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THREATS TO WILDERNESS ECOSYSTEMS: IMPACTS AND RESEARCH NEEDS¹

DAVID N. COLE AND PETER B. LANDRES

*Aldo Leopold Wilderness Research Institute, Forest Service, U.S. Department of Agriculture,
P.O. Box 8089, Missoula, Montana 59807 USA*

Abstract. One of the primary purposes of designated wilderness areas is protection of natural ecosystems. However, the ecological integrity of these most protected of public lands is threatened by direct and indirect effects of human activities both internal and external to wilderness. Accelerated research programs on threats to wilderness are needed to realize the purposes for which wilderness was established and to improve our understanding of natural ecosystems. This paper reviews current knowledge and critical research needs for some of the most significant threats to wilderness ecosystems: (1) recreational use and its management; (2) livestock grazing and its management; (3) fire management; (4) introduction of alien species; (5) diversion and impoundment of water; (6) emission of atmospheric pollutants; and (7) management of adjacent lands. Some of these threats cause highly disruptive localized impacts, whereas some have a more widespread effect. Other threats are highly significant because they threaten rare or irreplaceable ecological attributes. Ecological science needs to be applied to improve evaluations of wilderness conditions, improve efforts to protect wilderness ecosystems from further degradation, and improve efforts to restore the integrity of disturbed systems.

Key words: *ecological impact; ecological restoration; nature preservation; research needs; threats; wilderness.*

INTRODUCTION

The National Wilderness Preservation System currently consists of 42 X 10⁶ ha, about one-sixth of the federal lands in the United States, excluding military and Indian reservations. These officially designated wilderness lands are devoted to the protection of natural ecosystems, as well as the use and enjoyment of these areas as wilderness. They have tremendous long-term scientific and ecological value to society because they have the potential to provide the best remaining standards of relatively unmodified land in the United States. However, even the ecosystems in these most protected of public lands are threatened by direct and indirect impacts from human activities both internal and external to the wilderness. Moreover, attempts to manage both natural and anthropogenic disturbance within wilderness may exacerbate these impacts.

More and better information is needed on threats to natural ecosystems in wilderness, as well as other protected areas such as national parks and natural areas. Several recent papers on priorities for ecological science identify the need for more research on the detection, understanding, and management of human-induced stresses in natural ecosystems (Brussard 1991, Lubchenco et al. 1991). Unless accelerated research programs on human-induced stresses are developed in wilderness areas, it is unlikely that wilderness and similar large protected areas will realize their potential as a "base datum of normality . . . a laboratory for the

study of land-health" (Leopold 1949). The understanding of natural ecosystems and their variability that would accrue from research programs in protected areas should also contribute substantially to the basic foundation of ecological science.

Our goal in this paper is to present information and ideas that can help in the development of ecological research programs in wilderness. We begin by defining terminology and suggesting an organizational framework based on the concept of threats. To assist with prioritization of research needs, we advance criteria for evaluating the relative significance of these threats and the impacts they cause. We selectively review current literature on the most significant threats and impacts and conclude with a discussion of three arenas of research, of interest to ecologists, that are critical to the long-term protection of wilderness.

TERMINOLOGY AND ORGANIZATIONAL FRAMEWORK

This paper is organized around potential threats and their impacts on ecological attributes of wilderness. We define threats as modern human activities or consequences of those activities that have changed, or have the potential to change, wilderness conditions. Attributes are components of wilderness systems that are subject to change. Impacts are the changes in attributes caused by a threat.

Machlis and Tichnell (1985) define threats as "stresses perceived to have detrimental impacts upon valued components of . . . ecosystems," pointing out that their conception of a threat is highly anthropo-

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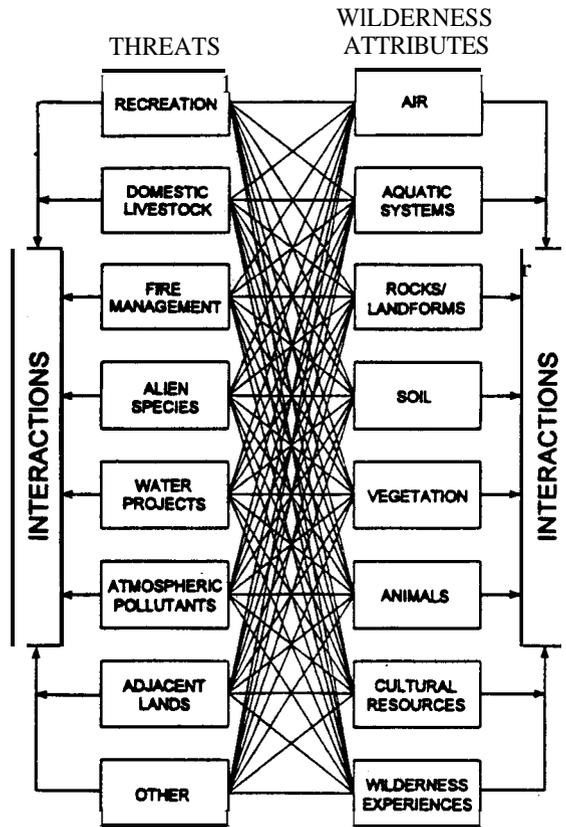


FIG. 1. Linkages between threats and wilderness attributes.

centric. We believe that the subjective difficulties inherent in defining threats are less pronounced in wilderness. The Wilderness Act states that the goal of wilderness designation is "to assure that an increasing population, accompanied by expanding settlement and growing mechanization, does not occupy and modify all areas . . . leaving no lands designated for preservation and protection in their natural condition." In wilderness, all "natural" components of ecosystems are valued. Therefore, we consider all modern human activities with the potential to cause deviations from "natural conditions" to be threats and all such deviations to be detrimental impacts.

Numerous lists of threats to parks and natural ecosystems have been generated. In Machlis and Tichnell's (1985) survey of parks worldwide, a total of 1611 different threats were reported. However, this list of threats included natural processes, inadequate management response to other threats (e.g., inadequate funding and personnel), and secondary effects (e.g., erosion was identified as a threat, although it was originally caused by livestock grazing). We believe that the vast majority of impacts to wilderness ecosystems can be traced to seven specific human activities: (1) recreational use and its management; (2) livestock grazing and its management; (3) fire management; (4) introduction and invasion of alien species; (5) diversion and

impoundment of water; (6) emission of atmospheric pollutants; and (7) management of adjacent lands.

Although it is convenient to describe the effects of individual threats on wilderness, threats usually act in combination, and the cumulative effects of multiple threats are often synergistic rather than additive. Thus, impacts on natural ecosystems will often exceed predictions based on analyses of the individual effects of several threats. For example, livestock grazing reduces grass cover and therefore the frequency of low intensity fires, exacerbating the effects of fire suppression (Madany and West 1983). Climate change, caused by the cumulative effect of disparate activities such as pollutant emissions and deforestation, may ultimately be the most pervasive and persistent human-induced impact on wilderness (Peters and Lovejoy 1992).

