore be less robust to the kinds of error we imposed in our simulation experiment. One exception to the generally high degree of overlap we observed involved omission errors. In both random and boundary error cases, omission error resulted in the lowest overlap with the true hotspot maps, and in the case of boundary error overlap actually declined with the number of species considered. A potential explanation for this result is that commission error tends to expand the general range of a species such that areas of overlap between species distributions are broadened with minimal impact on the underlying pattern of high species counts. On the other hand, omission error is more likely to erode areas of overlap between species distributions such that true areas of high species counts may look more like their neighbors, lessening the signal of the underlying pattern of species richness. This finding suggests that models that favor commission over omission error may lead to more robust hotspot identification.

An important limitation of this particular case study is that our findings may be conditioned on the ecological circumstances that define this region of the United States. Distinguishing whether the results we observed are general, or are simply specific to this particular geography, will require repeating our analysis in other geographic locales. Furthermore, repeating this analysis using other geographic approaches (e.g., optimization) and other species criteria (representation) would more completely inform the conservation planner about the likely impacts of error. Although we end this case study with a call for additional research on the impacts of error, these are not the only avenues for future work that will extend our conservation planning capability.

FUTURE RESEARCH NEEDS

Geographically based conservation planning to conserve biodiversity is deceptively simple: Information on the location and identity of species can be used to prioritize where limited conservation resources should be focused. However, designing effective biodiversity conservation strategies across extensive landscapes is remarkably complex. This complexity derives from a number of sources, and reductions in that complexity can be realized by extending research efforts into a number of areas that include improving data availability and quality, improving the inferential basis for spatially explicit representation of biodiversity, extending geographic approaches to planning, incorporating consideration of ecological processes, and improving the accessibility of geographically based conservation analyses to practitioners.

Data Availability and Quality

The species has been regarded by many as the fundamental unit of biodiversity (Huston 1993). Although species surveys are central to any geographically based assessment of biodiversity, important barriers to comprehensive biodiversity inventories remain. Perhaps the most obvious need for extending future work concerns the development of monitoring protocols that are economically feasible and ecologically tenable (see Haight and Gobster, this volume).

Part of the difficulty with the availability and quality of primary biodiversity data relates to substantial knowledge gaps in the systematics of some taxa and the fluid nature of taxonomic classifications over time. The emerging discipline of biodiversity informatics (see Bisby 2000), which focuses on the development of a comprehensive taxonomic accounting of all species, would help further efforts to monitor biodiversity patterns. However, even among taxa with relatively well-described taxonomies, most have no data from which to describe species occupancy pattern over the geographic scales necessary to support spatially explicit, landscape-wide analyses for conservation planning. We have the technical wherewithal to design and implement species monitoring programs, but we lack the financial resources to make comprehensive monitoring programs a reality in the near future. Even the simpler task of a taxonomically comprehensive accounting of species is decades, not years, away (Lawler 2001). Furthermore, there is a need to move from biodiversity data derived from accumulated records with an unknown statistical foundation, to data that have their basis in a probabilistic sample of both presence (recorded) and absence (not recorded) (e.g., Pollock et al. 2002). Failure to do so will continue to limit our use of formal inferential procedures (Anderson 2001) to estimate and predict important attributes of biodiversity.

Inferential Basis for Conservation Planning

Although taxonomically comprehensive and spatially extensive species inventories are not going to be available in the foreseeable future, conservation planners cannot sit idle waiting for ideal biodiversity data to appear while land-use decisions are made. For this reason, research that explores how to make the best use of extant data needs to continue. Of particular importance are efforts that extend our capacity to infer species occupancy across space (species distribution models) and to infer overall biodiversity patterns from a few taxa (surrogacy).

Given the incomplete spatial coverage of species surveys, distribution models permit planners to extend survey data to infer occupancy across large geographic areas—portions of which have not been surveyed (Guisan and Thuiller 2005). Although there has been an explosion of species distribution modeling approaches (see Scott et al. 2002), there are a number of modeling challenges that remain, including better representation of species movement to capture source-sink or metapopulation dynamics; determining if interspecific interactions need to be incorporated into species occupancy models; and the extension of species models to those that treat multispecies attributes as the response variable (see Noon et al., this volume). Certainly, more evaluative efforts like that of Elith et al. (2006) are needed to quantify the performance of these modeling approaches and to better inform planners about which approaches are appropriate given the situation specific to the planning context.

