CHAPTER

Social and Economic Considerations for Planning Wildlife Conservation in Large Landscapes

5

Robert G. Haight and Paul H. Gobster

People conserve wildlife for a variety of reasons. People conserve wildlife because they enjoy wildlife-related activities such as recreational hunting, wildlife viewing, or ecotourism that satisfy many personal and social values associated with people's desire to connect with each other and with nature (Decker et al. 2001). People conserve wildlife because it provides tangible benefits such as food, clothing, and other products. People conserve wildlife because they recognize that species are integral parts of larger ecosystems that perform a number of valuable services including nutrient cycling, water purification, and climate regulation (Daily 1997). People also conserve wildlife for its option value or potential to produce future benefits, such as new pharmaceuticals (Fisher and Hanneman 1986). Finally, people conserve wildlife for its existence value even if they will never see or use it (Bishop and Welsh 1992).

Because wildlife provides benefits to the public at large, government agencies and private organizations take responsibility for wildlife conservation. Programs for wildlife conservation typically protect species and habitat from human activities such as hunting, timber harvesting, or housing. As a result, conservation programs may impose substantial costs on other parts of society. Although it seems reasonable to evaluate conservation programs with an assessment of their benefits and costs, in practice, quantifying benefits is difficult, if not impossible. We are far from being able to obtain definitive estimates of wildlife benefits associated with nonconsumptive recreation activities, option values, existence values, and ecosystem services.

An alternative approach to evaluating conservation programs involves efficiency and trade-off analysis. Because public and private groups involved in wildlife conservation often have multiple objectives and limited resources to carry out their programs, efficiency analysis plays an important role in the evaluation of alternative programs. Efficiency analysis involves determining the strategy that maximizes a conservation objective given limited resources. Trade-off analysis involves the analysis of competing conservation goals in terms of how much of one goal must be given up to achieve another goal. Both types of analyses focus on the cost-effectiveness of alternative strategies and sidestep the difficult problem of estimating the total benefits of conservation.

In this chapter we describe key reasons why people conserve wildlife. We first examine contemporary attitudes and values associated with activities such as recreation, landscape restoration, and amenity migration. We then discuss ways to determine cost-effective habitat protection strategies and to identify the trade-offs among various conservation goals in case studies of habitat protection. We conclude with directions for future research. By "wildlife conservation," we mean a wide range of activities to protect and restore individual species and assemblages, from hands-on management of animals to land acquisition for habitat protection and restoration. Our definition of "large landscape" is similarly wide ranging and refers to conservation programs such as protecting old growth forest for northern spotted owl (*Strix occidentalis*) habitat on thousands of hectares in the western United States as well as programs to protect small habitat remnants from encroaching urban development in the Chicago, Illinois, USA metropolitan area. In these ways the largeness of landscapes is a social construction that depends on particular conservation goals. Finally, the term "human dimensions" describes the range of perceptions, attitudes, values, uses, and other interactions that people have with respect to natural resources such as wildlife (e.g., Decker et al. 2001).

PEOPLE-WILDLIFE INTERACTIONS AND TRENDS

We see three important trends in people-wildlife interactions: (1) direct interactions with wildlife through consumptive and nonconsumptive uses that are largely recreational in nature, (2) the restoration of landscapes and the wildlife that depends on them, and (3) indirect impacts on wildlife caused by "amenity migration" where people are increasingly purchasing and building seasonal or permanent homes on forested and other natural lands because of their amenity values. While these trends are occurring to varying degrees across the United States, we focus our discussion on data and examples within the Midwest.

Recreation

The transition from unregulated market and subsistence hunting to regulated recreational hunting at the turn of the 20th century helped many wildlife species to successfully rebound. But while an important part of contemporary recreational hunting is consumptive in nature and aimed at harvesting game, social scientists have come to understand how hunting also satisfies a wide range of human values. These include aesthetic values in viewing wildlife, personal values in the development and testing of self-reliance skills, social values such as camaraderie and the passing down of traditions across generations, and ecological values such as understanding ecological principles and developing an ethical relationship with wildlife (e.g., Dizard 2003). The balance of these values, however, can shift across time and location, affecting how hunting as a wildlife-related activity is engaged and perceived. For instance, urbanization and the severance of rural ties to the land can disrupt long-held social values and uses and is thought to be partly responsible for declines in hunting participation (Heberlein and Ericsson 2005). Income, education, and race/ethnicity are additional forces that are affecting a shift in expressed values, suggesting a continued decline in hunting participation in future years (Manfredo et al. 2003, Lopez et al. 2005).

Longitudinal statistics from the National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (U.S. Fish and Wildlife Service 2002*a*) reinforce these conceptual studies, and document that nationally the percentage of the U.S. population that hunts dropped from 10% in 1955 to 6% in 2001. Variations within these broad numbers are illustrated by statistics in Wisconsin, where in 2001, 9% of urban residents hunted compared to 24% of rural residents. Furthermore, the Wisconsin resident hunting population was 98% white non-Hispanic and 87% male versus 86% and 50%, respectively, for the entire state population; and half as many college graduates hunted (21%) compared to those with only high school diplomas (45%). The drop in participation over time should not discount the importance of hunting because in many states and localities hunting continues to be a major driver of social and economic activity. In Wisconsin, resident and nonresident hunters spent more than $800 million on licenses, equipment, trips, and other items in 2001 (U.S. Fish and Wildlife Service 2002*b*).

Manfredo and his colleagues suggest that this shift in the balance of values has drawn people away from recreational hunting and made them "protectionists" in their views of wildlife (Manfredo and Zinn 1996, Manfredo et al. 2003). But has this shift resulted in increased nonconsumptive recreation? The National Survey refers to observing, photographing, and feeding fish and wildlife as wildlife-watching activities, and distinguishes "residential" activities close to home from "nonresidential" activities more than a kilometer away from home. Here data for wildlife watching over the period 1980–2001 also show a significant drop in activity nationally, with an 18% decrease in the number of people that fed wildlife close to their home and a 19% decrease in those who took wildlife-watching trips away from home. Despite this drop, there are still many more who engage in nonconsumptive versus consumptive recreation, with 30% of U.S. residents engaging in some form of wildlife viewing. The large majority of this activity involves birds, especially people feeding and observing them in residential areas. The population of wildlife watchers is also much more broad-based than hunters. Again looking at statistics from Wisconsin, 47% of urban residents and 63% of rural residents participated in wildlife watching in 2001, and watchers were well distributed across gender, age, income, and education categories. While there may be some overlap in economic impact by those who watch wildlife and also hunt or fish during the same trip, the contribution of wildlife watchers is nonetheless considerable and in Wisconsin amounted to $1.3 billion in 2001 including more than $137 million for wild bird food alone (U.S. Fish and Wildlife Service 2002*b*).

These recreation data do not speak directly to planning wildlife conservation in large landscapes, but some key characteristics can be inferred. For Midwestern hunting, most of it focuses on species that favor early successional and mixed woodland-agricultural habitat rather than large, undisturbed landscapes. Bear hunting is one exception that has a small but dedicated cadre of participants and is concentrated in the large blocks of Northwoods forestland. Another exception might be waterfowl hunting, which often takes place on farmland but depends on significant wetland resources nearby. With residential bird watching and feeding as top activities, wildlife watching is also predominantly focused on fragmented habitat, but again there are important exceptions. Although places such as Yellowstone National Park may be better known destinations for watching charismatic megafauna such as grizzly bear (*Ursus arctos horribilis*) and timber wolves (*Canis lupus lycaon*) (e.g., Montag et al. 2005), the Midwest is also becoming known for this type of ecotourism. In Ely, Minnesota, the International Wolf Center has established itself as a center for "wolf country learning vacations" (International Wolf Center 2006).

Restoration

Landscape restoration is becoming a major means of land management as people increasingly value the existence of native species and an understanding of ecological principles (Gobster and Hull 2000). Landscape restoration involves the re-establishment of vegetation structure, native plant species, and natural disturbance processes such as fire that maintain plant communities; and the removal of roads, invasive species, and human activities such as cattle grazing or off-road recreation that are incompatible with the native ecosystem. Efforts to restore landscapes also involve the reintroduction of native wildlife species, which may play important roles in maintaining ecosystem structure and function. Consequently, large landscape restoration efforts can be controversial, as they involve a range of potentially conflicting management goals and human values (Gobster and Hull 2000).

Efforts to "re-wild" North America (Foreman 2004) include long-range visions for huge proposals such as The Wildlands Project for the Florida Everglades (Noss and Cooperrider 1994) as well smaller scale efforts that have been accomplished or are now underway. Often inherent in these proposals is the re-establishment of viable populations of large mammals (Maehr et al. 2001), but restoration can also focus on smaller birds, mammals, insects, and fish that are rare, threatened, or endangered.

Recent examples of habitat restoration in the Midwest illustrate the range of goals associated with large landscape planning for wildlife conservation. In 1980, the State of Missouri acquired the 1600 ha Prairie State Park for prairie restoration and bison reintroduction (Boyd 2003). The small herd of 78 bison (*Bison bison*) had high educational and symbolic value, but when the herd contracted brucellosis in 1990, the bison were removed because of the economic risk they posed to local livestock operations. A disease-free herd was reinstated into a fenced-in park, but the issue of disease transmission remains a key problem in reintroducing wild, free-ranging herds in large, unfenced landscapes such as Yellowstone National Park (Animal and Plant Health Inspection Service 2006).