Threats and attributes are linked in a complex manner (Fig. 1). Each threat impacts a variety of different attributes and each attribute is affected by various threats. Eventually, we need to understand the impacts of each significant threat on a variety of wilderness attributes. The categorization of attributes depicted in Fig. 1 emphasizes the wide range of scientific disciplines that need to contribute to wilderness research, including many subdisciplines of ecology as well as physical and social sciences. Concern for impacts on the wilderness experience or "psychological wilderness" is important, but beyond the scope of this paper. Other categorizations of attributes, based on distinctions between the composition, structure, and function of ecological entities or levels of biological hierarchy (e.g., genes, populations, communities, ecosystems, and landscapes), are also possible, depending on the purposes of investigation.

CRITERIA FOR EVALUATING SIGNIFICANCE

All changes in wilderness conditions caused by human activities are undesirable, but some are of more concern than others. We propose that the ecological significance of an impact is a function of both impact and attribute characteristics (Fig. 2). Impacts vary in their areal extent, longevity, and intensity. The significance of an impact increases as any of these characteristics increase. The significance of an impact is also determined by the rarity or irreplaceability of the affected attribute. For example, trampling can eliminate

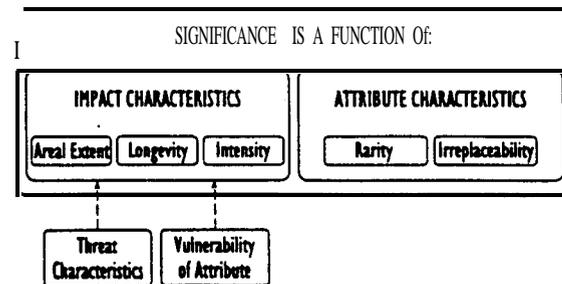


FIG. 2. Criteria that define the significance of an impact,

all individuals of a plant population. This is a locally intense impact, but it is not highly significant if the species is neither rare nor functionally irreplaceable. In contrast, if the population was the last in the region (rare) or if the plants were the only food source for another species (irreplaceable), the same impact would be more significant.

The intensity, longevity, and areal extent of impacts are determined by threat characteristics (intensity, areal extent, frequency, timing, predictability, and others) and the vulnerability (resistance, resilience) of the affected attribute (Fig. 2). Therefore, the significance of an impact is ultimately determined by characteristics of the threat and attribute. We draw tentative conclusions about which threats are likely to cause the most significant impacts and which attributes are most at risk.

THREATS TO WILDERNESS

In the discussions of individual threats that follow, we selectively review current knowledge about the most significant impacts, those that are most intense, most widespread, and most long lasting. We also highlight the resistance and resilience of ecological attributes and attempt to generalize about their relative vulnerability. High priority research needs are also identified.

Recreation use and its management

Recreational use is the most obvious, well-known, and most intensively managed threat to wilderness and parks. Total wilderness recreation use has increased about 10-fold in the past 40 yr. By 1992, total wilderness recreation use probably exceeded 18×10^6 visitor-days (a 12-h stay by one person). In mountainous parks and wilderness, the rate of increase in visitation has declined during the past two decades. In desert regions and Alaska, however, visitation has increased greatly in recent years, as has visitation during nontraditional use seasons, particularly winter visitation to mountain wilderness.

The primary ecological impacts of recreation are: (1) physical site alteration and disturbance of biota by trampling of humans and packstock; (2) the removal and redistribution of materials by packstock grazing and the collection and burning of wood in campfires; (3) disturbance of native animals by human presence and the importation of foreign substances, particularly food; (4) harvesting of animals and plants; and (5) pollution of waters by human waste and foreign substances. Intentional alteration by management is also commonplace. Construction of trails, campsites, and administrative facilities alters physical sites. Stocking of fish alters lakes and streams (Luecke 1990). Moreover, because so many wilderness areas are located at high elevations or in the desert, they are naturally stressed ecosystems that are not highly resilient.

Effects of wilderness recreation on most levels of

biological organization (i.e., genes, populations, communities, and ecosystems) have been documented. The relatively extensive recreation impact literature specific to parks and wilderness is reviewed by Cole (1987) and Knight and Cole (1995). Recreation impacts on soil and vegetation conditions generally are intense and persistent. For example, hiking and camping abrade vegetation and organic horizons and compact mineral soils (Cole 1987). Resultant impacts on soil structure (e.g., loss of macropores) and chemistry (e.g., reduced O_2) alter biological composition and function, but are poorly understood. Biotic-abiotic, plant-soil interactions are characterized by strong positive feedback linkages (Perry et al. 1989). When these linkages are disrupted, systems rapidly change in structure. Change can be irreversible or recovery can be slow, even when recreational disturbances are removed. Fortunately, these highly disruptive disturbances are also highly localized. In a heavily used portion of the Eagle Cap Wilderness, Oregon, Cole (1981) estimated that no more than 2% of the area had been altered by recreation use.

In contrast to impacts on vegetation and soil, the effects of recreation on aquatic ecosystems and vagile animals are often more extensive, even though point sources of impact remain highly localized. For example, inadequate disposal of human waste has been implicated in the spread of water-borne intestinal parasites (*Giardia* spp.), even in watersheds that receive little recreation use (Suk et al. 1987). In the Sierra Nevada, Verner and Ritter (1983) found that Brown-headed Cowbirds (*Molothrus ater*) were positively associated with recreational packstock stations. Given the ability of cowbirds to disperse widely, populations of songbirds that are subject to cowbird parasitism may be adversely affected even in places that packstock never reach.

Where the impacts of recreation use are highly localized, the most significant ecological impacts are likely to be those that affect rare species and assemblages. There are a few situations where endangered plant species are known to be threatened by recreational trampling. For example, Robbin's cinquefoil (*Potentilla robbinsiana*) populations have been extensively disturbed along popular hiking routes in the White Mountains of New Hampshire and Missouri bladderpod (*Lesquerella filiformis*) is threatened by recreational trampling of limestone glades in Missouri (Thomas and Willson 1992). Where packstock use is heavy and most existing meadows are grazed, meadow community types may be altered over their entire range. Finally, aquatic systems typically occupy a small proportion of the land surface and are highly attractive to certain uses. In some wilderness areas, recreation use, livestock trampling, associated pollution, and the introduction of fish may have altered essentially all lakes and water bodies. None of these situations has been

studied in sufficient detail to permit estimates of how prevalent or severe these problems are.

The recreational activities that have probably caused the greatest disruption of conditions detectable at large spatial scales are fishing, hunting, and the introduction and translocation of game fish and wildlife to improve fishing and hunting opportunities. For example, rainbow trout (*Salmo gairdneri*) planted in the upper reaches of the Colorado River have hybridized with native Colorado River cutthroat trout (*S. clarki pleuiticus*), substantially contaminating the genetic purity of the race (Behnke and Benson 1980). In addition to effects on genetic stocks, these activities have altered structural characteristics (ages, sexes, and sex ratios), behaviors, and distributions across entire regions.

Individuals, populations, species, communities, and landscapes all vary in their ability to resist being disturbed (resistance) and to recover from disturbance (resilience). Characteristics that contribute to resistance and resilience are best understood for stresses on plants caused by trampling. Plant characteristics that contribute to trampling resistance include short stature, a rosette, creeping, or caespitose growth form, flexible stems, and leaves that are small, thick, flexible, and able to fold under pressure (Cole 1987, Liddle 1991). Resilience is largely determined by the growth rate of individual plants and the extent to which perennating tissues are protected from disturbance (Cole 1995). Similar research is needed on the vulnerability of animals and of aquatic systems to recreational disturbance.

