Building a Wilderness Recovery Network

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Introduction

We have an opportunity unique to our generation: to halt a mass extinction. In order to accomplish this feat, conservation must be practiced on a truly grand scale. Simply put, the tide of habitat destruction must be stopped. Despite growing dangers of pollution, acid rain, toxic wastes, greenhouse effects, and ozone depletion, direct habitat alteration by humans remains the greatest of all threats to terrestrial and aquatic biodiversity, from Panama to Alaska and beyond. The effect of habitat alteration, generally speaking, is to create conditions unlike those under which many species native to an area evolved. Whereas some species thrive under the new conditions (cheatgrass, Norway rats, and cowbirds are familiar examples), other species are not so adaptable–they go extinct. Hence, the biodiversity crisis.

To stop the destruction of native biodiversity, major changes must be made in land allocations and management practices. Systems of interlinked wilderness areas and other large nature reserves, surrounded by multiple-use buffer zones managed in an ecologically intelligent manner, offer the best hope for protecting sensitive species and intact ecosystems. This article is about how to select and design such systems at a regional scale.

Below, I discuss the application of conservation biology to wilderness recovery and large-scale land protection strategy in general. After reviewing the ecological goals of such a strategy and discussing approaches to reserve selection and design, I outline the basic components of a wilderness recovery network: core reserves, buffer zones, and connectivity. The most important considerations in designing and managing such systems are representation of all ecosystems; population viability of sensitive species, especially large carnivores because they are usually most demanding; and perpetuation of ecological and evolutionary processes. My hope is that biodiversity activists and bioregionalists will be able to use this information in the design of ambitious wilderness recovery networks in their own regions.

Wilderness recovery, I firmly believe, is the most important task of our generation.

Application of Conservation Biology to Wilderness Recovery

Preservation of large, wild landscapes for their natural features is not a new idea, as the history of the national parks and wilderness movements in the United States attests (Fox 1981, Runte 1987). The introduction of science to the process of selecting and managing parks and other landscape-sized reserves, however, is both new and promising. Science alone, of course, is not sufficient; it must be guided by a land ethic (Leopold 1949).

Most national parks, wilderness areas, and other large reserves were selected on the basis of aesthetic and recreational criteria, or simply because they contained little of value in terms of extractable resources. The result is that high-elevation sites (rock and ice), wetlands, and other scenic but not particularly diverse lands dominate our system of protected areas; many ecosystem types are not represented, at least not in sizable areas (Davis 1988, Foreman and Wolke 1989, Noss 1990a). Because biology has been absent from design decisions, park boundaries do not conform to ecological boundaries and most parks and other reserves are too small to maintain populations of wide-ranging animals over the long term or to perpetuate natural processes (Kushlan 1979, Harris 1984, Newmark 1985).

Increasing discussion of "greater ecosystems" (Craighead 1979, Grumbine 1990), regional landscapes (Noss 1983), regional ecosystems (Keystone Center 1991), and ecosystem management (Agee and Johnson 1988) heralds a new way of looking at conservation, a way informed by ecological science. The basic idea underlying these new concepts is that most parks and other reserves are, by themselves, incomplete ecosystems. If parks or other reserves can be enlarged, and if the lands surrounding these areas are managed intelligently with the needs of native species and ecosystem processes in mind, a landscape as a whole may be able to maintain its ecological integrity over time.

If, on the other hand, surrounding lands are greatly altered from their natural condition, the chances that a reserve can maintain its integrity are slim. Animals with large home ranges (and therefore low population density) and other sensitive species will decline or fluctuate to extinction. Restoration may be needed to bring the complex of reserves and surrounding lands back to health. In any case, conservation biologists recognize that any system of parks, wilderness areas, and the public and private lands that envelop them must be managed as a whole in order to meet the goal of maintaining natural processes and native biodiversity over long spans of time.

Conservation biology and landscape ecology are both young sciences and show many signs of immaturity, such as theoretical confusion. However, the experience gained from myriad empirical case studies and observations, guided sometimes but not invariably by theory, has led to some general principles about how land might be "managed" (in a humble and non-manipulative sense of this term) to maintain biodiversity and ecological and evolutionary processes. The principles of conservation biology are not laws; we can expect them to be refined continually as the science matures. To put off implementing these principles until the science is completely developed, however, would be foolhardy; the forces that degrade natural ecosystems will not wait for the advice of scientists. Instead, the most prudent course for conservation is to proceed on the basis of the best available information, rational inference, and consensus of scientific opinion about what it takes to protect and restore whole ecosystems.

Ecological Goals

A conservation strategy is more likely to succeed if it has clearly defined and scientifically justifiable goals and objectives. Goal-setting must be the first step in the conservation process, preceding biological, technical, and political questions of how best to design and manage such systems. Primary goals for ecosystem management should be comprehensive and idealistic so that conservation programs have a vision toward which to strive over the decades (Noss 1987a, 1990b). A series of increasingly specific objectives and action plans should follow these goals and be reviewed regularly to assure consistency with primary goals and objectives (Stankey 1982). Four fundamental objectives are consistent with the overarching goal of maintaining the native biodiversity of a region in perpetuity (Noss 1991a,b):

1. Represent, in a system of protected areas, all native ecosystem types and seral stages across their natural range of variation.

2. Maintain viable populations of all native species in natural patterns of abundance and distribution.

3. Maintain ecological and evolutionary processes, such as disturbance regimes, hydrological processes, nutrient cycles, and biotic interactions, including predation.

4. Design and manage the system to be responsive to short- and long-term environmental change and to maintain the evolutionary potential of lineages.

Representation. Representation is one of the most widely accepted criteria of conservation. As an example, delegates of 62 nations at the Fourth World Wilderness Conference, in 1987, unanimously approved a resolution to preserve "representative examples of all major ecosystems of the world to ensure the preservation of the full range of wilderness and biological diversity" (Davis 1988). Perhaps the best way to represent all ecosystems is to maintain the full array of physical habitats and environmental gradients in reserves, from the highest to the lowest elevations, the driest to the wettest sites, and across all types of soils, substrates, and topoclimates (Hunter et al. 1988, Noss 1991a).

To accommodate seral stage diversity within vegetation types, reserves must either be large enough to incorporate functional natural disturbance regimes or be managed to supplement or mimic natural disturbances (Pickett and Thompson 1978, White and Bratton 1980). Because we do not know very well how to do the latter, as well as for ethical and aesthetic reasons, emphasis must be placed on maintaining the natural condition wherever it occurs.

Representation of all ecosystems and environmental gradients is the first step toward maintaining the full spectrum of native biodiversity in a region. Representation is subtly different from the conservation criterion of representativeness (see Margules and Usher 1981), where the best or typical examples of various community types are targeted for preservation. The latter concept is typological and static; it often results in the sequestration of "museum pieces" or specimens of nature (Noss and Harris 1986). Representation does not seek to preserve characteristic types of communities so much as to maintain the full spectrum of community variation along environmental gradients. It is understood that this variation is dynamic. The best example of a conservation program based on representation goals in North America is the Gap Analysis project directed by the U.S. Fish and Wildlife Service (Scott et al. 1991).