The U.S. Forest Service manages jack pine (*Pinus banksiana*) for the federally endangered Kirtland's warbler (*Dendroica kirtlandii*) on the Huron-Manistee National Forests in central Michigan. The warbler depends on large, dense stands of young jack pine, which in turn depend on fire for regeneration. The low, sandy plains provide ideal ecological conditions for warbler restoration efforts, and while many people value the idea of restoring endangered species, they may not be supportive of jack pine management, as the monotypic stands have low scenic value (Schroeder et al. 1993). Additionally, concerns about using fire as a management tool stem back to the 1980 Mack Lake fire, a prescribed fire that escaped and killed 1 person, destroyed 44 homes, and burned more than 8000 ha of forestland before it was brought under control (Simard et al. 1983).

In 1996, the U.S. Forest Service acquired 7700 ha of the former Joliet Arsenal in Will County, Illinois, and established the Midewin National Tallgrass Prairie with a goal of restoring the tallgrass prairie and other native plant communities. There was early public interest in reintroducing bison and elk (*Cervus elaphus*) to the site, but the 2002 Prairie Plan recommended this be deferred to a future date. The site is on the Chicago metropolitan fringe and nearby residential growth and expected recreational demand increase the complexity of reintroduction issues, and fencing, removal of toxics from the former arsenal, and prairie plant re-establishment are needed before reintroduction can be considered (U.S. Forest Service 2002).

In the 1970s, the gray wolf (*Canis lupus*) was listed as an endangered species in the eastern United States, and its recovery plan prohibited hunting and facilitated natural recolonization in parts of Minnesota, Wisconsin, and Michigan. While the wolf became a cherished symbol of the wilderness forests, its movement into agricultural areas was greeted with much less enthusiasm. Today, many rural residents view the wolf as a threat to livestock, poultry, and pets (Chavez et al. 2005).

Amenity Migration

Landscape fragmentation can seriously impact the ability of wildlife managers to sustain species that require large blocks of undisturbed habitat. Land ownership parcelization and development can have significant impacts on landscapes (Sampson and DeCoster 2000). This trend is occurring nationwide but is especially acute near regions of the country with substantial surface water resources, public lands, and other amenity resources. For example, recreation has long been an important use of the Lake States Northwoods, and access to lakes for summer fishing and forests for fall hunting has been a major driving force behind private land acquisition. Recently, human demographic change has fueled an increased demand for owning a piece of the Northwoods. This amenity migration is resulting in further subdivision of private lands and development in the form of seasonal and permanent homes (Gobster and Schmidt 2000, Hammer et al. 2004). In a study of stakeholder perceptions of parcelization and development in the Wisconsin Northwoods, Gobster and Rickenbach (2004) identified four areas of interest and concern that highlight many of the social, environmental, and economic impacts: patterns, drivers, effects, and response strategies.

Stakeholders identified a number of trends in parcelization and development patterns. These included new development and land subdivision along small lakes and rivers and in forest areas that had formerly not been considered amenity attractions. They also spoke of a number of places in the Northwoods where private lands were being advertised for sale bordering national forest and state wild river properties. These patterns of parcelization and development could compromise critical habitat areas needed for wildlife as well as constrict the effectiveness of large blocks of public land by eroding the buffer of undeveloped private forestland that now surrounds them. Finally, stakeholders were concerned that the size of private forestland parcels considered "big" is steadily eroding. In northern Wisconsin where once 30 or 40 ha was thought to be a sizeable piece of land to own, 15 ha is now considered large by many.

Human demographic change is a major driving force behind amenity migration, and as more of the baby boom generation retires, more of them are purchasing and developing seasonal and retirement homes in amenity areas such as northern Wisconsin. Another driver is globalization. Many stakeholders noted the substantial transfer of locally owned industrial forests to multinational corporations, and they feared this transfer would "cream off" attractive vacation properties and fragment these large blocks of private forestland. Last, stakeholders discussed changes in technology such as the mound septic system that has led to increased home building in wet and rocky areas formerly unsuited to development (see also LaGro 1996).

The effects of parcelization and development on wildlife were well summarized by one stakeholder: "If you come to it from the aspect of wildlife, period, it's probably not a bad thing because fragmented property can support all kinds of wildlife. But if you come to it from the position of the diversity of wildlife, or wildlife that was historically present in Wisconsin, then it's probably a growing problem and it'd be a bad thing." In this respect, other stakeholders mentioned direct impacts to species including wolves, bears, lynxes, goshawks, and woodland and grassland songbirds. They also talked about indirect effects including loss of habitat because of invasive plants and loss of songbirds because of cowbird parasitism. One participant mentioned that changing landowner values are leading to a decline in timber harvesting and a subsequent "mapleization of the north," where the loss of earlier successional trees such as oaks and hickories will affect important food sources to many animals.

A final area of interest involved land use strategies to minimize or mitigate the negative effects of parcelization and development. Those strategies include conservancy zoning, where individual landowners cluster development and leave the larger proportion of their land in relatively undisturbed forest cover, and incentive programs such as the Wisconsin Managed Forest Law, which provides a tax break to landowners who develop a conservation plan for their property that may include managing their land for wildlife values (Gobster and Rickenbach 2004). Cross boundary management among private and public landowners is a growing method in which large landscapes can be more effectively managed to meet wildlife goals (e.g., Harper and Crow 2006). Another type of government payment program involves land acquisition for reserves to protect wildlife habitat and provide open space for recreation activities. Reserve-based modeling approaches to large-scale conservation planning are discussed in detail in following sections.

COST-EFFECTIVE WILDLIFE CONSERVATION

A cornerstone of wildlife conservation planning is establishing and expanding habitat reserves (Noss and Cooperrider 1994). Reserves are typically public lands protected from development and managed in part with wildlife objectives. Reserves have a variety of forms including public parks dedicated to nonconsumptive wildlife viewing, wilderness areas in national forests, or multiple-use lands managed for key species.

As we discussed in the previous section, residents of small towns and large cities alike are concerned about the environmental impacts of rapid growth and large-scale conversion of undeveloped to developed land. One result is that local governments and private land trusts have instituted policies to acquire land or conservation easements to preserve undeveloped land within or on the fringe of towns and cities. From 1996 through 2004, voters approved 1062 of 1373 referenda for open space and parks and authorized the use of $26.4 billion (2000 constant dollars) to acquire open space or development rights (Nelson et al. 2007). Agency planners have a variety of objectives for open space acquisition, including habitat protection for wildlife as well as economic efficiency (Ruliffson et al. 2002). In response, biologists and economists have developed reserve selection and design models, which suggest cost-effective ways to protect open space to attain wildlife objectives.

Reserved-based modeling approaches to large-scale conservation planning have been around since the 1980s and are the subject of a rich and growing literature (Kingsland 2002; Flather et al., this volume; Noon et al., this volume). We discuss three broad categories of models: reserve selection models, reserve design models, and reserve design models with population dynamics (Table 5-1). Following Williams et al. (2005), we distinguish the terms "site," "reserve," and "reserve system." A site is a selection unit—a piece of land that

Table 5-1 Reserve-Based Modeling Approaches to Large-Scale Conservation Planning: A. Reserve Selection Models, B. Reserve Design Models, and C. Reserve Design Models With Population Dynamics

Problem	Objective	Reference
A. Reserve selection models		
Maximum species covering	Maximize number of species protected for a given budget	Church et al. 1996
Bi-criteria species covering	Maximize number of species protected and some other conservation objective	Church et al. 2000, Ruliffson et al. 2003
Maximum expected species covering	Maximize expected number of species protected for a given budget	Camm et al. 2002, Arthur et al. 2004
Dynamic species covering	Maximize expected number of species protected at end of horizon	Costello and Polasky 2004, Haight et al. 2005, Turner and Wilcove 2006
B. Reserve design models		
Reserve proximity	Minimize sum of pairwise distances between reserves	Önal and Briers 2002
Reserve connectivity	Maximize number of adjacent reserves	Nalle et al. 2002
Reserve compactness	Minimize boundary length of reserves	Fischer and Church 2003
C. Reserve design models with population dynamics		
Metapopulation size	Maximize metapopulation size	Hof et al. 2001
Safe minimum standard	Maximize probability of metapopulation persistence	Montgomery et al. 1994, Moilanen and Cabeza 2002, Haight and Travis 2008
Surviving populations	Maximize expected number of surviving populations	Haight et al. 2004*a*

may be selected for protection. A site is undeveloped open space belonging to one or more cover types, including forest, grassland, pasture, or farm, that provide habitat for wildlife. A reserve is a single site or a contiguous cluster of sites that has been selected for protection. A reserve system is a set of multiple, spatially separated reserves. Reserve selection models identify sites to protect to maximize some measure of biological diversity (e.g., species richness). Reserve design models incorporate spatial attributes of the selected sites (e.g., connectivity) as conservation objectives. Reserve design models with population dynamics identify sites to protect to achieve objectives related to population size or persistence. We begin by discussing reserve selection models with an objective of maximizing species richness in the selected sites subject to a budget constraint. The problem is used to explain basic economic principles of cost-effectiveness, marginal cost, and trade-off analysis.

