

Continental Conservation

Scientific Foundations of Regional Reserve Networks

Edited by

Michael E. Soulé and John Terborgh

THE WILDLANDS PROJECT

ISLAND PRESS

Washington, D.C. • Covelo, California

FWS_LIT_010013

Copyright © 1999 The Wildlands Project

All rights reserved under International and Pan-American Copyright Conventions. No part of this book may be reproduced in any form or by any means without permission in writing from the publisher: Island Press, 1718 Connecticut Avenue, N.W., Suite 300, Washington, DC 20009.

ISLAND PRESS is a trademark of The Center for Resource Economics.

No copyright claim is made in chapters 2, 3, 6, and 7, which were written in part by government employees.

Library of Congress Cataloging-in-Publication Data

Continental conservation : scientific foundations of regional reserve networks / edited by Michael E. Soulé and John Terborgh.

p. cm.

Includes bibliographical references.

ISBN 1-55963-697-1 (cloth). — ISBN 1-55963-698-X (paper)

1. Nature conservation. 2. Biological diversity conservation.

I. Soulé, Michael E. II. Terborgh, John, 1936-

QH75.C685 1999

99 18457

333.95'16—dc21

CIP

Printed on recycled, acid-free paper 

Manufactured in the United States of America

10 9 8 7 6 5 4 3 2 1

FWS_LIT_010014

5

Core Areas: Where Nature Reigns

*Reed F. Noss, Eric Dinerstein, Barrie Gilbert,
Michael Gilpin, Brian J. Miller, John Terborgh,
and Steve Trombulak*

Experience on every continent has shown that only in strictly protected areas are the full fauna and flora of a region likely to persist for a long period of time. Hence the central component of a landscape design for conservation is the core area where human uses are greatly restricted and natural processes reign. We use the term "core area" to refer to areas where the conservation of biodiversity, ecological integrity, wilderness, or similar values takes precedence over other values and uses. In the context of continental reserve networks, core areas are designated largely to maintain existing natural qualities or to restore landscapes that have been degraded by human activities.

Some protected areas qualify as cores; others do not. Core areas might include national parks, designated wilderness areas on federal lands (the National Wilderness Preservation System), certain state or provincial parks and nature preserves (such as the wild forests of Adirondack Park in New York), reserves managed by The Nature Conservancy or other conservation groups, and, finally, other lands with an operational management plan that puts conservation of biodiversity or natural conditions as the highest priority (Noss and Cooperrider 1994). A distinguishing characteristic of core areas is limited human access—that is, low road density or, ideally, roadlessness. Many protected areas, including national parks in the United States and Canada (such as Banff in Alberta and Yosemite in California), vary in their degree of development and the extent to which biological conservation takes precedence over other values. Some portions of these parks are heavily developed with tourist facilities to the detriment of native wildlife. In these cases only the

wilder, less accessible portions of the parks meet our definition of core areas.

Natural condition and limited access to humans are important attributes of core areas, but an area need not be pristine to qualify for protection as a core. In most of the eastern and midwestern United States, for example, a strict insistence on pristine conditions and roadlessness would disqualify virtually all areas from protection within the federal wilderness system. Candidate areas for wilderness designation—and core areas generally—need not be free of roads, evidence of past timber harvesting or mineral extraction, or other signs of human activity. Nevertheless, further road building, logging, and other damaging activities should be prohibited after designation.

In this chapter we review the purposes of core areas in conservation networks. We discuss methods of selecting core reserves and considerations for reserve design and management. From our review we conclude that core areas are essential for meaningful conservation and that rigorous methods should be used to select and design them. Focal species can be very useful in reserve selection and design, but they must be selected carefully using scientifically defensible criteria (B. Miller et al. 1988). As no single species can be expected to provide an umbrella effect for all other species and phenomena in the region of interest, usually several focal species must be selected. Furthermore, many cores will require restoration and other active management, at least in the short term, to attain their biological goals.

Purposes of Core Areas

Core areas, when properly designed and managed, are places where nature can operate in its own way in its own time. They are also places where humans can encounter and understand nature in a relatively undefiled state. Lessons learned in studying these natural areas can be extended to management of other lands where resource exploitation and other intensive human uses take place.

Biological, Scientific, and Other Values

Conservationists involved in reserve design must address several questions: What conservation goals are achieved in core areas that cannot be achieved by multiple-use management? Why must some core areas be large? What are the consequences—biological and social—of not having core areas? Although these questions should be addressed case by case,

lessons learned through a number of case studies can inform new planning initiatives in their early stages.

Until recently most parks and other reserves were acquired for non-biological reasons such as scenery, recreation potential, and lack of conflict with resource production objectives. Another important use of reserves, pointed out by Leopold (1941) but not appreciated by many land managers, is as a "base datum of normality" for a "science of land health" (see also Arcese and Sinclair 1997). In the context of ecosystem management (Agee and Johnson 1988; Grumbine 1994; Christensen et al. 1996), cores can serve as reference or control areas that can be compared to other areas receiving management treatments (for example, different kinds of logging). Although replication is not usually possible in such experiments and, too, most core areas are not entirely free of human influence, management experiments with no controls are dangerous. Too many ecosystem management experiments today are being conducted without controls. Similarly, reference areas have proved essential in setting standards for indices of biotic integrity for streams with varying degrees of degradation (Karr 1991). The larger and less accessible the core area (which for aquatic ecosystems should include entire watersheds whenever possible), the better it will function as a control that is relatively free from human influences.

To provide adequate controls for ecosystem management experiments, an ideal reserve network would have landscape-scale reference areas for every major vegetation type or other ecosystem class native to the region. The present distribution of core areas is highly biased toward unproductive lands, however, and many types of ecosystems are poorly represented. For example, the GAP project for Idaho identified twenty-nine out of seventy-one vegetation types that were either not represented in protected areas or had less than 10,000 hectares represented (Caicco et al. 1995). Among forests, alpine types were well represented, but montane and low-elevation types were poorly represented. Similarly, in Sweden alpine landscapes were well represented but species-rich river landscapes were not (Nilsson and Götmark 1992). A preliminary gap analysis of abiotic zones (physical habitat classes) in the Klamath-Siskiyou region of northwestern California and southwestern Oregon found that 65 percent of the current protected area includes just three classes of habitats representing cool, high-elevation sites and areas with poor soils. Fully 47 percent of the coldest, poorest soil class is protected (some 26 percent of the total area in wilderness) compared to only 0.1 percent of the low-elevation, warm, best soil class (K. Vance-Borland, Conservation Biology Institute, unpublished data). For core areas to function effectively as con-

trols, as critical habitats for key species, and as reservoirs of biodiversity generally (that is, as coarse filters), they should span entire environmental gradients and contain viable examples of all types of habitats (Noss 1987; Noss and Cooperrider 1994).

An essential function of core reserves is to serve the needs of species that are hypersensitive to human activities. Many of these species are large mammals—including both herbivores (for example, mountain sheep and goats, monkeys, tapirs) and carnivores (for example, wolverines). Large birds, such as eagles and both terrestrial and arboreal frugivores, are also highly sensitive in many cases, as are other species vulnerable to human persecution (such as snakes), hunting, or collecting for the pet or horticultural trade. For these species, indicators of human access—particularly road density—are often the most accurate predictors of habitat effectiveness (Lyon 1983; Thiel 1985; Noss and Cooperrider 1994: 54–57).

Recreational trails, even if only for foot travel, may also pose threats to some species (Knight and Gutzwiller 1995; S. G. Miller et al. 1998). As trail access increases, population density or reproductive success of sensitive species may decline. The sensitivity of large animals to human activity is often a learned response stemming from hunting pressure, legal or illegal. Paradoxically, some species considered classic wilderness animals, such as the grizzly bear, are often fairly tolerant of humans—the problem is that humans are intolerant of the bears (Mattson et al. 1996; Noss et al. 1996). In most regions where grizzly bears coexist with people, human-caused mortality is the leading category of bear deaths (Woodroffe and Ginsberg 1998). In one troubling case, grizzly bear mortality within a designated wilderness area exceeded mortality in the surrounding multiple-use lands because of heavy recreational hunting within the wilderness (Mace and Waller 1998). Larger wilderness areas with less trail access—or prohibition of hunting—would function better as true refugia for such animals.