The assumption that biodiversity patterns from a few well-studied taxa can represent the pattern among the throng of little-known taxa is necessary given that most biodiversity remains nameless to science. This assumption has been the focus of widespread empirical testing with equivocal outcomes. Although a number of studies have found little evidence supporting the surrogacy assumption (Flather et al. 1997, Ricketts et al. 1999, Ceballos and Ehrlich 2006, Noon et al., this volume), others observed sufficient positive co-occurrence patterns among taxa (Lamoreux et al. 2006) to provide hope that we can simplify the biodiversity conservation challenge by focusing on a small subset of species to derive tenable conservation plans. In particular, Lennon et al. (2004) recently found evidence that variation in species richness across the landscape is associated with relatively few, more common, species. Such contradictory findings point to an important research need—namely, to identify those ecological circumstances when it is tenable to use occurrence patterns of a few taxa to represent the pattern for other taxa (Marcot and Flather 2007).

Extending Geographic Approaches

Improvements in data, whether empirical or model-based, will make substantial improvements in the accuracy of conservation plans. However, data by themselves will not address all limitations associated with geographically based conservation planning; we also need research to extend the capability of the analytical approaches themselves. An admittedly partial list includes explicit incorporation of scale effects, extending the capability of optimization, and consideration of landscape context effects.

Much of what is done to conserve biodiversity takes place at very local scales (Oldfield et al. 2004, Turner et al. 2006). This contrasts considerably with many recent analyses of global conservation priority, which are often several orders of magnitude larger in their analysis grain. Although this disparity is acknowledged (Harris et al. 2005, Burgess et al. 2007), the potential implications of the disparity between coarse-grain prioritization to fine-grained implementation has not been fully evaluated. Our analysis of biodiversity patterns in the southwestern United States suggests that conservation priorities can be scale dependent and that large-grain analyses may not efficiently identify where fine-grained implementation should occur (but see Probst and Gustafson, this volume). However, there is a need to repeat our multiscale comparisons across a broad range of ecosystems to determine if the lack of spatial coincidence in multiscale priority setting that we observed is a general pattern.

Another area of future research concerns the need for continued improvements in applying optimization analyses to conservation planning problems. The literature points to an inherent trade-off in optimization approaches, namely that there is a choice between obtaining an exact optimal solution to simple conservation problems or an approximate optimum to ecologically complex conservation problems (Fig. 4-6). Incorporation of mechanistic ecological detail is a strength of simulation modeling where system responses are explored numerically. Simulation models, however, are hard pressed to prescribe how lands should be managed to obtain efficient conservation objectives. The latter is a strength of optimization, but optimization suffers from constraints associated with obtaining analytical solutions to complex ecological circumstances. For this reason there is potential for fruitful research that explores the joint use of these strategies. A combination of approaches may offer planners the ecological detail from simulation approaches and the analytical power of optimization to prescribe the best solution (Williams et al. 2005, Hof and Flather 2007).

Traditionally, geographically based conservation planning has represented landscapes in a binary fashion with places of biodiversity conservation focus being embedded in a benign matrix of concealed heterogeneity. After plan implementation, these conserved lands are cordoned off in an attempt to isolate them from the threats attributable to “human enterprises” (*sensu* Vitousek et al. 1997). However, research is showing that this matrix is far from benign. The matrix of semi-natural and intensely managed lands can have significant impacts on designed conservation networks (Williams et al. 2006). On the one hand, human uses within and outside the conservation area boundary can erode the biodiversity elements featured in the conservation areas (Liu et al. 2001). On