Reserve Selection Models

Reserve selection models are based on information about the distribution of species or other conservation features (e.g., habitat types) among sites and targets for protecting those features. For convenience, we will use species as the feature of interest. Each site is described by a list of species that it contains, and a species is covered or represented if at least one site that contains the species is selected for protection. Early models selected the minimum number of sites that represented all species from a list of target species (e.g., Margules et al. 1988). Selecting sites to minimize the cost of protecting all species is called the species set covering problem, an analogue of the location set covering problem from facility location science (ReVelle et al. 2002). Recognizing that resources may limit the number of sites selected for protection, later models maximized the number of species or conservation features that could be represented within a given number of sites (e.g., Church et al. 1996). This latter type of model is called the maximal species covering problem (ReVelle et al. 2002), and it provides case-specific policy guidance on sets of sites that efficiently achieve conservation goals and trade-offs between conservation goals. Cabeza and Moilanen (2001), ReVelle et al. (2002), and Rodrigues and Gaston (2002) summarize applications of reserve selection models.

Maximum Species Covering Problem.— Here, we describe an application of the maximal species covering problem in a case study in the Midwestern United States. The application is in the Lake County portion of the Fox River watershed northwest of the city of Chicago (Fig. 5-1). In response to rapid population growth and conversion of open space to housing and commercial development, Lake County planners are interested in acquiring land to protect rare animals and plants and provide equitable access to recreation. It is important to note that the focus was on rare animals and plants rather than all animals and plants. To help planners identify cost-effective sets of sites, we formulate a maximal species covering problem and analyze the cost of increasing the number of species represented in the selected sites.

The analysis is conducted using data for 31 privately owned open-space sites in the Lake County portion of the Fox River watershed (see Haight et al. 2005 for details). The sites vary in size from 1 to 313 ha, with a median of 29 ha (Table 5-2). Each site is described by a list of rare plants and animals present. Collectively, 27 rare species occur in the 31 sites, and species richness of individual sites varies from 1 to 9 species. Because the budget constraint places an upper bound on total area of sites selected, we expect that smaller sites with

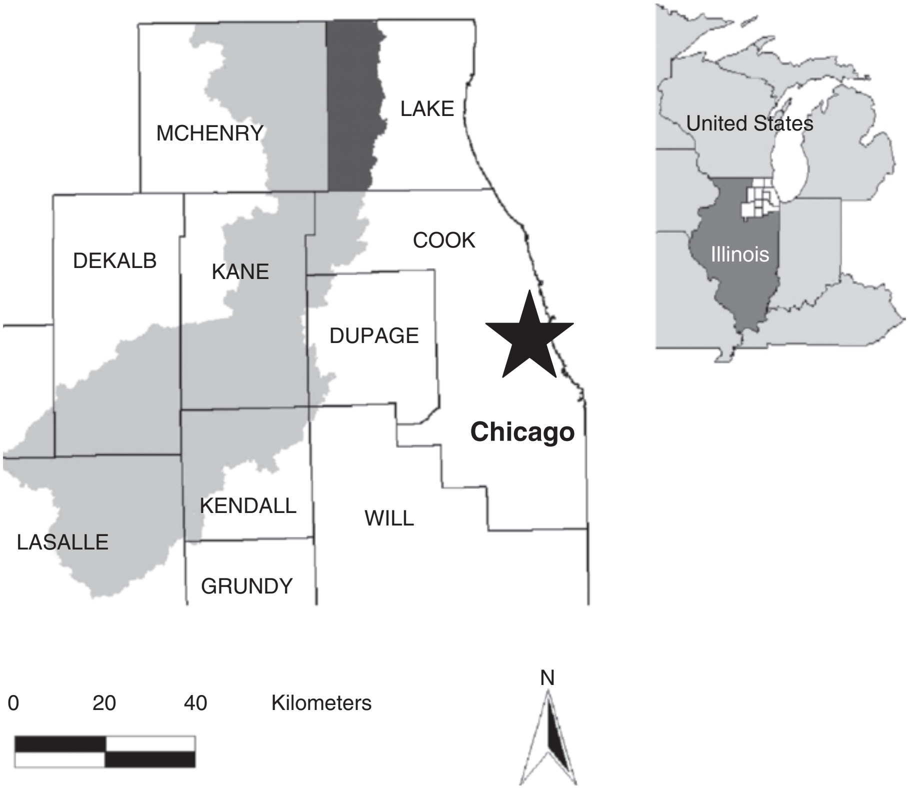

FIG. 5-1

Fox River watershed (shaded gray) in counties of northeastern Illinois, USA. The study area (shaded black) is the northeastern portion of the watershed located in Lake County.

more species may be preferable, and we list the number of species per unit area in Table 5-2.

The maximal species covering problem is a linear-integer programming problem with a cost constraint that limits resources spent for site protection. The problem is solved in seconds using commercial software on a laptop computer. The model has the following notation:

i, I = index and set of species in need of protection,
j, J = index and set of potential reserve sites,
B = upper bound on budget,
c_j = cost of protecting site j,
M_i = set of sites that contain species i,
x_j = 0–1 variable: 1 if site j is selected for protection, 0 otherwise,
y_i = 0–1 variable: 1 if species i is represented in at least one protected site, 0 otherwise.

Table 5-2 Attributes of Open-Space Sites in the Fox River Watershed of Lake County, Illinois, USA

Site	Area (ha)	Number of Species	People With Access (1000s)	Species per ha	Rank	People With Access per ha (1000s)	Rank
1	37	1	0.0	0.03	26	0.00	30
2	40	2	8.0	0.05	19	0.20	18
3	65	2	2.7	0.03	25	0.04	24
4	24	2	9.3	0.08	13	0.39	13
5	9	1	2.9	0.11	10	0.31	15
6	47	3	3.3	0.06	16	0.07	23
7	1	5	1.8	5.00	1	1.80	1
8	16	1	17.6	0.06	17	1.09	5
9	39	4	36.1	0.10	11	0.93	6
10	121	5	9.3	0.04	23	0.08	22
11	141	2	3.3	0.01	28	0.02	27
12	29	2	0.0	0.07	14	0.00	31
13	22	1	33.8	0.05	21	1.55	3
14	9	5	2.7	0.56	3	0.30	16
15	84	7	21.4	0.08	12	0.26	17
16	23	1	9.1	0.04	22	0.39	12
17	5	4	3.1	0.82	2	0.64	8
18	14	3	32.9	0.21	6	2.33	2
19	13	2	6.0	0.16	8	0.48	10
20	30	2	26.7	0.07	15	0.88	7
21	7	1	10.5	0.14	9	1.44	4
22	189	9	2.7	0.05	20	0.01	28
23	313	2	32.5	0.01	31	0.10	20
24	80	1	2.4	0.01	29	0.03	26
25	10	2	5.8	0.20	7	0.57	9
26	142	1	5.8	0.01	30	0.04	25
27	92	2	35.2	0.02	27	0.38	14
28	17	4	0.2	0.23	5	0.01	29
29	24	1	2.7	0.04	24	0.11	19
30	7	2	2.9	0.29	4	0.42	11
31	37	2	3.1	0.05	18	0.08	21

The model is formulated as follows:

$$\text{Maximize}: \sum_{i \in I} y_i \tag{1}$$

$$\sum_{j \in M_i} x_j \geq y_i \qquad \text{for all } i \in I \tag{2}$$

$$\sum_{j \in J} c_j x_j \leq B \tag{3}$$

$$x_j, y_i \in \{0, 1\} \qquad \text{for all } i \in I \text{ and } j \in J \tag{4}$$

The objective (Eq. 1) is to maximize the number of species that are represented or covered in the set of selected sites. Eq. 2 enforces the logic of covering: a species is considered covered ($y_i = 1$) if at least one site that contains the species is selected for protection. Eq. 3 is the budget constraint that limits how much can be spent on site protection. Eq. 4 describes the integer restrictions on the decision variables.

The cost constraint (Eq. 3) is a key part of the maximal species covering problem because it represents the decision maker's goal of staying within a budget. In our application, we use areas of sites as proxies for site costs because we do not know the dollar value of every site. We therefore assume that the decision maker has an overall area budget for selecting sites. Solving the problem for a given value of the budget level B allows the determination of an efficient set of sites, where efficiency means that there are no other sets of sites that provide a higher level of species coverage and stay within the budget. Solving the problem with increasing levels of B allows construction of a cost curve showing the cost of increasing the number of species covered.

We determine the optimal sets of sites to protect for budgets ranging from 1 to 618 ha and plot the cost curve in Fig. 5-2. The slope of the cost curve is the marginal cost of species protection, which is the area required to protect an additional species. Marginal cost is small (4 ha per species) as coverage increases from 5 to 20 species, moderate (34 ha per species) in the range of 20 to 25 species, and large (195 ha per species) for levels of species coverage greater than 25.

As the budget increases, the optimal set of sites is not always a matter of adding another site to the previously selected set. For example, to increase species coverage from 20 to 22 species, one site can be added to the list of protected sites (Table 5-3). However, increasing species coverage above 22 species involves dropping one site and adding up to four others. Nevertheless, there is consistency in sites selected for protection. Six sites (7, 8, 17, 18, 21, and 30) are selected whenever the budget is greater than 50 ha. These sites are small (<16 ha), rank in the top 10 in terms of species per ha, and contain endemics (Table 5-2).