Apart from maintaining populations of space-demanding and disturbance-sensitive species, large core areas with limited access will likely stand as the last biological fortresses against the invasion of exotic species (see Chapter 4). Exotic plants are less likely to invade undisturbed natural vegetation than vegetation disturbed by human activity. Whether this failure is generally attributable to competition from native species or to an inability to tolerate the natural process regime is not known. We do know that roads are often the avenues along which exotic plants (such as purple loosestrife), pathogenic fungi (such as Port Orford cedar root rot fungus), and certain animals (such as the gypsy moth) invade natural

landscapes (Schowalter 1988; Noss and Cooperrider 1994; Forman 1995). Invasive exotics may disperse along disturbed roadsides or travel as seeds or spores in the tires, grills, or other parts of vehicles. In a car washing facility in Canberra, Australia, more than 18,500 seedlings representing almost 260 plant species were germinated from mud washed from vehicles (Wace 1977). The larger the core area and the lower the road density, the fewer invasions of exotics are expected.

We are not emphasizing here the economic values of protected areas because we do not believe they should be the main consideration in designating or managing cores. Relying on economic arguments to justify protection of core areas only encourages the practice of placing human interests above nature—a practice that created the biodiversity crisis in the first place. Nevertheless, we should recognize that protected areas often contribute greatly to local economies by providing pollination, protecting soil from erosion, regulating local climate, and protecting watersheds. For example, La Tigre National Park above Tegucigalpa, Honduras, produces about 40 percent of the water supply for the city—and the water is of high quality, requiring only limited processing (McNeely 1988). Tourism income is a salient economic benefit to many people living within or near protected areas.

Other values of core areas, not readily found in multiple-use or intensive-use zones, include providing opportunities for people to experience the wild. For many, wilderness is inspiring and renewing. Because it is often an uncomfortable and even dangerous place, wilderness can also be humbling. This humbling provides a positive benefit to the individual psyche and to civilization. Perhaps more important, people who feel a deep respect for nature—often bred in wild areas—are less likely to do violence against it.

Criticism of Core Areas

Despite the widespread recognition among conservation biologists of the values of large, wild core areas, not everyone agrees. In recent years the trend in international conservation has been away from national parks and other core areas—which are often construed as imperialistic and elitist—toward projects called “sustainable development,” “sustainable use,” “ecodevelopment,” or “integrated conservation-development projects” (ICDPs). This humanistic theme springs from the belief that protected areas in developing countries will survive only to the extent that they address human concerns (Western and Pearl 1989). A key assumption is that traditional communities use their resources in a sus-

tainable manner (McNeely 1988). Most of these projects, however, have failed to maintain the integrity of core areas and appear to be leading to continued, often rapid, losses of biodiversity. (See Redford and Sanderson 1992; Robinson 1993; Kramer et al. 1997; Wells et al. 1999.)

In North America, moreover, there are people who dismiss the value of core areas and try to keep the amount of protected area to an absolute minimum. This tendency is not limited to the inane "wise use" movement and conservative politicians but has become the dominant attitude within some land managing agencies. Ecosystem management, as interpreted by federal and many state agencies in the United States, is distinctly utilitarian. In assuming that human stewardship of nature can be wise and fully cognizant of natural phenomena, ecosystem management demonstrates the "arrogance of humanism" that Ehrenfeld (1978) described. Most ecosystem management projects allocate little area to reserves. In the USDA Forest Service, for example, many scientists and managers have advocated what has been called a "landscape without lines" approach to ecosystem management—without any zoning and meaningful protected areas—and instead seek to manage for all uses across the entire landscape. The critical assumption in this approach is that managers have learned from past mistakes and are now capable of managing forests, rangelands, and other environments in an intelligent way compatible with biodiversity conservation. Ironically, the validity of this dubious claim can only be tested if, in fact, unmanipulated, landscape-scale core areas are available to serve as controls.

Two Forest Service scientists who have led the charge against protected areas recently explained their rationale as follows: "We think the reserve model is not flexible and often inefficient because it focuses too much on allocating the landscape to specific uses. . . . We should try to minimize balkanization of the landscape with permanent land allocations and standard prescriptions, and attempt to manage the landscape as an integrated whole" (Everett and Lehmkuhl 1997). This philosophy is exemplified by the interagency Interior Columbia Basin Ecosystem Management Plan released in 1997 and covering some 24 million hectares—including 70 percent of USDA Forest Service roadless areas in the Lower 48 states (J. Jontz, Western Ancient Forest Campaign, pers. comm.). This egregious plan lacks any recommendations for new protected areas and instead proposes relatively minor changes in management practices. Curiously this project, as well as the earlier Northwest Forest Plan approved by President Clinton, was able to dispense with alternatives that included the establishment of meaningful protected areas by fashioning these alternatives to exclude all management (including restora-

tion and prescribed burning). Hence the preservation alternatives in these planning processes were just straw men easily dismissed as ecologically unrealistic.

British Columbia provides an example of agency biologists questioning the applicability of the core-corridor-buffer model for grizzly bears. Hence the provincial management agency, with the belief that designating discrete cores would leave the intervening matrix without sufficient protection, has opted instead to manage the entire matrix for bears and humans. If the entire matrix were truly managed conservatively for protection of bears and other sensitive animals, this approach would indeed have merit. We doubt that this is feasible, however, and see no reason why protection of core areas and conservative management of the matrix should be seen as mutually exclusive. Both elements should be part of a comprehensive conservation strategy. Strictly protected core areas would provide a safety valve in the event that matrix management fails. As shown by Woodroffe and Ginsberg (1998) in their review of extinctions of large carnivores in protected areas that are too small, reserve border areas are often population sinks because they expose these animals to persecution by humans. Moreover, even if it works for bears, matrix management is not likely to provide the other values of core areas.

Attacks by resource managers on core areas come at a time when social scientists are dismissing normative ethics, wilderness, and even scientific facts as mere social constructs (Soulé and Lease 1994). This new brand of nihilistic relativism not only discards the values upon which conservation is based but also rejects the science that underpins conservation biology. Nevertheless, it is important to note that even those who recently have criticized the wilderness concept as ethnocentric, historically naive, and anachronistic, such as philosopher J. Baird Callicott (1994-95) and historian William Cronon (1995), have acknowledged the necessity of allocating lands to conservation purposes. Thus they implicitly denote a value for core areas that transcends philosophical debates over the extent to which our images of nature are social constructions.

Conservation strategies that lack meaningful core areas are naive, arrogant, and dangerous. Such approaches assume a level of ecological knowledge and understanding—and a level of generosity and goodwill among those who use and manage public lands—that are simply unfounded. Nearly all conservation biologists would reject ecosystem management plans that offer no meaningful protection to imperiled species and habitats. Yet many in government seem to have embraced the view that enlightened management can substitute for strict protection.

Hence scientists must educate the public and decision makers about the perils of the landscape-without-lines approach and point out the need for a greatly expanded system of core areas.

Reserve Selection

We distinguish two phases in the establishment of core areas: reserve selection and reserve design. Selection has to do with identifying the key sites that should be included in a reserve network as core areas. Reserve design addresses the spatial configuration of the reserve network and includes consideration of lands, including corridors and buffers, that fall outside strict core areas (Noss and Cooperrider 1994; Scott and Csuti 1997).

Selecting lands to designate as core areas is not as straightforward as it might seem. In some regions the last undisturbed parcels are strikingly evident against a background of agricultural fields, suburbs, strip malls, or clearcuts. In other regions the natural remnants are larger and more options are available. In still other regions the landscape matrix remains as wildland, but with a broad gradient from highly degraded to nearly pristine habitat. In each of these scenarios the choice of areas to protect presents different challenges and opportunities. Selection also depends on whether the land is public or community land versus land owned by private individuals or corporations. Money for acquiring private land as core areas is limited (but see *Wild Earth*, vol. 8, no. 2), and political considerations limit the amount of public or private land that can be devoted to strict conservation.

In many cases land designated as a nature reserve has been dedicated for opportunistic or other nonscientific reasons—virtually guaranteeing that options to establish reserves will be exhausted before all elements of biodiversity are protected (Pressey et al. 1993). It is essential, then, that acquisitions target the lands of highest biological and ecological value: areas that are irreplaceable or otherwise stand to contribute most to conservation goals. New acquisitions should generally complement other acquisitions, both old and new, in terms of the specific elements protected so that the full range of biodiversity can be protected as efficiently as possible. Moreover, several occurrences of each species should be preserved because a single storm, epidemic, or other catastrophe could wipe out more populations.