Viable Populations. Simply representing a species in a reserve or series of reserves does not guarantee that it will be able to persist in those areas (or anywhere) indefinitely. The representation objective must be complemented by the goal of maintaining viable populations of every species. Population viability is a central concern in conservation biology (Shaffer 1981, Soulé 1987). A viable population is one that has a high probability (say, 95 or 99%) of persisting for a long time (say, for 100 to 1,000 years). Population viability analysis is complex, with estimates depending on the mathematical model used, its assumptions, and values used for key population parameters such as population density and birth and death rates. With a few interesting exceptions, viable populations are generally on the order of thousands of individuals (Thomas 1990).

Fortunately, one does not have to worry about each of the thousands of species that may live in a region in order to meet the ambitious goal of maintaining viable populations of all native species. Rather, "conservation should not treat all species as equal but must focus on species and habitats threatened by human activity" (Diamond 1976). Concerns about population viability should be directed toward species at most risk of extinction in the region. Vulnerable species typically include those with small populations (limited or patchy distribution or low density), large home ranges, poor dispersal abilities, low reproductive potential, as well as those subject to exploitation or persecution or dependent on habitats that are themselves rare or threatened (Noss 1991a). These are the species that require our attention; many others tolerate or even thrive on human disturbance and can get along quite well without conservation assistance. For a regional wilderness recovery strategy, large and wide-ranging carnivores—bears, wolves, jaguar, puma, wolverine—are ideal primary target species.

Although answers to population viability questions are species-specific, some general principles for managing landscapes for vulnerable species are emerging. Thomas et al. (1990:23), in their conservation strategy for the northern spotted owl, listed five reserve design concepts "widely accepted among specialists in the fields of ecology and conservation biology." I generalize their guidelines below to multiple species, adding a sixth guideline that applies to species, such as large carnivores, that are especially sensitive to human disturbance (and, therefore, greatly in need of protection).

1. Species well distributed across their native range are less susceptible to extinction than species confined to small portions of their range.

2. Large blocks of habitat, containing large populations of a target species, are superior to small blocks of habitat containing small populations.

3. Blocks of habitat close together are better than blocks far apart.

4. Habitat in contiguous blocks is better than fragmented habitat.

5. Interconnected blocks of habitat are better than isolated blocks; corridors or linkages function better when habitat within them resembles that preferred by target species.

6. Blocks of habitat that are roadless or otherwise inaccessible to humans are better than those with roads and accessible habitat blocks.

Maintaining Ecological and Evolutionary Processes. One general theme of ecosystem management is that process is at least as important as pattern (Noss and Harris 1986). In other words, our concern for maintaining particular species, communities, places, and other entities must be complemented by a concern for the ecological and evolutionary processes that brought those entities into being and that will allow them to persist and evolve over the eons. Fundamental processes critical to ecosystem function include cycling of nutrients and flow of energy, disturbance regimes and recovery processes (succession), hydrological cycles, weathering and erosion, decomposition, herbivory, predation, pollination, seed dispersal, and many more. Evolutionary processes, such as mutation, gene flow, and differentiation of populations, must also be maintained if the biota is to adapt to changing conditions.

Allowing for Change. Maintaining ecological and evolutionary processes implies that change must be allowed to occur, hopefully without a net loss of biodiversity. A glaring deficiency of many conservation plans is their failure to recognize and accommodate change in nature. Conservation strategy has implicitly assumed that natural communities are unchanging entities (Hunter et al. 1988) and has sought to freeze in time snapshots of nature and associations of species that may have been apart for longer periods of their evolutionary histories than they have been together. The meaning of "preservation" must be revised to emphasize processes and to interpret local patterns in the context of global biodiversity over long time periods.

Short-term (years to centuries) ecological change occurs as a consequence of natural disturbance and succession. Disturbance–recovery cycles are among the most important of all ecological processes and have had a profound effect on the evolution of species (for example, many plant species are adapted to or even dependent on frequent fire). Only very large reserves or natural landscapes will be able to accommodate disturbance regimes characterized by stand replacement and large patch sizes without losing diversity (Pickett and Thompson 1978, Shugart and West 1981). In the Greater Yellowstone Ecosystem, for example, the lodgepole pine forests that cover much of the area are characterized by high-intensity, stand-replacing fires that recur naturally every two to three centuries; apparently, the landscape is not in equilibrium (Romme and Knight 1982, Romme and Despain 1989). Yellowstone National Park by itself is too small to exist in anything close to steady state with a natural fire regime—one more reason for managing the entire 19 million acres of the Greater Yellowstone Ecosystem as a whole.

Long-term (decades to millennia) change occurs largely as a result of changing climate. The response of plants and animals to climate change over time has primarily been to migrate with shifting climate zones. Communities did not migrate as intact units, however. Rather, plants and animals migrated at rates and in routes that were highly individualistic (Davis 1981, Graham 1986). The conservation strategy of maintaining all physical habitats (soil types, slope aspects, etc.) and intact environmental gradients, with corridors or other forms of connectivity linking habitats across the landscape, is perhaps the best way to accommodate change without losing biodiversity.

Approaches to Land Conservation

How might a regional land conservation program meet the objectives of representing all ecosystems, maintaining viable populations, maintaining natural processes, and allowing for change? Four approaches emphasized in recent years appear promising: (1) identify and protect populations of rare and endangered species; (2) maintain healthy populations of species that play critical roles in their ecosystems (keystone species) or that have pragmatic value as "umbrellas" (species that require large wild areas to survive, and thus if protected will bring many species along with them) or "flagships" (charismatic species that serve as popular symbols for conservation); (3) protect high-quality examples of all natural communities; and (4) identify and manage greater ecosystems or landscapes for both biodiversity conservation and sustainable human use.

These four approaches have obvious relationships to the objectives posed above. Unfortunately, they have sometimes been presented as competing rather than complementary strategies. Advocates of one approach may get very attached to it and fail to see its limitations or the merits of other approaches. In practice, the familiar strategy of protecting sites that harbor rare species or natural communities has worked quite well for plants and animals with small area requirements, but has been less successful in protecting wide-ranging animals and has been unable to capture landscape mosaics and other higher-order expressions of biodiversity (Noss 1987b). Empirical evidence has demonstrated that the small reserves selected through the site-by-site approach are heavily assaulted by external influences and often fail to retain the natural qualities for which they were set aside.

On the other hand, many so-called "ecosystem" or "landscape" approaches have lacked scientific rigor and objectivity and have failed to target those elements of biodiversity that are truly most threatened. Furthermore, most attempts to use "sustainability" as a management paradigm (Salwasser 1990) have been anthropocentric, biased toward commodity production, and seriously flawed from a biological standpoint, (Noss 1991c and in press).

