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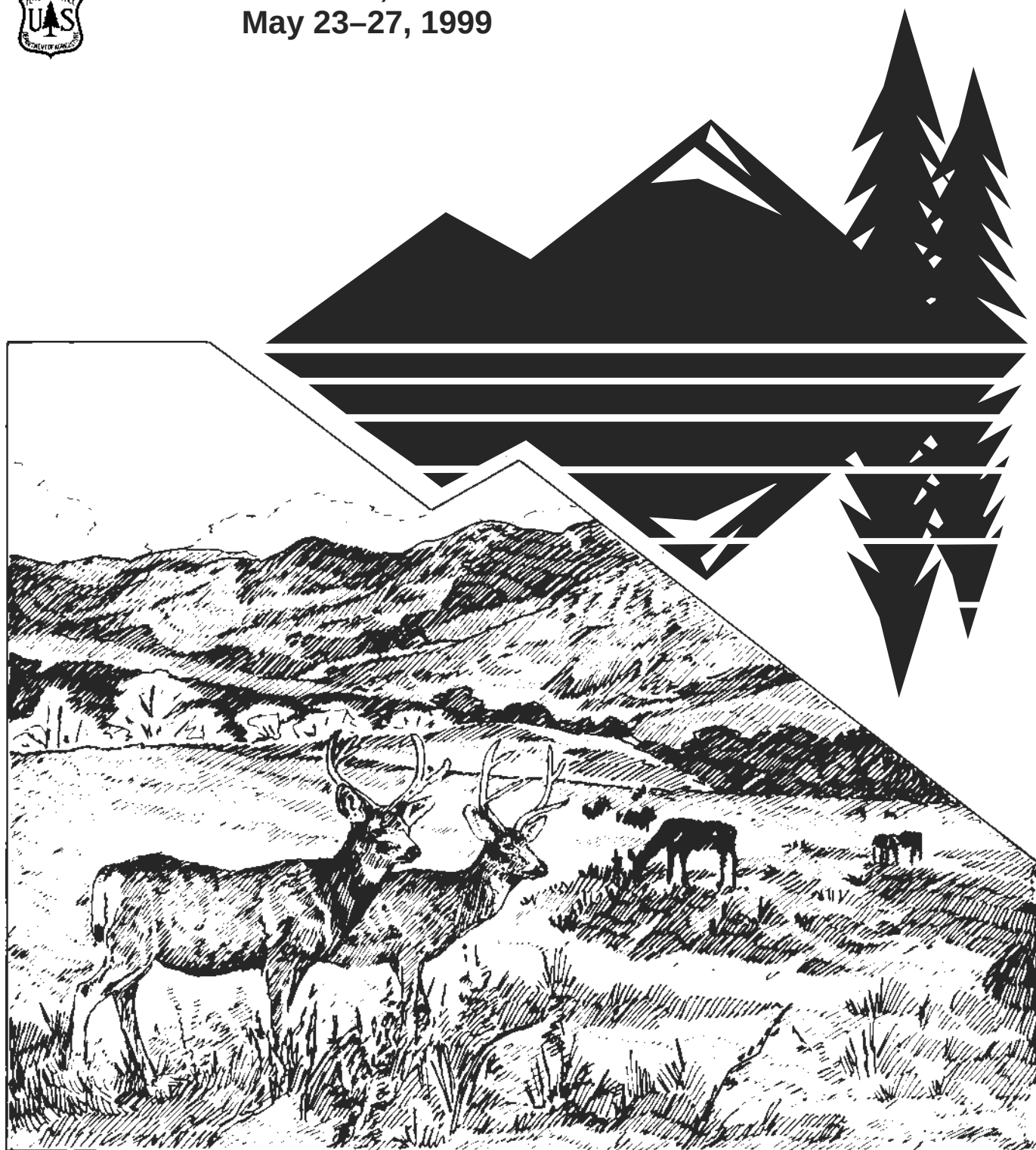
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Wilderness Science in a Time of Change Conference

Volume 5: Wilderness Ecosystems, Threats, and Management

Missoula, Montana
May 23–27, 1999



Abstract

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Forty-six papers are presented on the nature and management of threats to wilderness ecosystems. Five overview papers synthesize knowledge and research on wilderness fire, recreation impacts, livestock in wilderness, nonnative invasive plants, and wilderness air quality. Other papers contain the results of specific research projects on wilderness recreation impacts and management, wilderness restoration, fire and its management, and issues related to air, water, and exotic species.

Keywords: air quality, campsites, fire, fish stocking, invasive species, livestock, recreation impact, restoration, trails

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Wilderness Science in a Time of Change Conference

Volume 5: Wilderness Ecosystems, Threats, and Management

**Missoula, Montana
May 23-27, 1999**

Compilers

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Stephen F. McCool
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Preface

The Wilderness Science in a Time of Change Conference was held in Missoula, Montana, May 23 through 27, 1999. The conference was conceived to be both a followup and an expansion of the first National Wilderness Research Conference, held in Fort Collins, Colorado, in 1985. That conference brought together most of the scientists in the world who are working on issues related to the management of wilderness and resulted in literature reviews and compilations of research that remain critical references today (Lucas 1986, 1987). Our intent was to bring scientists together again, along with wilderness managers, to produce an updated compendium of the current state-of-knowledge and current research. In addition, we sought to increase the array of scientific disciplines represented at the conference and to expand the range of topics beyond the challenges of managing wilderness. Finally, we hoped to use plenary talks to highlight controversy, divergent viewpoints, and management dilemmas—to challenge participants' belief systems—in the hopes that this would stimulate interaction and personal growth.

Well over 400 people participated in the conference. Conference attendees included a roughly equal mix of people from federal land managing agencies and from academia. There were also several representatives from state, local, and tribal governments. There were more than 30 attendees from 16 different nongovernmental organizations, as well as a number of private individuals, consultants, and members of the press. About 20 participants were from Canada, with about 20 more participants from other countries. We succeeded in attracting people from diverse disciplines, united in their interest in wilderness. As usually is the case, a large proportion of the researchers who attended specialize in the social science aspects of outdoor recreation. However, attendees also included other types of social scientists, philosophers, paleontologists, and life scientists interested in all scales of analysis from cells to the globe.

The conference consisted of plenary talks to be presented before the entire conference, as well as more narrowly focused presentations organized around three conference themes and presented in concurrent sessions. The conference's plenary talks were organized into four sessions: (1) global trends and their influence on wilderness, (2) contemporary criticisms and celebrations of the idea of wilderness, (3) the capacity of science to meet the challenges that wilderness faces and to realize the opportunities that wilderness presents, and (4) concluding talks related to conference themes.

The bulk of the conference was organized around three themes. The first theme was "Science for Understanding Wilderness in the Context of Larger Systems." The emphasis of this theme was better understanding of the linkages between wilderness and the social and ecological systems (regional, national, and international) in which wilderness is situated. The emphasis of the second theme, "Wilderness for Science: A Place for Inquiry," was better understanding of what we have learned from studies that have utilized wilderness as a laboratory. The third and most traditional theme was "Science for Wilderness: Improving Management." The emphasis of this theme was better understanding of wilderness visitors, threats to wilderness values, and means of planning for and managing wilderness.

We organized three types of sessions under each of these three themes. We invited 18 speakers to present overview papers on specific topical areas under each theme. Many of these speakers developed comprehensive state-of-knowledge reviews of the literature for their assigned topic, while others developed more selective discussions of issues and research they judged to be particularly significant. In addition, conference participants were given the opportunity to contribute either a traditional research paper or to organize a dialogue session. Most of the research papers (131 papers) were presented orally, but 23 additional papers were presented in a poster session. The 14 dialogue sessions were intended to promote group discussion and learning.

The proceedings of the conference is organized into five separate volumes. The first volume is devoted to the papers presented during the plenary sessions. Subsequent volumes are devoted to each of the three conference themes, with two volumes devoted to wilderness management, the theme with the most papers. Within each theme, papers are organized into overview papers, research papers, and papers from the dialogue sessions. The format of dialogue session papers varies with the different approaches taken to capture the significant outcomes of the sessions. Research papers include papers presented orally and on posters. Within each theme, research papers are organized into broad topical areas. Although the initial draft of each proceedings paper was reviewed and edited, final submissions were published as submitted. Therefore, the final content of these papers remains the responsibility of the authors.

We thank the many individuals and institutions on the lists of committee members and sponsors that

follow. They all contributed to the success of the conference.

Planning Committee: Joan Brehm, Perry Brown, David Cole, Wayne Freimund, Stephen McCool, Connie Myers, and David Parsons.

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References

- Lucas, Robert C., comp. 1986. Proceedings, national wilderness research conference: current research; 1985 July 23-26; Fort Collins, CO. Gen. Tech. Rep. INT-212. Ogden, UT: Intermountain Research Station. 553 p.
- Lucas, Robert C., comp. 1987. Proceedings, national wilderness research conference: issues, state-of-knowledge, future directions; 1985 July 23-26; Fort Collins, CO. Gen. Tech. Rep. INT-220. Ogden, UT: Intermountain Research Station. 369 p.

—The Compilers

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5. Air, Water, and Exotic Species

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Wilderness Ecosystems, Threats, and Management

David N. Cole
Stephen F. McCool

The Wilderness Act of 1964 gave wilderness managers a difficult and challenging mandate. Wilderness areas are to be kept in a wild and natural state—relatively free of human influence and human control. Their value is dependent on the degree to which they remain unmodified—a contrast to the highly modified world in which most of us live. However, even the ecosystems in these most protected of public lands are threatened by human activities both internal and external to wilderness (Cole and Landres 1996). Impacts from these activities vary in both intensity and areal extent. Recreation use, often heavy and highly concentrated, has turned many sites into compacted, erosion-prone places, stripped of vegetation and topsoil. Livestock grazing impacts, while absent in a majority of wilderness areas, have been profound where they occur, with impacts from current grazing practices often less pronounced than those of the past (Vankat and Major 1978). The impacts of fire suppression, while less intense, are widespread. Huge acreages of wilderness have already experienced profound changes in vegetation structure as a result of this activity. Air pollution effects may be even more pervasive and problems with exotic invasions are increasing all the time.

As recognition of the prevalence and severity of human impact in wilderness increases, pressure to restore wilderness ecosystems—to compensate for human influence—mounts. Some managers are advocating intentional manipulation of wilderness ecosystems—from thinning of vegetation and management-ignited fire to liming of water bodies and genetic manipulation. This raises the serious dilemma of whether it is best to emphasize naturalness or wildness in wilderness—whether to minimize human influence or human control (Cole 1996).