Most parks, wilderness areas, and other reserves, then, have been acquired for nonbiological reasons. The application of biological goals and scientific principles to reserve selection began early in the 1920s and

1930s—most conspicuously within committees of the Ecological Society of America chaired by Victor Shelford. (See Shelford 1926, 1933; Kendeigh et al. 1950–1951; Noss and Cooperrider 1994.) Science still has little leverage on the designation of many core areas. (Federal assessments of wilderness study areas in the United States focus largely on recreation visitor days.) Nevertheless, the influence of science is growing along with the increasing popularity of conservation biology. In recent decades, scientific approaches to core area selection have proceeded along three separate tracks: special elements, representation, or focal species. We review each of these tracks in turn.

Special Elements

Special elements are species, places, and other entities considered to have high conservation value. Perhaps the best-known special elements are the “elements of diversity” ranked and tracked by the natural heritage programs (conservation data centers) established by The Nature Conservancy (Jenkins 1985, 1988; Noss 1987). The elements of greatest concern are those species and plant communities ranked as “critically imperiled globally” (G1) and “imperiled globally” (G2) based on their rarity and the severity of threats. “Element occurrences” are mapped locations of these elements. Conservation planners working at a regional or broader scale—the scales addressed in this book—generally look not for individual occurrences but for geographic clusters of occurrences. These clusters are often called hotspots, especially when they represent centers of endemism (Myers 1988). The assumption is that local land trusts and governments, or in some cases state agencies or The Nature Conservancy, can best protect the often small sites in which isolated endangered plants or other elements occur. Besides rare species and plant communities, there are other special elements that might be mapped in the process of selecting core areas: roadless areas and other relatively pristine sites; old-growth forests; unique geological sites; streams, lakes, and watersheds of high value for native fisheries or aquatic biodiversity; resource hotspots such as ice-free bays, artesian springs, and mineral licks; disturbance initiation or export areas (for example, peaks that receive lightning strikes and ridges that carry the resulting fires into new watersheds); sites recognized as sacred by indigenous peoples; sites inherently sensitive to development; and sites adjacent to existing protected areas.

This strategy has its weaknesses, however. In some regions hotspots may not stand out because a large number of endemic species are distributed throughout the region (as in much of Mexico). Moreover, the

strategy assumes that everything we want to protect is still present and just waiting to be protected. In fact, some special elements—such as old-growth forests throughout most of North America—have been lost and will have to be restored. Nevertheless, remnant stands of old growth are of high value for providing blueprints for restoration and old-growth associated species to inoculate restoration zones with native species; hence they are worthy special elements.

Representation

Protection of special elements, which often comprise the rare and unique in nature, does not assure that all species and habitats in a region will be adequately protected in core areas. In particular, species groups that are poorly known or inventoried may be missed. Hence the representation (or “coarse filter”) strategy, which seeks to protect intact examples of each vegetation or habitat type in a region, can be considered complementary to special element protection. The assumption is that if we protect examples of all habitat types (complete environmental gradients), we will capture occurrences of a vast majority of species. (A further assumption, often implicit, is that gradients in species composition parallel gradients in physical habitat variables or that vegetation types, because they reflect environmental gradients, are surrogates for biodiversity.) Although the effectiveness of the coarse-filter approach has never been quantified (it would require complete species lists for all taxa in a region and would have to be repeated in different regions), it is reasonable. The gap analysis projects in the United States, Canada, and several other countries are examples of this approach (Scott et al. 1987, 1993).

The literature on representation, much of it originating in Australia, is large and dense and need not be reviewed here. Guidelines for assessing representation at an ecoregional scale have been presented by Noss (1996), and algorithms for efficiently selecting a complementary set of sites that represent all habitats in a region have been presented or reviewed by Pressey and Nicholls (1989), Bedward et al. (1992), Pressey et al. (1993, 1996), Church et al. (1996), Csuti et al. (1997), and many others. Questions have been raised, however, about the value of these approaches without consideration of the long-term viability of the selected sites. This concern has led leading researchers to develop new models that focus on retention as well as representation of biodiversity in reserves (Pressey 1998).

Focal Species

The third means of selecting core areas is based on the needs of particular focal species. Focal species analysis answers many questions that cannot be addressed by considering special elements or representation. Whereas the locations of special elements and underrepresented habitats point to particular sites and landscapes that require protection, focal species analysis identifies additional high-value habitats and addresses two questions: How much area is needed? And in what configuration should habitat areas be designed? These questions are the bridge from reserve selection to reserve design.

Focal species serve many roles in conservation planning. A list of focal species that serves one conservation need may overlap only slightly with lists of species intended for other needs. Localized (endemic) species may be important special elements for identifying irreplaceable core areas in the selection process. Endemic species, however, may not be useful for other reasons such as low public interest and minuscule area and connectivity requirements.

People often assume that a particular focal species is an umbrella species—so that if its needs are met, all other species in the region will also persist. This is not necessarily so, however, though species with large area requirements are most likely to serve the umbrella function. There are other categories of focal species. Species that play pivotal roles in an ecosystem, disproportionate to their abundance, are known as keystone species. (See Chapter 3 for further discussion of focal species.) Some focal species are highly charismatic with the public and hence are useful as “flagships” stimulating broad support for conservation proposals. Some focal species are useful as ecological indicators because their responses to environmental change tell us something valuable about the integrity of the community at large. Although these species are sometimes of little use in reserve selection or design, they are valuable in monitoring and adaptive management (Lambeck 1997; Noss et al. 1997).

One of the first steps in using focal species for reserve selection is a clear description of their intended purposes. What species are being considered and why were they selected? What assumptions were made in the selection of those species? How will these focal species contribute to the general goals of the reserve network? What type and quality of data for each species are available? Because no single species can serve as a surrogate for the needs of all others, many biologists have recommended

using a suite of species (Noss and Cooperrider 1994; Lambeck 1997; Noss et al. 1997).

Large carnivores often are chosen as the major focal species in regional and continental conservation strategies. In many cases this choice is well justified. Although large carnivores may or may not qualify as keystone species, apparently they often control populations of other species—such as ungulates and opportunistic mesopredators—that might otherwise have negative influences on the native biodiversity of an area. (See Chapter 3; Wilcove et al. 1986; Soulé et al. 1988; Terborgh 1988; Noss et al. 1996.) And carnivores, through their public appeal, can attract support for a conservation plan. In using terrestrial carnivores as umbrella species, it may be most practical to use the movements and spatial needs of females to estimate the number of animals a reserve could hold. Male carnivore movements are extensive, highly variable, and related mainly to social status, behavioral spacing mechanisms, and hormonal production (Ewer 1973; Powell 1979). Female carnivores are usually more valuable demographically and will raise their young in areas where critical resources are concentrated and easiest to obtain (Lindzey 1982; King 1989; Miller et al. 1996). Because they must satisfy elevated energy requirements with minimal time away from their young, they are restricted to optimal habitat. Therefore their home range sizes accurately represent the quality of that habitat (King 1989; Lindstedt et al. 1986). In highly fragmented habitat, however, considering only female needs can result in low mating success and high male mortality (Beier 1993). Moreover, it is usually male raptors that protect the breeding territories. The degree of emphasis placed on females depends on the natural history of the species chosen.

To what extent can a focal species serve as an umbrella for other species? This is a topic of considerable discussion in conservation biology. In calculating the umbrella function of the grizzly bear in Idaho, Noss et al. (1996) compared ambitious recovery zones (potential core and buffer areas) proposed by Shaffer (1992) to the distribution of vegetation types and vertebrate species in the Idaho Gap Analysis database (see Scott et al. 1993). Shaffer's proposed recovery zones covered 34 percent of the state of Idaho (compared to 8 percent in a U.S. Fish and Wildlife Service proposal). For two-thirds of the state's vegetation types, Schaffer's proposal would have protected at least 10 percent of their statewide ranges. In addition, it would have protected 71 percent of the mammal species, 67 percent of the birds, 61 percent of the amphibians, but only 27 percent of the reptiles native to Idaho. Thus large carnivores are likely to be better umbrellas for some communities, taxa, or species assemblages than for

others. The proportion of species richness protected by considering individual species is likely to decline as one moves from a homogeneous landscape to a heterogeneous one with high beta-level diversity (see Chapter 2).