These four approaches to conservation must be pursued in concert if the full spectrum of biodiversity is to be protected. Again, this can only be accomplished by representing all ecosystems (from small habitat patches to large landscape mosaics), maintaining viable populations of all native species (plant and animal, big and small), maintaining ecological and evolutionary processes, and accommodating change. The most difficult challenge is to meet all these objectives while still allowing for some kinds of human use. Most conservation biologists agree that compatible human uses of the landscape must be considered and encouraged in large-scale conservation planning. Otherwise, the strategy will have little public support. However, the native ecosystem and the collective needs of non-human species must take precedence over the needs and desires of humans, for the simple reason that our species is both more adaptable and more destructive than any other. Putting the needs of one species (humans) above those of all other species combined, as exemplified by the sustainable development theme, is one of the most pernicious trends in modern conservation.

Regionalization is a central issue in The Wildlands Project (also known as the North American Wilderness Recovery Project). Trying to make sense of the distribution of biodiversity and planning reserves across all of North America at once would be overwhelming. Regionalization on the basis of physiography, biogeography, land use, and other large-scale patterns helps assure that every physically and biotically distinct region is represented in a broad conservation strategy. Omernik (1986), for example, has produced a map portraying 76 ecoregions in the 48 conterminous states, and the Canadian Parks Service recognizes 39 terrestrial natural regions (Hummel 1989). Ecoregions or bioregions are a convenient scale for planning and often inspire feelings of belonging and protectiveness in their more enlightened human inhabitants. Many grassroots groups around the continent have defined bioregions and developed conservation plans for them. The Wildlands Project exists essentially to coordinate and provide technical support for these regional efforts.

Regionalization of reserve networks should be a hierarchical process; that is, we should consider regions within regions in our planning efforts. We can contemplate our homeland as a nested series, with our local watershed functioning as an interdependent part of a larger river watershed (a hydrologic unit), which in turn is part of an ecoregion or bioregion (for example, the Blue Ridge Mountains), then a biogeographical province (eastern deciduous forest), a continent, and eventually, the biosphere. Putting this nested hierarchy idea into practice means local nature reserve systems should be linked together into regional systems, which in turn are connected by inter-regional corridors that ultimately span continents. These hierarchical connections will help promote the multiple functions of connectivity discussed later in this article.

Reconnaissance and Selection

How do we choose reserves in a regional land conservation strategy? The process involves field inventory, remote sensing interpretation, and biogeographical research to determine the spatial distribution of biodiversity and wild areas, followed by an evaluation of which areas are most important to protect. The next step, drawing lines on maps, is not as easy as might be expected. Each line on a reserve design map represents a decision about areas to protect and areas to leave out. Within the near future, unfortunately, not every acre can be protected or restored. Decisions must be made quickly about which areas are most valuable ecologically, before they are altered irrevocably. Such decisions should not result in any area being "trashed." Ideally, all lands should be managed, at least in part, for biodiversity. But some areas deserve and require more rigorous protection than others. We call this process of picking and choosing "conservation evaluation" (Usher 1986).

Conservation evaluation is legitimate because biodiversity is not distributed uniformly across the landscape. Certain areas, sometimes called "hot spots," are unusually high in sheer number of species or contain concentrations of rare or endemic species or unusual natural communities. Areas of high physical habitat diversity, such as topographically complex landscapes with many distinct soil types, are often hot spots. Sites in a landscape also vary in conservation value as a result of historical influences, including past human activities. Roadless areas, especially when large (see Foreman and Wolke 1989), are of great importance because they harbor reclusive species and are often inherently sensitive to physical disturbance due to steep terrain or highly erodible soils (which made them difficult to exploit economically and explains why they are still roadless). Parking lots and corn fields, on the other hand, would score low in a conservation evaluation. Some degraded sites, however, may be priorities for restoration due to their locations relative to other landscape features, such as lying within a corridor that links hot spots across a landscape.

Core reserves and primary corridors in a regional network should enclose and link biologically critical areas (i.e., those that contribute to the goals discussed above) in a continuous system of natural habitat whenever possible. Some critical steps in selecting core reserves (the most strictly protected areas) and primary linkages in a wilderness recovery network, are as follows (Foreman 1976; Noss 1987a, 1991a, b,d; Foreman and Wolke 1989):

1. Select areas that, on the basis of field reconnaissance and interpretation of maps, aerial photographs, or satellite images, appear to be roadless, undeveloped, or otherwise in essentially natural condition. Center proposed core reserves on these undeveloped areas. A map of land ownership will show which of these areas are on public lands.

2. Add those landscapes with roads that are relatively undeveloped and restorable, especially when adjacent to or near roadless areas. Addition of such areas is important to increase core reserve size and to link roadless areas into larger complexes or networks.

3. Map the distribution of rare species and community types in your region, using state natural heritage program databases (these also exist for some Canadian provinces and Latin American countries). The heritage programs use a five-point scale of global and statewide endangerment developed by The Nature Conservancy, with rank 1 signifying the most imperiled elements. Map occurrences of all species, subspecies, varieties, and communities that rank 3 (very rare and local throughout range or found locally

in a restricted range) or higher at a global scale (G3 or T3, G2 or T2, and G1 or T1; the G indicates global status and the T indicates status of taxonomic subcategories). Add species that are imperiled or critically imperiled statewide (S2 and S1), though they may be less rare globally. Request a computer printout from the heritage program with data on each occurrence, including township, range, section and other location information. Map occurrences on mylar overlays on maps ranging from 1:100,000 to 1:250,000 scale (e.g., Forest Service 1/2 inch = 1 mile maps are 1:126,720). Local analyses should use 1:24,000 scale (the familiar 7.5-minute quadrangle maps) or larger. If you use a Geographic Information System (GIS), you can request a disk with longitude/latitude coordinates of occurrences. In some regions, mapping the distribution of rare species and communities might be the most practical first step in the network design process.

4. Draw polygons around clusters or constellations of rare species and community types. If not encompassed in core reserves proposed in steps 1 and 2, add these polygons to the system. Some hot spots will be naturally isolated (for instance, caves, serpentine barrens, or kettlehole bogs), so linking them by corridors is unnecessary.

5. Obtain information from the U.S. Fish and Wildlife Service GIS gap analysis (if completed for your state or states) on unprotected and underprotected vegetation types and centers of species richness in your region (see Scott et al. 1991). The purpose of gap analysis is to provide information on representation of ecosystems and species in protected areas. A similar representation study is being conducted in Canada by World Wildlife Fund-Canada (A. Hackman, personal communication). Locate areas that contain vegetation types and centers of species richness (areas where the ranges of many species overlap) that are not adequately protected in existing reserves. Add these areas to your network of sites if not already encompassed through steps 1-4.

6. You have now determined the general locations of your core reserves and some of the linkages between them. Next, you need to define boundaries more precisely, add more corridors so that all sites that would be naturally linked are reconnected, and envelop the entire network in a matrix of buffer zones (Figure 1). To do these things, you must zoom in to the landscape scale (say, 1:24,000 or larger, if feasible). Refer to detailed road maps, land ownership maps, land-use information including grazing allotments, proposed timber sales, and mineral rights, wildlife maps such as ungulate winter range and dispersal corridors, and additional data, as available (Foreman 1976, Noss 1991b,d). This information also tells you about threats to sites which must be averted. Using this information and knowledge of the land, based on field reconnaissance and maps, adjust proposed boundaries.