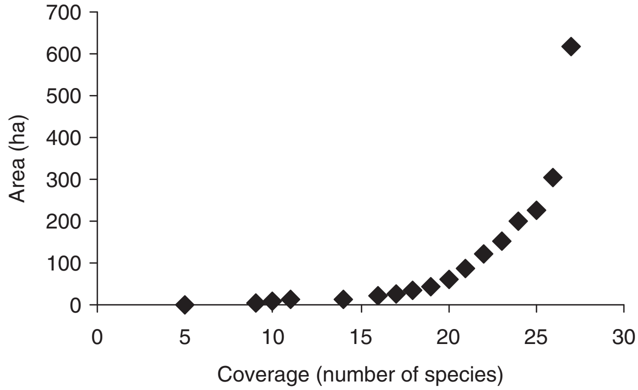

FIG. 5-2

Cost curve showing area protected versus number of species covered for the site selection options in the Fox River watershed of Lake County, Illinois, USA.

Table 5-3 Optimal Sets of Sites Selected for Increasing Area Budgets in the Fox River Watershed of Lake County, Illinois, USA

Objective Values		Sites Protected											
Species	**Area (ha)**	**3**	**7**	**8**	**14**	**15**	**17**	**18**	**20**	**21**	**22**	**23**	**30**
20	59		X	X	X		X	X		X			X
22	123	X	X	X	X		X	X		X			X
25	228	X	X	X		X	X	X	X	X			X
27	618	X	X	X			X	X		X	X	X	X

Bi-Criteria Covering Problem.—Metropolitan planners may have a variety of objectives for land acquisition including habitat protection for rare species, public accessibility, and economic efficiency (Ruliffson et al. 2002). In this section, we extend the maximum species covering problem to handle a second objective of maximizing the accessibility of open space sites to urban populations in the county (Ruliffson et al. 2003, Haight et al. 2005).

Multiobjective site selection models are useful tools for investigating the opportunities for simultaneously meeting multiple conservation objectives (Rothley 1999, Church et al. 2000, Marianov et al. 2004). Analyses typically determine the trade-off between objectives—the pareto-optimal curve that displays the best value of one objective given a required achievement of the other. In addition, important information can be obtained by analyzing the site selection decisions associated with alternative solutions along the trade-off curve, including identification of sites that should be selected no matter what the decision maker's position on the relative importance of the two objectives (Schilling et al. 1982). We show how to display model solutions in terms of both the objectives and decisions in our multiobjective analysis.

There are 34 towns in western Lake County. Based on the 2000 U.S. Census, the towns collectively held 222,000 people, and individual towns were home to 1,000 to 30,000 people. We assume that people in a town have access to a site if the site is within 3.2 km (2.0 miles) of the town. Based on the average distance between each town and each site, we know towns that are within the required distance of each site, and based on the population of each town, we list the total population with access to each site (Table 5-2). Almost all sites have at least 2,000 people within 3.2 km, and five sites have more than 30,000 people within 3.2 km. We also compute the number of people with access per unit area as an approximate index of site desirability.

In addition to the notation listed previously, the bi-criteria site selection model has the following:

k, K = index and set of towns,
Q_1 = number of species represented in the protected sites,
Q_2 = number of people with access to the protected sites,
r_k = number of people in town k,
N_k = set of sites that are within 3.2 km of town k,
w = objective weight: $0 \leq w \leq 1$,
z_k = 0–1 variable: 1 if town k has at least one protected site within 3.2 km, 0 otherwise.

The model is formulated as follows

$$\text{Maximize}: wQ_1 + (1-w)Q_2 \tag{5}$$

$$Q_1 = \sum_{i \in I} y_i \tag{6}$$

$$Q_2 = \sum_{k \in K} r_k z_k \tag{7}$$

$$\sum_{j \in N_k} x_j \geq z_k \qquad \text{for all k} \in K \tag{8}$$

$$\sum_{j \in M_i} x_j \geq y_i \qquad \text{for all } i \in I \tag{9}$$

$$\sum_{j \in J} c_j x_j \leq B \tag{10}$$

$$x_j, y_i, z_k \in \{0,1\} \qquad \text{for all } i \in I, j \in J, k \in K \tag{11}$$

The objective (Eq. 5) is to maximize the weighted sum of the two objective functions: the number of species represented in protected sites (Eq. 6) and the number of people with access to protected sites (Eq. 7). Public access is the number of towns with access weighted by population size (r_k). The weight w represents the decision maker's position on the relative importance of the two objectives. When w is closer to one, more weight is placed on maximizing the number of species covered. When w is closer to zero, more weight is put on maximizing the number of people with access to protected sites. Eq. 8 is the condition under which town k has access to protected sites (i.e., $z_k = 1$): at least one site that is within 3.2 km of town k must be selected for protection. Eqs. 9–11 are the species coverage definition, the budget constraint, and the integer restrictions on the decision variables, respectively.

The analysis focuses on how the optimal set of protected sites varies as we trade off species representation and public access under different budget levels. We compute optimal site selections for problems in which the objective function weight is decreased from 1.0 to 0.0 in increments of 0.05 subject to area constraints of 81 ha and 200 ha. The curves showing the trade-offs between species representation and public access have concave shapes in which species representation drops as public access increases (Fig. 5-3). The points on each curve represent nondominated sets of sites and their relative performance with respect to the two objectives under a given level of budget. For each nondominated set of sites, improvement in one objective cannot be achieved without simultaneously causing degradation in the value of the other objective. As a result, the points on each trade-off curve represent a frontier beyond which no better solutions can be found.

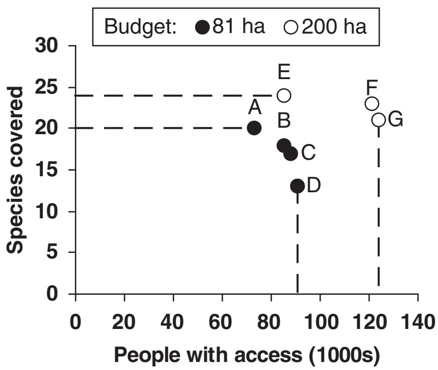

FIG. 5-3

Trade-offs between open-space protection objectives of maximizing species coverage and maximizing public access under different area budgets in the Fox River watershed of Lake County, Illinois, USA.

Among the nondominated solutions for a given budget, the best depends on the decision maker's preference for the two objectives. If species representation is most important and the budget is 81 ha, the choice is alternative A, in which species representation is 20 (74% of the maximum representation without a budget constraint) and public access is 73,000 people (33% of the maximum accessibility). The dashed horizontal line between the y-axis and point A indicates that we found a number of solutions that had the same species representation as alternative A but with less public access. A move from alternative A to alternative B or C increases public access more than 20% and reduces species representation 15%. The highest level of public access under this budget (point D, 91,000 people) is obtained with a 35% reduction in species representation. The dashed vertical line from point D to the x-axis indicates that there are a number of solutions with the same level of public access as alternative D but with less species representation. Increasing the budget from 81 ha to 200 ha shifts the trade-off curve up and to the right while reducing the trade-off between the objectives (Fig. 5-3).

To complement the trade-off curves, it is important to look at the site selection decisions and identify core sites, which are sites selected for protection regardless of the weights given to the objective functions. With a budget of 81 ha, three core sites (7, 18, and 20) are protected in all four solutions (Table 5-4). With a budget of 200 ha, there are four additional core sites (8, 15, 17 and 21). The core sites are typically small (<30 ha) and have relatively large numbers of species and people with access (Table 5-2). As a result, the

Table 5-4 Objective Function Values and Sites Selected for Protection for Nondominated Solutions with Area Budgets of 81 ha (Solutions A,B,C,D) and 200 ha (Solutions E,F,G) in the Fox River Watershed of Lake County, Illinois, USA

	Objective Value		**Sites Protected**												
Solution	**Species**	**People (1000s)**	**3**	**7**	**8**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**30**	**Area Protected (ha)**
A	20	73		X		X			X	X		X	X	X	73
B	18	85		X	X	X			X	X		X			75
C	17	88		X	X				X	X		X	X	X	80
D	13	91		X	X					X	X	X	X		81
E	24	85	X	X	X		X		X	X			X	X	198
F	23	121		X	X	X	X	X	X	X		X	X	X	196
G	21	124		X	X		X	X	X	X	X	X	X		193

core sites rank in the top half in terms of species per hectare and people with access per hectare, two indices of desirability (Table 5-2).

There is a lot of overlap in the sets of sites selected for protection in the alternative solutions under each budget (Table 5-4). As a result of the overlap in composition, the choice between alternative solutions on a trade-off curve involves shifting a small portion of the area budget between a few sites. For example, with a budget of 81 ha, moving from alternative A to alternative C shifts about 11% of the budget from protecting site 14 to protecting site 8. Moving from alternative C to alternative D involves a shift of about 15% of the budget from protecting sites 17 and 30 to protecting site 19. Because adjacent solutions on the trade-off curve often differ in only a few sites, decisions about which alternative to select can focus on the strengths and weaknesses of those few sites.