Until recently, most large carnivore populations ranged widely. In the Indian subcontinent tigers have declined but are still found in fifty-nine landscape units that contain a mixture of sanctuaries and government forests. Because their distribution overlaps with the most important areas for biodiversity in these landscapes, they make an ideal umbrella species for conservation planning (Dinerstein et al. 1999; Wikramanayake et al. 1998). But basing core areas on the present and truncated distribution of Amur tigers in the Russian Far East, which has been proposed, would exclude many hotspots of rare and endemic invertebrates and plants. A similar example in Africa is offered by Berger (1997), who shows that using the spatial needs of an existing herd of twenty-eight black rhinos did not assure healthy populations of six other herbivores. When Berger modeled the spatial needs of a hundred rhinos, the protection given to the six herbivores increased significantly. Kerr (1997) has found that basing reserves on the present distribution of complete carnivore communities did not offer good protection to several invertebrate taxa. Thus basing a reserve's location on present distribution of a species whose range has been reduced drastically by human activities, or using a species without a good understanding of the region's biogeography, can lead to poor decisions.

In general, then, we contend that an umbrella species approach is more suitable to questions of how much land is necessary in a reserve network and how that land should be configured (an issue of reserve design) than for where to place core areas—especially noteworthy considering the truncated distribution of many large carnivores today.

Combining the Three Tracks

Although the three tracks of conservation planning are usually pursued in isolation, they are most useful when united into a single comprehensive approach (Noss 1996; Noss et al. 1999). (See Figure 5.1.) One promising approach to reconciling the three tracks is to develop a series of reserve design options for a region, each reflecting a different emphasis given to each of the three tracks. Conservation planners may decide that certain conservation criteria are not especially relevant to their region or congruent with their conservation goals. If rewilding (Soulé and Noss 1998) is the fundamental goal of a regional plan, for example, planners

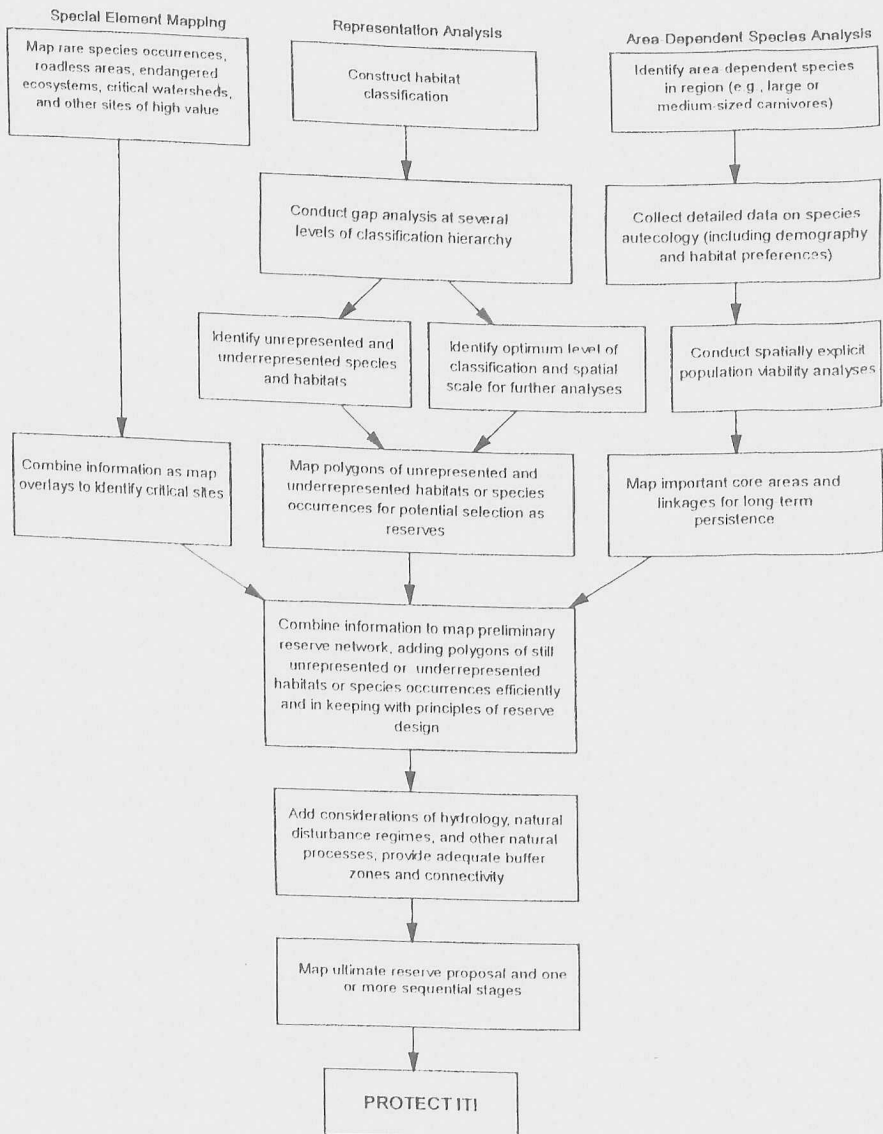


Figure 5.1. Three tracks of reserve selection and design, which rarely have been combined in practice. Adapted from Noss (1996).

may decide to dispense with representation analyses and mapping of rare species occurrences and concentrate instead on a restricted class of special elements—roadless areas, landscapes with high potential for road closures, habitat quality for prey species, connectivity—and a short list of focal species that are most area-demanding and sensitive to human activities (such as large mammalian predators). In regions lacking large, native carnivores but with high physical and biological heterogeneity (such as southwestern Australia) or in centers of endemism (such as many tropical and Mediterranean-climate regions), protection of endemic species and representation of plant communities may take precedence over rewilding. Hence the three-track approach is inherently flexible and adaptable to different goals and circumstances.

Reserve Design

The design phase of reserve establishment follows on the heels of reserve selection and inevitably results in additional lands being recommended for protection. Ideally, core areas should be large enough to contain natural disturbance regimes and viable populations of top carnivores (at least over a span of decades). Where sufficiently large wildlands are not available, as in most of the Lower 48 states, smaller areas can serve as cores but will require greater investments in management as well as additional land allocated to buffer zones and linkages. These considerations highlight the need to think of core areas within the broad context of regional landscape design, which incorporates buffer areas (Chapter 7) and other human uses, as well as the critical issue of connectivity (Chapter 6). Nevertheless, a system of small cores is unlikely to attain ambitious conservation goals. Buffer areas, connectivity, and broad issues of reserve network design are addressed elsewhere in the book.

Focal Species

Focal species, as we have seen, have a variety of potential uses in conservation planning. Large carnivores and other area-dependent species are particularly useful in designing reserve networks because of their demanding requirements for secure habitats and movement corridors. The needs of other appropriate focal species (see Lambeck 1997; Noss et al. 1997; B. Miller et al. 1998) should also be considered. Some biologists believe that the ideal approach is to conduct a rigorous population viability analysis (PVA) for each species based on real demographic and

genetic data. To conserve all the biodiversity in a region, viability assessments of many focal species may be required. A multispecies model would add biological logic describing the interactions between pairs of the species, such as predation or interspecific competition for space. To protect species X, for example, it might be required that the overlap between the units of species X and Y be no more than 10 percent. Moreover, the cost of land acquisitions may be a consideration involving both economic and political factors. Using the techniques of discrete linear programming, a theoretically optimal reserve network can be designed. (Heuristic algorithms that offer "near-optimal" solutions with greater simplicity and practicality may be preferred in many cases; see Pressey et al. 1996; Csuti et al. 1997.) This analytic process does not result in a final map or proposal but rather a set of options and decision tools that planners can use in a stepwise fashion to move toward the final solution.