Science is needed to provide a foundation for appropriate management of wilderness ecosystems. Rich research traditions in the fields of wilderness recreation impact and fire have contributed to relatively firm scientific bases for dealing with these threats. Air quality programs, strengthened by the mandates of the Clean Air Act, are also relatively well developed. Most other threats to wilderness ecosystems have received even less attention. This problem is aggravated, moreover, by the fact that many scientists who work

on large undisturbed ecosystems make little attempt to apply their knowledge to wilderness management.

Managers need research on the nature and significance of a wide variety of anthropogenic impacts, as well as an understanding of factors that influence impact characteristics. They need an improved understanding of natural conditions and processes and the extent to which existing conditions deviate from natural conditions. They need practical indicators and techniques for assessing conditions and monitoring deviation from natural or acceptable conditions. Armed with this knowledge, managers should be in an improved position when deciding where and what management is appropriate.

This volume is devoted to research on human activities that threaten the integrity of wilderness ecosystems, impacts of those activities, and management approaches that minimize these impacts. It is organized into seven sections. The first section provides five overview papers, one on each of five major threats. Yu-Fai Leung and Jeff Marion provide a comprehensive overview of the field of recreation ecology and update the synthesis of recreation impact research provided in the proceedings of the first wilderness science conference (Cole 1987). Jim Agee synthesizes the rich research tradition on fire and its management in wilderness, again updating a review developed for the first science conference (Kilgore 1987). Research on air quality issues and their management in wilderness, another topic covered in the first science conference (Schreiber and Newman 1987), is reviewed by Kathy Tonnessen. The final two overview papers provide research syntheses and perspectives on threats that were not addressed at the first science conference. Mitch McClaran examines livestock management in wilderness, while John Randall covers management of alien plants.

The second section consists of research papers on recreation impacts and their management. While some of these papers improve our understanding of the fundamental nature of recreation impacts, many are devoted to assessment and management of impacts. Papers in the third section deal with wilderness restoration. Most of these papers are concerned with restoration of sites damaged by recreation use. Papers on restoration of fire in wilderness are included in the fourth section, along with other research papers on fire regimes, impacts associated with suppression of fires, and appropriate fire management in wilderness. The few research papers presented on air, water, and exotic species issues are collected in the fifth section. Broad papers on wilderness management and planning are collected in the sixth section. The final section consists of the one dialogue session included in this volume, a session devoted to the dilemma of manipulative restoration of wilderness ecosystems.

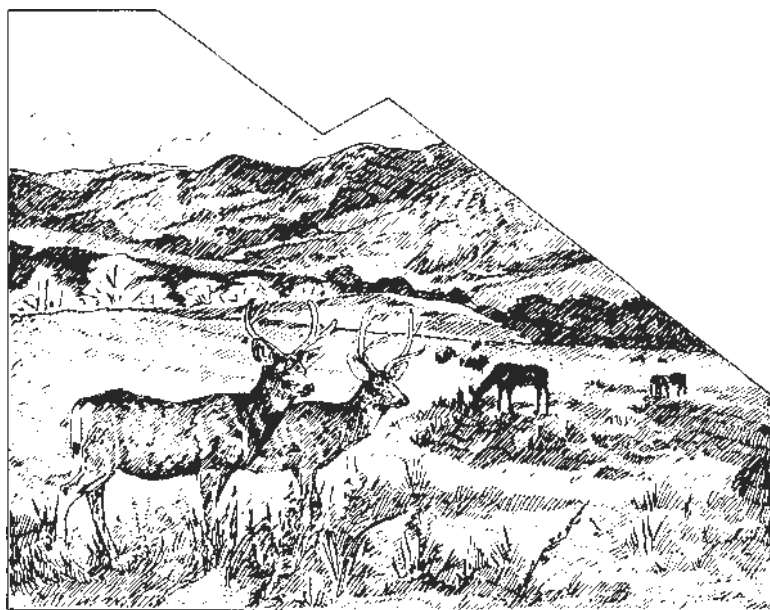
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References

- Cole, David N. 1987. Research on soil and vegetation in wilderness: a state-of-knowledge review. In: Lucas, Robert C., comp. Proceedings—National wilderness research conference: issues, state-of-knowledge, future directions; 1985 July 23-26; Fort Collins, CO. Gen. Tech. Rep. INT-220. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 135-177.
- Cole, David N. 1996. Ecological manipulation in wilderness: an emerging dilemma. *International Journal of Wilderness*. 2: 12-16.
- Cole, David N.; Landres, Peter B. 1996. Threats to wilderness ecosystems: impacts and research needs. *Ecological Applications*. 6: 168-184.
- Kilgore, Bruce M. 1987. The role of fire in wilderness: a state-of-knowledge review. In: Lucas, Robert C., comp. Proceedings—National wilderness research conference: issues, state-of-knowledge, future directions; 1985 July 23-26; Fort Collins, CO. Gen. Tech. Rep. INT-220. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 70-103.
- Schreiber, R. Kent; Newman, James R. 1987. Air quality in wilderness: a state-of-knowledge review. In: Lucas, Robert C., comp. Proceedings—National wilderness research conference: issues, state-of-knowledge, future directions; 1985 July 23-26; Fort Collins, CO. Gen. Tech. Rep. INT-220. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 104-134.
- Vankat, John L.; Major, Jack. 1978. Vegetation changes in Sequoia National Park, California. *Journal of Biogeography*. 5: 377-402.

1. Overviews



Wilderness Fire Science: A State-of-Knowledge Review

James K. Agee

Abstract—Wilderness fire science has progressed since the last major review of the topic, but it was significantly affected by the large fire events of 1988. Strides have been made in both fire behavior and fire effects, and in the issues of scaling, yet much of the progress has not been specifically tied to wilderness areas or funding. Although the management of fire in wilderness has been slow to recover from the fires of 1988, science has progressed most significantly in its ability to deal with fire at a landscape level. Major challenges include better understanding of the regional context and function of wilderness areas, as well as understanding and incorporating fire patchiness, variability and synergistic disturbance factors into predictive models. If more precise models are to be applied accurately in wilderness, better weather databases are essential.

Wilderness fire has presented both managers and scientists with considerable challenges over the 30 years that wilderness fire programs have been operational. Wilderness fire, in its purest form, should be “wild” fire: unfettered by the constraints of humans. We have never prescribed a “let-it-blow” policy for tornadoes and hurricanes, a “let-it-erupt” policy for volcanoes or a “let-it-grind” policy for glaciers. Why, then, did we need a “let-it-burn” policy for fires, or surrogate strategies like prescribed fire? Humans and fire have an inseparable history (Pyne 1995). We have been able to control fire for human purposes for thousands of years and find it very difficult to “let wild fire loose” (Pyne 1989). There are some good reasons for this reluctance, including the issues of safety to humans and damage to resources and property. As much as we have tried, we have not been able to find areas large enough to “let wild fire loose,” and this has been at the root of the challenges to research and management over three decades. It remains a primary challenge today.

The literature on fire in wildlands is immense. As in every field, some of it is hardly worth printing, while some is insightful and informing. In this review, I cannot cover even the entire latter category, and do not attempt a complete literature review by any means. My objective is to summarize the major trends in wilderness fire science since its inception, with a focus on recent times, and to define scientific challenges for the future. Fortunately, there are a number of major conference proceedings that

have synthesized fire research over the past decades and allow somewhat cursory coverage in this review. In chronological order, they include: Fire Regimes and Ecosystem Properties: Proceedings of the Conference (Mooney and others 1981); Proceedings – Symposium and Workshop on Wilderness Fire (Lotan and others 1985); National Wilderness Research Conference: Issues, State-of-Knowledge, Future Directions (Lucas 1986a) and National Wilderness Research Conference: Current Research (Lucas 1986b); Fire and the Environment: Ecological and Cultural Perspectives (Nodvin and Waldrop 1991); and Proceedings: Symposium on Fire in Wilderness and Park Management (Brown and others 1995a). In addition, there is the once-annual and now-periodic Tall Timbers Fire Ecology Conference proceedings which contain significant material related to wilderness areas. Several books are available that provide specific geographic or disciplinary information about fire: Fire and Ecosystems (Kozłowski and Ahlgren 1974); Fire Ecology of the United States and Southern Canada (Wright and Bailey 1982); and Fire Effects on Ecosystems (DeBano and others 1998). Some regional treatments have been possible where information is abundant: Fire and Vegetation Dynamics: Studies from the Boreal Forest (Johnson 1992); and Fire Ecology of Pacific Northwest Forests (Agee 1993).

Definitions of fires have changed over the past decades, most recently in 1997. I have attempted to be faithful to the new terminology where possible, but doing so is awkward. The first natural fires allowed to burn were called “let-burn” fires, but that phrase conjured up an impression of no management at all. It was changed to “prescribed natural fires” in the 1970s as part of a tripartite division of fires: wildfires, which were unwanted fires of any origin, prescribed fires, which were manager-ignited fires, and prescribed natural fires. All of these fires were called wildland fires, as they occurred in wildlands, in contrast to structural fires. In the mid-1990s, Federal fire policy was reviewed, and a new terminology was created. Prescribed fire remained a separate category, and all other fires were classed as “wildland fires,” which was somewhat confusing as that phrase referred previously to all fires in wildlands. The wildland fire category was subdivided into (1) wildfires (unwanted wildland fires) and (2) wildland fires that might be managed (those of natural origin burning within a predetermined zone and within prescription limits of some type): the old prescribed natural fire. Unfortunately, there has been no formal phrase adopted for these fires: Prescribed natural fire is now defined by what it is not (not a prescribed or unwanted wildland fire). A logical name such as “managed wildland fire” is not very descriptive or formally used, so I will continue to call these fires “prescribed natural fires,” or “pnf.”

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Historical Evolution of Fire Science Applied to Wilderness

The recognition of ecological process as a major management objective for parks and wilderness came of age in the 1960s. Before then, of course, there were national parks and monuments managed by the National Park Service (NPS) and designated wild areas and primitive areas, as well as considerable unroaded but unclassified lands, managed by the Forest Service. Fire was suppressed in all of these units, except for experimental burning in Everglades National Park (Robertson 1962). Three major public policy shifts occurred in one decade: the Leopold Report (1963), the Wilderness Act (1964) and Department of the Interior fire policy (1968) that recognized natural processes, including fire, as valid objectives of management. The Leopold Report was generated by a wildlife controversy in Yellowstone National Park, but its chair, A. Starker Leopold, broadened the report to a grand vision of the purposes of national parks (Leopold and others 1963).