Maximum Expected Species Covering Problem.—In many cases information about the presence and absence of species in sites is uncertain, and presence is expressed as a probability of occurrence. The species covering problem can be extended to handle this information and maximize the expected number of species covered subject to a budget constraint. Let p_{ij} be the probability that species i exists at site j where the probabilities are independent across sites. Defining v_i as the probability that species i is not covered in the sites selected for protection, we can write

$$v_i = \prod_{j \in J} \left(1 - p_{ij}\right)^{x_j} \qquad \text{for all } i \in I \tag{12}$$

where x_j is the 0–1 decision variable for whether site j is selected for protection. Eq. 12 follows from the fact that a selected set of sites fails to cover a given species i if that species is absent from all the selected sites. The independence assumption allows us to write v_i as a product of absence probabilities over all sites. The problem is to determine the values of the site selection variables to maximize the expected number of species covered subject to a budget constraint:

$$\text{Maximize}: \sum_{i \in I} (1 - v_i) \tag{13}$$

$$\sum_{j \in J} c_j x_j \leq B \tag{14}$$

$$x_j \in \{0, 1\} \qquad \text{for all } j \in J \tag{15}$$

The problem in Eqs. 12–15 is nonlinear and cannot be converted to an equivalent linear integer program because the objective function is the sum of terms that involve the products of the decision variables x_j. Nevertheless, a linear approximation of the nonlinear problem can be formulated and solved using commercial software (Camm et al. 2002), and the model has been illustrated using probabilistic occurrence data for 403 terrestrial vertebrates in 147 candidate sites in western Oregon, USA (Arthur et al. 2004).

Dynamic Species Covering Problem.—The maximum species covering problems described so far have taken a static approach to conservation planning. Static models are designed to select cost-effective sets of sites to protect biodiversity given current information about species occurrence and site availability. The models assume that decisions are made all at once and protection takes place rapidly before site degradation or loss. This may be a reasonable first-pass approach to the immediate problem of slowing biodiversity loss; however, planning is a dynamic process that incorporates new information as it unfolds. Researchers have begun to develop methods to address sequential site selection problems with budget restrictions and uncertainties about site degradation and loss with the objective of maximizing the expected number of species covered in protected sites at the end of a planning horizon.

One approach to sequential site selection is building a stochastic dynamic programming model that includes periodic budget constraints and uncertainty about future site availability (Costello and Polasky 2004). The optimal solution includes the set of sites to protect now along with a policy or rule that describes the sites to protect in the future depending on species already protected and sites that are available. Unfortunately, dynamic programming is computationally intensive and has been used to solve problems with fewer than about 10 sites, far less than can be handled with heuristic algorithms. Simple rules for site selection based on current gaps in species coverage and current threats to habitat loss perform reasonably well on small problems in comparison with optimal policies obtained from dynamic programming (Costello and Polasky 2004), and similar heuristics have been applied to large, practical problems (Meir et al. 2004, Turner and Wilcove 2006).

Another approach to sequential site selection involves a two-period linear-integer model in which uncertainty about future site availability is represented with a set of probabilistic scenarios (Snyder et al. 2004, Haight et al. 2005). The decision variables include the set of sites to protect now and sites to protect in the second period, depending on availability. The linear-integer formulation allows solution of realistic-sized problems with commercial software on personal computers. Furthermore, the formulation can be expanded to model multiple objectives and constraints that allow for budget allocation between periods.

Reserve Design Models

A weakness of site selection models is their ignorance of the effects of size and spatial arrangement of reserves on species dynamics, and as a result, there is no guarantee that species represented will persist (Cabeza and Moilanen 2001). One way to improve the models is to include spatial objectives for reserve design that are related to species persistence. For example, some species require large areas of compact and contiguous habitat for survival, whereas other species can survive in disjunct habitat patches as long as they are relatively close together. Reserve proximity, contiguity, and compactness can be formulated as spatial objectives in linear-integer programming models (see Williams et al. 2005 for a review), and we give examples of each type of model in the following sections. For ease of presentation, each model includes a spatial objective combined with a species coverage constraint. By varying the level of the constraint, trade-offs between the spatial and coverage objectives can be obtained. The models can be easily expanded with budget constraints and other conservation objectives.

Reserve Proximity Problem.—A reserve system in which the reserves are close together may be preferred to facilitate movement of individuals between reserves. Shorter migration distances facilitate recolonization of areas where a species has become locally extinct and help prevent the loss of genetic diversity because of inbreeding. One way to reduce the distances between reserves is to minimize the sum of distances between all pairs of selected sites. Letting d_{jk} be the distance between sites j and k and u_{jk} be a 0–1 variable for whether both sites j and k are selected, the problem can be written as follows:

$$\text{Minimize}: \sum_{j \in J} \sum_{k > j} d_{jk} u_{jk} \tag{16}$$

$$U_{jk} \geq x_j + x_k - 1 \qquad \text{for all } j, k \in J, k > j \tag{17}$$

$$\sum_{j \in M_i} x_j \geq R \qquad \text{for all } i \in I \tag{18}$$

$$x_j \in \{0, 1\}, u_{jk} \in \{0, 1\} \qquad \text{for all } j, k \in J \tag{19}$$

The objective (Eq. 16) minimizes the sum of the pairwise distances between selected sites subject to a species coverage constraint (Eq. 18) that requires each species i to be represented in at least R selected sites. Eq. 17 enforces the definition of u_{jk} by requiring both $x_j = 1$ and $x_k = 1$ for $u_{jk} = 1$. Önal and Briers (2002) apply this formulation to the problem of selecting a subset of 131 pond sites in Oxfordshire, United Kingdom, to protect 256 invertebrate species.

Reserve Connectivity Problem.—Another objective of reserve design is to maximize the structural connectivity of the selected sites. Structural connectivity refers to the physical contiguity of sites and is desirable to create larger reserves or corridors between reserves. In situations in which the landscape is subdivided into contiguous polygons representing candidate sites, structural connectivity can be promoted by selecting sites for protection that are adjacent to each other. The objective is to maximize the number of adjacent pairs of selected sites:

$$\text{Minimize}: \sum_{j \in J} \sum_{k \in A_j, k>j} u_{jk} \tag{20}$$

$$u_{jk} \geq x_j + x_k - 1 \qquad \text{for all } j \in J, k \in A_j, k > j \tag{21}$$

$$\sum_{j \in M_i} x_j \geq R \qquad \text{for all } i \in I \tag{22}$$

$$x_j \in \{0,1\}, u_{jk} \in \{0,1\} \qquad \text{for all } j,k \in J \tag{23}$$

where the set A_j represents all sites that are adjacent to (share a boundary with) site j. Nalle et al. (2002) employed a similar formulation to the problem of selecting a subset of 4181 sites in Josephine County, Oregon, to protect examples of 13 habitat types.

Reserve Compactness Problem.—The shape of reserves in a reserve system may be important for species survival, and many authors advocate creating compact reserves that are nearly circular and have low edge-to-area ratios. Compact reserves are better for edge-intolerant species such as tropical songbirds that prefer large areas of interior habitat for nesting. In reserve design models, compactness is measured by the total length of the boundaries (perimeters) of all the reserves. Total boundary length is the difference between the length of the boundaries of all the selected sites and two times the length of the shared boundaries between the selected sites. Letting b_j be the length of the boundary of site j and sb_{jk} be the length of the shared boundary between sites j and k, the problem of minimizing total boundary length can be written as follows:

$$\text{Minimize}: \sum_{j \in J} b_j x_j - 2 \sum_{j \in J} \sum_{k \in A_j, k>j} sb_{jk} u_{jk} \tag{24}$$

$$u_{jk} \geq x_j + x_k - 1 \qquad \text{for all } j \in J, k \in A_j, k > j \tag{25}$$

$$\sum_{j \in M_i} x_j \geq R \qquad \text{for all } i \in I \tag{26}$$

$$x_j \in \{0,1\}, u_{jk} \in \{0,1\} \qquad \text{for all } j,k \in J \tag{27}$$

In the objective function (Eq. 24), the boundary length of the reserve system is calculated by adding the boundary lengths of the selected sites and then subtracting twice the length of the boundaries shared by selected sites that are adjacent. Fischer and Church (2003) utilized this model to analyze trade-offs between total area and compactness of reserve systems to protect examples of 55 plant community types in northern California forests.

Reserve Design Models with Population Dynamics

While the reserve design models discussed in the preceding sections include spatial objectives, they do not model species' population dynamics. In this section, we discuss reserve design models that explicitly incorporate population dynamics. We begin with a discussion of deterministic reserve design models that aim to maximize the size of the metapopulation (collection of subpopulations residing in separate sites) based on estimates of population growth and dispersal (Hof and Bevers 2002). Then, we describe two types of reserve design problems that incorporate stochastic models of population dynamics. These are important because the fields of wildlife management and conservation biology have a long history of developing stochastic models of population viability, which help managers predict the likelihood that wildlife populations survive under various levels of habitat protection (Boyce 1992; Beissinger and Westphal 1998; Beissinger et al., this volume). There are two broad types of viability models: Demographic models predict the birth, death, and migration of individuals in one or more localized populations (e.g., Liu et al. 1995); and incidence function models predict the extinction of local populations and colonization of empty habitat patches (Hanski 1994). Both types of models incorporate uncertainty in one or more demographic parameters, and Monte Carlo methods are used to sample from the underlying distributions and simulate populations many times for different combinations of parameter values. Thus, stochastic population models yield probabilistic results, which are typically summarized by performance measures such as the probability that the ending metapopulation size exceeds a threshold or the expected number of surviving populations.