7. As part of your final proposal, indicate specific actions that must be taken to secure the system. These actions include land and mineral rights acquisitions, wilderness or other reserve designations on public lands, road closures, road modifications (such as underpasses to allow migration of animals beneath highways), cancellation of grazing leases and timber sales, tree planting, dam removals, stream de-channelization, and other restoration projects (Noss 1991d).

The issue of appropriate size or scope of a regional wilderness recovery network, some aspects of which will be discussed later in this article, is thorny. Each region must be assessed individually. I suggest that at least half of the land area of the 48 conterminous states should be encompassed in core reserves and inner corridor zones (essentially

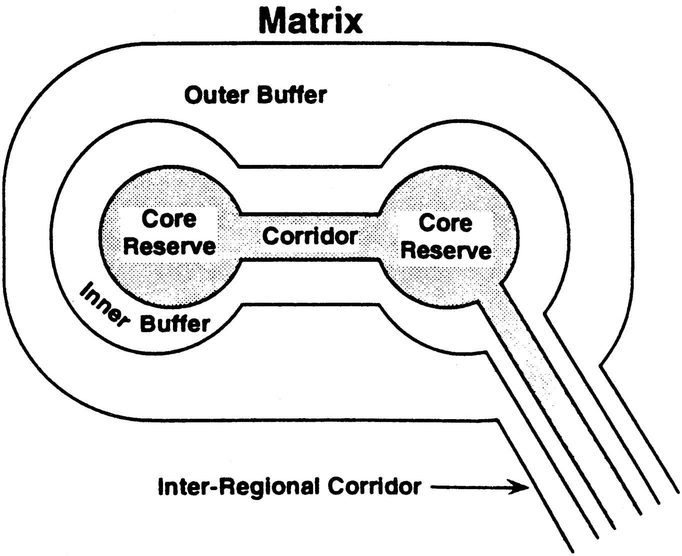

Figure 1. A regional wilderness recovery network, consisting of core reserves, connecting corridors or linkages, and buffer zones. Only two core reserves are shown, but a real system may contain many reserves. Inner buffer zones would be strictly protected, while outer zones would allow a wider range of compatible human uses. In this example, an interregional corridor connects the system to a similar network in another bioregion. "Matrix" refers to the landscape surrounding the reserve network, but this is only true in the first stages of a wilderness recovery project in regions now dominated by human activity. Eventually, a wilderness network would dominate a region and thus would itself constitute the matrix, with human habitations being the islands. In regions where wildlands is already the matrix, the inverted model should be implemented right away.

extensions of core reserves) within the next few decades; I also believe that this could be done without great economic hardship. Areas with more wild land remaining, such as much of Canada, Alaska, and parts of Mexico and Central America, should have higher targets. Some regions, such as the Midwestern Till Plains and Northeastern Coastal Zone, will take longer to restore to 50% wilderness, perhaps on the order of centuries. Nonetheless, half of a region in wilderness is a reasonable guess of what it will take to restore viable populations of large carnivores and natural disturbance regimes, assuming that most of the other 50% is managed intelligently as buffer zone.

Other authors, using different criteria, have arrived at similar estimates of what it might take to protect ecological integrity in a region. Odum and Odum (1972) suggested that managing half of southern Florida as natural area and half as cultural land was optimal. Earlier, Odum (1970) estimated that managing 40% of the state of Georgia as natural, 10% as urban-industrial, 30% in food production, and 20% in fiber production would maximize ecological services while maintaining the current standard of living. I would offer a more ambitious long-term goal, pending human population reduction, that at least 95% of a region be managed as wilderness and surrounding multiple-use wildlands. The following sections provide detailed ecological criteria for designing a wilderness recovery network.

Components of a Wilderness Recovery Network

A wilderness recovery network is an interconnected system of strictly protected areas (core reserves), surrounded by lands used for human activities compatible with conservation that put biodiversity first (buffer zones), and linked together in some way that provides for functional connectivity of populations and processes across the landscape. These basic concepts are common to many conservation strategies, including the biosphere reserves of the Man and the Biosphere (MAB) program (UNESCO 1974, Hough 1988, Batisse 1990, Dyer and Holland 1991), and the multiple-use module idea that applies these concepts at various spatial scales (Harris 1984, Noss and Harris 1986, Noss 1987a).

Below, I discuss core areas, buffer zones, and connectivity as they apply to wilderness recovery. I follow with a brief discussion of the "bigness" issue, that is, determining how large a reserve or reserve system must be to maintain its native biodiversity over time.

Core Areas. The backbone of a regional reserve system is formed by those protected areas managed primarily to maintain or restore their natural values. The selection of core reserves should be based on the criteria and objectives discussed above: representing all ecosystems, maintaining viable populations of all native species, maintaining ecological and evolutionary processes, and being responsive to change. Core reserves should collectively encompass the full range of communities, ecosystems, physical habitats, environmental gradients, and natural seral stages in each region. Design and management guidelines for specific core reserves require considerable site-specific research.

Buffer (Multiple-Use) Zones. A system of core reserves is necessary but not sufficient to maintain biodiversity. In most regions, strictly protected areas will not occupy enough land, in the short term, to meet the conservation goals suggested in this article (see Brussard 1991). For a largely wild region, such as much of the western United States and Canada, the multiple-use public lands that envelop reserves should be managed in a way more sensitive to natural ecosystems and processes than what is now the custom (to put it mildly). To the extent that buffer zones are managed intelligently, core reserves have a better chance of maintaining viable populations and regional landscapes will be richer in native biodiversity than if reserves are surrounded by intensive land use.

I use the terms "multiple-use zone" and "buffer zone" interchangeably (Noss 1991a). The former term, although tainted by misuse by public agencies and special interest groups, may be preferable because such zones can indeed provide for many human uses and function as much more than buffers. Multiple-use public lands adjacent to reserves should serve as at least marginal habitat for vulnerable species and should insulate reserves from intensive land uses. A reserve properly insulated from high-intensity land use by one or a series of buffer zones is, to a measurable degree, functionally enlarged as a conservation unit. In many cases, private lands will need to be acquired and added to national forests and other public lands in order to serve as effective buffers.

Physical and biotic edge effects can be serious problems for small reserves with high perimeter/area ratios (Noss 1983); buffer zones have been recommended to mitigate edge effects in these situations (Harris 1984, Noss 1987a). Among forest communities, deleterious edge effects are best documented for closed-canopy forest types. Forest interior species may be sensitive to a variety of edge-related environmental changes. Increased blowdown potential may extend at least two tree-heights into a stand (Harris 1984, Franklin and Forman 1987). Some kinds of external influences, such as invasions of weedy species, penetrate much farther—perhaps 5 km or more into a forest (Janzen 1986). Weedy, exotic species of plants and animals are often abundant in human-disturbed environments; buffer zones may help screen these pests away from reserves. Core reserves, if designed according to the criteria discussed in this article, will generally be large enough that edge effects from their boundaries should not be a significant problem. Edge effects from internal fragmentation, such as that caused by roadbuilding and clearcutting, will be a threat until artificially disturbed habitats are restored.