The report recognized that the primitive landscapes of America were, in large part, products of disturbance, including fire, and that in the long run, management would only be successful if it was to manage these disturbances, rather than just suppress them. The authors were somewhat pessimistic that this could ever occur, but dreamed of recreating the "...vignette of primitive America...at least on a local scale." The report was very radical for its time and was circulated by the DOI for a year before Secretary of the Interior Udall accepted it. That year, 1964, was the same year the Wilderness Act passed and was signed into law by President Johnson. It defined wilderness as an area "...untrammeled [unaffected] by man," "...affected primarily by the forces of nature..." and "...managed to preserve its natural conditions." The Leopold Report and the Wilderness Act provided similar guidance to scientists and managers. Clearly, the natural force missing from almost every park and wilderness area was fire: How could it be reintroduced to these systems? There was no regulatory guidance for an operational application of fire management until 1968, when the DOI released its new fire policy, based on the concepts of the Leopold Report. This new policy not only recognized prescribed fire as a legitimate action, but also sanctioned the use of natural fires where appropriate.

Within the same year, a fire management program was instituted at Sequoia and Kings Canyon National Parks, accompanied by a research program that investigated the effect of these programs on fuels, flora and fauna (Kilgore and Briggs 1972). Yosemite National Park followed in 1970. These parks had primarily low- and moderate-severity fire regimes (c.f. Agee 1993), where fire historically was fairly frequent and few of the fires were of stand-replacement intensities over large areas (mixed-conifer/pine, red fir). Higher-severity chaparral areas were avoided in the initial years. The broad granite terrain of these parks also helped contain fires to individual valleys: Long wind-driven intense fire runs were uncommon there. The early research there (Agee 1973; Biswell 1961, 1967; Hartesveldt 1964; Kilgore 1971a,b, 1972, 1973; Parsons 1976, 1978; van Wagtenonk 1972, 1974, 1978) clearly showed that prescribed fire could be valuable in moving ecosystems back to more natural conditions, without unacceptable resource damage, and that

prescribed natural fire could be successfully managed (Kilgore and Briggs 1972). Although Forest Service research had been helpful to the NPS scientists and managers, in both research and application the NPS was a leader by the early 1970s (van Wagtenonk 1991a).

Yellowstone National Park began a prescribed natural fire program in 1972 (Romme and Despain 1989). Research and monitoring there found two seemingly apparent patterns: (1) Fires tended to burn primarily in old-growth forest (Sweaney 1985) and naturally extinguished themselves at the boundary of younger forest; and (2) very large fires were characteristic of the distant past (Romme 1982). Romme's work was somewhat consistent with the monitoring, in that he found older forest to be more flammable than younger forest. But his reconstruction of the Yellowstone landscape since the early 1700s suggested an ecosystem never in equilibrium or stability at any park scale, due to large events at infrequent intervals. The implications of these findings were never addressed by the fire management plan for Yellowstone, although they were available almost a decade before the fires of 1988.

The Forest Service began a similar wilderness fire program in the Selway-Bitterroot Wilderness in northern Idaho in 1972. This area contained forest types in moderate- and high-severity fire regimes (Brown and others 1995b), and the second fire that was allowed to burn (Fritz Creek 1973) escaped, burning about 500 ha outside of the management unit (Daniels 1974). The fire had been monitored during the burn, and research work was initiated after the smoke had cleared (Mutch 1974). The program was continued, although it was later described by the agency as meeting with "moderate" success (Towle 1985). In 1978 the Forest Service adopted a nationwide "appropriate response" suppression strategy that more clearly allowed this type of integrated fire management. A naturally occurring ignition, under this policy, could be declared a wildfire, but limited resources might be directed to suppress it. Manager-ignited prescribed fire was not allowed in designated Forest Service wilderness through the mid-1980s.

The adoption of wilderness fire management plans that incorporated prescribed fire or prescribed natural fire blossomed in the 1980s. Associated with this increase were extensions of plans into primarily high-severity fire regimes and the increase in both prescribed fire and prescribed natural fire (Botti and Nichols 1995). Management was clearly moving faster than research, partly because of limited funding for park and wilderness research, and the limitations of science to address operational concerns.

Limitations of the Science Through the Mid-1980's

The primary limitation posed by science for wilderness until the fires of 1988 was the dissolving paradigm of successional theory. The fading of a firm theoretical model (classical Clementsian climax theory) to apply to disturbance in natural ecosystems allowed managers to view reintroduction of fire as a "good" thing without much attention to either what fire was doing or where it might go. Ecological problems with some fire programs were difficult to solve because of a lack of records on where burns occurred

and a lack of monitoring of the fires' effects on resources (Thomas and Agee 1986).

The classical view of shifting paradigms (Kuhn 1970) was that after an accepted model of science (a paradigm) was created, evolving research would accumulate evidence suggesting the current paradigm was too simple or just wrong. Eventually, a relatively rapid shift towards a more robust model would occur, and that new paradigm, in turn, would eventually be rejected in favor another, more robust paradigm. In plant ecology, the major paradigms of the century themselves underwent a succession similar to initial floristics (Egler 1954; Agee 1993), where many of the species (theories) represented in the successional sequence are present in early succession but display differential dominance over time. The major plant ecology theories were all proposed within a decade early in the 20th century, but exhibited differential dominance over time.

The classical view of plant succession (the theory that attained initial dominance) persisted much of the 20th century: the Clementsian view of regional convergence towards a vegetation life-form created by autogenic succession in the presence of stable climate (Christensen 1988, 1991). Although competing models were proposed early (Gleason 1917, Tansley 1924), the Clementsian model was not seriously challenged until Odum (1969) proposed an ecosystem model that had a number of tautological premises. Among them were assumptions that diversity and stability increased with ecosystem development (time since disturbance). Odum's paper generated a number of rebuttals (such as Drury and Nisbet 1973) that suggested that ecosystems did not have emergent properties, that various forms of diversity might peak in early succession and that stability might in some cases be maintained by disturbance. Rather than producing a more robust paradigm, these challenges to the existing order recognized that ecology is a science of place and time. Grand unified theories are unlikely to apply (Christensen 1988). Much of the new theory was developed by ecologists who had worked in disturbance-prone ecosystems, and they recognized the multiple pathways that succession might take after disturbance, a function of both the disturbance and the "players" or organisms at the site. Disturbance, rather than a binary presence-absence variable, became a complex combination of characteristics (White and Pickett 1985).

Wilderness fire scientists welcomed these challenges to the classical theory. The incorporation of disturbance into new theory provided a scientific niche for the presence of fire in wilderness: Disturbance had a place in natural landscapes (White 1979). It was now possible to more clearly explain the previously baffling myriad of successional trajectories after disturbance. But as the challenges were comforting in one sense, they were discomfiting in another. To what the new theory added in recognizing fire as a natural factor, it removed in discarding the notion of convergence toward stable ecosystem states (Christensen 1991). This created two managerial challenges: (1) The issue of what to preserve became much more complex, as ecosystem classification resulted in much less convergence of community types; and (2) The stable end point toward which we should manage suddenly disappeared, leaving managers groping for a definition of a natural ecosystem state or states. This latter point had crucial significance for wilderness fire.

This question took form in 1980s wilderness as a debate between structure and process as appropriate goals for park and wilderness management. In a somewhat simple synopsis, the process argument stated that every past landscape was a snapshot of a variable ecosystem, and that ecosystem would vary into the future. Reintroducing the process of fire would eventually restore an uncertain but natural future set of ecosystem states (Parsons and others 1986). This view was supported by some of the early interpreters of the Wilderness Act (Worff 1985a,b). The structure argument (Bonnicksen and Stone 1982) stated that in any ecosystems where an unnatural structure had developed, reintroducing fire without attention to current structure could not result in a restored natural ecosystem. To some extent, the debate depended on where one was (Agee and Huff 1986): after all, ecology is a science of place. But the question remained even where scientists were viewing the same place. The argument became most heated in the Sierra Nevada/Cascades low-severity fire regimes, where almost everyone agreed on the degree of ecological change but differed on the need for structural approaches to restoration (Bancroft and others 1985; Bonnicksen 1985).

Added to the uncertainty of a desired future condition was the uncertainty of the disturbance regime. In the 1960s, the recognition of fire as a natural factor was sufficient to encourage management implementation. In the 1970s and 1980s, more information began to emerge about fire regimes. White and Pickett (1985) defined a number of characteristics important for understanding the effects of disturbance (such as frequency, magnitude, seasonality, extent, etc.), but for fire regimes, the primary one investigated was frequency, and primarily for low-severity fire regimes. Kilgore's review of wilderness fire (1986) for the first conference on wilderness focused primarily on frequency within broad fire regime types. More than 40 references to fire frequency were made by generalized fire regime types. The fire regime types did carry implications for fire intensity, but little was known about extent, season or synergism with other disturbances. Variability and patchiness, now known to be very important, were largely unquantified. Some information on variability in fire frequency was presented in terms of ranges of fire frequency. Complex fire regimes in the moderate severity fire regimes had little information available on patch size, proportions of different severity or other aspects of the fire regime.

Standards for monitoring were largely lacking during this period. Success was often gauged by area burned by prescribed fire and/or prescribed natural fire. Even though uncertainty about the operational goals of fire management (fuel reduction, ecological effects, etc.) persisted, there was little information that could be used to track progress towards any goal. Concerns about visual effects of prescribed fire in giant sequoia groves led to establishment of an independent committee to review the fire program at Sequoia and Kings Canyon National Parks (Christensen and others 1987; Cotton and McBride 1987). The committee recommended development of a detailed monitoring system for fires by the National Park Service.

Stand-level dynamic models incorporating disturbance began to emerge in the 1970s, but they suffered from the absence of established subroutines for stand growth, fire effects, or fire behavior. Most were derived from the JABOWA-type

gap models that grew stands on a small area (Botkin and others 1972). The first model, FYRCYCL, was developed at Yosemite (van Wagtenonk 1972) and was far ahead of its time in using historical fire weather to drive the fire portion of the model. Another early model was SILVA (Kercher and Axelrod 1984), which was an improvement on FYRCYCL in the stand growth routine but less elegant in its fire behavior and fire weather. Fire effects on trees were estimated from scorch height (a function of fireline intensity) and tree diameter. However, many of the weather inputs were held constant, so a crude simulation at best of the fire regime was possible. CLIMACS (Dale and Hemstrom 1984) was another fire model parameterized for the Pacific Northwest. Its stand growth subroutines were robust but it treated disturbance as an external effect that required the user to define exactly which size classes and species were removed from a particular disturbance. It was verified for only one forest type in the region.

Two models linking fire behavior and fire effects were developed during this period. Peterson and Ryan (1986) developed an algorithm that integrated stand-level characters and fire behavior (including estimated flame residence time) into a probability of mortality that was a function of volume of crown kill and the ability of a given bark thickness to withstand lethal heat. The model requires estimation of burning time in order to compare time of lethal heat to critical time for cambial kill (based on bark thickness), and burning time was not commonly available to users. Ryan and Reinhardt (1988) used empirical data to develop a similar mortality function based on crown scorch volume and bark thickness.