Most recreation research has focused on immediate impacts on aboveground, terrestrial systems. More research is needed on belowground processes, biotic-abiotic interactions linked by soil biota, and how disrupted processes and interactions can be repaired so that recreationally damaged sites can be restored. Research is needed on the long-term effects of both non-consumptive recreation (unintentional harassment) and consumptive recreation (hunting, fishing, introductions, stockings, and translocations) at large spatial scales. Short-term effects on individuals are obvious, but we know little about effects on populations or about the prevalence of cascading effects through food chains, energy flows, and nutrient cycles, effects that may only appear after a time lag. The difficulties of studying belowground processes and effects at large spatial and temporal scales will challenge recreational ecologists, much as they are challenging other ecologists: Finally, our understanding of recreational impacts on aquatic systems in wilderness is so rudimentary that a simple assessment of the prevalence and intensity of such impacts is a top research priority. Most recreational impacts are so poorly understood that effective indicators of impact have not been identified, and monitoring programs cannot be initiated.

Domestic livestock grazing

Contrary to popular assumption, domestic livestock grazing is generally permitted in wilderness if it was an established use prior to designation. Active sheep or cattle allotments are present in ~35% of designated wildernesses (Reed et al. 1989), mostly in the western United States. Even in many places where domestic livestock grazing is not permitted today, such as most national parks, heavy grazing often occurred in the late 19th and early 20th centuries (the "hooved locusts" of John Muir's Sierra Nevada), and packstock grazing continues today. The general effects of livestock on ecosystems are relatively well understood (Fleischner 1994, Noss and Cooperrider 1994). Most impacts derive ultimately from trampling, grazing, and defecation, as well as from various actions taken to manage livestock (e.g., fencing and water development). The results are direct impacts, such as defoliation, abrasion, and death of plants, compaction and destabilization of soils, and redistribution of nutrients, as well as indirect impacts such as changes in geomorphology, water characteristics, and animal populations. Direct contact between livestock and wildlife can also cause behavioral change in animals and transmission of disease.

Studies of grazing impacts within wilderness and parks are extremely limited, however. Effects of historic grazing, such as gully formation and lowering of water tables, changes in the composition of herbaceous vegetation, increases in the density of forested stands, and the expansion of trees into areas that formerly were treeless have been documented in wilderness and parks in California (Vale 1977, Vankat and Major 1978, DeBenedetti and Parsons 1979, Odion et al. 1990), Oregon (Reid et al. 1980), and Utah (Madany and West 1983). Beymer and Klopatek (1992) documented how current grazing in Grand Canyon National Park has detrimental effects on cryptogamic crusts and has caused shifts in herbaceous species composition. In the Sierra Nevada, heavy historic sheep grazing has also been blamed for reduced wildlife populations, resulting from both forage consumption and transmission of diseases, particularly to mountain sheep (*Ovid canadensis*) (Ratliff 1985).

Grazing has affected wilderness at many levels of biological organization, but to differing degrees. The most significant effects of grazing are probably manifested at the species and community-ecosystem levels. Although there are cases where grazing has substantially reduced the abundance of rare plant species, the most significant effects at the species level are probably indirect effects on animals (West 1993). For example, both the remaining current range of the endangered Gila trout (*Oncorhynchus gilae*), and the native habitat of the narrow endemic golden trout (*O. aguabonita*) lie primarily in wilderness areas with a long history of livestock grazing (Knapp and Dudley 1990, Propst et al. 1992). For *O. aguabonita*, Knapp and Dudley (1990)

found a positive correlation between growth rates of trout and amount of aquatic vegetation, amount of bank vegetation, and stream stability, attributes that are negatively affected by livestock grazing (Kaufmann and Krueger 1984).

Impacts on plants are probably most significant at the community level. An extreme example of historic grazing impact was the apparently irreversible conversion of perennial bunchgrass communities in California to annual grasslands, dominated by species that are not native to California (Vat&at and Major 1978). Numerous studies document changes in the function, structure, and composition of plant communities wherever grazing occurs (Milchunas and Lauenroth 1993). For instance, studies in arid and semiarid regions report that grazing has a greater effect on nonvascular plants that inhabit soil crusts than on vascular components of the community (Brotherson et al. 1983, Marble and Harper 1989). Shifts in relative abundance among both vascular and nonvascular species and growth forms occur. Losses of nonvascular plants may disproportionately alter nutrient cycling, given the ability of cyanobacteria, both free-living and as symbionts in lichens and mosses, to fix nitrogen (West 1990). Lesica and Shelly (1992) report a case where the survival of Sapphire rockcress (*Arabis fecunda*), a rare regional endemic in Montana, is enhanced on cryptogamic soil crusts. Grazing impacts may be most significant in riparian ecosystems of arid and semiarid areas because these ecosystems are unusually important biologically, relatively rare, and because concomitant geomorphological changes (such as gulying and lowering of water tables) suggest that impacts may be largely irreversible (Ohmart 1994).

Much is known about the resistance of species and growth forms to grazing disturbance (Rosenthal and Kotanen 1994), but our understanding of the resistance of communities and ecosystems is rudimentary. Milchunas and Lauenroth (1993) performed a meta-analysis of results from 236 sites worldwide and concluded that "within levels not considered abusive overgrazing, the geographical location where grazing occurs may be more important than how many animals are grazed or how intensively an area is grazed." They found that reductions in net primary production attributable to grazing were particularly pronounced in ecosystems without a long history of intense grazing pressure. Most grazing in wilderness and roadless areas occurs in the mountain and desert regions of the western United States, in communities that have not evolved with intensive grazing by native herbivores (Mack and Thompson 1982). This suggests that the effects of grazing in wilderness may be particularly severe, unless the intensity, timing, and distribution of grazing can be adequately controlled.

Although a substantial body of research on grazing impacts has developed, most of it is focused on the responses of individual species or community assem-

blages to various intensities and systems of grazing. The ecology of natural rangeland ecosystems remains poorly understood. Research is needed to improve our understanding of how natural ecosystems have changed in response to domestic livestock grazing. This work should also contribute to an understanding of how and why ecosystems vary in their sensitivity to grazing and how excessively altered sites might be restored. This is likely to require long-term, multidisciplinary research, including paleoecological studies. Despite the evidence that grazing substantially compromises wilderness values, it is unlikely that grazing will be discontinued in many wilderness areas (McClaran 1990). We can, however, attempt to identify those places where grazing is most inappropriate and develop grazing management objectives and guidelines that are more compatible with the goals of wilderness than the goal of maximizing sustainable animal production (the most common goal outside wilderness). We also must develop practical techniques for monitoring success at achieving and maintaining these objectives.

Fire management

Naturally ignited fire significantly affects the composition, structure, and functioning of many different types of ecosystems. Fire occurs relatively quickly, but the effects may last over long time spans, and influence spatial patterns of vegetation and the distribution of wildlife over large regions. Because fire disturbance is such a key process, the suppression of fire has profound effects, from altered species distributions and population viability to modified spatial patterns of habitat types and seral stages across a landscape. These effects have long been recognized (e.g., Weaver 1943), and prescribed fire is now considered necessary to maintain naturalness within wilderness (Parsons et al. 1986).