Ideal methodologies are sometimes impractical. Reserve design often cannot afford to wait until a PVA for each species of concern is completed. When the data, time, and funds to conduct PVAs are limited, other means of assessing the status and trends of focal species populations should be pursued (Noss et al. 1997; Carroll et al. 1998). Several approaches have been developed to make use of presence/absence information, which may be the only information available for many lesser-known species (Wiens 1989; Howe et al. 1991; Kariëva et al. 1996; Hanski 1996; Hanski et al. 1996). These alternatives to intensive demographic studies and PVA are less data-hungry and can be used in many planning situations where timing is critical (Noss et al. 1997). An example is the spatially explicit incidence function model applied by Hanski et al. (1996) to a metapopulation of an endangered butterfly (the Glanville fritillary, *Melitaea cinxia*). The model does not require detailed demographic information but uses "snapshot" presence/absence data from a collection of habitat patches to estimate biologically important metapopulation parameters. The model predicted patch occupancy well and allowed for quantitative predictions about metapopulation dynamics.

Recognizing the uncertainty inherent in modeling, habitat should always be added to the reserve design to provide something more than the defined minimum required to provide for viable populations of each focal species. And it is important to recognize that the minimum viability unit for a large carnivore or other wide-ranging species is often considerably larger than the planning region—hence the need for interregional connectivity and coordination.

Principles of Reserve Design

Reserve design, like all facets of conservation planning, is still as much art as science. The best algorithms for selecting core areas and determining their optimal distribution leave unanswered many practical questions of how to tie them all together into a design that offers a high probability of meeting conservation goals within a complex social and political framework. In practice, the conservation planner must rely on biological knowledge, empirical generalizations, political savvy, and intuition.

Various empirical generalizations—often called principles—of reserve design have emerged over the last two and a half decades. (See Terborgh 1974; Willis 1974; Diamond 1975; Wilson and Willis 1975; Diamond and May 1976.) Most of these principles deal with the requirements of area-sensitive focal species. These rules, and the controversy that has developed over them (Simberloff and Abele 1976), were to some extent reconciled by Soulé and Simberloff (1986), who agreed that reserves should be both large and multiple. Principles that have withstood the test of time and seem to apply to many cases have been proposed by Thomas et al. (1990) and Wilcove and Murphy (1991), with some modification by Noss (1992) and Noss et al. (1997). Among the principles that apply to reserve design are these:

- Species well distributed across their native range are less susceptible to extinction than species confined to small portions of their range.
- Large blocks of habitat, containing large populations, are better than small blocks with small populations.
- Blocks of habitat close together are better than blocks far apart.
- Habitat in contiguous blocks is better than fragmented habitat.
- Interconnected blocks of habitat are better than isolated blocks. (This principle addresses multiple blocks, often in different landscapes, whereas the preceding principle addresses the intactness of single blocks.)
- Blocks of habitat that are roadless or otherwise inaccessible to humans are better than roaded and accessible blocks.

All these rules have exceptions and their application to specific cases is not usually straightforward. They should not be applied uncritically. Proper interpretation of these principles can only be made by competent biologists familiar with the life forms and landscapes in question. Nevertheless, they provide a useful starting point for reserve design. In combination with site-specific and species-specific data and the judgment of

experienced biologists, these principles can contribute to a design that has a high probability of meeting conservation goals.

The International Context

We cannot overstate the importance of envisioning core areas within a broader geographic, biological, and social context. In developing countries, there are problems with poverty, lax protection of wildlife, and a large number of people who depend directly on the land for a meager survival. Few protected areas in developing countries have been effectively zoned to contain core areas as described in this chapter. Indeed, many reserves in Latin America have human settlements within their boundaries. Unless a part of a reserve is remote or virtually inaccessible, it is likely to be used by humans. Areas viewed by biologists as critical habitat for sensitive species, breeding areas for source populations of endangered species, or sites where key processes still occur are viewed by locals as a place to harvest trees and bush meat, cut grass, gather non-timber forest products, trap animals for trade, poach, or set fires to clear the forest for future hunting trips.

Recent efforts to link strict protection of protected areas with local use in many Asian settings have largely failed—delivering neither conservation nor development (Dinerstein et al. 1999; Wells et al. 1999). To conserve populations of endangered species that are habitat specialists or threatened by the illegal wildlife trade, there is no substitute for well-protected core areas. Although tigers are still distributed over large landscapes in India, for example, biologists suspect that tigers are only breeding in protected areas where there is still an adequate prey base; adjacent unprotected habitats are population sinks. Ecodevelopment projects that designate core areas without supporting strict protection measures inside them are building on quicksand (Dinerstein et al. 1999). A practical way to reduce pressure on core areas while increasing their effectiveness is to create economic incentives within adjacent buffer zones. To be effective, incentives must be commensurate with the spatial and temporal scale of the threats to core areas and directly related to those threats. If poaching is widespread across a core zone that is surrounded by villages, for example, incentives that go to a single village will do little to reduce poaching.

We suggest several approaches for enhancing protection of core areas drawn from ongoing projects in Nepal, some of which also apply to North America:

1. Hire local people to protect buffer zone areas adjacent to the core area. This gives them a direct stake in conservation.
2. Pass legislation that calls for the return of a large fraction of revenues to local development projects in the reserve buffer zone that are generated by park entry fees, concessions, and other enterprises that depend on a healthy core area.
3. Promote alternatives to harvesting timber for firewood and building materials by managing areas in the buffer zone for the benefit of local people—and whenever possible allow them to take over management of these lands.
4. Link reserve management efforts to rural development projects that introduce new environment-friendly technologies such as methane gas digestors and stall feeding of livestock.
5. Fence and restore buffer zones and encourage community-based ecotourism efforts in buffer zones and corridors. This development can return significant sums to locals and divert tourism pressure from the core area.
6. Link the concept of core areas to flood control, watershed protection, or other ecological services wherever appropriate and explain their value through public outreach programs.
7. Link core areas with spiritual or religious symbolism and incorporate areas where taboos on hunting or entry prevail.
8. Monitor the effects of incentives by using indicators of encroachment, poaching, and habitat degradation in core areas adjacent to buffer zones where incentive programs have been introduced.

Ultimately, in both industrialized and developing nations, core areas will succeed only when they enhance the economic value of adjacent buffer zones or other parts of the conservation network to local human populations or when the livelihoods of local people are directly tied to intact or well-managed core areas.

Management of Core Areas

Management is anathema to conservationists who prefer a vision of nature untainted by human hands. It would be nice if there were a simple management plan that could reinstate the prehuman condition of biological evolution, but science is not able to provide such a prescription, at least not under the constraints imposed by our civilization.

Toward Minimal Management

Short of complete, landscape-wide restoration, we espouse a policy of creating protected areas designed, whenever possible, to give free reign to abiotic and biotic processes critical to maintaining biodiversity. Our prescriptions include the reestablishment of natural fire regimes and river flows; the reintroduction of top carnivores, large ungulates, and other locally extirpated species; and, wherever possible, the elimination of alien species. These prescriptions will provide the closest attainable approximation to the original regime while recognizing that some processes, such as dispersal and migration, lie beyond current management capabilities. By letting natural processes prevail to the maximum practical extent, optimal conditions for the maintenance of biodiversity are provided at minimal cost in hands-on management. To minimize errors of commission, direct interventions are to be avoided wherever possible.

But achieving a regime where natural processes predominate will, in most of North America, take time and—paradoxically—human effort. Hence management is often necessary, but it should be driven by ecological sensibility. Management of core areas may be of two kinds: temporary (until an area is restored to the point where it can take care of itself) or perpetual. Because so little of North America is truly pristine and so many lands have been managed so poorly in the past, virtually all newly established core areas will need at least temporary management. Intensive, hands-on restoration will be required in many landscapes where natural processes and species composition have been severely altered by agriculture, logging, fire suppression, livestock grazing, elimination of large predators, invasions of exotic species, and other insults. A recent study of the causes of species imperilment in the United States determined that a large proportion of imperiled species is threatened by fire suppression and alien species (Wilcove et al. 1998). The authors conclude that “both types of threats must be addressed through active, ‘hands-on’ management . . . such as by pulling up alien plants and trapping alien animals or using prescribed fire to regenerate early successional habitats.”

Restoration

Restoration is a complex topic and, in some cases, the methods are still in the developmental stage. Chapter 4 discusses this topic in more detail. The first step is to stop whatever practice is degrading the natural ecosystem. Sometimes this may be enough to restore natural processes and

conditions. In many cases, however, the wounds are deeper and restoration demands active intervention. Restoration techniques—including slope recontouring, soil preparation, planting, thinning, burning, and hydrological manipulation—are highly experimental and specific to certain plant communities and site conditions; detailed and systematic descriptions, however, are beyond our scope here. In some cases—in landscapes that were logged but not stocked with tree plantations, for example, and where sources for recolonization of native plants and animals remain—restoration might amount to simply closing roads (and ripping and revegetating roadbeds, as necessary), stepping back, and letting nature take its course. But the greater the disruption of natural processes by humans, the more human effort is required for restoration if we want native species to return to “normal” patterns of abundance and distribution within several human generations.