Metapopulation Size Problem.—A simple way to model change in the size of a metapopulation is to estimate the growth rate (per capita reproduction minus net mortality) of each subpopulation and a matrix of dispersal parameters that govern movement of individuals between subpopulations. Given these parameters, Bevers and Flather (1999) formulated a system of linear difference equations for metapopulation dynamics and explored the effects of patch size, number, and spatial arrangement on the size of hypothetical metapopulations. Because the model is a system of linear equations, it can be put into a linear programming model for site selection to maximize metapopulation size subject to budget constraints. Hof et al. (2001) described an application to black-tailed prairie dog (*Cynomys ludovicianus*) conservation in the Buffalo Gap National Grassland in South Dakota, USA. First, they identified 601 patches of prairie dog habitat covering approximately 20,000 ha and defined choice variables for the amount of each patch that is zoned for prairie dog colonies. Then, they developed a model of the prairie dog population in which each subpopulation grows exponentially until patch carrying capacity is reached, emigration is limited to subpopulations that exceed patch carrying capacity, and the number of dispersers that reach each patch depends on inter-patch distances. Finally, they explored the effects of budget constraints on total population size over an 8-year horizon and suggested priority locations for habitat expansion. While this reserve design model contains some basic elements of population dynamics, it ignores features of population models (e.g., age-dependent birth and mortality rates, density-dependent emigration rates, and parameter uncertainty) that are difficult to formulate in linear programs. Later we discuss ways to incorporate stochastic demographic models of population viability into reserve design problems.

Safe-Minimum-Standard Problem.—In the United States, the Endangered Species Act requires the U.S. Fish and Wildlife Service and other agencies to prepare recovery plans for threatened and endangered species. Recovery plans usually include population size goals assuming that species are viable when those goals are attained. Recognizing that population dynamics and species survival are uncertain, scientists have defined population viability in probabilistic terms as the likelihood of survival over some time period (e.g., Boyce 1992, Beissinger and Westphal 1998). We define population viability as a safe minimum standard—the likelihood that population size exceeds a minimum size target at the end of the planning horizon—and we assume that site selection affects the probability of exceeding the target (also see Millspaugh et al., this volume). Then, we can estimate the trade-off between higher probabilities and the costs of attaining them. These cost curves, first developed by Montgomery et al. (1994) and Haight (1995), quantify important components of the social costs and benefits of species protection.

Suppose we have a set of disjunct sites that can support subpopulations of an endangered species and a limited budget for habitat protection. By "disjunct," we mean that sites are physically separated from each other; however, individuals can move between sites. The objective is to determine the sites to protect to maximize the viability of the metapopulation. A metapopulation is considered viable if its size is greater than a predefined minimum population size (the safety standard). Because of uncertainty in population dynamics, population size at the end of the horizon is uncertain and the viability objective is probabilistic. As before, we define 0–1 decision variables x_j for all $j \in J$ for site protection. In addition, we define random variable $N(x)$ as the size of the metapopulation in ending period T as a function of the decision variables and parameter n as the target population in period T. The safe-minimum standard problem is

$$\text{Maximize : prob}[N(x) \geq n] \tag{28}$$

$$\sum_{j \in J} c_j x_j \leq B \tag{29}$$

$$x_j \in \{0, 1\} \qquad \text{for all } j \in J \tag{30}$$

The objective (Eq. 28) is to maximize the probability that the metapopulation exceeds a predetermined size target at the end of the management horizon subject to a budget constraint (Eq. 29) and binary restrictions on the decision variables (Eq. 30). A solution is a cost-effective set of sites to protect to maximize the likelihood of exceeding the population size target. By increasing the budget B and re-solving the problem, we can estimate the cost of attaining higher levels of certainty of attaining the target. The model explicitly recognizes that species survival is not certain and that the decision to save a species is not an all or nothing choice. Rather, the model measures the performance of a conservation plan in terms of the probability of attaining a population size target and allows determination of the cost of attaining higher probabilities.

This is a difficult optimization problem because the objective function is estimated using a stochastic population model, which typically has nonlinear relationships and random variables that cannot readily be put into classical integer and mixed-integer programming formulations. Instead, tools are needed to join simulation and optimization to find good approximations of optimal reserve design. One approach is simulation optimization in which the probability of metapopulation persistence is estimated via stochastic simulation until a suitable approximation of the optimal reserve design is found. A disadvantage of simulation optimization is computational intensity: Multiple replications of the stochastic population model may be required to obtain a useful estimate of the probability of persistence for each set of sites evaluated. Simulation optimization strategies are beginning to be developed and tested with incidence function models (Moilanen and Cabeza 2002) and demographic models (Haight and Travis 2008) of population dynamics. An excellent application of the safe-minimum-standard problem is a study of the cost of protecting old growth forest for northern spotted owl habitat in the Pacific Northwest (Montgomery et al. 1994).

Surviving Populations Problem.—In some cases, populations of an endangered species exist in disjunct sites that are isolated enough that migration between sites is inconsequential. If we have information for each site about the relationship between risk of population extinction and the amount of habitat in the site, we can formulate a model for determining the amount of habitat to add to each site to maximize the expected number of populations that survive over the management horizon. Here, the decision variable x_j is the amount of habitat to add to site j and the parameter a_j is the amount of already-protected habitat. In addition, we define $N_j(a_j + x_j)$ as a random variable for the population size in site j in ending period T as a function of the total amount of habitat in the site, and n as the minimum viable population size. Then, prob $[N_j(a_j + x_j) < n]$ is defined as extinction risk and the optimization problem

$$\text{Maximize}: \sum_{j \in j} 1 - \text{prob}\left[N_j\left(a_j + x_j\right) \leq n\right] \tag{31}$$

$$\sum_{j \in J} c_j x_j \leq B \tag{32}$$

$$x_j \geq 0 \qquad \text{for all } j \in J \tag{33}$$

is to maximize the expected number of populations that survive over the management horizon (Eq. 31) subject to a budget constraint on the total cost of added habitat (Eq. 32). The probability of extinction of each population depends on the amount of habitat, which is the sum of the already-protected habitat and the newly added habitat.

Haight et al. (2004*a*) used this formulation to address a problem of allocating a fixed budget for habitat protection among disjunct populations of the endangered San Joaquin kit fox (*Vulpes macrotis mutica*) in California to maximize the expected number of surviving populations. A key part of the problem is estimating prob[N_j $(a_j+x_j) \leq n$] as a function of the amount of habitat in each site. They used response surface analysis in the following way. First, a stochastic demographic model of a disjunct kit fox population was used to predict extinction risk in 100 years in habitat patches of increasing size. For each patch area, the estimator of extinction risk was the percentage of 1000 independent simulations in which population size was less than 10 individuals in 100 years. Then, the predictions were used to estimate a relationship between extinction risk and patch area. The risk-area relationship was a logistic function estimated using a form of logistic regression called the minimum logit chi-squared method (Maddala 1983). Logistic regression describes a binary response as a function of one or more explanatory variables. In this case, the binary response was extinction or persistence of a population in a habitat patch, and the explanatory variable was patch area. The minimum logit chi-squared method of estimation is appropriate when there are multiple observations of the binary response for each level of the explanatory variable. Risk-area curves were estimated for each of eight populations and then incorporated into the optimization model (Eqs. 31–33). The results included priorities for reserve expansion under increasing budgets and a cost curve showing funding required for incremental increases in the number of surviving populations.

Discussion of Modeling Approaches

Reserve selection and design models provide guidance to planners about cost-effective ways to achieve wildlife objectives and trade-offs. The type of model to use depends on the scope of the problem, the management objective, and the information available. With an objective of maximizing the number of species within protected sites, reserve selection models provide information to decision makers about sets of sites that protect the most species within the budget for acquiring land, and the models provide the marginal cost of increasing the number of species protected. Sometimes the marginal cost of protecting the last species within the scope of the problem is very high (e.g., Fig. 5-2), which suggests that funding could be invested in other conservation projects with greater benefits. While reserve selection models provide a first-pass solution, they ignore reserve design features such as proximity, connectivity, and shape that may affect species dynamics and persistence. These design features can be included as objectives and analyzed in terms of their trade-offs with species representation under a given budget. A limitation of reserve selection and design models is their ignorance of species dynamics, and there is no guarantee that species will persist in the resulting reserve system. Reserve design models can be formulated with species dynamics, but they are complicated by the difficulties of adequately representing birth, death, and migration as functions of available habitat and by the computational intensity of finding optimal or near-optimal reserve designs. Nevertheless, applications have addressed reserve design problems for single species for which there is considerable knowledge of population dynamics.