Fire regimes (location, intensity, frequency, size, and pattern) vary from high-frequency, low-intensity surface fires (e.g., in warm and dry ponderosa pine, *Pinus ponderosa*, forests) to low-frequency, high-intensity stand-replacing fires (e.g., in cool and moist western red cedar, *Thuja plicata*, forests) (Kilgore 1987, Kilgore and Heinselman 1990). The effects of fire suppression have so far been most pronounced in ecosystems with high-frequency fire regimes because technological advancements allowed active fire suppression only since the 1930s (Pyne 1987). It is more difficult to discern fire suppression impacts in ecosystems with low-frequency fire regimes (100 to >300 yr) because these systems are likely still within the bounds of natural variation.

One of the most visible effects of fire suppression has been on plant species composition and distribution pattern in ecosystems that typically experience high-frequency, low-intensity fires (reviewed by Kilgore 1987). With fire suppression, shade-intolerant species (e.g., ponderosa pine) are typically outcompeted by shade-tolerant species (e.g., Douglas fir, *Pseudotsuga*

menziesii). Changes in species composition, density, and vertical layering have additional ecological effects, such as creating nearly ideal habitat for western spruce budworm (*Choristoneura occidentalis*) outbreaks (Carlson et al. 1985). By altering vegetation, fire suppression changes the faunal composition of an area and the viability of species. Kirtland's Warbler (*Dendroica kirtlandii*), for example, nests only in fire-maintained jack pine (*Pinus banksiana*) woodlands that are between 6 and 21 yr old (reviewed in Botkin 1990). By altering vegetation composition and promoting the buildup of woody debris, fire suppression also affects several ecosystem-level processes in wilderness. Low-intensity surface fires, for example, volatilize few nutrients, generally increasing nutrient mobilization and soil fertility. In contrast, high-intensity fires driven by accumulated fuels volatilize many more nutrients, effectively removing them from the soil (Agee 1993). Barrett (1988) showed how fire suppression altered forest successional pathways in western coniferous forests, especially on cooler, moister north- and east-facing stands.

Fire suppression also affects the landscape-level distribution of vegetation patches and habitat types in ecosystems that experience stand replacing or crown fires. In these systems, fires typically do not move across a landscape uniformly because of variation in topography, moisture, temperature, wind, and fuels. A mosaic of vegetation patches remains after fire. Over time, the result is a complex mosaic of different seral stages, varying in the physiognomy and species composition of live vegetation and the quantity and characteristics of woody debris (e.g., see Turner et al. 1994a). Fire suppression in these landscapes freezes this vegetation mosaic (Bonnicksen and Stone 1982), while the age class structure of the forest shifts towards older stands (Vankat and Major 1978). Regional floral and faunal diversity is reduced as late seral stands increase in size and abundance, and the difficulty of restoring fire is increased as fuels increase in flammability and homogeneity. Although there is much circumstantial support for fire suppression reducing landscape heterogeneity (e.g., see Turner et al. 1994a, Turner and Romme 1994), empirical tests of these inferences that separate the impacts of fire suppression from other altered disturbances regimes (e.g., from insects and disease) are generally lacking.

Restoring natural processes that occur over large areas and hundreds of years will be a formidable management challenge for both ecological and political reasons. Restoring fire (whether natural- or management-ignited) in high-frequency, low-intensity systems that now contain large amounts of fuel may cause stand-replacing crown fires that bum the old, large trees within a forest. As protected areas increasingly provide the last refugium for certain species or communities, a single fire may cause the extirpation or extinction of a population, species, or community.

The more we learn about fire regimes in the distant past, the more difficult it becomes to define natural fire regimes. For example, Swetnam (1993) examining a 2000-yr record, concluded that "... giant sequoia (*Sequoiadendron giganteum*), fire regimes were clearly nonstationary; fire frequencies and sizes constantly changed through time." Romme (1982) reached a similar conclusion for the subalpine forests of Yellowstone National Park, as did Baker (1989) for the Boundary Waters Canoe Area. Politically, restoring fire will be difficult because of the risk to property and visitors both within wilderness as well as on adjacent lands, coupled with the notion that fire destroys a forest. Furthermore, smoke production from management-ignited fires may violate Clean Air Act standards and be obnoxious to people living adjacent to wilderness.

Although we know more about fire and the effects of its management than many other threats to wilderness conditions, most of this knowledge is at the level of individual stands. The importance of fire and its suppression demands that we expand this understanding to large spatial scales and long time frames and to synergistic interactions between fire and other disturbances. Critically needed are methods for accurately quantifying natural fire regimes over large areas (e.g., 10 000 ha and greater), and models for explaining and predicting spatial and temporal variation of fire regimes in different types of environments (e.g., see Clark 1990). We need to understand how physical attributes (e.g., slope, aspect, elevation), historical climate conditions, and past disturbances interact to influence current fire probabilities, and the influence of these probabilities on landscape-level vegetation patterns as they change over short- and long-term time frames. Fire regimes are typically described in terms of a "mean fire-return-interval," but this traditional descriptor does not adequately convey either temporal or spatial variation. More precise and accurate descriptors need to be developed, perhaps as probability functions for both spatial and temporal occurrence (Turner et al. 1994b). In addition, impacts of dynamic vegetation patterns on wildlife distribution, abundance, and viability are poorly understood, even for game species where considerable effort has already been concentrated. Research on the consequences of fire restoration following decades of fire suppression is just starting and needs to be greatly expanded. This knowledge will be crucial to managers charged with maintaining and restoring the process of fire in wilderness.

Introduction and invasion of alien species

Nonnative, or alien species pose a significant threat to wilderness and other protected lands by their direct and indirect impacts to native species, and by their effects on broader scale ecological patterns and processes. Impacts from aliens on indigenous plants and animals have been recognized since at least 1860

(Mack 1989); virtually every wilderness and other type of nature reserve has alien species.

Alien species affect all levels of biological organization. Abbott (1992) demonstrated interspecific hybridization between native and alien plants (in *Helianthus* and *Senecio* spp.), compromising the genetic integrity of the native species. Many studies found changes in the distribution or community composition of native species following invasion of alien pathogens, plants, invertebrates, and vertebrates (reviewed by Mooney and Drake 1986, Drake et al. 1989). Non-native species may cause dramatic effects that ripple throughout a community. Spencer et al. (1991), for example, showed how intentional stocking of a non-native shrimp caused a series of changes in lake trophic structure, eventually displacing nesting Bald Eagles (*Haliaeetus albicilla*). Unintentionally brought from Eurasia in 1910, white pine blister rust (*Cronartium ribicola*) is epidemic over most of the range of whitebark pine (*Pinus albicaulis*). Declining whitebark pine populations indirectly affect other species, such as grizzly bears, which feed extensively on whitebark pine seeds (Hoff and Hagle 1990, Mattson and Jonkel 1990). Alien species also affect ecosystem-level processes, such as nutrient cycling (reviewed by Vitousek 1990). Successional pathways, for example, are significantly altered by the alien musk thistle (*Carduus macrocephalus*) invading and dominating a pinon-juniper woodland in Mesa Verde National Park after a hot fire (Floyd-Hanna et al. 1993).