One of the major developments useful in fire behavior analysis was adaptation of the Rothermel spread model (1972) to a variety of stylized fuel models (Albini 1976), including those applicable to wilderness. A PC-version known as BEHAVE was made available in 1984 (Burgan and Rothermel 1984), with later improvements in several areas (Andrews 1986). This model allows prediction of surface fire behavior for given fuel, weather and topographic predictions. At high levels of input variables, fire behavior expressed as fireline intensity or flame length can be interpreted as leading to erratic fire behavior, but crown fire models during this period were limited to empirical studies in boreal forests (Van Wagner 1977).

Most of the growth in operational fire management plans in the 1980s was in parks and wilderness areas with moderate- to high-severity fire regimes, suggesting that these plans contained sufficient research information on effects and behavior of fire to indeed make these “prescribed” natural fire plans. In most cases, this information was very generalized. Boundaries of prescribed natural fire zones were rather arbitrarily drawn inside the boundaries of the preserves, with little attention to the main direction of spread for intense fires or their historical or projected eventual size. Historical size could be estimated from fire history research, but technology to project fire behavior days or weeks in advance was not available. In other areas, such as the chaparral of California, research in high-severity fire regimes did occur but focused on ecological effects of fire (Baker and others 1982; Parsons 1976; Rundel and Parsons 1979, 1980) and much less on behavioral aspects. Limited research in the Pinnacles Wilderness

(Agee and others 1980) focused more on behavior than ecology.

Social science research was encouraged during this period, focusing on visitor perceptions and acceptance of wilderness fire. Visitors who understood the role of fire in wilderness generally supported the policies (Cortner and others 1984; Rauw 1980; Stankey 1976; Taylor and Daniel 1984; Taylor and Mutch 1986). The economics of fire in wilderness remained clouded due to the blending of fire management activities outside and inside wilderness which made separation of costs difficult, and the different ways that agencies accounted for prescribed natural fire versus wildfire in the pre-Yellowstone fires era. The Forest Service and some regions of the National Park Service required upfront budgeting for monitoring activities; when that budget was expended, the fire was reclassified as a wildfire (Agee 1985, Daniels 1991). Another complication is the contrast between classical “least-cost-plus-loss” approaches, which assumes all resource change is a loss, and evaluation of resource change when fire could be viewed either as a cost or benefit. Mills (1985) defined the major obstacle to appropriate economic analysis of fire in wilderness as understanding the “natural state” objective of wilderness which would then allow resource change to be viewed as cost or benefit. Ecologists, as noted above, had been little help in agreeing on a consensus definition useful for economic analysis.

The Wilderness Fire workshop held in Missoula in 1983 (Brown and others 1985) defined the major issues apparent at that time. Over 100 papers and posters were presented at the conference, and five major issues were addressed: (1) the “natural fire” issue—what is natural; (2) the “Indian fire” issue; (3) the “lightning (prescribed natural fire) versus human (prescribed fire)” issue; (4) the “fire size and intensity” issue; and (5) the “unnatural fuel buildup” issue. There were no resolutions of these issues at that time, but considerable discussion of each. Clearly, the issue of “naturalness” was paramount in the first three topics. Are the origins or effects of fire the basis for “natural?” Native Americans burned many of the landscapes of their day, often repeatedly, and these effects had a large influence on vegetation as far back as we can reconstruct it (Arno 1985; Gruell 1985; Kilgore 1985; Lewis 1985). How should this be incorporated into current fire planning for wilderness? The lightning versus human ignition issue is tied to the previous questions and to the last question as well. Arguments about how close a prescribed fire can mimic a natural ignition (Despain 1985), the need for caution in using prescribed fire in wilderness (Daniels and Mason 1985), the need to focus on fire effects (Van Wagner 1985) and the need to keep human hands off wilderness (Worff 1985a) all surfaced in this discussion. The management-caused fuel buildup in some ecosystems was suggested to be reason enough for prescribed fire programs to restore more natural conditions (Brown 1985; van Wagtenonk 1985).

Yellowstone: The Revolution of 1988

A revolution is defined as a drastic change of any kind, and that describes the events of the summer of 1988. Yellowstone’s fires were at the center of the controversy because of their

visibility, but other fire events occurred that same year under similar circumstances.

Yellowstone's Fire Program

Yellowstone's prescribed natural fire program began in 1972 and was considered by the Park to be a successful program before 1988. An average of 30 fires per year burned between 1972 and 1987 (Despain and Romme 1991), and about half were monitored. The monitoring of the fires during this time indicated that fuels were a major determinant of where fires burned, with weather influencing the behavior of the fires. Most fire starts and fire spread occurred in older lodgepole pine (*Pinus contorta*) stands, and fires appeared to naturally extinguish themselves at the edges of younger stands. The monitoring results might have been interpreted to mean that as more natural fires burned, the Park would be buffered from extreme events by the patch mosaic of fuels (Sweeney 1985). However, work by Romme (1982) had suggested that a very large event had occurred in the early 1700s over at least part of the Park.

The summer of 1988 brought many fires and little precipitation compared to the 1972-1987 record, a very short period of comparison for a high-severity fire regime of hundreds of years. It is not surprising that conditions of the extreme event were not forecast, and two-thirds of the 1972-1987 period July and August precipitation was well above long-term averages (Despain and Romme 1991). When the fires of 1988 began to spread, they were pushed by a series of cold fronts, which resulted in substantial increases in fire area in short periods of time, capped by the runs of early September that resulted in fire area growth of tens of thousands of ha per day.

By the end of the summer, over 300,000 ha (750,000 ac) of the Park, and similar areas around it, had burned in a spectacular series of fire runs. Roughly half of the area burned was from direct or indirect human causes (camper, firewood, power line), reviving the argument of whether nature cared who started the fire (Van Wagner 1985). Park researchers defended that area as "natural" by claiming that natural fire starts in each area occurred later in the same year and, under the extreme conditions of 1988, would have resulted in similar spread patterns (Despain and Romme 1991). Yet that argument remains a weak *ex post facto* attempt to justify the argument that we were witnessing a "natural" event of unparalleled magnitude in recent history. Certainly the scale had precedent (Pyne 1982), but human activities altered the pattern and extent of the fires of 1988 (Christensen and others 1989).

Canyon Creek

The Canyon Creek fire burned in the Bob Marshall Wilderness. Ignited by lightning on June 25, 1988, it was designated a prescribed natural fire and was allowed to burn (Daniels 1991). It stayed at less than 1 ha (2.5 ac) for 26 days, but in late July grew to 4,000 ha (10,000 ac) in three days, burning in a mosaic pattern so that about a third of the encompassed area actually burned. After 65 days of active management, the fire escaped the wilderness boundary and grew from about 25,000 ha (60,000+ ac) to almost 100,000 ha (250,000 ac) in 16 hours, at the same time the Yellowstone

fires were rapidly expanding. Full suppression action was ordered for the fire.

Prophecy Fire

The Prophecy fire burned at Crater Lake National Park, Oregon, in August 1988. It began in the eastern boundary area of the Park, but was within the approved natural fire zone. Crater Lake had managed natural fires for a decade in the moderate-severity red fir type, and these burns had remained in prescription. The Prophecy fire was pushed by strong westerly winds and moved out of the Park to cover about 400 ha of Forest Service land to the east. These winds may not have been unusual, but the absence of weather stations in the area meant that this fire weather, and the associated fire behavior, would not be predicted. The fire crowned through a sparsely vegetated climax lodgepole pine type that was thought to rarely support such behavior (Agee 1981, Gara and others 1985).

Sifting Through the Ashes

By late summer of 1988, the political climate of an election year, combined with the perceived multi-regional, multi-agency failure of the natural fire program, resulted in the suspension of all such programs until completion of a review and implementation of any review recommendations. Local policy reviews of the Yellowstone situation (Christensen and others 1989) and a major national fire policy review (Philpot and Leonard 1989) were completed before the end of the year. The local review focused on ecological issues and proposed both research and management recommendations for Yellowstone. For research, the review recommended an ecosystems approach, a landscape or geographic context for individual projects and provision for long-term studies (Christensen and others 1989). For management, the local review recommended that an ecological blueprint evolve on a wilderness-specific basis, to articulate clearly the range of landscape configurations locally acceptable and to guide fire management planning. The national review (Philpot and Leonard 1989) suggested that the natural fire policy was in general a sound policy, but that it had been implemented without sufficient prescription criteria. Most of the plans that did not meet current policy were in national parks (Wakimoto 1989).

The Flame Flickers: Politics and Philosophy After Yellowstone

The political landscape has been as important as the natural landscape in directing wilderness fire science. The events of 1988 essentially shut out wilderness fire, and the recovery of management programs over the past decade has been relatively slow. No one wanted to be the supervisor of the next Yellowstone event. Some wildernesses, such as Yosemite and Sequoia-Kings Canyon, which pioneered both prescribed fire and prescribed natural fire, had their programs reinstated almost immediately, as they met the criteria of the 1988 national fire policy review even before 1988. Other suspended programs have never been reinstated. The result was a significant and immediate decline

in numbers of fires and area burned (fig. 1; Parsons and Landres 1998). Although area contained with prescribed natural fire zones increased by seven percent between 1988-92, area burned by prescribed natural fires decreased by 94 percent (Botti and Nichols 1995), largely due to conservative management criteria, including funding. At the same time, prescribed fire activity doubled over its pre-1988 levels (Botti and Nichols 1995), but this is largely due to increases for one unit (Big Cypress National Preserve).

The conservative management criteria were all based on control (flame length) or external issues (smoke, availability of regional forces). Not a single criterion was based on meeting objectives for wilderness management. Given that planning context, major reductions in numbers of programs and fires allowed to burn are not at all surprising. Yet the operational management plans were not to blame. Without an ecological blueprint for what was desired in wilderness, it was not only much easier but more defensible to define conditions where fire was not wanted than to define conditions where it was.

The consolidation of research scientists in the Department of the Interior also affected wilderness fire science. The management agencies (such as the NPS) lost their ability to fund research, because that function was now in the newly created National Biological Survey. The brief life of both the National Biological Survey and its replacement, the National Biological Service, resulted in financial chaos for research scientists, and funding for fire research has continued to be problematic in the Geological Survey, where these scientists now reside.

The political developments and problems of wilderness fire management began to erode the "era" of wilderness fire (Pyne and others 1996). Pyne correctly foresaw the 1990s as a new era of urban intermix fires, and it was ushered in with the horrific Oakland fire of 1991 (Ewell 1995). Pyne's declaration was rooted in the belief that the philosophical questions posed by the marriage of fire and wilderness had never been resolved and that technical approaches could not resolve them. Yet in the end, technical approaches must be employed to foster operational fire management programs, even if the philosophical issues remain unresolved.