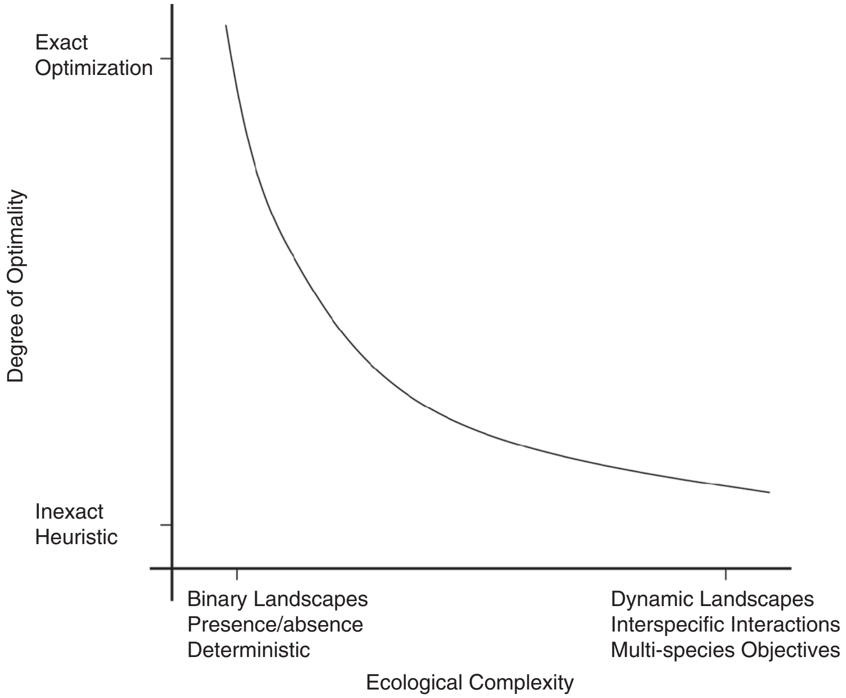

FIG. 4-6

The trade-off between degree of optimality and ecological complexity that can be addressed with exact and inexact optimization analyses for conservation planning (adapted from Hof and Flather 2007).

the other hand, the so-called matrix is not devoid of biodiversity and therefore makes a contribution to the overall mix of species that inhabit a conservation network (Ricketts 2001). Treating the matrix as biologically depauperate may be justified from the standpoint of accounting for the worst-case scenario. However, such a strategy may in fact be setting the conservation bar too high and limiting our flexibility in considering reasonably good solutions in the real world where strong economic pressures make implementation of "the best" strategy politically infeasible (Possingham et al. 2000). For this reason, there is a need for geographic approaches that take a broader perspective—one that considers the landscape whole as well as the conserved parts. This is easier said than done, but it is a perspective that is extending landscape ecology from its traditional patch-based focus to one that treats the landscape as a continuum (Fischer and Lindenmayer 2006)—a shift that is also seen as facilitating the incorporation of ecological process into conservation planning and land management.

Incorporating Ecological Process

Geographic approaches to conservation planning have been dominated by the examination of patterns of noteworthy biodiversity features across the landscape (Pressey 2004). These biodiversity patterns manifest from a complex interaction of ecological processes (e.g., species dispersal, spatially explicit demographics, disturbance, succession, interspecific interactions, primary productivity) with human uses of the landscape (e.g., subsistence, land use conversion, resource extraction). These interactions are the drivers of ecosystem dynamics and the transitory nature of species occupancy patterns across the landscape (Sarkar et al. 2006). The observed temporal turnover in species begs the question: "Can static conservation designs protect the full complement of species suggested by the planning analyses?" (Cabeza and Moilanen 2001).

Unfortunately, conservation science has yet to determine the best approaches for including the underlying ecological and socioeconomic processes into geographical approaches to conservation planning (Williams et al. 2005). For this reason, there is a growing recognition that the biodiversity patterns used to justify a particular conservation design may not be conserved over time. This realization was the motivation behind the efforts of Leroux et al. (2007) to examine the effects of disturbance on the effectiveness and efficiency of static conservation designs. The risks associated with ignoring dynamics varied across conservation targets with some targets (population levels of a focal species) failing to be maintained in a dynamic world, whereas other targets (vegetation representation) were conserved with a high probability under most circumstances (Leroux et al. 2007:1963).

Demonstrating the potential consequences for failing to consider ecosystem dynamics in conservation designs is important, but the planning tools to incorporate such effects analytically are still in their infancy. Given the impending changes to ecosystems in response to climate change (Thuiller 2007), this perhaps represents one of the most important gaps in our ability to effectively plan for the long-term persistence of species across broad landscapes (Groves 2003). Closing that gap will require continuing efforts to incorporate both pattern and process in conservation planning (Ferrier 2002), including an ability to explicitly consider the dynamic nature of ecosystems and the scheduling of conservation strategies over time (Meir et al. 2004).

Accessibility to Practitioners

Geographic-based approaches have a long conservation history, and recent developments in computer technology and analytical capability have allowed impressive advances to the science of biodiversity conservation. These advances notwithstanding, several investigators have made a somewhat disturbing observation—namely, that much of this advance has remained in the ivory towers of academia with little impact on applied conservation (Prendergast et al. 1999).