Syntheses on non-native species in nature reserves (Macdonald et al. 1987, Thomas 1988, Usher et al. 1988) suggest some general patterns in the magnitude of invasion by these species. First, oceanic island reserves have a higher number of alien vascular plant and terrestrial vertebrates than comparable mainland reserves (Macdonald et al. 1988, Usher 1988). Loope and Mueller-Dombois (1989) concluded that oceanic islands are "predisposed" to invasions because of their history of isolation from the mainland and a relatively small number of native species. Second, reserve size, at least in mediterranean-type ecosystems, is inversely related to the proportion of invasive alien vascular plants in the flora (Macdonald et al. 1988). And third, nature reserves in extreme climates (very hot, cold, or dry) have fewer alien species than reserves with more moderate climates (Loope et al. 1988, Macdonald et al. 1989). Comparable generalizations about the impacts of alien species are lacking.

Impacts of non-native species range in severity, from merely coexisting within the extant community, to displacing rare or common species and disrupting ecosystem functions. The invasion patterns discussed above suggest that wildernesses with a longer history of isolation, smaller size, surrounded by very different types of vegetation, and occurring in moderate climates (e.g., coastal, southern, southeastern areas, and lower altitudes of relatively high-elevation wilderness) will

be most susceptible to invasion and establishment of non-native vascular plants and terrestrial animals. Impacts to lentic and lotic systems are especially significant because of the relative rarity of these ecosystems in most wildernesses. For example, there is ample evidence that stocking non-native fish changes the abundance and distribution of phyto- and zooplankton, invertebrates, fish, and amphibians in lakes and streams (e.g., Courtenay and Moyle 1992, Chess et al. 1993). At present, our understanding of the interaction among invasion processes, invader life history attributes, and site conditions is insufficient to predict impacts or resistance (the inverse of invasibility) and resilience in native terrestrial or aquatic communities.

Other than intentional introductions of species such as livestock or fish, little is known about the vectors or mechanisms that bring alien species into wilderness. A variety of activities occurring outside protected lands may facilitate the spread of aliens into the protected area through typical wind, water, or animal dispersal (Saunders et al. 1991). These activities include soil disturbance, habitat fragmentation and alteration, building of roads, and introduction of species for erosion control and other purposes. Within a wilderness, disturbed areas created by people, packstock, or livestock often provide germination and establishment sites. In addition, these three vectors bring aliens into wilderness on clothing, hair, or in feces. For example, Tyser and Worley (1992) found large numbers of alien plants along trail and road corridors in Glacier National Park, and Macdonald et al. (1989) found significant correlations between the number of visitors to an area and the number of alien vascular plants in North American and African nature reserves.

Despite widespread occurrence and long-standing recognition of non-native species and their impacts, substantive basic and applied questions remain. When should an alien species be considered "naturalized" (Westman 1990, Lodge 1993), and when is eradication or control both necessary and feasible? For example, it is often suggested that aliens need to be eradicated when they replace native species or eliminate their habitat. But the omnipresence of aliens and their high eradication costs suggest that philosophical, social, and practical dimensions of this debate need to be melded with purely ecological considerations. Given the relative remoteness and naturalness of wilderness, what are the vectors and mechanisms by which alien species establish and spread throughout an area? Do models of spread adequately predict patterns of alien species' invasion within the context of environmental conditions in wilderness? Under what conditions do aliens induce long-term changes in ecosystems? And last, how can wilderness be managed to maintain natural disturbance regimes, or how can these regimes be restored, when disturbance increases the chance of surrounding aliens establishing in the protected area? Research within wilderness on these questions will provide ecologists basic

knowledge about invasion processes and impacts, and allow agencies to more effectively manage the onslaught of non-native species.

Water impoundments, diversions, and regulated flows

Wilderness ecosystems are affected by a variety of water projects, including dams, diversions, and even cloud seeding. Dams, reservoirs, or water conveyances (ditches, canals, pipelines) built prior to wilderness designation are present in about 15% of designated wildernesses (Reed et al. 1989). In most of these areas, dams were built at existing lakes to raise water levels and control the timing of runoff. Of even greater concern are the wildernesses and other protected areas that are affected by diversions or dams located beyond their boundaries but that control the hydrology of the supposedly protected areas located downstream. Fortunately, most wilderness watersheds contain their own headwaters and thus are minimally affected by water projects outside their own boundaries. However, Everglades National Park, (Kushlan 1987, Davis and Ogden 1994) and Grand Canyon National Park are examples of parks that have been highly altered by dams and diversions. Moreover, external water diversions and regulated flows will have a greater effect on designated wilderness in the future as more downstream areas are designated as wilderness, particularly on Bureau of Land Management lands. One portent of this increasing concern is the current controversy over federal reserved water rights for wilderness (Marks 1987).

The impacts of water projects within wilderness are most significant where fluvial systems are dammed and where dam operations result in extreme maximum and minimum flows. The creation of a reservoir represents a complete alteration of existing ecosystems. Lotic, riparian, and adjacent upland ecosystems are extinguished locally and replaced by lacustrine ecosystems, new riparian ecosystems, and wetlands associated with deltas that form where tributaries enter the reservoir. The potential effects of diverting low-order tributary streams, such as those within most wildernesses, are illustrated by several studies conducted adjacent to a wilderness in the eastern Sierra Nevada (Smith et al. 1991, Stromberg and Patten 1992). These studies suggest that low-flow conditions have significantly altered various attributes of riparian vegetation, including physiology (reduced stomatal conductance and water potential), morphology (smaller, thicker leaves and reduced total leaf area), population structure (fewer very old and large black cottonwood trees, greater mortality of cottonwood trees), and vegetation structure (lower canopy foliage density and tree density).

Mainstem dams generally alter the temperature, sediment load, and flow regime of the rivers they impound. This can have dramatic effects on lotic ecosystems in downstream protected areas. The lotic ecosystem of the Colorado River in Grand Canyon National Park, for

instance, has been so highly altered by an upstream dam that Johnson and Carothers (1987) consider it an exotic ecosystem. Nineteen introduced fish species have been recorded in the Colorado River in Grand Canyon, while four of the eight original natives are locally extinct and a fifth only breeds at the mouth of one tributary. However, the intensity of impact caused by the upstream dam declines rapidly with distance from the river. Ecosystems located just 50 m from the present high-water level have been virtually unaffected by the upstream dam. The riparian ecosystem, adjacent to and influenced by the lotic system, is now a mix of alien and indigenous organisms and processes. Alien species have been introduced and community structure and composition have changed. Riparian vegetation has colonized remnant floodplain deposits that were formerly kept bare by periodic scouring, while dense woody vegetation growing upslope on predam flood terraces is now apparently senescent. However, the dam-related changes have caused no known species extirpations, and they have increased the diversity and abundance of native plants and vertebrates.