Reestablishing populations of keystone species (Chapters 3 and 4; Mills et al. 1992; Power et al. 1996) is a critical element in the restoration of natural processes in core areas. Even though management increasingly emphasizes an ecosystem perspective (Smith 1984; Scott et al. 1987; Noss and Cooperrider 1994; Pickett 1997), many individual species will require detailed study and specific attention (Noss et al. 1997). Managing keystone species provides an excellent opportunity to influence ecosystems through single-species management (Miller et al. 1994). Species that exert considerable influence on ecosystem processes include beavers, prairie dogs, bison, gopher tortoises, and many large carnivores.

Control of exotic plants and exotic or feral animals is often needed in core areas, especially small core areas with nearby roads or human settlements. Even in vast wildlands, however, feral ungulates in the form of sheep, goats, pigs, cattle, horses, and donkeys are established in many places in North America. Control campaigns and open hunting can reduce populations, but they rarely result in complete eradication. The optimal long-term solution is to allow native top predators to act as control agents. Even if exotic ungulates do invade core areas, predators may be able to keep their numbers in check so that their impact on native vegetation is kept to a minimum. But in arid rangeland regions—where large mammalian carnivores were generally not abundant prior to European settlement or, in some cases, appear to have been entirely absent—the direct removal of large, exotic herbivores (cattle, horses) that are not effectively controlled by coyotes may be necessary.

Ideally, core areas should contain the finest remaining examples of pristine native habitat. But in large areas of the continent, pristine natural

habitats no longer exist or are, at best, too small and fragmented to sustain the full spectrum of diversity-maintaining processes. Lack of sufficient primary habitat should not be a pretext, however, for abandoning the effort to preserve native biodiversity. Core areas large enough to sustain critical processes should be established, wherever possible, because nature tends to restore itself with time. Top predators and large ungulates can be successfully reestablished in a disturbed landscape if conditions otherwise are amenable. Restoration of a natural process regime can be promoted by removing roads and dams, relaxing fire suppression, and curtailing hunting, grazing, logging, and mining. Whether the recovery of a natural process regime will eventually lead to the reduction or even elimination of exotic species remains a major untested hypothesis. Nevertheless we predict that reducing human disturbances, and directly removing exotics where they are found, are crucial for solving the exotics problem.

Management may be required in perpetuity for smaller, isolated core areas—say, those less than 400,000 hectares. Perpetual management may involve law enforcement, boundary maintenance, public education and visitor management, exotic species control, prescribed burning, and other habitat management to improve conditions for native species and communities, intensive population management (including translocations) of severely imperiled species, and more. The smaller the reserve, the more management per unit area is required (Bratton and White 1980; Noss 1983)—and the more it costs (Soulé 1984). This is one of the key reasons for establishing big reserves: they are usually more cost-effective to manage. And because less management and associated human presence are required, big reserves are more capable of sustaining conditions where visitors confront real wilderness rather than highly manipulated artifacts of nature.

Monitoring and Adaptive Management

Monitoring is an essential component of management. The key to managing ecosystems is understanding biological relationships, processes, and change (Noss and Cooperrider 1994). Monitoring is also the only way to ascertain if the reserve design principles are correct. Moreover, monitoring and research aimed at addressing problems such as overabundance of ungulates and decline of browsing-sensitive species (McShea et al. 1997) will be beneficial to other parks and reserves.

We should not regard reserve designs or management plans as final products. A plan should be an organized method of pursuing goals and

opportunities and a way to improve our knowledge about what is needed to protect nature. If the process works, we will be able to respond swiftly and effectively to new problems as they arise. An adequate adaptive management framework, in the sense of Holling (1978), allows conservationists to solve problems and take advantage of opportunities. We emphasize that reserves of all kinds require law enforcement. Selective road removal and other controls of human access will ease the enforcement burden and reduce mortality for species hunted or persecuted by humans.

The Need for Management

Conservation is biologically and socially complex and occurs in a context of uncertainty, change, and public scrutiny. Planning conducted in such an atmosphere is vulnerable to challenges that cut across institutional and geographical lines (Miller et al. 1996). Stakeholders often have disparate values, attitudes, and agendas. In such an arena, one group may think a problem is critical whereas a second faction shies away from it because it conflicts with their ideology. Consensus is almost impossible when groups have conflicting values. For this reason, a conservation plan is best created by a group of people with similar—though not necessarily identical—goals. Strong leadership and perseverance can make or break a plan.

The authors of this chapter, and indeed the entire book, share biocentric values. We agree that the native biodiversity of North America should be maintained and restored to the greatest extent possible. Hence we believe that conservation planning should occur on a regional to continental scale and that the present extent of protected area in North America is far too small to meet well-accepted conservation goals. We also recognize that given the historical evidence, rigorously protected core areas are an essential ingredient of a successful conservation strategy. Although some of us are species biologists and others are more interested in communities and ecosystems, we conclude that focal species are of great utility in the selection and design of core areas and their enveloping networks. These focal species, which must include a variety of taxa, should be selected and their habitat needs modeled on the basis of the best available information.

Perhaps one of our more controversial conclusions is that most core areas will require restoration—and, often, continued active management—if they are to function effectively. Although, in time, nature will restore itself, many species could be lost under a policy of benign neglect

(Soulé et al. 1979). Most conservation biologists appreciate the need for ecological management, but we have not communicated the message to nonscientists, including grassroots activists. Land management agencies such as the USDA Forest Service routinely exploit the aversion of conservationists for management. In both the president's Forest Plan for the Pacific Northwest and the Interior Columbia Basin Ecosystem Management Plan, for example, the alternative of calling for a substantial network of protected areas was a blatant straw man—a classic preservationist, hands-off alternative with no management of any kind allowed in reserves. Given the obvious need for ecological restoration in many of the forests and rangelands of these regions, evaluation panels easily rejected the preservation alternative, preferring the alternatives with active management and minimal protection. Motivated by similar values, anti-conservation journalists for *Science* and *Newsweek* magazines, in their articles on The Wildlands Project and the Yellowstone-to-Yukon initiatives, respectively, incorrectly labeled reserve proposals “hands off” or as human exclusion zones. The purpose of these caricatures was to discredit conservation biology and arouse distrust of conservation planning in the minds of the vocal segment of the public that favors both intensive resource production and recreation on public lands.

Conservation biologists must make one thing clear: where human activities have led to ecological degradation, it is a human responsibility to repair the damage. This is true both inside and outside core areas. Many of these reserves will need active management in perpetuity, as they are simply too small and isolated for fire, predator/prey relationships, and other ecological processes to operate within the range of variability that native species experienced over their evolutionary histories. Aggressive control or eradication programs are also necessary for many invasive, exotic species. It may be a bitter irony that human management is now needed to maintain some degree of naturalness in protected areas. This realization, however, does not detract from our ultimate vision of a wild, self-sustaining biosphere.