Multiple-use zones have functions other than ameliorating edge effects. If maintained in low road density, they can protect core reserves from poaching and other harmful human activities that otherwise would be intense near reserve boundaries. They may also protect developed areas from depredating large mammals (such as grizzly bears and wolves) that will hopefully thrive in core reserves. Outer zones of vegetation resistant to high-intensity fire (such as grasslands), supplemented by fire lanes on the perimeter, may protect private forests and settlements from fires originating in core reserves.

An ideal function of multiple-use zones is to provide supplementary habitat to native species inhabiting a core reserve, thus increasing population size and viability. To the extent that multiple-use zones can be restored and managed to increase habitat area for those species most vulnerable to extinction, they will enlarge the effective area of the reserve. In some cases, animals that depend on several different habitat types, perhaps on a seasonal basis, will require areas not represented in a reserve to meet a portion of their annual life-history needs. Obvious examples are elk and deer that make seasonal migrations between high-elevation summer ranges and low-elevation winter ranges (Adams 1982). Core reserves can be created or enlarged to protect the most critical migration corridors, but many other movement areas will need to be protected by buffer zones.

Population dynamics across reserve boundaries can be complex. The notion of "source" versus "sink" habitats is germane here. As discussed by Pulliam (1988), source habitats are those that can support a net population increase, whereas "sink" habitats have in situ death rates higher than birth rates—they are "black holes" for wildlife. Populations are maintained in sink habitats only when subsidized by source habitats. Population density, therefore, may be a misleading indicator of habitat quality (Van Horne 1983). Concentrations of socially subordinate individuals (for instance, female and subadult male bears, or juvenile songbirds) in sink habitats may lead to mistaken impressions about habitat quality in those areas.

Although most of the population may exist at any given time in the sink habitat, conservation of the source habitat is absolutely essential to the survival of the whole population (Pulliam 1988, Howe et al. 1991).

The source-sink dichotomy (really a continuum) is relevant to the planning of buffer zones, because whenever habitat quality or population density for a species differs across a boundary, we can expect net movement of individuals across that boundary. This gradient-aligned dispersal is in addition to any movements made by animals that use resources on both sides of the boundary.

The developed landscape is often a sink, relative to reserve habitat, for native species (Janzen 1986, Schonewald-Cox and Bayless 1986, Buechner 1987). In the absence of well protected buffer zones, surplus animals produced in a park or other reserve may disappear into the developed landscape matrix, seldom reproducing and often dying there. Areas near roads and developments are well-known population sinks for Yellowstone grizzly bears, even within the National Park (Mattson and Knight 1991a). Across the Greater Yellowstone Ecosystem, illegal shooting and management "removals" are the major causes of mortality for the grizzly and are associated with real or perceived threats to humans or livestock, particularly sheep (Knight et al. 1988, Mattson 1990). Road closures and removal of sheep allotments are probably essential to grizzly bear recovery in this region (Mattson and Reid 1991).

If, on the other hand, lands surrounding core reserves are managed for the benefit of a sensitive species and contain habitat of moderate or high quality for that species, those lands may be minor sinks or no sink at all. If death rates in the buffer are approximately equal to birth rates, there will be no drain on the reserve population. Furthermore, a recent model suggests that sink habitats can actually contribute to metapopulation persistence (Howe et al. 1991). Although the highest priority is to identify and protect source habitats where annual reproduction exceeds mortality, a large fraction of a species' population may exist in sink habitats and those areas may extend the survival time of the metapopulation as a whole (a metapopulation is a collection of local populations linked by dispersal; Levins 1970). A buffer zone of marginal habitat quality, even if technically a sink, can be managed to reduce mortality and contribute to metapopulation persistence. Dispersal is a key factor in metapopulation persistence (Figure 2) and can be enhanced if buffer zones are managed to minimize road density, artificial openings, and other potential barriers.

Another advantage of buffer zones around reserves may be to allow plants and animals to shift their distributions in response to disturbances and other changes. In the long term, or perhaps rather quickly (within the next few decades, if prevailing models of anthropogenic global warming prove true), organisms will need to shift their ranges in response to climate change (Peters and Darling 1985). Buffer zones or habitat corridors between reserves will help organisms make these distributional shifts and avoid extinction (see connectivity discussion, below).

In order to protect species sensitive to legal or illegal hunting or persecution, such as grizzly bear, jaguar, and wolf, buffer zones must have low road density (say, no more than 0.5 miles of road per square mile). Research has shown that road densities as low as 0.8 or 0.9 miles per square mile may make habitat unsuitable for large carnivores and omnivores (Brody 1984, Thiel 1985, Mech et al. 1988). Road access is a major threat to wildlands throughout North America (Diamondback

Metapopulation Dynamics

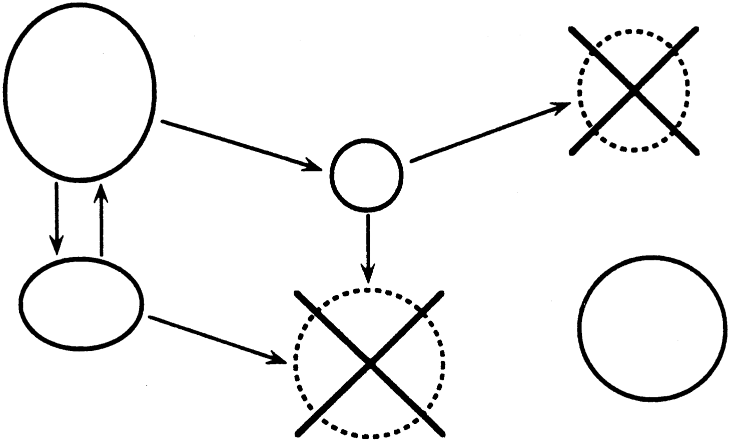

Figure 2. A hypothetical example of metapopulation dynamics. Subpopulations are connected by dispersal, which may keep local populations from going extinct (the "rescue effect") and thus stabilizes the metapopulation. In this example, two subpopulations (each marked by an "x") have recently gone extinct. Dispersal from other subpopulations allows for these areas to be recolonized. The subpopulation in the lower right is not receiving any immigrants, perhaps because developments or other barriers lie between it and other subpopulations. Should this isolated subpopulation go extinct, it can only be recolonized by restoration of dispersal corridors or active reintroduction by humans.

1990). Road closures are one of the most effective ways to make multiple-use lands function as buffers.

Connectivity. A fundamental principle for designing regional reserve systems is connectivity. Unless many millions of acres in size, individual core reserves will not be able to function alone as whole ecosystems, in the sense of maintaining viable populations of large animals and ecological and evolutionary processes (see the following section on bigness). In the long term, regions themselves must be functionally interconnected to allow for long distance dispersal and migration in response to climate change. In order to maintain their ecological integrity, many or most core reserves will have to be functionally joined to other protected areas.