Mainstem dams also disrupt the movement of migratory fish. Consequently, wildernesses with substantial populations of migratory fish can be affected by downstream dams as well as upstream dams. These effects are probably most pronounced in areas within the basin of the frequently dammed Columbia River.

The most significant threat to wilderness comes from water projects that are external to wilderness and that operate with little consideration of effects on natural ecosystems. Although effects may not be areally extensive, the aquatic and riparian systems that are affected are often rare and of high ecological value, particularly in arid regions. One critical applied question that needs to be addressed is if and how dam operations and resultant flow levels, temperatures, and sediment loads can be manipulated to minimize downstream impacts. A related question concerns minimum instream flow levels and the temporal distribution of flows needed to maintain the geomorphological and biological integrity of riparian and lotic systems.

The magnitude of the changes that occur when predam lotic and riparian natural systems are converted to postdam exotic and naturalized systems places managers of protected areas in a difficult position. Given the degree to which preservation objectives have been irretrievably lost, it is not obvious what the objectives for these novel ecosystems should be. It is tempting to conclude that changes that increase species richness or that increase the abundance of a rare species or community are desirable and to develop management programs that attempt to maintain such conditions. However, even "desirable" changes represent a divergence from predisturbance conditions and are at odds with the original intent inherent to wilderness or protected area designation. Ultimately, social and political considerations will determine the objectives for highly al-

tered ecosystems in wilderness and other protected areas, but ecological science can contribute to more informed decisions. Science can sharpen thinking about the trade-offs involved and the long-term consequences of alternative courses of action.

Emission of air pollutants

Photochemical oxidants, heavy metals, and acid deposition are major classes of pollutants produced by both point and nonpoint sources, and are readily transported by local winds. These pollutants affect virtually all wildernesses throughout the conterminous United States, leading Schrieber and Newman (1987) to conclude that "maintaining the pristine condition of air quality in wildernesses may be impossible."

Air pollutants cause ecological impacts across all levels of biological organization. A vast literature exists on these impacts to terrestrial and aquatic ecosystems (at least 37 books published since 1980). These impacts were reviewed for fish (Haines 1981), lakes (Schindler 1988), wildlife (Schrieber and Newman 1988), soil microflora (Fritz 1992), and forests (Taylor et al. 1994). Guidelines for evaluating air pollutant impacts to physical, chemical, and biological conditions within wilderness were developed by Fox et al. (1987, 1989), and specific indicators and standards developed for Pacific Northwest (J. Peterson et al. 1992) and California (D. Peterson et al. 1992) wildernesses.

Air pollution impacts to terrestrial and aquatic resources within wilderness are summarized by Schrieber and Newman (1987) Grigal (1988), Schofield (1988), and Blankenship (1990). In the western United States, nonpoint sources of ozone, nitrogen, and sulfur pollutants from urban areas pose the greatest threat to wilderness. For example, most wildernesses along the Cascade and Sierra Nevada mountains are downwind from metropolitan areas that are continual sources of pollutants. In these areas, several tree species (e.g., *Pinus ponderosa*, *P. jefferyi*) and lichens show visible injury, reduced photosynthetic capacity, premature leaf senescence, reduced growth, and increased susceptibility to pathogens, all caused by air pollutants (Sigal and Nash 1983, D. Peterson et al. 1992). These areas also have little buffering capacity in their soils (D. Peterson et al. 1992, J. Peterson et al. 1992) suggesting that aquatic ecosystems would be very susceptible to impacts of acid deposition. However, of 455 lakes sampled for acidity in western wildernesses by the Environmental Protection Agency (EPA) during 1984 and 1985, only 1 had a pH <5, and this lake was fed by a sulfur hot spring (Blankenship 1990). So far, it appears that aquatic ecosystems in the western United States have escaped significant injury from air pollutants. However, the EPA did not sample any lakes in southern California wildernesses such as the San Gabriel or San Geronio, which are subject to high levels of air pollutants.

Wildernesses in the eastern United States are thought

to be most influenced by point sources of pollution (Schrieber and Newman 1987), which cause extensive impacts to terrestrial and aquatic plants and animals (Hutchinson and Meema 1987). In addition to point sources, metropolitan areas and heavy industry produce ample nonpoint pollution as well. For example, Bennett (1985) found extensive vegetation damage from nonpoint source ozone and sulfur dioxide in 10 national parks, while Armentano and Loucks (1983) found ozone and acid-forming pollutants in most of the national parks throughout the Great Lakes region.

Because of their relative remoteness, wildernesses are most likely affected by chronic, rather than acute air pollution. Moreover, a variety of air pollutants combine to form interacting, or synergistic, stresses. Enhanced ultraviolet-B radiation (especially at higher elevations), for example, combined with acidic deposition is a possible cause of amphibian population declines (Blaustein et al. 1994). These chronic stresses cause general loss of vigor and reproductive capacity, and increase susceptibility to disease or pathogens in many plants and animals. These impacts may be especially severe on rare species and populations that are declining or are at the margins of their distribution. Carey (1993) hypothesized that acid precipitation, working synergistically with other environmental changes, caused immunosuppression in declining boreal toad (*Bufo boreas boreas*) populations in the West Elk Wilderness in Colorado, leading to their extirpation. Other critical impacts resulting from air pollutants in wilderness include altered nutrient cycling (Raloff 1995) and changes in primary productivity caused by changes in soil pH, soil microflora activity, and leaching rates of nutrient ions (Blankenship 1990, Clayton et al. 1991). And air pollutant impacts to aquatic ecosystems may be especially significant because of the general rarity of these habitats in wilderness.

Air pollutant impacts are especially significant in Class I areas, where federal land managers have "... an affirmative responsibility to protect air quality related values" as mandated by the Clean Air Act as amended in 1977 (Public Law 95-95). Class I status confers the highest level of air quality protection in the United States. These areas consist of wildernesses that were >2024 ha when the Clean Air Act was passed in 1977 (~12% of the area in the present National Wilderness Preservation System). All wildernesses designated after this date are accorded Class II status, allowing greater deterioration of air quality. In a survey of personnel managing Class I areas across the United States, Schrieber and Newman (1987) found that 78% of these areas had "identifiable" influences from air pollution.

The most significant and long-lasting effects of air pollutants in wilderness are likely from impacts to soil- and water-based processes. We need a better understanding of the impacts to processes such as decomposition and nutrient turnover and availability in wil-

derness environments, especially in poorly buffered soils. This information would be crucial to evaluating air pollutant impacts to primary productivity and the factors affecting ecosystem resistance and resilience. Research on indicators of air pollutant impacts on communities and ecosystems is critically needed, especially for aquatic ecosystems and amphibians. Basic monitoring data on pollutant levels are lacking in many areas, requiring extrapolation and inference with unknown levels of precision and accuracy. It is also uncertain how air quality data in remote locations, where mechanized transportation is prohibited, can be collected without compromising wilderness values. For example, the Bureau of Land Management developed a remote air monitoring station that can be horse packed into an area, but when assembled is ~10 m high and is considered an unacceptable intrusion by some people.