References

- Agee, J. K., and D. Johnson (eds.) 1988. *Ecosystem management for parks and wilderness*. Seattle: University of Washington Press.
- Arcese, P., and A. R. E. Sinclair. 1997. The role of protected areas as ecological baselines. *Journal of Wildlife Management* 61:587–602.
- Bedward, M., R. L. Pressey, and D. A. Keith. 1992. A new approach for selecting fully representative reserve networks: Addressing efficiency, reserve design,

- and land suitability with an iterative analysis. *Biological Conservation* 62:115–125.
- Beier, P. 1993. Determination of minimum habitat areas and habitat corridors for cougars. *Conservation Biology* 7:94–108.
- Berger, J. 1997. Population constraints associated with the use of black rhinos as an umbrella species for desert herbivores. *Conservation Biology* 11:69–78.
- Bookbinder, M. P., E. Dinerstein, A. Rijal, H. Cauley, and A. Rajuria. Eco-tourism's support of biodiversity conservation. *Conservation Biology* 12:1399–1404.
- Bratton, S. P., and P. S. White. 1980. Rare plant management—after preservation what? *Rhodora* 82:49–75.
- Caicco, S. L., J. M. Scott, B. Butterfield, and B. Csuti. 1995. A gap analysis of the management status of the vegetation of Idaho. *Conservation Biology* 9:498–511.
- Callicott, J. B. 1994–95. A critique of and an alternative to the wilderness idea. *Wild Earth* 4(4):54–59.
- Carroll, C., P. Paquet, R. Noss, and J. Strittholt. 1998. Modeling carnivore habitat in the Rocky Mountain region: A literature review and suggested strategy. Unpublished report. Corvallis, Ore.: Conservation Biology Institute.
- Christensen, N. L., A. M. Bartuska, J. H. Brown, S. Carpenter, C. D'Antonio, R. Francis, J. F. Franklin, J. A. MacMahon, R. E. Noss, D. J. Parsons, C. H. Peterson, M. G. Turner, and R. G. Woodmansee. 1996. The report of the Ecological Society of America Committee on the Scientific Basis for Ecosystem Management. *Ecological Applications* 6:665–691.
- Church, R. L., D. M. Stoms, and E. W. Davis. 1996. Reserve selection as a maximal covering location problem. *Biological Conservation* 76:105–112.
- Cronon, W. (ed.). 1995. *Uncommon ground: Toward reinventing nature*. New York: Norton.
- Csuti, B., S. Polasky, P. H. Williams, R. L. Pressey, J. D. Camm, M. Kershaw, A. R. Kiestler, B. Downs, R. Hamilton, M. Huso, and K. Sahr. 1997. A comparison of reserve selection algorithms using data on terrestrial vertebrates in Oregon. *Biological Conservation* 80:83–97.
- Diamond, J. M. 1975. The island dilemma: Lessons of modern biogeographic studies for the design of natural preserves. *Biological Conservation* 7:129–146.
- Diamond, J. M., and R. M. May. 1976. Island biogeography and the design of natural reserves. In R. M. May (ed.), *Theoretical ecology: Principles and applications*. Philadelphia: Saunders.
- Dinerstein, E., A. Rijal, M. Bookbinder, B. Kattel, and A. Rajuria. 1999. Tigers as neighbors: Efforts to promote local guardianship of endangered species in lowland Nepal. In J. Seidensticker, P. Jackson, and S. Christie (eds.), *Riding the tiger: Conserving tigers in a human-dominated landscape*. Cambridge: Cambridge University Press.
- Ehrenfeld, D. 1978. *The arrogance of humanism*. New York: Oxford University Press.
- Everett, R. L., and J. F. Lehmkuhl. 1997. A forum for presenting alternative view points on the role of reserves in conservation biology? A reply to Noss (1996). *Wildlife Society Bulletin* 97:575–577.

- Ewer, R. F. 1973. *The carnivores*. Ithaca: Cornell University Press.
- Forman, R. T. T. 1995. *Land mosaics: The ecology of landscapes and regions*. Cambridge: Cambridge University Press.
- Grumbine, R. E. 1994. What is ecosystem management? *Conservation Biology* 8:27–38.
- Hanski, I. 1996. Metapopulation ecology. In O. E. Rhodes, R. K. Chesser, and M. H. Smith (eds.), *Population dynamics in ecological space and time*. Chicago: University of Chicago Press.
- Hanski, I., A. Moilanen, T. Pakkala, and M. Kuussaari. 1996. The quantitative incidence function model and persistence of an endangered butterfly population. *Conservation Biology* 10:578–590.
- Holling, C. S. 1978. *Adaptive environmental assessment and management*. London: Wiley.
- Howe, R. W., G. J. Davis, and V. Mosca. 1991. The demographic significance of sink populations. *Biological Conservation* 57:239–255.
- Jenkins, R. E. 1985. Information methods: Why the heritage programs work. *Nature Conservancy News* 35(6):21–23.
- . 1988. Information management for the conservation of biodiversity. In E. O. Wilson (ed.), *Biodiversity*. Washington: National Academy Press.
- Karieva, P., D. Skelly, and M. Ruckelshaus. 1996. Reevaluating the use of models to predict the consequences of habitat loss and fragmentation. In S. T. A. Pickett, R. S. Otsfeld, M. Schachak, and G. E. Likens (eds.), *The ecological basis of conservation: Heterogeneity, ecosystems, and biodiversity*. New York: Chapman & Hall.
- Karr, J. R. 1991. Biological integrity: A long-neglected aspect of water resource management. *Ecological Applications* 1:66–84.
- Kendeigh, S. C., H. I. Baldwin, V. H. Cahalane, C. H. D. Clarke, C. Cottam, I. M. Cowan, P. Dansereau, J. H. Davis, F. W. Emerson, I. T. Haig, A. Hayden, C. L. Hayward, J. M. Linsdale, J. A. MacNab, and J. E. Potzger. 1950–51. Nature sanctuaries in the United States and Canada: A preliminary inventory. *Living Wilderness* 15(35):1–45.
- Kerr, J. T. 1997. Species richness, endemism, and the choice of areas for conservation. *Conservation Biology* 11:1094–1100.
- King, C. M. 1989. *The natural history of weasels and stoats*. Ithaca: Comstock Publishing Associates and Cornell University Press.
- Knight, R. L., and K. Gutzwiller (eds.). 1995. *Wildlife and recreationists: Coexistence through research and management*. Washington, D.C.: Island Press.
- Kramer, R., C. van Schaik, and J. Johnson. 1997. *Last stand: Protected areas and the defense of tropical biodiversity*. New York: Oxford University Press.
- Lambeck, R. J. 1997. Focal species: A multi-species umbrella for nature conservation. *Conservation Biology* 11:849–856.
- Leopold, A. 1941. Wilderness as a land laboratory. *Living Wilderness* 6(July):3.
- Lindstedt, S. L., B. J. Miller, and S. W. Buskirk. 1986. Home range, time, and body size in mammals. *Ecology* 67:413–418.
- Lindzey, F. G. 1982. Badger. In J. A. Chapman and G. A. Feldhamer (eds.), *Wild mammals of North America*. Baltimore: John Hopkins University Press.

- Lyon, L. J. 1983. Road density models describing habitat effectiveness for elk. *Journal of Forestry* 81:592–595.
- Mace, R. R., and J. S. Waller. 1998. Demography and population trend in grizzly bears in the Swan Mountains. *Conservation Biology* 12:1005–1016.
- Mattson, D. J., S. Herrero, R. G. Wright, and C. M. Pease. 1996. Science and management of Rocky Mountain grizzly bears. *Conservation Biology* 10:1013–1025.
- McNeely, J. A. 1988. *Economics and biological diversity: Developing and using economic incentives to conserve biological diversity*. Gland, Switzerland: World Conservation Union.
- McShea, W. J., H. B. Underwood, and J. H. Rappole. 1997. *The science of overabundance: Deer ecology and population management*. Washington, D.C.: Smithsonian Institution Press.
- Miller, B., G. Ceballos, and R. Reading. 1994. Prairie dogs, poison, and biotic diversity. *Conservation Biology* 8:677–681.
- Miller, B., R. P. Reading, and S. Forrest. 1996. *Prairie night: Black-footed ferrets and the recovery of endangered species*. Washington, D.C.: Smithsonian Institution Press.
- Miller, B., R. Reading, J. Stritholt, C. Carroll, R. Noss, M. E. Soulé, O. Sanchez, J. Terborgh, D. Brightsmith, T. Cheeseman, and D. Foreman. 1998. Focal species in the design of reserve networks. *Wild Earth* 8(4):81–92.
- Miller, S. G., R. L. Knight, and C. K. Miller. 1998. Influence of recreational trails on breeding bird communities. *Ecological Applications* 8:162–169.
- Mills, L. S., M. E. Soulé, and D. F. Doak. 1992. The history and current status of the keystone species concept. *BioScience* 43:219–224.
- Myers, N. 1988. Threatened biotas: "Hot spots" in tropical forests. *Environmentalist* 8:187–208.
- Nilsson, C., and E. Götmark. 1992. Protected areas in Sweden: Is natural variety adequately represented? *Conservation Biology* 6:232–242.
- Noss, R. F. 1983. A regional landscape approach to maintain diversity. *BioScience* 33:700–706.
- . 1987. From plant communities to landscapes in conservation inventories: A look at The Nature Conservancy (USA). *Biological Conservation* 41:11–37.
- . 1992. The Wildlands Project: Land conservation strategy. *Wild Earth* (special issue)1:10–25.
- . 1996. Protected areas: How much is enough? In R. G. Wright (ed.), *National parks and protected areas*. Cambridge, Mass.: Blackwell.
- Noss, R. F., and A. Y. Cooperrider. 1994. *Saving nature's legacy: Protecting and restoring biodiversity*. Washington D. C.: Island Press.
- Noss, R. F., M. A. O'Connell, and D. D. Murphy. 1997. *The science of conservation planning: Habitat conservation under the Endangered Species Act*. Washington, D.C.: Island Press.
- Noss, R. F., H. B. Quigley, M. G. Hornocker, T. Merrill, and P. C. Paquet. 1996. Conservation biology and carnivore conservation in the Rocky Mountains. *Conservation Biology* 10:949–963.
- Noss, R. F., J. R. Stritholt, K. Vance Borland, C. Carroll, and P. Frost. 1999. Sci-