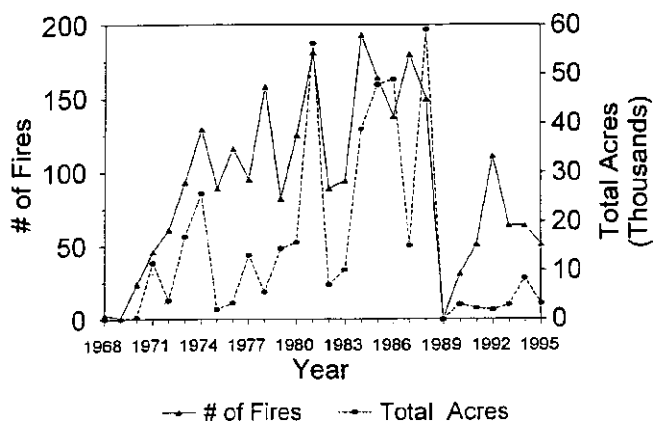


Figure 1—Trends of numbers of fires and area burned since the inception of prescribed natural fire programs in 1968. Note the pronounced drop after 1988 (Parsons and Landres 1998).

Science Since Yellowstone

The science of wilderness fire has progressed remarkably in the past decade, withstanding the political issues and a largely fragmented research approach. There have been few large research programs directed specifically toward wilderness fire, partly because of the fragmented, multi-agency management of wilderness and a lack of research focus that is characteristic of many other large, national-scope projects (Long Term Ecological Research, International Biological Program, NASA's space program, etc.). The NPS Global Change program is one larger program that has produced some substantial implications for wilderness fire. Yet many of the technical developments have resulted from locally funded projects, or from research done for other purposes.

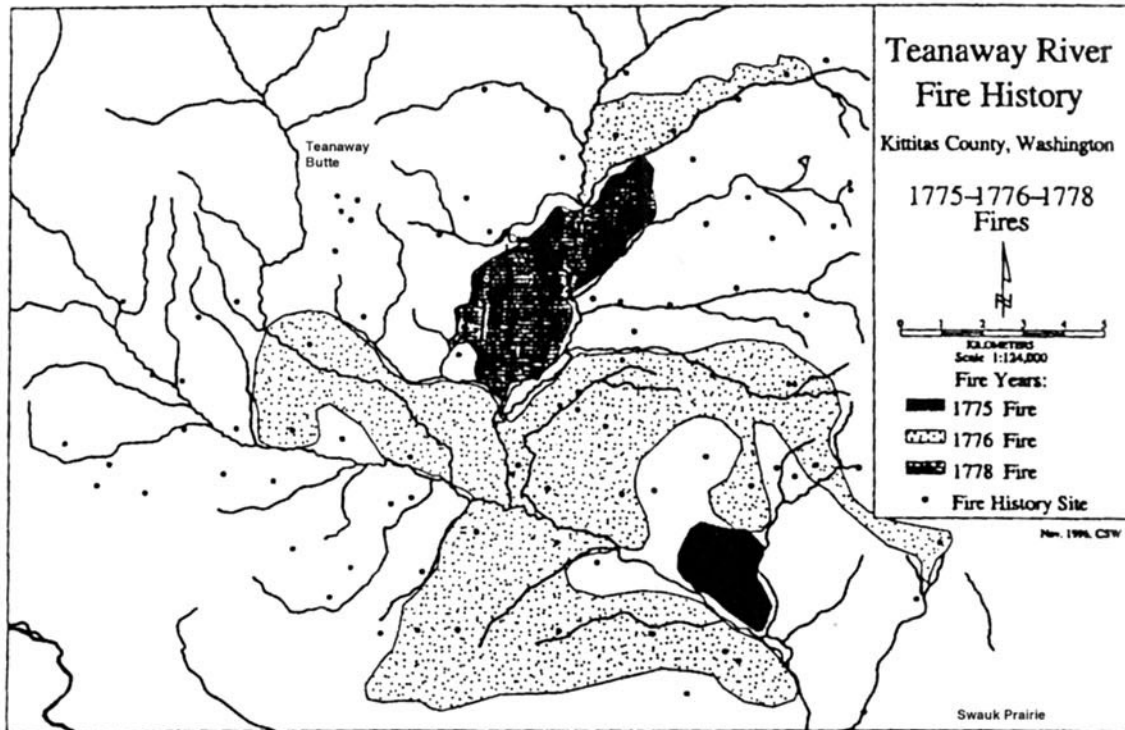
Drivers of Wilderness Fire

That fuel, weather and topography drive the behavior of an individual fire has long been known (Barrows 1951, Brown and Davis 1973). Yet the factors driving wilderness fire regimes continue to be debated: Are fuels or weather more important? Our research of the past decade suggests that the answer not only differs by fire regime, but to some extent on the interaction of fuels and weather. Swetnam and Betancourt (1990) linked a set of regional cross-dated fire histories in ponderosa pine (*Pinus ponderosa*) forests to high (La Nina) and low (El Nino) phases of the Southern Oscillation. During the El Nino phases, precipitation in the Southwest is much higher and fire activity is much less. At the same time, tropical and subtropical areas receive less precipitation as those storms are moving further north. Large areas burned in Borneo (Davis 1984) and Australia (Rawson and others 1983) during a large El Nino event in the early 1980s. This link between global climate and local variability in fire regime shows a trend that links wilderness to the rest of the world.

In high-severity fire regimes, arguments about the relative influence of fuels and weather continue (Weir and others 1995, Wierzchowski and others 1995). In Canadian boreal and subalpine forests, prescribed fire has been used operationally under the assumption that decades of fire exclusion have changed these forest types, that younger stands have not been created during that period and that older forests were more flammable. Bessie and Johnson (1995) concluded that weather was the primary driving factor in large fire behavior; and since large fires constitute almost all the area burned, fuel conditions are relatively unimportant. They generalized these conclusions to all forest types, a conclusion rebutted by Agee (1997). He suggested that under extreme weather in low-severity fire regimes, fire size may well have increased, but that fire severity may not have been markedly increased. Fuel conditions have been shown to affect fire behavior and extent in low- (Wright 1996) and moderate-severity (van Wageningen 1995) fire regimes (fig. 2).

In some high-severity fire regimes, fire return intervals may be so long that very unusual synergistic influences may occur and mask more simple correlations of fire with flammability-stand age or weather-climate patterns. In the Olympic Mountains, Henderson and others (1989) mapped a very large forest fire event (fig. 3) circa 1700 A.D. that had been

A



B

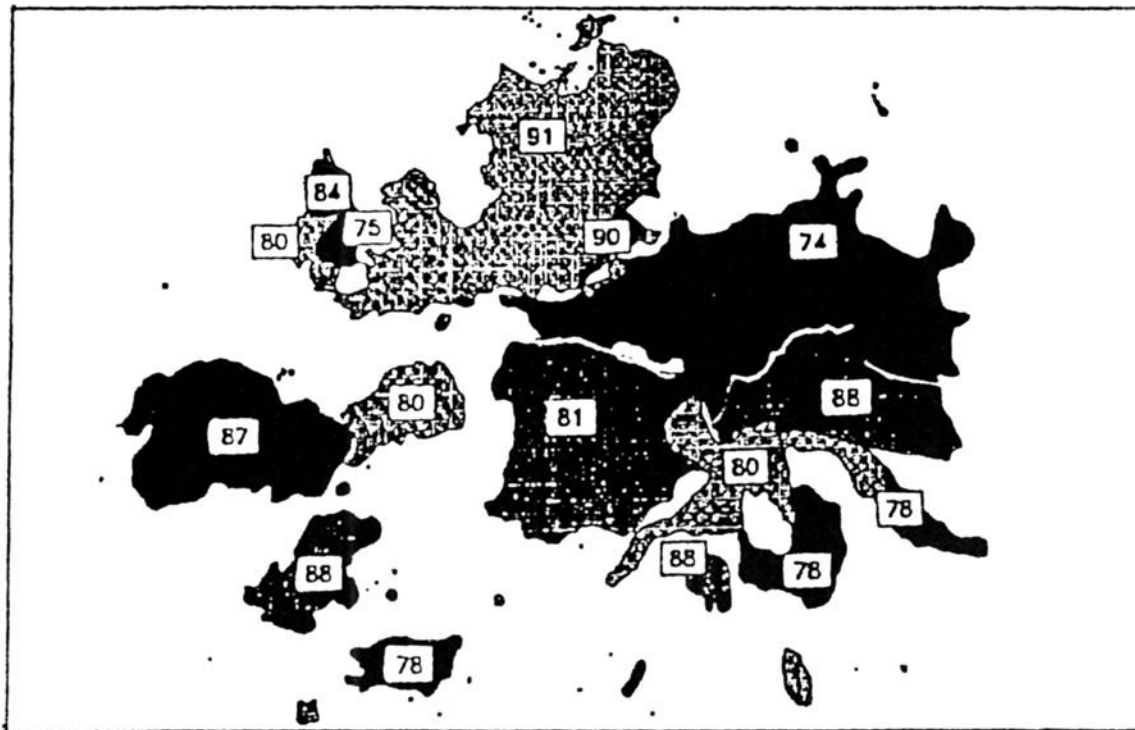


Figure 2—A. Reconstructed fires of 1775-1778 in mixed-conifer forests of eastern Washington (Wright 1996). Fires occurring with 1-2 years of one another in this low-severity fire regime appear to be extinguished when they enter recently burned areas. B. Monitored fires 1974-1991 in Yosemite National Park show similar mosaics (van Wagtenonk 1995). These appear to be more stable patterns than in high-severity fire regimes where process overwhelms pattern under severe weather (Romme and Turner 1991).

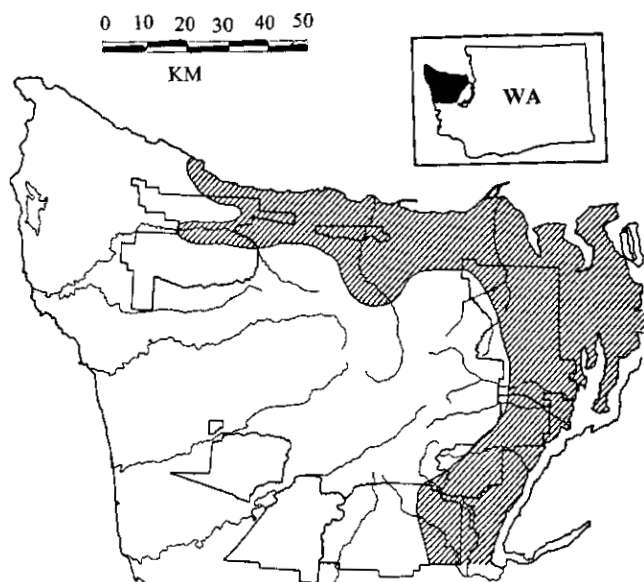


Figure 3—The large fire of ca. 1700 in the Olympic Mountains (Henderson and others 1989) appears to have occurred after a large earthquake in 1700. This quake, occurring in January, may have been associated with considerable treefall and copious dead fuel, needing only an ignition source to become very large.

partially identified by Fonda and Bliss (1969). Fire cycle models based on climate (Agee and Flewelling 1983) were unable to reproduce similar fire events, and it was thought that very unusual patterns of lightning frequency or foehn winds may have occurred in the past. Recently, a very large historic earthquake along the Washington coast was reconstructed from tree ring records of trees buried beneath sea level by submergence of coastal lands at the time of the quake (Yamaguchi and others 1997). This date was consistent with records of a major tsunami that hit the east coast of Japan on January 26, 1700. At a time when soils are saturated, this earthquake likely felled many stands of trees around the peninsula, and this additional dead fuel may have driven the large fire activity that apparently occurred. Lightning frequency, drought or foehn winds, the usual combinations of factors associated with large fires, may have remained quite average during this period.