Habitat fragmentation, one of the greatest of all threats to biodiversity (Noss 1983 and 1987a, Harris 1984, Wilcox and Murphy 1985, Wilcove et al. 1986), is a process where large blocks of natural habitat are broken up into smaller and isolated pieces. Connectivity is in many respects the opposite of fragmentation. A reserve system with high connectivity is one where individual reserves are functionally united into a whole that is greater than the sum of its parts (Noss and Harris 1986).

As suggested above, properly managed buffer zones in which a constellation of reserves is embedded may provide adequate habitat connectivity. Key qualities of buffer zones that provide for animal movement are low road density and minimal development, clear-cutting, or other forms of habitat fragmentation. In some cases, however, distinct corridors of suitable habitat may be needed to link core reserves or reserve complexes into a functional network. These corridors may range in scale from short connectors a few dozen meters wide to regional corridors one hundred miles or more in length and many miles in width (Noss 1991d and 1993). I use the term "linkages" to emphasize the many types and functions of connectivity.

Linkages as Habitat. Some types of corridors are distinct in the natural landscape, riparian corridors being a good example. Riparian forests are highly productive and often very rich in species. As an illustration of how many animals may depend on riparian forests, in the Blue Mountains of Oregon and Washington 285 (75%) of the 378 species of terrestrial vertebrates either depend on or strongly prefer riparian zones over other habitats (Thomas 1979). Riparian forests are immensely valuable in their own right, aside from any role they may play as conduits for wildlife movement.

Wide protected corridors are basically extensions of core reserves. The width of corridor needed to contain an adequate amount of forest interior habitat and minimize edge effects is uncertain and depends on habitat quality both within and outside the corridor (Noss 1993). For example, the edge effect of increased blowdown risk extends at least two tree-heights into a forest (Harris 1984). If forest trees average 40 m in height, a corridor would have to be at least 360 m (approximately one-quarter mile) wide to maintain a modest 200 m wide strip of interior forest. Another consideration for determining optimal corridor width is the territory or home range size of target species expected to use the corridor. Because this issue also affects the ability of a corridor to promote dispersal, I discuss it below in the dispersal section.

Linkages for Seasonal Movements. The conservation function most commonly associated with corridors is to allow movement of animals between reserves. For wide-ranging animals, a small core reserve may not encompass a single annual home range. Some large carnivores have annual ranges of 1,000 or more sq km, and elk and mule deer may travel over 100 km in linear distance between summer and winter ranges (Noss 1991a and 1993). Maintaining safe travel opportunities for these species is largely a matter of protecting them from human predation; wide, roadless corridors will best serve this purpose.

Vertebrates often use traditional migration routes between summer and winter range. Elk generally use forested travel lanes, when available, for migratory movements (Adams 1982). Elk migration has been disrupted by removal of security cover by logging in many regions, for example on the Targhee National Forest near Island Park, Idaho. Travel corridors used by grizzly bears include ridgetops, saddles, and creek bottoms (LeFrance et al. 1987); grizzlies avoid crossing clearcuts and other large openings (D. Mattson, personal communication). Traditional wildlife migration routes should be incorporated into corridors between reserves. Habitat nodes or staging areas for migratory animals also should be identified and protected.

Linkages for Dispersal. Dispersal refers to the movement of organisms away from their place of origin, such as the movement of subadult animals out of the parental home range. Many species are distributed as metapopulations (Figure 2). Dispersal can counteract the isolating effects of habitat fragmentation, but only if adequate dispersal habitat remains. For a regional metapopulation of a species to persist, movement of individuals between patches must be great enough to balance extirpation from local patches (den Boer 1981). Late successional species tend to be poorer dispersers and more vulnerable to extinction in fragmented landscapes than species associated with early successional stages (den Boer 1990). Therefore, dispersal corridors are most important for late successional species and for species, such as large carnivores or ungulates, likely to be killed by humans or vehicles in developed landscapes.

Dispersal is more often successful when habitat in a corridor or other linkage is similar to the habitat in which a species lives (Wiens 1989), with some exceptions (Bleich et al. 1990). Just how similar it must be is a question yet to be answered. Thomas et al. (1990) predicted, on the basis of a collective best guess, that maintaining 50% of the landscape matrix between proposed habitat conservation areas in forest stands averaging at least 11 inches dbh and 40% canopy closure would provide adequate dispersal habitat for the northern spotted owl. Other scientists might have opted for more stringent standards, for example, 75% of the matrix, more canopy closure, lower road density, and less edge to protect owls from shooting and great horned owl predation. In any case, maintaining matrix suitability, as in the multiple-use zoning strategy reviewed above, is another way to provide connectivity between core reserves. For those species most sensitive to human harassment, barrier effects of roads, or edge effects, the prudent strategy is to maintain wide corridors with roadless core zones and true interior habitat (Noss 1993).

Corridors that maintain resident populations of animals are more likely to function effectively as long-distance dispersal conduits for those species (Bennett 1990). Minimum corridor widths, then, might be based on average home range or territory diameters of target animals (Harrison 1992). Consider the grizzly bear, with an average male lifetime home range of approximately 3,885 sq km (1,500 square miles) in the Greater Yellowstone Ecosystem (Mattson and Reid 1991). A male lifetime home range may contain, at any one time, one or two adult males, and up to a few females; thus, it would provide an adequate width for an inter-regional corridor.

If the population of grizzlies in the Greater Yellowstone Ecosystem is to be connected to other populations, which seems to be necessary to assure population viability, then wide corridors with resident grizzlies must connect Yellowstone with the Northern Continental Divide Ecosystem (about 200 miles away) and the wildlands of central Idaho (Picton 1986, Metzgar 1990). Considering rectangular lifetime home ranges twice as long as wide, a between-population corridor for grizzly bears should be at least 44.25 km (27.5 miles) wide. A corridor based on annual or seasonal home ranges would be much narrower but also less secure; it is best to risk erring on the side of caution. Because road densities above about 0.5 miles of road per square mile of habitat may be a threat to grizzlies (Bader 1991), road closures would be required to make inter-regional corridors safe. Figure 1 portrays a wide inter-regional corridor of the type discussed here and others are shown in the statewide network proposed for

Florida (Figure 3; Noss 1985, 1987a).

Linkages for Long-Distance Range Shifts. A final function of connectivity is to provide for long-distance migration of species in response to climate change. Models of anthropogenic global warming predict dramatic shifts in vegetation in most regions. In the Greater Yellowstone Ecosystem, for example, the upper and lower tree lines are expected to move considerable distances (Romme and Turner 1991). Human activities have imposed a new set of barriers on the landscape that, in addition to natural barriers, may interfere with long-distance movements. Unfortunately, if rates of global warming in the next few decades are as fast as predicted, many species will be unable to migrate quickly enough, even along ideal corridors. In Yellowstone, as elsewhere, species with short and rapid life histories, such as introduced weeds, will probably adjust well to climate change, as will broadly distributed species such as lodgepole pine. On the other hand, whitebark pine and many alpine species, which already show limited and discontinuous distributions, are at high risk of extirpation (Romme and Turner 1991).