Although there are clearly exceptions to this observation—most notably, the success that Australian ecologists have had in infusing spatially explicit conservation science into the planning process (see Margules and Pressey 2000)—it is cause for concern that the interaction between research and management on conservation planning remains limited (Cabeza and Moilanen 2001, Flather et al. 2002). Making these geographically based conservation approaches more accessible to conservation practitioners will fail if researchers rely solely on published journal articles to communicate advances. Rather, it will require concerted efforts directed toward meaningful and long-term collaboration on real applied problems and will also require research that demonstrates what is gained by using these planning tools. Two areas that we see as particularly important are evaluative monitoring and characterizing uncertainty.

We are not implying a complete absence of research that has demonstrated the value of conservation planning approaches. However, much of this demonstration has been done with simulated data and evaluation of hypothetical scenarios (Cabeza and Moilanen 2001). Clearly, there are several key logistical constraints. Implementation of conservation plans can take many years, and monitoring to detect species responses, particularly as it relates to detecting extinctions, can take generations (Reed et al. 2003). However, unless we attempt long-term monitoring of implemented conservation plans, we will remain exposed to an apathy of inaction (Prendergast et al. 1999) or the relegation of conservation areas to those of low economic value (Prance 2000). Fortunately, real-world examples that are attempting to use many of the methodologies outlined here are beginning to appear. For example, in the U.S., The Nature Conservancy has seemingly taken the lead in implementing research in large-scale conservation planning and prioritization (Valutis and Mullen 2000, Groves et al. 2002), and has completed a number of conservation assessments in various regions, e.g., the Pacific Northwest coast (Vander Schaaf et al. 2006). Still other conservation planning efforts such as the Northwest Forest Plan (Forest Ecosystem Management Assessment Team 1993), Greater Yellowstone Ecosystem, Wyoming (Noss et al. 2002), Florida (Oetting et al. 2006), South Africa (Smith et al. 2006), and Canada (Beazley et al. 2005) are utilizing these methods to evaluate existing conservation networks and proposed land acquisition for biodiversity conservation.

A related research need is a more comprehensive effort toward the characterization of uncertainty. As noted by Margules and Pressey (2000:251), conservation planning is "...riddled with uncertainty," and our case study has shown how this uncertainty can affect conservation planning. Geographical displays of species distributions or priority conservation areas are often presented as if they are known, and this tendency impedes consideration of uncertainty in the planning process. Given the burgeoning number of modeling approaches that are now available to either predict species occurrence or to prescribe efficient conservation designs, comparative evaluations (like that of Elith et al. 2006) of what works and under which set of circumstances remain an important research opportunity.

Commensurate with research efforts to quantify our uncertainty, we also need research that asks: “How accurate do we need to be?” We know that the propagation of errors from a multitude of sources can be substantial (Elith et al. 2002). What we know less about is how robust our conservation plans are to these various sources of uncertainty and whether robust conservation designs can be derived from approaches that explicitly incorporate uncertainty in the underlying data (e.g., Bini et al. 2006, Moilanen et al. 2006). Ultimately, this issue comes down to determining when moderately precise models will be adequate and when increased accuracy will be necessary (Wiens 2002).

Until we make these analyses more accessible to those that engage in conservation planning by demonstrating their utility and quantifying their uncertainties in ways that go beyond measures of deviance, kappa statistics, or area under receiver operating curves, these approaches will be less accessible to practitioners than they might otherwise be. And ultimately, this rapid development of sophisticated geographically based planning tools will do little to advance conservation decisions if these tools are not understood or used appropriately by those burdened with the responsibility for recommending what, where, and how biodiversity conservation occurs on the landscape.

SUMMARY

Establishment of conservation areas, whether focused strictly on biodiversity conservation or on conservation allowing some degree of multiple-use resource management, continues to be an important regional strategy in the conservation of contemporary biodiversity resources. Given that financial resources for species conservation are limited, responsible stewards must decide where on the landscape management actions should be implemented to maximize conservation benefits. On the surface this seems a simple objective, but unequivocal approaches to attain that objective have eluded conservation scientists for a number of reasons. Two important factors that affect landscape planning for biodiversity conservation are scale and error. Because scale affects our detection and description of biological diversity across the landscape, it also affects our choice of where to focus conservation efforts. Similarly, error in measurement and prediction of species occupancy across the landscape contributes to uncertainty in biodiversity patterns and to the conservation designs derived from those patterns. This chapter focused on the implications of scale and error effects to geographically based conservation planning. We provided an overview of geographic conservation approaches before examining scale and error effects in detail using data from a case study in the southwestern United States. Finally, we provided suggestions on how conservation practitioners can address scale and error in conservation plans and offered our thoughts on future research needs.

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