Where fire return intervals are quite long, these “surprises” may be a major factor in the disturbance dynamics. Not only may extreme events be driving the system, but they may be events that we have not yet uncovered. Lertzman and others (1998) showed through simulating fire regime parameters that substantial variability may result, even in the absence of an underlying physical or ecological pattern. They recommend caution in attributing causality of fire regime drivers that are not motivated by independently generated hypotheses.

Fine-Tuning the Fire Regime

When early fire management programs began in wilderness, general knowledge of the fire regime was considered adequate. Research inside and out of wilderness has led to a

more precise understanding of the fire regime, but it is still not possible to generate many parameters of a fire regime by simply knowing, for example, what forest type is being considered. Where more precise information has been generated, it usually shows variability in frequency, intensity or extent. Synergistic effects are known to be more important than previously considered, although our ability to predict them is still poor. And the general implications for management have been clouded by the complexity of these emerging fire regimes. Faced with considerable ranges in variability, which combination is appropriate for a certain place now? Research on fire regimes has allowed us to place bounds on uncertainty, but it has also generally driven us away from relying on simple statistics like the mean. Programs have evolved from rather uniform burns to those incorporating considerable variability (Bancroft and others 1985; Parsons and Nichols 1986).

Fire frequency has always been a primary parameter of the fire regime. Kilgore’s wilderness fire review (1986) has over 40 citations on fire frequency in selected wilderness ecosystems, and he recognized that more examples could be cited. But information on other fire regime parameters was lacking. Since that time, we know even more about fire frequency in wilderness. These new data have allowed us to understand the distribution of fire frequency, not just its central tendency. A remarkable achievement was the reconstruction of giant sequoia (*Sequoiadendron giganteum*) fire regimes back over millennia (Swetnam 1993). The mean fire-return interval shifted significantly for this low-severity forest type over periods of centuries, and inferences about fire intensity were made from correlations of tree-ring growth with fire occurrences and percentages of sample trees scarred from an individual fire. Landscape juxtaposition of forest types was found to be important in determining fire frequency. In the north Cascades, where wet, west Cascades forest types are mixed with dry, east Cascades types due to a rainshadow effect west of the Cascade crest, the wet types had fire-return intervals well below those measured elsewhere in the Cascades for those types. The dry, eastside forest types had fire return intervals well above those measured in the eastern Cascades (Agee and others 1990).

Fire intensity remains difficult to reconstruct from historic fire regimes. Reconstruction of growth on trees experiencing fire, and defining age classes of trees likely to establish in fire-generated gaps, have been used to infer historic intensities. In giant sequoia groves where the history of prescribed fire includes some fairly hot burns, reconstruction of tree-ring growth showed that fire generally increased growth, but some variable response was evident (Mutch and Swetnam 1995). A delayed growth response was found where very intense fires had occurred and scorched the foliage of the sequoias. Sequoia regeneration was tied to fire-generated gaps where sunlight could penetrate to the forest floor. These data were used to infer past fire intensities. For example, a fire in 1297 A.D. was inferred to be relatively intense due to the increase in tree growth on giant sequoias (fig. 4), suggesting a release from competition and substantial regeneration that occurred locally (Stephenson and others 1991). A recent article suggests that high-intensity fire also was characteristic of ponderosa pine stands (Shinneman and Baker 1997). However, these stands in the Black Hills are transitional to boreal forest; white spruce

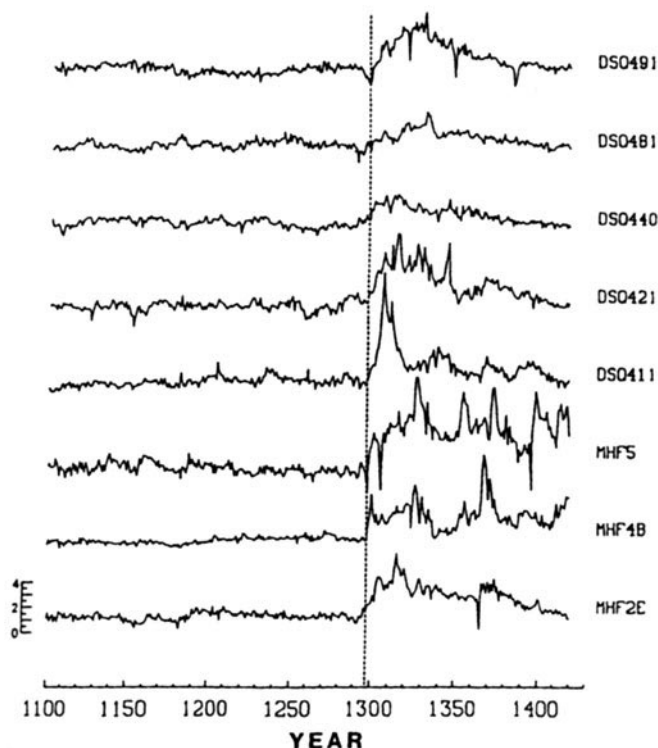


Figure 4—Patterns of tree-ring growth from giant sequoias (Stephenson and others 1991) show a pronounced growth effect after a reconstructed fire of 1297 A.D. Unlike many previous fires, this one must have been severe and reduced competition, as all trees show a growth release. The pattern of unusual growth continues for a century, and some fires are associated with decreases in growth for sample trees. This suggests severe fires did occur in sequoia groves, and reminds us of the variability in fire regime for very long-lived organisms like giant sequoias.

(*Picea glauca*) is a common understory species, and a complex mix of fire regimes (e.g., Agee and others 1990) should be expected where types are in transition. Tree regeneration is closely linked to fire severity; in moderate-severity fire regimes, severity will have significant effects on tree species likely to establish (Chappell and Agee 1996).

Quantifying season of burning has been important because of the opportunity to ignite prescribed fires over a broad seasonal range. What is most natural? Historical seasonality has been evaluated primarily for low-severity fire regimes by defining the placement of the fire scar for a particular year in the earlywood to latewood of the annual ring. In Southwest ponderosa pine stands, most scars are in the earlywood, defining spring as the most common season for fires (Baisan and Swetnam 1990), although some areas exhibit more even distribution of fires across the growth season (Grissino-Mayer and Swetnam 1995). In the Pacific Northwest, the same species exhibits mostly late-season fires (Wright 1996). Heyerdahl (1997) showed that there was considerable seasonal variation in the Blue Mountains of Oregon and Washington. Southerly Blue Mountain stands had a longer snow-free season and more scars within the growing portion of the annual ring than stands of the same species composition in the northern Blue Mountains, which had a shorter growing season and a concentration of scars after growth for the year had ceased.

Synergism, or the interaction of fire with other disturbances, was recognized by White and Pickett (1985) as an important parameter of disturbance regimes. Very little quantification of this effect was evident for fire regimes before the late 1980s. Interaction with insects has long been recognized as a major second-order fire effect (Fischer 1980), but defining the degree of interaction is difficult, as many other factors are important (Amman and Ryan 1991). After the Yellowstone fires of 1988, the major tree species in the area (lodgepole pine, Douglas-fir [*Pseudotsuga menziesii*], Engelmann spruce [*Picea engelmannii*], and subalpine fir [*Abies lasiocarpa*]) were attacked by a variety of insects; between 28-65% of the trees living after the fire were infested and killed (Amman 1991). Most of the bark beetle-attacked trees had basal damage from the 1988 fires.

At Crater Lake National Park, Swezy and Agee (1991) found that low-intensity but long-duration fires, caused by forest floor buildup due to fire exclusion, killed many of the fine roots after late spring burns. Low vigor, old-growth pine trees had an increased level of insect attack and mortality after these fires, and fall burning was recommended as a better season, based on surveys of trees burned in spring and fall.

Disease can also be an important synergistic factor. In the western United States, perhaps the most important synergism between fire and disease is the introduced white pine blister rust (Kendall and Arno 1990). This disease causes cankers on the stems of young pines and kills them. When fire kills older trees, recolonization of whitebark pine (*Pinus albicaulis*), often mediated by Clark's nutcrackers (*Nucifraga columbiana*) (Tomback 1982) may be short circuited. In mountainous terrain, snow avalanches can create persistent snow avalanche paths and alter other processes such as landsliding and future fire spread (Butler and others 1991).

Models

The past decade has witnessed an explosion in personal computing power and with that growth, an accompanying expansion of models attempting to explain fire behavior and effects. These models have particular relevance to wilderness fire because they allow forecast of spatially explicit fire sizes, as well as fire effects.

One of the more important models for fire effects has been the individual tree model FOFEM (First Order Fire Effects Model; Reinhardt and others 1997). It scales mortality to the stand level by aggregating individual tree effects to the stand level based on the Ryan and Reinhardt (1988) mortality algorithm. This model has gone through four iterations in the past decade and will continue to be updated periodically. It is national in scope and provides information in addition to tree mortality on fuel consumption, mineral soil exposure and smoke. Synergistic effects, which tend to be difficult to predict as second-order interactions, are not predicted by FOFEM. Nevertheless, it has served as the basis for tree mortality prediction in several important models.

A variety of individual-based gap models have been developed since the 1970s (Hinckley and others 1996; Urban and others 1991), but few have concentrated on incorporating fire. FIRESUM (Keane and others 1989) was an improved gap model that incorporated stand growth and disturbance

for inland Northwest conifers. The fire algorithms were complex, but the stand-level results were greatly influenced by the initializing stand condition; an individual tree dying of old age, for example, had a large influence on the basal area output over the simulation period.

While the science of gap modeling grew, the ability to represent wilderness landscapes in geographically referenced form also increased. Geographic information systems (GIS) represented a way to evaluate often inaccessible landscapes in digital form. The development of better software packages and more powerful personal computers allowed robust analyses to occur at relatively low expense. Fire applications, such as analysis of historic fire incidence by vegetation type, fuel inventories, prescribed burn units, lightning strike incidence analysis and fire regime analysis, were done (van Wagtenonk 1991b). Links of these types of analyses to fire growth simulators were beginning (Bevins and Andrews 1989). Development of accurate input layers for the current generation of fire area growth models remains relatively poor (Keane and others 1998).