Mountainous regions with broad elevational spans are better suited for adaptation to climate change than flatter regions. A 3°C rise in temperature, as is predicted under greenhouse warming, translates to a

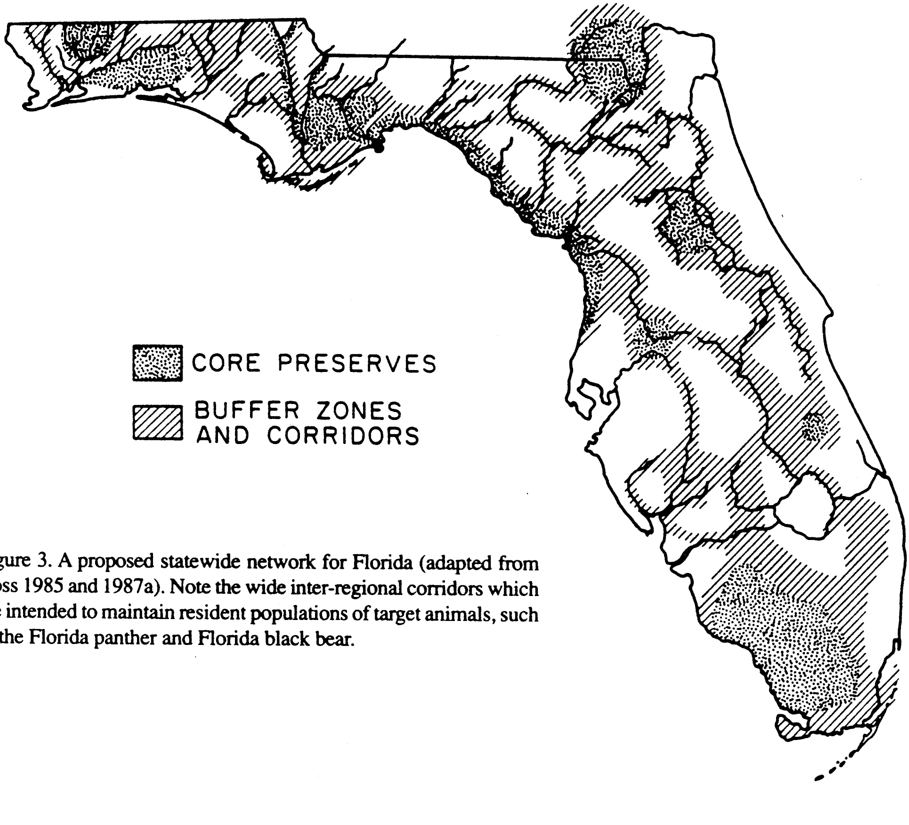

Figure 3. A proposed statewide network for Florida (adapted from Noss 1985 and 1987a). Note the wide inter-regional corridors which are intended to maintain resident populations of target animals, such as the Florida panther and Florida black bear.

latitudinal range shift of roughly 250 km (155 miles), but an elevational range shift of only 500 m (1,640 ft.) (MacArthur 1972). Perhaps the best way to facilitate adaptive migration of species in response to climate change is to maintain intact environmental gradients, as discussed earlier in this article. Complete, unfragmented elevational gradients, for example from foothill grasslands and shrub steppe up to alpine tundra, will offer the best opportunities for upslope migration of species in response to global warming.

The Issue of Bigness

The question that has most occupied conservation biologists for the last two decades has been, "How large does a reserve need to be to maintain its diversity over time?" Researchers have sought answers in various ways and have discovered many reasons why large reserves are preferable to small ones. The desirability of large reserves, all else being equal, is one of the few almost universally accepted principles of conservation biology (Soulé and Simberloff 1986, Thomas et al. 1990).

Some of the best reasons for large reserves are quite practical: per unit area, they are usually cheaper to buy and require less management effort to maintain their natural qualities than smaller reserves (Pyle 1980, White and Bratton 1980, Noss 1983). Due to the species-area relationship and its many potential causes (Connor and McCoy 1979), larger reserves also contain more species than smaller reserves in the same biogeographic region. Island biogeographic theory suggests that large islands or nature reserves contain more species because they experience higher colonization rates and lower extinction rates than smaller areas (MacArthur and Wilson 1967, Diamond 1975). But perhaps the most compelling arguments for large reserves have to do with population viability and habitat diversity in the face of environmental change.

Reserve Size and Population Viability. Estimates of minimum viable population sizes and corresponding reserve sizes are alarmingly high. Small populations are vulnerable to extinction due to a number of factors, including environmental change, demographic stochasticity, social dysfunction, and genetic deterioration (Shaffer 1981, Soulé 1987). All populations fluctuate over time; small populations are more likely to fluctuate down to zero. A recent review of empirical studies (Thomas 1990) concluded that an average population of 1,000 individuals must be maintained in order to assure population viability of species with average levels of fluctuation in abundance. Bird and mammal species with highly variable populations may require average populations of about 10,000 individuals for long-term persistence. In some cases, however, populations can persist for long periods at surprisingly small sizes, even less than 50 individuals (e.g., Walter 1990). It seems wise, however, to strive for large populations of vulnerable species whenever possible.

Habitat quality, social behavior, and other factors will determine how minimum population estimates translate to reserve size estimates. Schonewald-Cox (1983) estimated that reserves of 10,000 to 100,000 ha (25,000 to 250,000 acres) might maintain viable populations of small herbivorous and omnivorous mammals, but that large carnivores and ungulates require reserves on the scale of 1 to 10 million ha (2.5 to 25 million acres). Using a minimum viable population size of 50 (which is reasonable only under very short planning horizons), it has been estimated that grizzly bear populations in Canada require an average of 49,000 sq km (12.1 million acres); wolverines, about 42,000 sq km (10.4 million acres); and wolves, about 20,250 sq km (5 million acres)

(Hummel 1990). For a minimum viable population of 1,000 (see Thomas 1990), the figures would be 242 million acres for grizzly bears, 200 million acres for wolverines, and 100 million acres for wolves. And, of course, it is not prudent to manage down to the minimum!

Such immense areas could not be contained today within individual reserves, but only within regional and inter-regional systems of inter-linked reserves, for example, the Greater Yellowstone Ecosystem linked to the Northern Continental Divide Ecosystem and on to the Canadian Rockies; the Florida network (Figure 3) linked to a network that parallels the Appalachian Trail to Maine (Sayen 1987, Hunter et al. 1988); and a southern Arizona network linked to the rest of the Southwest and to Mexico. Regional and inter-regional systems of protected areas connected by wide corridors appear to be necessary to maintain viable and well-distributed populations of most large carnivores, hence the importance of these species as targets for wilderness recovery planning.