- ence-based conservation planning in the Klamath-Siskiyou ecoregion. *Natural Areas Journal* 19.
- Pickett, S. T. A. 1997. The ecological basis of conservation. In S. T. A. Pickett, R. S. Ostfeld, M. Shachak, and G. E. Likens (eds.), *The ecological basis of conservation: Heterogeneity, ecosystems, and biodiversity*. New York: Chapman & Hall.
- Powell, R. A. 1979. Mustelid spacing patterns: Variations on a theme by Mustela. *Zeitschrift für Tierpsychologie* 50:153–165.
- Power, M. E., D. Tilman, J. A. Estes, B. A. Menge, W. J. Bond, L. S. Mills, G. Daily, J. C. Castilla, J. Lubchenco, and R. T. Paine. 1996. Challenges in the quest for keystones. *BioScience* 46:609–620.
- Pressey, R. L. 1998. If the world were a logical place . . . Principles for the effective location of scarce conservation resources. Paper presented at annual meeting of the Society for Conservation Biology, Sydney, Australia, 16 July.
- Pressey, R. L., and A. O. Nicholls. 1989. Application of a numerical algorithm to the selection of reserves in semi-arid New South Wales. *Biological Conservation* 50:263–278.
- Pressey, R. L., C. J. Humphries, C. R. Margules, R. I. Vane-Wright, and P. H. Williams. 1993. Beyond opportunism: Key principles for systematic reserve selection. *Trends in Ecology and Evolution* 8:124–128.
- Pressey, R. L., H. P. Possingham, and C. R. Margules. 1996. Optimality in reserve selection algorithms: When does it matter and how much? *Biological Conservation* 76:259–267.
- Redford, K. H., and S. E. Sanderson. 1992. The brief, barren marriage of biodiversity and sustainability? *Bulletin of the Ecological Society of America* 73:36–39.
- Robinson, J. G. 1993. The limits to caring: Sustainable living and the loss of biodiversity. *Conservation Biology* 7:20–28.
- Schowalter, T. D. 1988. Forest pest management: A synopsis. *Northwest Environmental Journal* 4:313–318.
- Scott, J. M., and B. Csuti. 1997. Noah worked two jobs. *Conservation Biology* 11:1255–1257.
- Scott, J. M., B. Csuti, J. D. Jacobi, and J. Estes. 1987. Species richness: A geographical approach to protecting future biodiversity. *BioScience* 37:782–788.
- Scott, J. M., F. Davis, B. Csuti, R. Noss, B. Butterfield, C. Groves, J. Anderson, S. Caicco, F. D'Erchia, T. C. Edwards, J. Ulliman, and R. G. Wright. 1993. Gap analysis: A geographical approach to protection of biological diversity. *Wildlife Monographs* 123:1–41.
- Shaffer, M. L. 1992. *Keeping the grizzly bear in the American West: A strategy for real recovery*. Washington, D.C.: Wilderness Society.
- Shelford, V. E. (ed.). 1926. *Naturalist's guide to the Americas*. Baltimore: Williams & Wilkins.
- . 1933. Ecological Society of America: A nature sanctuary plan unanimously adopted by the society, 28 December, 1932. *Ecology* 14:240–245.
- Simberloff, D., and L. G. Abele. 1976. Island biogeography theory and conservation practice. *Science* 191:285–286.
- Smith, E. M. 1984. The Endangered Species Act and biological conservation. *Southern California Law Review* 57:361–413.

- Soulé, M. E. 1984. Application of genetics and population biology: The what, where, and how of nature reserves. In *Conservation, science, and society*. Paris: UNESCO.
- Soulé, M. E., and G. Lease (eds.). 1994. *Reinventing nature? Responses to postmodern deconstruction*. Washington, D.C.: Island Press.
- Soulé, M. E., and R. E. Noss. 1998. Rewilding and biodiversity: Complementary goals for continental conservation. *Wild Earth* 8(3):18–28.
- Soulé, M. E., and D. Simberloff. 1986. What do genetics and ecology tell us about the design of nature reserves? *Biological Conservation* 35:19–40.
- Soulé, M. E., B. Wilcox, and C. Holtby. 1979. Benign neglect: A model of faunal collapse in the game reserves of East Africa. *Biological Conservation* 15:259–272.
- Soulé, M. E., D. T. Bolger, A. C. Alberts, J. Wright, M. Soric, and S. Hill. 1988. Reconstructed dynamics of rapid extinctions of chaparral-requiring birds in urban habitat islands. *Conservation Biology* 2:75–92.
- Stanley, T. R. 1995. Ecosystem management and the arrogance of humanism. *Conservation Biology* 9:255–262.
- Terborgh, J. 1974. Preservation of natural diversity: The problem of extinction prone species. *BioScience* 24:715–722.
- . 1988. The big things that run the world: A sequel to E. O. Wilson. *Conservation Biology* 2:402–403.
- Thiel, R. P. 1985. Relationship between road densities and wolf habitat suitability in Wisconsin. *American Midland Naturalist* 113:404–407.
- Thomas, J. W., E. D. Forsman, J. B. Lint, E. C. Meslow, B. R. Noon, and J. Verner. 1990. A conservation strategy for the northern spotted owl. Portland, Ore.: USDA Forest Service, USDI Bureau of Land Management, USDI Fish and Wildlife Service and USDI National Park Service.
- Wace, N. 1977. Assessment of dispersal of plant species—the car borne flora in Canberra. *Proceedings of the Ecological Society of Australia* 10:166–186.
- Wells, M., S. Guggenheim, A. Khan, W. Wardojo, and P. Jepson. 1999. *Investing in biodiversity: A review of Indonesia's integrated conservation and development projects*. Washington, D.C.: World Bank.
- Western, D., and M. C. Pearl (eds.). 1989. *Conservation for the twenty-first century*. New York: Oxford University Press.
- Wiens, J. A. 1989. *The ecology of bird communities*. Vol. II: *Processes and variations*. New York: Cambridge University Press.
- Wikramanayake, E. D., E. Dinerstein, J. G. Robinson, U. Karanth, A. Rabinowitz, D. Olson, T. Mathew, P. Hedao, M. Conner, G. Hemley, and D. Bolze. 1998. An ecology based method for defining priorities for large mammal conservation: The tiger as a case study. *Conservation Biology* 12:865–878.
- Wilcove, D. S., and D. D. Murphy. 1991. The spotted owl controversy and conservation biology. *Conservation Biology* 5:261–262.
- Wilcove, D. S., C. H. McLellan, and A. P. Dobson. 1986. Habitat fragmentation in the temperate zone. In M. E. Soulé (ed.), *Conservation biology: The science of scarcity and diversity*. Sunderland, Mass.: Sinauer.
- Wilcove, D. S., D. Rothstein, J. Dubow, A. Philips, and E. Losos. 1998. Quantifying threats to imperiled species in the United States. *BioScience* 48:607–615.

- Willis, E. O. 1974. Populations and local extinctions of birds on Barro Colorado Island, Panama. *Ecological Monographs* 44:153-169.
- Wilson, E. O., and E. O. Willis. 1975. Applied biogeography. In M. L. Cody and J. M. Diamond (eds.), *Ecology and evolution of communities*. Cambridge, Mass.: Belknap Press.
- Woodroffe, R., and J. R. Ginsberg. 1998. Edge effects and the extinction of populations inside protected areas. *Science* 280:2126-2128.