FIRE-BGC (Keane and others 1996a) was developed by marrying some of the algorithms of FIRESUM with FOREST-BGC, a physiologically based model (Running and Gower 1991) that has been scaled up to a landscape approach. As applied to wilderness ecosystems in Glacier National Park (Keane and others 1996a) and the Bob Marshall Wilderness (Keane and others 1996b), the model links many across-scale interactions, but it has the universal problem of marrying not only diverse spatial scales, but those of time as well (Keane and others 1996a). Temporal information at scales from annual (stand growth equations) to hourly (fire growth equations) complicate current modeling efforts.

Disturbance propagation across landscapes has been modeled in two general ways: percolation-type models and deterministic models. The percolation models suffer from the fact that fire does not move across a landscape with equal probabilities of spread in all directions. The deterministic models suffer from data deficiencies (Van Wagner 1987). Both have increased our knowledge of fire effects and behavior at broader scales.

The percolation models have increased our knowledge about the influence of landscape pattern on process (fire) (Turner 1989). Most of the percolation work has been in high-severity fire regimes, where the binary process of a cell being occupied or not by disturbance fits the high-severity nature of the disturbance. Work in the 1980s suggested that disturbance in heterogeneous landscapes was dependent on the structure of the landscape, as well as disturbance frequency and intensity (Turner and others 1989). This evolved to a more complex view that disturbance probability affecting percolation can change over time, particularly where fire weather becomes extreme (Turner and Romme 1994). Under extreme conditions, process is relatively independent of pattern (Agee 1998; Romme and Despain 1989). Nonequilibrium systems will be the result (Baker 1989, Turner and Romme 1994); scientific advances in landscape theory have resulted in a tougher job for managers by increasing the envelope of uncertainty. Percolation-type models have suggested that landscapes altered by past intervention in fire regimes, or those subject to climate change in the past (for example,

Clark 1988) or the future, will take 0.5 to 2 rotations of the new disturbance regime for the landscape to adjust to that new regime (Baker 1989, 1994).

In contrast to the ecological gap and disturbance models, fire behavior models received less attention over the same period, yet our inability to predict fire spread and intensity has had much more effect on wilderness fire programs than imprecision in predicting ecological effects. Fortunately, substantial progress has been made in landscape modeling of fire behavior. A nonspatial model (RERAP) was developed to determine probabilities that a prescribed natural fire would exceed an acceptable size (predetermined by the user) before a fire ending event (precipitation) would halt spread (Carlton and Wittala, no date). However, it has not been widely used in wilderness fire management. A fire growth simulator (Bevins and Andrews 1989) was developed by the Forest Service, and a similar model was being developed by the National Park Service (Finney 1995). These efforts merged in the mid-1990s at the Missoula Fire Sciences Laboratory.

The model currently holding most promise for wilderness fire behavior is FARSITE, a spatially and temporally explicit fire growth model (Finney 1998). The model was initially developed to help predict spread of wilderness fires, but it has shown great applicability to wildlands in general. The landscape "themes" or data layers require information on elevation, aspect, slope, fuel model and canopy cover, with optional themes for crown fire behavior: crown height, crown base height and crown bulk density. Daily and hourly weather streams are required over the simulation period. Surface fire, spotting and crown fire behavior are simulated, subject to the limitations of models that currently exist for those types of fire behavior. Fires spread in the model using Huygens' principle, where the fire front is expanding based on elliptical wavelets, the shape of which depends on the fuel model and local wind-slope vectors (fig. 5). Backing and flanking fire spread is estimated from the forward rate of spread, as the current fire spread model (Rothermel 1972) only predicts the forward rate of spread. Finney (1998) discusses the limitations of FARSITE.

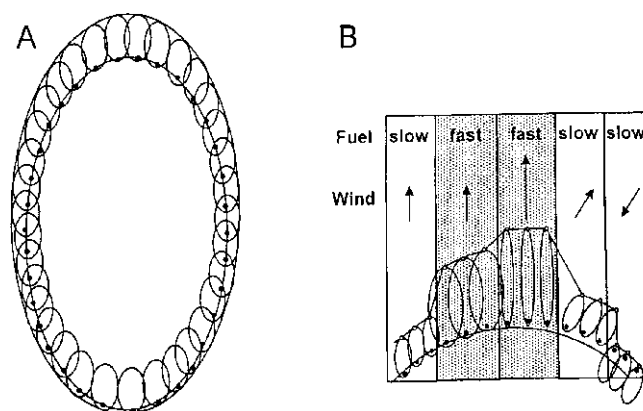


Figure 5—The fire growth algorithm of FARSITE uses a series of ellipses (Finney 1998). A. Under constant weather and fuels, these "wavelets" are of constant shape and size. B. Non-uniform conditions show the dependency of wavelet size on the local fuel type but wavelet shape and orientation on the local wind-slope vector.

Given accurate input data, the model is consistent with expectations for fire growth of surface fires. Spotting and crown fire spread are not possible to verify, although simulations do produce patterns that resemble phenomena observed on real fires. Outputs for FARSITE are geographically referenced, and flame length or fireline intensity per cell can be exported to fire effects and stand growth models to simulate landscapes over time (for example, Keane and others 1996 a,b). For wilderness applications, FARSITE could be applied to generate behavior under worst-case conditions to evaluate possible escape scenarios over a summer for a prescribed natural fire, and could be linked to ecological effects. If adjacent fuelbreaks are proposed adjacent to wilderness as a rationale for loosening prescriptions for fire within wilderness (Agee 1995), FARSITE can be used to evaluate effectiveness of the fuelbreak (van Wagtenonk 1996) and spatial effects on fire control efficiency (Finney and others, in press).

Few wilderness areas have databases that allow application of FARSITE. Yosemite National Park was on-line early due to the presence of an advanced geographic information system (J. van Wagtenonk, personal communication). Where FARSITE data layers (elevation, aspect, slope, fuel model, canopy cover, height to crown base, crown bulk density and canopy height) have been generated, accuracy levels are sometimes so low (Keane and others 1998) that application of the FARSITE model is bound to produce uncertain results, even if weather variables were perfectly predicted.

One of the major lessons learned in the 1988 fires was that the Rothermel fire spread model was not particularly robust in predicting the behavior of fires that contained a large degree of crown fire activity (Thomas 1989). Most of the quantification of conditions where crown fire occurred was derived from boreal forests of Canada (Van Wagner 1977). Crown fire assessments were possible (Alexander 1988) but not routinely employed by wilderness fire managers. After the 1988 fire season, it was apparent that better understanding of crown fire behavior was needed. Rothermel (1991) evaluated crown fire potential in northern Rocky Mountain forests, and his derivation of crown fire spread was empirically derived as 3.34 times the surface fire rate of spread of NFFL fuel model 10. Links of forest structure (Agee 1996) and weather conditions (Scott and Reinhardt, in press), using the Van Wagner and/or Rothermel approaches, have been made and are incorporated into the landscape model FARSITE (Finney 1998). Nevertheless, all involved in this research recognize the imperfect level of our understanding, and the difficulty of experimentation with crown fire only slows progress.

Monitoring

One of the deficiencies of wilderness fire programs in the early 1980s was inadequate monitoring of the fires. Most programs did have monitoring programs "on the books," but funding was often inadequate. Close monitoring occurred on early prescribed natural fires (for example, Daniels 1974), but as programs expanded, research and monitoring activities became a bit more haphazard. No standards existed for how or what to monitor or how intensive monitoring should be. Some programs had few records of where they had used prescribed fire or what prescriptions were applied (Swezy and

Agee 1991), as some programs were satisfied that fire was "back on the land" and effects were therefore natural. The review of visual effects of fire in giant sequoia groves led to a recommendation that a formal monitoring program be implemented for all national park fire management programs (Christensen and others 1987). This led to the development of the Western Region Fire Monitoring Handbook (NPS 1990), which was widely adopted through the NPS fire programs when funding mechanisms changed (Botti and Nichols 1995). Fire monitoring teams are base-funded, using emergency pre-suppression dollars, and supplemented with additional personnel during periods of high fire activity.

Levels of monitoring activity are defined, recognizing that not every fire requires the same degree of monitoring. Parks that cannot comply with the guidelines do not have a fire program. Level 1 covers reporting of all fires, and levels 2, 3, and 4 call for monitoring of fire conditions, short-term effects and long-term change, respectively. The levels are cumulative, so that requirements for one level include all those above it. Monitoring at all four levels is required for prescribed fires, while prescribed natural fires may include levels through 2, 3, or 4. While the monitoring is not research, meta-analysis of these data might be so considered, and the monitoring may suggest hypotheses that can be experimentally tested (NPS 1990).

This monitoring protocol has been widely adopted in the National Park Service and other state and federal agencies (Reeburg 1995). Training courses have been developed and implemented, and periodic refinements are expected.

Meeting the Challenges of 1986

The last state-of-knowledge review (Kilgore 1986) defined directions for future research in two broad categories: techniques/methods research and new information needs. I have chosen to rate our progress subjectively in those areas, ranking both the quality of the question and the degree of progress we have made since then.

Techniques/Methods Research

1. *Develop criteria by which managers can judge whether an ecosystem has been impacted in a major way by past fire suppression/exclusion.* Kilgore (1986) suggested that both fuels and forest structure must be addressed. We have made significant strides in this area, but the technology was available for fuels well before 1986 (Van Wagner 1977). For low-severity fire regimes, criteria for estimating surface fire intensity (Albini 1976; Burgan and Rothermel 1984; Rothermel 1972), torching potential and crown fire spread potential (Van Wagner 1977) have suggested that many ecosystems are capable of severe fire behavior where that behavior was once rare (Agee 1996). For high-severity fire regimes, the introduction of the concept of nonequilibrium systems has so broadened the sets of possible ecosystem states that the impact of fire exclusion, where fire return interval was historically >100 years, has become fuzzier.

2. *Develop minimum impact methods for determining fire history in wilderness and park ecosystems.* A good fire history technique requires a minimum impact method, but it still may be intrusive to wilderness character. In low-severity

fire regimes, there is no substitute for wedge samples that record tree-ring widths and scars. In high-severity fire regimes, stand ages can be sampled with little visual impact, whether one uses the natural fire rotation, negative exponential or Weibull methods (Heinselman 1973, Johnson and Van Wagner 1985). It appears unlikely that correlation of broad descriptors such as forest type will be sufficient to predict the central tendency and variation in fire regime in a given park or wilderness.