A big limitation of reserve-based models is their assumption of a static time horizon: Decisions are made all at once, and habitat protection takes place rapidly before site degradation or loss. Researchers are beginning to address sequential site selection problems to optimize conservation objectives subject to budget constraints and uncertainties about site degradation and loss (Costello and Polasky 2004, Meir et al. 2004, Snyder et al. 2004, Turner and Wilcove 2006). The idea is to develop adaptive decision rules for selecting sites to protect depending on sites already protected, those currently available, and available funding. Decision rules like these can be compared with rules used in practice to see if efficiency gains can be obtained.

FUTURE DIRECTIONS

Government agencies and private organizations design and evaluate wildlife conservation programs based in part on their benefits and costs to society. While it is relatively easy to quantify the costs of conservation programs in terms of foregone economic activity, we are far from able to obtain definitive estimates of wildlife benefits. One place where we can make progress is gathering and analyzing data on participation in recreational activities related to wildlife in large landscapes. This is especially true for watching large mammals and bird species that form the basis of eco-tourism and eco-learning programs. Information on economic expenditures as well as the nonmarket values of such experiences could go far to help quantify the benefits of wildlife conservation efforts. Regional and national level analyses in the United States would be particularly helpful; information tends to be very general or anecdotal in nature. For wildlife-based ecotourism, most detailed studies focus on national parks or adventures in Africa and Latin America than on opportunities in places like the Midwestern U.S. (Krüger 2005).

Recognizing that estimates of benefits of wildlife conservation are not available, planners with well-defined conservation objectives evaluate the cost-effectiveness of alternative conservation plans and the trade-offs among their objectives using a variety of analytical models, including reserve selection and design models discussed previously. While reserve selection and design models focus on one important element of conservation planning, they ignore activities such as fire management, invasive species detection and eradication, and vegetation management that restore and enhance habitat for targeted species. In some cases habitat restoration is the only available management option because creating and expanding reserves is neither feasible nor affordable. Investment models with a wider range of conservation actions are needed to evaluate and prioritize reserve expansion versus other habitat restoration activities that are beneficial and possibly more cost effective. While such investment models are beginning to appear (e.g., Wilson et al. 2007), their success will depend on the participation of stakeholders and experts who help identify conservation objectives, threats to achieving those objectives, mitigation activities, economic costs, and local constraints on implementation.

In terms of the human dimensions of restoration, more research is needed on how to anticipate and work through conflicts that juxtapose restoration with other human values. This is especially true for restoration projects in urbanized landscapes. In places like Lake County, Illinois, discussed previously, trade-off modeling and conflict resolution and negotiation might help stakeholder groups better understand how options such as consolidation of acquisitions or restoration practices such as prescribed burning might be balanced with issues such as spatial equity in the distribution of open space or aesthetic considerations.

Finally, we need a better understanding of the patterns, drivers, and impacts of amenity migration as they pertain to wildlife conservation in large landscapes. Recent work examining the influence of housing density on landscape fragmentation (Hawbaker et al. 2006), bird populations (Lepczyk et al., 2008), and fire (Haight et al. 2004) is a good first step. Further interdisciplinary collaboration can merge this work with wildlife modeling efforts to look at potential impacts of housing and urban development on different types of species that depend on large landscapes.

SUMMARY

We described social and economic considerations for wildlife conservation planning in large landscapes. The social value of wildlife for recreational hunting provided an important justification for early landscape conservation efforts, but in more recent decades there has been a shift toward an appreciation of the value of wildlife for recreational viewing. However, human-wildlife conflicts have increased, and parcelization and development of open space provided by private forests, grasslands, pastures, and farms have inhibited wildlife conservation planning. Because people are concerned about the loss of open space, local governments and private land trusts have instituted policies to acquire land or conservation easements to preserve undeveloped land within or on the fringe of towns and cities. Planners have a variety of objectives for land acquisition, including wildlife habitat protection and restoration as well as economic efficiency. In response, biologists and economists have developed reserve selection and design models, which suggest cost-effective ways to protect open space to attain wildlife objectives. We describe three reserve-based modeling approaches to large-scale conservation planning: reserve selection models, reserve design models, and reserve design models with wildlife population dynamics. Models are presented with real-life applications and used to explain basic economic principles of cost-effectiveness, marginal cost, and trade-off analysis.

LITERATURE CITED

Animal and Plant Health Inspection Service. 2006. Brucellosis—Cattle and Bison. <http://www.aphis.usda.gov/vs/nahps/brucellosis/cattle.htm> accessed 29 December 2006.

Arthur, J. L., J. D. Camm, R. G. Haight, C. A. Montgomery, and S. Polasky. 2004. Weighing conservation objectives: Maximum expected coverage versus endangered species protection. *Ecological Applications* 14:1936–1945.

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

Bevers, M., and C. H. Flather. 1999. Numerically exploring habitat fragmentation effects on populations using cell-based coupled map lattices. *Theoretical Population Biology* 55:61–76.

Bishop, R. C., and M. P. Welsh. 1992. Existence values in benefit-cost analysis and damage assessment. *Land Economics* 68:405–417.

Boyce, M. S. 1992. Population viability analysis. *Annual Review of Ecology and Systematics* 23:481–506.

Boyd, D. P. 2003. *Conservation of North American bison: Status and recommendations*. Thesis, University of Calgary, Alberta, Canada.

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

Camm, J. D., S. K. Norman, S. Polasky, and A. Solow. 2002. Nature reserve site selection to maximize expected species covered. *Operations Research* 50:946–955.

Chavez, A. S., E. M. Gese, and R. S. Krannich. 2005. Attitudes of rural landowners toward wolves in northwestern Minnesota. *Wildlife Society Bulletin* 33:517–527.

Church, R. L., R. A. Gerrard, A. D. Hollander, and D. M. Stoms. 2000. Understanding the tradeoffs between site quality and species presence in reserve site selection. *Forest Science* 46:157–167.

Church, R. L., D. M. Stoms, and F. W. Davis. 1996. Reserve selection as a maximal coverage location problem. *Biological Conservation* 76:105–112.

Costello, C., and S. Polasky. 2004. Dynamic reserve site selection. *Resource and Energy Economics* 26:157–174.

Daily, G. C., editor. 1997. *Nature's services: Societal dependence on natural ecosystems*. Island Press, Washington, D.C., USA.

Decker, D. J., T. L. Brown, and W. F. Siemer, editors. 2001. *Human dimensions of wildlife management in North America.* The Wildlife Society, Bethesda, Maryland, USA.

Dizard, J. 2003. *Mortal stakes: Hunters and hunting in contemporary America*. University of Massachusetts Press, Amherst, Massachusetts, USA.

Fischer, D. T., and R. L. Church. 2003. Clustering and compactness in reserve site selection: An extension of the biodiversity management area selection model. *Forest Science* 49:555–565.

Fisher, A. C., and W. M. Hanneman. 1986. Option values and the extinction of species. *Advances in Applied Microeconomics* 4:169–190.

Foreman, D. 2004. *Rewilding North America: A vision for conservation in the 21st century.* Island Press, Washington, D.C., USA.

Gobster P. H., and R. B. Hull, editors. 2000. *Restoring nature: Perspectives from the social science and humanities*. Island Press, Washington, D.C., USA.

Gobster, P. H., and M. G. Rickenbach. 2004. Private forestland parcelization and development in Wisconsin's Northwoods: Perceptions of resource-oriented stakeholders. *Landscape and Urban Planning* 69:165–182.

Gobster, P. H., and T. L. Schmidt. 2000. The use of amenity indicators in anticipating private forest-land parcelization: A look at the Lake States' Northwoods. Pages 171–180 *in* L. A. DeCoster and R. E. Sampson, editors. *Fragmentation 2000: A conference on sustaining private forests in the 21st century.* The Sampson Group, Annapolis, Maryland, USA.

Haight, R. G. 1995. Comparing extinction risk and economic cost in wildlife conservation planning. *Ecological Applications* 5:767–775.

Haight, R. G., B. Cypher, P. A. Kelly, S. Phillips, K. Ralls, and H. P. Possingham. 2004. Optimizing reserve expansion for disjunct populations of San Joaquin kit fox. *Biological Conservation* 117:61–72.

Haight, R. G., S. A. Snyder, and C. S. ReVelle. 2005. Metropolitan open-space protection with uncertain site availability. *Conservation Biology* 19:327–337.

Haight, R. G., and L. E. Travis. 2008. Reserve design to maximize species persistence. *Environmental Modeling and Assessment* 13:243–253.

Hammer, R. B., S. I. Stewart, R. L. Winkler, V. C. Radeloff, and P. R. Voss. 2004. Characterizing dynamic spatial and temporal residential density patterns from 1940–1990 across the North Central United States. *Landscape and Urban Planning* 69:183–199.

Hanski, I. 1994. A practical model of metapopulation dynamics. *Journal of Animal Ecology* 63:151–162.

Harper, C., and T. Crow. 2006. *Cooperating across boundaries: Partnerships to conserve open space in rural America*. U.S. Forest Service, Report number FS-861, Washington, D.C., USA.

Hawbaker, T. J., V. C. Radeloff, M. K. Clayton, R. B. Hammer, and C. E. Gonzalez-Abraham. 2006. Road development, housing growth, and landscape fragmentation in Northern Wisconsin: 1937–1999. *Ecological Applications* 16:1222–1237.

Heberlein, T. A., and G. Ericsson. 2005. Ties to the countryside: Accounting for urbanites attitudes toward hunting, wolves, and wildlife. *Human Dimensions of Wildlife* 10:213–227.