Management of adjacent lands

All wilderness and other protected areas are surrounded by lands that are more intensively developed and used for human purposes such as timber cutting, mining, livestock grazing, home building, and mechanized recreation. Wildernesses are vulnerable to impacts from adjacent lands because they are "open systems," as are all ecosystems, affected by the flow of biological, chemical, and physical matter and energy into and out of the area. Because of this "openness" the mosaic of surrounding lands and their condition influences the composition, structure, and functions within a protected area. As our human population continues to grow, wilderness will become progressively more insular and likely to be altered by adjacent lands.

Isolation of nature reserves has been discussed for nearly two decades (e.g., Diamond 1975). spawning numerous debates and a large literature (reviewed by Saunders et al. 1991). Isolation effects observed in nature reserves include "relaxation" or loss of species (Miller and Harris 1977. Soule et al. 1979. Burkey 1995). rippling impacts caused by the loss of key species (Bierregaard et al. 1992). increased fluctuations in population density leading to increased probabilities of extirpation (Soule et al. 1988). loss of genetic variation (Frankel and Soule 1981). and increased invasibility from alien species (Janzen 1983). These isolation effects may largely be due to altered ecological flows either into or out of wilderness, both with adverse consequences.

Activities on adjacent lands that increase harmful flows or decrease beneficial flows into wilderness are equally detrimental (Janzen 1986). Increased harmful flows into wilderness come from many activities. Adjacent mining or other extractive industry, for example, increases the likelihood of air and water pollutants moving into wilderness (Stottlemeyer 1987). Water pollutants from agricultural fields have been carried into

wildlife refuges causing severe declines in bird populations. Predators and parasites from adjacent lands can also spread into projected areas. Andren and Angelstam (1988) found significantly more predation on experimental nests that were within 50 m of the forest edge, while Brown-headed Cowbirds (*Molothrus ater*) can fly up to nearly 7 km to parasitize nests (Rothstein et al. 1984). Domestic and feral livestock easily roam into wilderness. sometimes with disastrous impacts to native flora and fauna, as shown by the impacts caused by feral pigs in Hawaiian national parks (Stone and hope 1987).

De-created beneficial or required ecological flows into protected areas may cause subtle yet widespread and long-lasting impacts. Withdrawing water for irrigation and power generation severely disrupts hydrologic flow regimes in the Everglades and Grand Canyon national parks, causing profound changes in vegetation and wildlife communities (Johnson and Carothers 1987, Kushlan 1987). Fire suppression on lands actively managed for timber production alters the pattern of naturally occurring fire spread into protected areas (Kilgore and Heinselman 1990). In the Selway-Bitterroot Wilderness, for example, Habeck (1985) found that fire suppression in surrounding warm and dry (i.e., short fire return-interval) lands increased fuels, increasing the probability of fire spreading into and burning cool-moist (i.e., long fire return-interval) western red cedar (*Thuja plicata*) forests. Population viability of species within wilderness is threatened by reduced immigration of individuals and genetic variation from surrounding areas that have been modified. Fahrig and Merriam (1994) reviewed impacts of altered landscape structure on population viability, concluding that immigration is more important than demographics in affecting regional population abundance.

Beneficial ecological flows out of wilderness may be restricted or adversely affected by activities on adjacent lands. For example, several ungulate species seasonally migrate out of wilderness into surrounding lower elevations. Activities on adjacent lands that modify critical habitat attributes needed by elk (*Cervus elaphus*) during winter may cause population reductions (Lyon and Ward 1982). Likewise, detrimental flows out of wilderness may be enhanced. Dispersal of animals out of protected areas to prey on livestock or travel along roads often results in their death, shown for wolves (Mech et al. 1988) and several other predators and large vertebrates. These increased detrimental flows out of wilderness may contribute to reductions in density and viability of populations residing within wilderness.

Adjacent lands will always influence protected areas, but some of these effects may be mitigated by increasing the size of protected areas and improving the location of administrative boundaries. Ideally, protected areas would be sufficiently large to encompass the full range of disturbance regimes and natural flows into and out of the area (Schonewald-Cox and Bayless 1986,

White 1987). In reality, most protected areas are smaller than the "minimum dynamic area" (Pickett and Thompson 1978) needed to maintain recolonization sources and minimize extirpation of species. Increasing the size of protected areas, establishing buffer zones around them, or linking them together may mitigate some of these isolation effects, but all these options are politically difficult, and the conservation benefits of landscape linkages are still debated (Saunders et al. 1991, Simberloff et al. 1992). Improving the location of administrative boundaries by placing them, for example, along biologically meaningful landscape breaks, may also reduce isolation effects. Theberge (1989), for example, offers 15 guidelines for drawing "ecologically sound boundaries" for protected areas. However, any administrative boundary, even in large reserves (e.g., > 100 000 ha) will likely cross or dissect some natural disturbance paths, sources of recolonization, and migration or dispersal routes. Each of these actions discussed above may ameliorate, but would not likely eliminate isolation effects.

New and continuing activities on adjacent lands provide both reason and opportunity to design experimental treatments and monitoring programs to quantify the mechanisms and impacts of ecological isolation. The most important experiments would increase understanding of how flows of plants, animals, and disturbances sustain wilderness character (Romme and Knight 1982, Noss 1983). Specific research topics include studies on: (1) evaluating the conservation effectiveness of corridors and landscape linkages (e.g., movement of animals, plants, aliens, pathogens, pests, and disturbances within a corridor; impacts of edge effects on movement; ecological cost-benefit analyses of linkages among protected areas vs. increasing reserve size); (2) understanding the mechanisms of altered "permeability" or flow into and out of protected areas caused by administrative boundaries and the disparate activities on either side of these boundaries; (3) quantifying the attributes of wilderness ecosystems (such as size and degree of contrast across boundaries) associated with a landscape's minimum dynamic area and levels of resistance and resilience in the face of adjacent land impacts; and (4) evaluating the effects of landscape-level vegetation patterns on disturbance regimes and population viability in ecosystems that exhibit high levels of natural fragmentation (e.g., see Andren 1994), such as occur in coniferous forests throughout much of the U.S. Rocky Mountain and Intermountain regions.

Other threats

A number of other modern human activities have affected natural ecosystems in wilderness, including mining, gas and oil drilling, poaching, subsistence hunting and gathering, aerial overflights, and attempts to control insects and disease. Of these, mining has probably caused the most intense but localized impacts,

particularly on aquatic systems (e.g., Deniseger et al. 1986). There are active mining claims in about 9% of wilderness areas (Reed et al. 1989), and many additional areas remain affected by historic mining. Other historic impacts in wilderness include the effects of floating logs for railroad ties down streams on channel morphology and riparian vegetation (Young et al. 1994).

Another perceived impact, particularly germane to this paper, is that resulting from scientific use of wilderness (Franklin 1987, Parsons and Graber 1990). The process of data collection in wilderness often requires the use of permanent markers, mechanized equipment, or destructive sampling, techniques that can compromise wilderness conditions, particularly the psychological wilderness. In the past, managers have frequently not permitted worthwhile research in wilderness on the grounds that it compromises wilderness. We hope wilderness managers will more often recognize situations where the long-term benefits of wilderness science exceed the short-term impacts.