Reserves making up a habitat system for large carnivores should be predominately wilderness, but should include appropriately managed buffer zones. In order to protect these species, which are very sensitive to human predation and harassment (Thiel 1985, Mattson et al. 1987, McLellan and Shackleton 1988, Knight et al. 1988, Craighead et al. 1988, Mattson and Knight 1991a,b), open roads and other means of human access must be tightly restricted. Recognizing (on paper) the threats posed by open roads, the Gallatin National Forest in Montana has implemented an open road density (ORD) standard of 0.5 miles of road per square mile in critical grizzly bear and big game habitat. The 0.5 ORD standard is assumed to maintain a habitat effectiveness of at least 70%, an accepted minimum for population viability of grizzlies and elk (Bader 1991). Road closures to reduce the density of roads to an acceptable level (less than 0.5 miles per square mile) in each region will be among the most difficult actions politically, but most necessary ecologically.

Reserve Size & Disturbance Regimes

Maintaining habitat diversity and the full range of species associated with different seral stages requires that natural disturbance regimes be taken into account when considering reserve size. Disturbances are patchy in time and space, so that a landscape can be viewed as a "shifting mosaic" of patches in various stages of recovery from disturbance (Bormann and Likens 1979). The mosaic appears to shift because new disturbances occur in some portions of the landscape at the same time as formerly disturbed areas are growing back into forest or other mature vegetation. Reserves that are small relative to the spatial scale (patch size) of disturbance may experience radical fluctuations in the proportions of different seral stages over time, which in turn threaten populations that depend on certain stages. Many nature reserves are smaller than the area likely to be disturbed by a single wildfire or windstorm, and therefore are quite vulnerable to loss of habitat diversity and associated species.

If a core reserve is to maintain a relatively stable mix of seral stages and species over time, it must be large enough that only a relatively small part of it is disturbed at any one time. Another requirement is that a source of colonists (that is, a reproducing population of the same species) exists within the reserve or within a reasonable dispersal distance so that populations can be reestablished on disturbed sites (see Figure 2). Disturbance patch sizes and spatial distribution, successional dynamics, potential refugia (areas within the reserve, or nearby, that are not likely to be disturbed), and dispersal capacities of species, are the ecological factors to keep in mind when planning reserves around natural disturbance regimes.

Pickett and Thompson (1978) used these criteria to define a "minimum dynamic area" as "the smallest area with a natural disturbance regime, which maintains internal recolonization sources, and hence minimizes extinction." In theory, a minimum dynamic area should be able to manage itself and maintain habitat diversity and associated native species with no human intervention. Shugart and West (1981) estimated that landscapes must be some 50-100 times larger-than-average disturbance patches to maintain a relative steady state ("quasi-equilibrium") of habitats. In a steady-state landscape, the proportions of different seral stages in the overall landscape would be relatively constant over time, even though the sites occupied by various seral stages would change. A steady state may never be reached in some ecosystem types, such as those regularly experiencing large, catastrophic fires (Baker 1989). Romme and Knight (1982) concluded that Yellowstone National Park is not large enough to exist in equilibrium with its disturbance regime, and that a steady state for the Greater Yellowstone Ecosystem as a whole is unlikely.

Very large but infrequent fires are characteristic of many landscapes in the central and northern Rocky Mountains. Surveys by Ayres (1901) in the Lewis and Clarke Reserve of Montana (which included what are now the Bob Marshall, Great Bear, and Scapegoat Wilderness Areas) showed that over 300,000 ha (750,000 acres) burned in the area in one year, 1889, and up to 136,000 ha in a single fire. About 100,000 ha burned in the Canyon Creek Fire in 1988 (Losensky 1990). Similarly, fires in the Coast Range of Oregon have burned as much as 200,000 ha (Spies and Cline 1988). In the Northwest, fires become smaller and less severe, but considerably more frequent, along a transect from the Washington Cascades to northern California (Swanson et al. 1990, Morrison and Swanson 1990).

Although most fires are mosaics, a minor portion of the affected acreage being of stand-replacement intensity, the immense scale of many natural disturbances provides a strong argument for establishing large reserves. Active fire suppression is simply not a reasonable option in these cases. Experience and research have shown that fire is a natural part of these systems and essential to their overall diversity; moreover, many fires are impossible to suppress (Christensen et al. 1989).

A core reserve, by itself, need not encompass a minimum dynamic area. The concept implies that all natural seral stages be maintained over time and that dispersal distances between similar habitats are surmountable by native species; but there is no reason to insist that a steady state of seral stages be maintained, for this may rarely occur in nature (Pickett and White 1985). The steady-state concept is useful, however, in the sense that reserves large enough to be close to steady state will likely experience lower extinction rates than reserves where habitat conditions fluctuate wildly over time. Larger landscapes buffer the effects of disturbance on diversity of habitats and species (Shugart and Seagle 1985). Thus, the scale of management planning, including core reserves and surrounding multiple-use lands, should encompass something approximating a minimum dynamic area whenever possible; the complex as a whole can be managed to maintain habitat diversity.

Conclusions

This article has reviewed some considerations for designing wilderness recovery networks at a regional scale. The spotlight has been on

North America, but projects of the type described here are urgently needed worldwide. I have emphasized terrestrial ecosystems for the simple reason that this is my area of expertise. However, protection and restoration of entire regional landscapes, as promoted by The Wildlands Project, are intended to maintain aquatic and terrestrial ecosystems alike. Nonetheless, many aquatic biota will require special recovery techniques, such as de-channelization of streams and elimination of dams and water diversion structures, in order to be healthy again. Furthermore, marine ecosystems, particularly near shore, are in serious jeopardy in many regions and need comprehensive recovery strategies of their own.

I have highlighted the needs of large carnivores in this article because they are often acutely sensitive to human activity and hence are among the best indicators of wilderness condition. However, the stated goals of The Wildlands Project should make clear that not just carnivores, but all of biodiversity is the target of our efforts. Many sensitive assemblages (for example, neotropical migrant songbirds, anadromous fish, freshwater bivalve mollusks, and declining amphibian species) will require focused recovery work for many years to come. Importantly, ecosystem-level protection does not imply that we neglect individual species or assemblages on the brink of extinction; endangered species legislation should be strengthened and rigorously enforced to help imperiled taxa.

No substitute exists for detailed on-the-ground knowledge of the ecology and natural history of a region. General theory and insights gained from other regions are helpful, but do not transfer directly to areas with different biotas and histories. A long-term conservation plan for a region should be hypothesis-driven and adaptive; that is, we should scientifically test various approaches and techniques to see how well they work, then adjust our management to reflect new knowledge. Activists should enlist the participation of ecologists and other scientists most familiar with a region; if the latter will not themselves get actively involved in a project (some are afraid of tarnishing their cherished credibility as impartial observers), they may at least provide information and guidance. If all else fails, become an expert yourself on the ecology of your region!

The discussions above should make clear that planning on a bioregion-by-bioregion basis is incomplete. Because of the huge areas required to support viable populations of some animals and the necessity for all species to be able to migrate long distances with climate change, inter-regional and inter-continental planning is mandatory. The Wildlands Project will facilitate planning among regions and provide access to critical information, both scientific and tactical, to activists and planners worldwide. We now need, all of us, to put this information and strategy into action.

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