Hof, J., and M. Bevers. 2002. *Spatial optimization in ecological applications*. Columbia University Press, New York, New York, USA.

Hof, J., M. Bevers, D. W. Uresk, and G. L. Schenbeck. 2001. Optimizing habitat location for black-tailed prairie dogs in southwestern South Dakota. *Ecological Modelling* 147:11–21.

International Wolf Center. 2006. IWC homepage. <http://www.wolf.org/wolves/index.asp>. Accessed 29 December 2006.

Kingsland, S. E. 2002. Creating a science of nature reserve design: Perspectives from history. *Environmental Modeling and Assessment* 7:61–69.

Krüger, O. 2005. The role of ecotourism in conservation: Panacea or Pandora's box? *Biodiversity and Conservation* 14:579–600.

LaGro, J. A. 1996. Designing without nature: Unsewered residential development in rural Wisconsin. *Landscape and Urban Planning* 35:1–9.

Lepczyk, C. A., C. H. Flather, V. C. Radeloff, A. M. Pidgeon, R. D. Hammer, and J. Liu. 2008. Human impacts on regional avian diversity and abundance. *Conservation Biology* 22: 405–416.

Liu, J., J. B. Dunning, and H. R. Pulliam. 1995. Potential effects of a forest management plan on Bachman's sparrow (*Aimophila aestivalis*): Linking a spatially explicit model with GIS. *Conservation Biology* 9:62–75.

Lopez, R. R., A. Lopez, R. N. Wilkins, C. C. Torres, R. Valdez, J. G. Teer, and G. Bowser. 2005. Changing Hispanic demographics: Challenges in natural resource management. *Wildlife Society Bulletin* 2005 33:553–564.

Maddala, G. S. 1983. *Limited-dependent and qualitative variables in econometrics*. Cambridge University Press, New York, New York, USA.

Maehr, D., R. F. Noss, and J. L. Larkin, editors. 2001. *Large mammal restoration: Ecological and sociological challenges in the 21st century.* Island Press, Washington, D.C., USA.

Manfredo, M. M., T. L. Teel, and A. D. Bright. 2003. Why are public values toward wildlife changing? *Human Dimensions of Wildlife* 8:287–306.

Manfredo, M. J., and H. C. Zinn. 1996. Population change and its implications for wildlife management in the new west: A case study of Colorado. *Human Dimensions of Wildlife* 1:62–74.

Margules, C. R., A. O. Nicholls, and R. L. Pressey. 1988. Selecting networks of reserves to maximize biological diversity. *Biological Conservation* 43:63–76.

Marianov, V., S. A. Snyder, and C. S. ReVelle. 2004. Trading off species protection and timber production in forests managed for multiple objectives. *Environment and Planning* 31:847–862.

Meir, E., S. Andelman, and H. P. Possingham. 2004. Does conservation planning matter in a dynamic and uncertain world? *Ecology Letters* 7:615–622.

Moilanen, A., and M. Cabeza. 2002. Single-species dynamic site selection. *Ecological Applications* 12:913–926.

Montag, J. M., M. E. Patterson, and W. A. Freimund. 2005. The wolf viewing experience in the Lamar Valley of Yellowstone National Park. *Human Dimensions of Wildlife* 10:273–284.

Montgomery, C. A., G. M. Brown, Jr., and D. M. Adams. 1994. The marginal cost of species preservation: The northern spotted owl. *Journal of Environmental Economics and Management* 26:111–128.

Nalle, D. J., J. L. Arthur, and J. Sessions. 2002. Designing compact and contiguous reserve networks with a hybrid heuristic algorithm. *Forest Science* 48:59–68.

Nelson, E. J., M. Uwasu, and S. Polasky. 2007. Voting on open space: What explains the appearance and support of municipal-level open space conservation referenda in the United States? *Ecological Economics* 62:580–593.

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

Önal, H., and R. A. Briers. 2002. Incorporating spatial criteria in optimum reserve network selection. *Proceedings of the Royal Society of London Series B* 269:2437–2441.

ReVelle, C. S., J. C. Williams, and J. J. Boland. 2002. Counterpart models in facility location science and reserve selection science. *Environmental Modeling and Assessment* 7:71–80.

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

Rothley, K. D. 1999. Designing bioreserve networks to satisfy multiple, conflicting demands. *Ecological Applications* 9:741–750.

Ruliffson, J. A., P. H. Gobster, R. G. Haight, and F. R. Homans. 2002. Niches in the urban forest: Organizations and their role in acquiring metropolitan open space. *Journal of Forestry* 100:16–23.

Ruliffson, J. A., R. G. Haight, P. H. Gobster, and F. R. Homans. 2003. Metropolitan natural area protection to maximize public access and species representation. *Environmental Science and Policy* 6:291–299.

Sampson, N., and L. DeCoster. 2000. Forest fragmentation: Implications for sustainable private forests. *Journal of Forestry* 98:4–8.

Schilling, D. A., A. McGarity, and C. ReVelle. 1982. Hidden attributes and the display of information in multiobjective analysis. *Management Science* 28:236–242.

Schroeder, H. W., P. H. Gobster, and R. Frid. 1993. *Visual quality of human-made clearings in Central Michigan conifers*. U.S. Forest Service, North Central Forest Experiment Station, Research paper NC-313, St. Paul, Minnesota, USA.

Simard, A. J., D. A. Haines, R. W. Blank, and J. S. Frost. 1983. *The Mack Lake Fire*. U.S. Forest Service, North Central Forest Experiment Station, General Technical Report NC-83, St. Paul, Minnesota, USA.

Snyder, S. A., R. G. Haight, and C. S. ReVelle. 2004. A scenario optimization model for dynamic reserve site selection. *Environmental Modeling and Assessment* 9:179–187.

Turner, W. R., and D. S. Wilcove. 2006. Adaptive decision rules for the acquisition of nature reserves. *Conservation Biology* 20:527–537.

U.S. Fish and Wildlife Service. 2002*a*. *2001 National survey of fishing, hunting, and wildlife-associated recreation*. U.S. Fish and Wildlife Service, Report FHW/01-NAT, Washington, D.C., USA.

U.S. Fish and Wildlife Service. 2002*b*. *2001 National survey of fishing, hunting, and wildlife-associated recreation—Wisconsin*. U.S. Fish and Wildlife Service, Report FHW/01-WI Rev, Washington, D.C., USA.

U.S. Forest Service. 2002. *The Prairie Plan* (*Record of Decision, Final Environmental Impact Statement, and Land and Resource Management Plan*), Midewin National Tallgrass Prairie. U.S. Forest Service, Wilmington, Illinois, USA.

Williams, J. C., C. S. ReVelle, and S. A. Levin. 2005. Spatial attributes and reserve design models: A review. *Environmental Modeling and Assessment* 10:163–181.

Wilson, K. A., E. C. Underwood, S. A. Morrison, K. R. Klausmeyer, W. W. Murdoch, B. Reyers, G. Wardell-Johnson, P. A. Marquet, P. W. Rundel, M. F. McBride, R. L. Pressey, M. Bode, J. M. Hoekstra, S. Andelman, M. Looker, C. Rondinini, P. Kareiva, M. R. Shaw, and H. P. Possingham. 2007. Conserving biodiversity efficiently: What to do, where and when. *PLoS Biology* 5:1850–1861.

Models for Planning Wildlife Conservation in Large Landscapes

Joshua J. Millspaugh
Department of Fisheries and Wildlife Sciences,
University of Missouri, Columbia, Missouri

Frank R. Thompson, III
U.S. Forest Service Northern Research Station,
Columbia, Missouri

ELSEVIER

AMSTERDAM • BOSTON • HEIDELBERG • LONDON
NEW YORK • OXFORD • PARIS • SAN DIEGO
SAN FRANCISCO • SINGAPORE • SYDNEY • TOKYO
Academic Press is an imprint of Elsevier

Academic Press is an imprint of Elsevier
30 Corporate Drive, Suite 400, Burlington, MA 01803, USA
525 B Street, Suite 1900, San Diego, California 92101-4495, USA
84 Theobald's Road, London WC1X 8RR, UK

This book is printed on acid-free paper. ∞

Cover photo credit for Greater sage-grouse: Gary Kramer.
Cover photo credit for sagebrush landscape: USDI Bureau of Land Management.

Library of Congress Cataloging-in-Publication Data
Millspaugh, Joshua J.
Models for planning wildlife conservation in large landscapes / Joshua Millspaugh, Frank R. Thompson.
p. cm.
Includes bibliographical references and index.
ISBN 978-0-12-373631-4 (hard cover : alk. paper) 1. Wildlife conservation–Simulation methods. 2. Habitat conservation–Simulation methods. 3. Landscape ecology–Simulation methods. I. Thompson, Frank R. (Frank Richard) II. Title.
QL82.M55 2009
639.901′1–dc22
2008018400

British Library Cataloguing-in-Publication Data
A catalogue record for this book is available from the British Library.

ISBN: 978-0-12-373631-4

For information on all Academic Press publications visit our
Web site at *www.books.elsevier.com*

Printed in the United States of America
09 10 11 12 9 8 7 6 5 4 3 2 1