DISCUSSION

The prevalence of large areas with a high degree of integrity and continuity means that wilderness harbors substantial information of benefit to science and society (Graber 1985, Noss 1991). In particular, information contained in wilderness can contribute to understanding how natural ecosystems operate, to understanding landscape processes, and to assessing global change. This information is crucial to improve management of lands inside wilderness, as well as management of lands outside wilderness. Moreover, the value of this information will increase greatly in the future, as the last of the lands not specifically devoted to nature preservation are developed and modified.

All seven of the threats we describe have already had a significant impact on wilderness ecosystems, and all levels of biological organization have been significantly affected by at least some of these threats. Fish introductions, for example, have reduced genetic diversity; dam construction has reduced native species diversity; and fire suppression has reduced the structural diversity of landscapes.

Threats and the impacts they cause vary in both intensity and areal extent. Some threats, such as recreational use and river impoundments, cause locally extreme changes in system structure and function. Air pollution, in contrast, has had extremely widespread but less intense effects. The threats that have probably caused the most extensive and intensive impacts are domestic livestock grazing, fire suppression, the introduction of alien species, and adjacent land practices. Not all wilderness lands are equally vulnerable to these threats, however. For each of these threats, certain wilderness areas are virtually unaffected, as are certain ecosystem types within affected wilderness areas. Domestic livestock grazing has probably been most dis-

TABLE 1. A basic agenda for research on threats to wilderness ecological systems.

Evaluation
1) Define and describe natural conditions.
2) Determine methods for assessing deviation from natural conditions.
Protection
3) Describe anthropogenic impacts to natural conditions and the significance of these impacts.
4) Determine how characteristics of threats and attributes influence the intensity, longevity, and areal extent of impact.
5) Determine the effectiveness of alternative management strategies in avoiding impacts.
Restoration
6) Determine how to manipulate ecosystems to restore them to natural conditions.

ruptive in riparian ecosystems in the arid southwest (Fleischner 1994), while it is currently of little concern in National Park Service wildernesses or outside of rangeland ecosystems in the western United States. Fire suppression effects have been most severe in ecosystems with fire regimes characterized by high-frequency, low-intensity fires. Alien species incursions have been particularly significant at lower elevations, on islands, and in freshwater aquatic systems. Adjacent land practices probably have had the greatest impact on small wilderness areas, particularly those in more densely populated and industrialized parts of the country.

A theme common to many individual threats is that impacts to aquatic and riparian ecosystems in wilderness are particularly significant. The most intense types of impact, such as recreation use, domestic livestock grazing, and water impoundment are concentrated in these ecosystems. Moreover, riparian systems often harbor an exceptional proportion of the regional biotic diversity. In many regions, particularly the arid Southwest, most of the riparian habitat outside of protected areas has already been substantially impoverished.

Many of the wildernesses most at risk are small wildernesses, particularly those at low elevations and located in the eastern United States. In these small protected areas, disturbance regimes are unlikely to be intact, and the effects of adjacent land management practices and land fragmentation are likely to be severe. Vulnerability to alien invasions is high, particularly in low elevation wildernesses, and the concentrations of air pollutants and recreation use are high in areas close to urban centers.

If natural conditions continue to erode in the face of anthropogenic threats, wilderness will be protected in name only. We challenge ecologists to more actively participate in developing the knowledge needed to protect natural conditions in designated wilderness. This knowledge is also relevant to the theoretical interests of ecologists (Huenneke 1995). We suggest the following basic agenda for ecological research in wilderness, organized around the three goals of evaluation, protection, and restoration of wilderness conditions (Table 1).

Research is needed to help managers evaluate the extent to which wilderness conditions deviate from natural conditions or some other desired state. Much of this research must be predicated on a better understand-

ing of natural conditions. Philosophical questions must be resolved about how to define naturalness (Anderson 1991). Studies of past and present ecological conditions must be conducted to improve our ability to describe natural conditions. The spatial and temporal variation of natural systems suggests the need to describe natural conditions in probabilistic and variable rather than deterministic and static terms. The need to describe natural or desired conditions at large spatial and temporal scales is particularly challenging. Managers also need practical indicators and techniques for assessing and monitoring deviation from desired or natural conditions. Ecological research can help specify which system components to assess, at which spatial and temporal scales. Research can also help managers make trade-offs between the practical advantages of monitoring indicator or keystone species and the shortcomings of this approach (Landres et al. 1988, Mills et al. 1993).

Research is also needed to help managers protect wilderness systems from further impact. More studies are needed on the effects of human activities on wilderness attributes. Fundamental questions remain about synergistic relationships between different threats and cumulative impacts to multiple attributes. A critical need is to better understand impacts on ecosystem processes and impacts occurring at large spatial and temporal scales. This research should improve our assessment of the significance of different impacts to wilderness systems. More research is needed on how variation in disturbance characteristics and ecosystem resistance and resilience influence the intensity, longevity, and areal extent of impacts. This understanding should suggest strategies for mitigating impacts. Once implemented, these management strategies are large-scale experiments that need to be evaluated and modified accordingly (Underwood 1995).

Finally, because all wildernesses have already been compromised to some extent, research is needed to help managers restore natural conditions and processes. Restoration may simply involve removal of a disturbance agent; however, in many cases it will require active manipulation of ecosystems. Active restoration has been called the "acid test" of our ecological understanding (Bradshaw 1987). For restoration to be successful, ecological understanding must be sufficient to identify both how ecosystem function has been ad-

versely affected and how proper function can be reestablished. Ecological research is needed to guide restorations at all scales from small sites to disturbance regimes that operate over areas larger than entire wildernesses. Particularly where restoration involves intentional manipulation of ecosystem linkages and processes, basic research is needed to complement more applied studies that evaluate the success of restorations.

Both philosophical and practical concerns are raised when restoration involves intentional manipulation within wilderness ecosystems. Attempts to restore one ecosystem process or component may exacerbate impacts to other components, and restoration goals may conflict with one another. For example, attempts to restore fire may increase vulnerability to invasions by alien plants. In other situations, attempts to restore ecosystem processes within an area can compromise attempts to conserve biological diversity at larger spatial scales. For example, in response to the establishment of alien salt-cedar (*Tamarix chinensis*) riparian vegetation along the Colorado River in Grand Canyon, several rare riparian birds, such as Bell's Vireo (*Vireo bellii*), have expanded their distribution into the canyon where they had not occurred prior to Grand Canyon Dam (Johnson and Carothers 1987). Attempts to eradicate alien salt-cedar and restore predam flow regimes would reduce the presence of these rare birds in the region. Managers of protected areas will increasingly face such trade-offs in their attempts at restoring natural conditions.

Restoration also raises the issue of the appropriateness of intentional manipulation of wilderness conditions. A primary scientific value of wilderness ecosystems is their utility as reference areas: for comparison with more intensely managed landscapes and for discerning impacts from subtle and long-term threats such as global climate change. This value is compromised when these ecosystems are intentionally modified, even for the purpose of restoring natural conditions and processes. Increasingly, wilderness managers will have to choose between two undesirable courses: permitting ecosystems to deviate further from their natural condition or manipulating ecosystems to match our image of those natural conditions and the processes that shaped them. Perhaps the ultimate challenge to ecologists is to develop the knowledge needed to decide when and how to intentionally manipulate ecosystems to compensate for the unintentional effects of anthropogenic disturbance.

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