3. *Develop cost-effective techniques for restoring natural conditions over extensive areas of a wilderness or national park and demonstrate these methods.* We have made no progress in this area, as it is largely a management-oriented question, and area burned has declined so much that little information would even be available to analyze.

4. *Develop standard techniques to help managers monitor performance of their wilderness fire plans.* The NPS (1990) monitoring plan has been largely successful in providing a basic outline for monitoring requirements for both prescribed fires and prescribed natural fires.

5. *Develop the capability to predict August behavior of natural fires ignited in July in wilderness areas.* FARSITE has given us the technical capability to provide this prediction capability, but it requires very precise, short-term weather data that are lacking in most wilderness areas.

6. *Develop special techniques for simulating the natural role of fire in wilderness areas where allowing natural (lightning-caused) fires to burn is impractical and where ignitions outside the wilderness no longer burn into the wilderness.* We have made no progress in this area in the United States; in Canada, this technique is still controversial due to the uncertainties expressed in the first technique discussed above (Weir and others 1995, Weirzchowski and others 1995).

New Information Needs

1-3. *Using the best data available, determine the "natural" fire history, fire behavior and fire effects for key short-return-interval wilderness ecosystems. Document with case studies, in key short-return-interval ecosystems, how significantly current conditions depart from "natural" in terms of fuels and forest structure. Decide how precise we must be in restoring fuel levels and forest structure to key short-return-interval ecosystems before we allow natural fires to burn again.* I have combined the first three because they all are related, although not exclusively, to the giant sequoia fire restoration controversy of the mid-1980s. These questions have not been addressed in other ecosystems to the same extent they have in sequoia groves, where there has been a considerable investment in science (Stephenson 1991; Stephenson and others 1991; Stohlgren 1993). The answers have generally been that there are wide boundaries on what is considered natural and that, in the process of restoration, care can be taken to avoid effects that, even though natural and perhaps essential for sequoia reproduction, cause public concern, high bark char being an example (Cotton and McBride 1988).

4. *What is the relative importance of aboriginal ignitions in determining intervals between fires and both intensities and severities of fire?* This question is impossible to answer,

and no effort has been expended in any quantification of an answer.

5. *Determine whether scheduled fairly high-intensity prescribed burns can approximate the ecological effects of high-intensity, stand-replacing fires under less explosive burning conditions.* See #6 under Techniques/Methods.

6. *Determine fire effects relationships to habitat needs of endangered wilderness wildlife species such as the grizzly bear.* An entire conference was devoted to rare and endangered species issues and fire (Greenlee 1997). Although production function relations were largely lacking (such as fire at 'x' level results in 'y' response from wildlife or plants), it was clear that many of these species have some tolerance to fire, and some may be dependent on it.

7. *Determine how suppression of fire has impacted key insect and disease populations in certain forest types.* Both insects and disease may attain outbreak or epidemic conditions where plant vigor is low. Due to factors beyond fire suppression, long-term reconstructions may be needed to tease out the "fire exclusion" effect from natural variation over time (see, for example, Swetnam and others 1995). We have made some progress here, but future progress is needed and likely.

Challenges for the Future

The next wilderness conference may well have state-of-knowledge papers that will critique progress after this conference. I have chosen a more restrictive set of challenges than did Kilgore, and I hope for a higher degree of success, at least through the semantic ruse of having fewer categories.

Fire Island: Wilderness in Linked Landscapes

Natural resources planning has increasingly moved to tiered approaches at various scales to account for species and process issues that are important from broad to fine scale. Park and wilderness areas, because of their relatively unspoiled ecological conditions, can be considered core areas. Often, these areas have the highest ecological integrity of regional landscapes and serve as buffers to managed landscapes (Quigley and others 1996). Conservation strategies based on incorporating or simulating historic disturbance processes are thought to have a high probability of maintaining ecological integrity (Dale and others 1999), and they are the basis of the coarse-filter/fine-filter conservation approach (Hunter 1990). In this approach, quite consistent with wilderness management, the ecological processes, including fire, are allowed to interact as naturally as possible, and they therefore help to maintain the conditions that provided the biological diversity of the ecosystem.

Where this coarse filter fails, then species-specific fine filter plans are implemented. Many wilderness areas are surrounded by managed landscapes where past management has placed species at risk. Those species often have required fine-filter plans to maintain certain vegetation structures on the landscape. A good example is the northern spotted owl in the Pacific Northwest, which favors old-growth forest. Most of the remaining old-growth is in parks

and wilderness, and these areas provide a core of habitat for the owl. Where the old-growth is in a natural high-severity fire regime, fire will destroy owl habitat locally. While this may be consistent with a coarse-filter conservation strategy for wilderness, it may be incompatible with a fine-filter conservation strategy for owls. In spotted owl habitat with low- and moderate-severity fire regimes, lower intensity fires may be necessary for and compatible with maintaining old-growth structure.

There are likely to be increasing numbers of fine-filter plans that may conflict with coarse filter conservation strategies (Agee 1999), and fire will likely be a key issue. In a complex natural resources management environment, fire will have to be judged on both its short- and long-term effects at scales well beyond the wilderness boundary. Wilderness is not an island, but science has not yet comfortably placed wilderness in the ecological context of neighboring landscapes.

Ecology and Behavior

The nature of scientific challenges will differ by fire regime. While both ecological and behavioral issues remain for all fire regimes, the ecological ones appear largely in low- and moderate-severity fire regimes, while the behavioral ones dominate the high-severity fire regimes. What we have learned about fire regimes in the last decade is that they are more complex than previously described, and management plans need to address these complexities. Additional research in the parameters of fire regimes (both central tendencies and distributions) will help fine tune future management planning.

One of the more profound lessons we have learned over the past decade is that patchiness and variability are important ecological determinants of fire effects. For example, simulations such as FIRESUM (Keane and others 1989) show Douglas-fir to be almost absent where the fire-return interval in inland Northwest ecosystems is 10 years. That result is because fire return interval is fixed, and the simulated fire burns every piece of the simulated landscape. Real fires are patchy and variable, and we do find Douglas-fir on these landscapes, although it is subordinate to ponderosa pine. One of the major challenges of the next decade is to realistically incorporate patchiness into our simulation models. At landscape levels, FARSITE (Finney 1998) may be able to accommodate patchiness if the cell size is designed to be sufficiently small. Each cell will still burn as a homogeneous unit, but the variability on the landscape will be more realistically simulated. Perhaps a link to fire weather will allow a scaling of fire coverage by cell: At high fuel moisture and low wind, more patchiness will be allowed; as fire weather becomes more severe, less patchiness will result.

Incorporating synergism into future models will be important. This was tried in rudimentary fashion by earlier models (such as Keane and others 1990), and is in progress in currently developing watershed simulations (K. N. Johnson and J. Sessions, Oregon State University, personal communication). Linking fire effects to those of wind, insects and disease will be important to realistic ecological models of the future. It is possible to conceive a realistic model. Wind effects are largely a function of stand structure and topographic location. Insects are target (species)-specific organisms, both

at endemic levels and at epidemic levels once a basal area threshold is exceeded. Organisms causing disease are similarly focused on target species and may be more or less important in various potential vegetation types.

Fire behavior models for wilderness are now far ahead of the databases available to test the models. Continued work on refining methods to collect accurate GIS layers will be necessary. The models will need better criteria for transition from surface to crown fires, and better ways of modeling crown fire behavior. Uncertainty will always remain, due to unpredictable events such as low-level jet stream movement or the movement of plume-driven fires, but the envelope of uncertainty can be significantly narrowed from where it is today. None of these models will work well without good weather information.

Fire Weather and Climate

The accuracy of fire behavior models is highly dependent on good fire weather information. Recent fire model applications (Keane and others 1996a) continue to note the lack of good weather information for wilderness. In some areas, there is no local information at all. Our future requirements are not only for longer-term local weather, but for very specific parameters on hourly time steps, if we want to accurately predict future fire (within limits, of course) or even reconstruct historic events (Cohen 1991). The network of fire weather stations where long-term fire weather data are collected is largely outside of park and wilderness areas, so extrapolation of these data to local conditions is necessary. This limits the ability to simulate future activity of currently active fires or gaming of possible future fires. Expansion of fire weather stations within wilderness, technically feasible now with RAWs (remote automated weather stations), is needed for both research and management purposes.

The existing ability to project climate change, due to either natural change or global warming from human activities, is poor. Effects on distribution of vegetation and possible drivers of that change, such as fire are also largely unknown and speculative. Better models will start with better climate projections, which now appear to deal with temperature much better than moisture, and even then are not very reliable at subregional scales. Current projections of increases in area burned (such as Flannigan and Van Wagner 1991; Romme and Turner 1991) are largely speculative. Global warming may increase fire activity, but coastal evidence suggests major fires during global cooling episodes (for example, Agee 1993). The primary research need is better climate scenarios, followed by research on effects of such climate shifts on structure (vegetation) and process (disturbance, broadly defined).

The Need for Courage

Natural resources science often does not provide specific answers to operational problems. At best, it may provide limits or boundaries on uncertainty, or it may increase the uncertainty of the manager's domain. This may be very pleasing to a scientist, but it may leave the manager with a

longer list of what might go wrong. In wilderness fire science, the political triggers are much more oriented to fire behavior than to fire effects. The consequences of the long-term effects of fire exclusion, or the severity of an individual fire, are much less likely to be on the manager's radar screen than a fire that escapes a wilderness boundary. While the scientific community has made progress in both the ecological and behavioral domains of wilderness fire, we have still a long way to go. Ironically, one of the important ways we can learn from wilderness fire is to do it and accept the uncertainty in the process. Continued progress can occur in the laboratory and in the computer, but the land is where wilderness fire science must be evaluated. Wilderness fire managers face a real challenge, as even the most successful "people managers" will always be failures at managing the weather. There always will be subtle pressures to avoid a commitment to wilderness fire programs. Successful wilderness fire management will require continued generations of courageous managers (Daniels 1991, Kilgore and Nichols 1995). The success of wilderness fire science depends on it.

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References

- Agee, J. K. 1973. Prescribed fire effects on physical and hydrologic properties of mixed-conifer forest floor and soil. University of California Water Resources Center Contribution 143.
- Agee, J. K. 1981. Initial effects of prescribed fire in a climax lodgepole pine forest: Crater Lake National Park. National Park Service Cooperative Park Studies Unit Rep. CPSU/UW 81-4. College of Forest Resources, University of Washington, Seattle.
- Agee, J. K. 1985. Cost-effective fire management in national parks. In: Brown, J. K.; Kilgore, B. M.; Fischer, W. C.; Mutch, R. W. tech coords. Proceedings: Symposium and workshop on wilderness fire; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 193-198.
- Agee, J. K. 1993. Fire ecology of Pacific Northwest forests. Washington, D.C.: Island Press.
- Agee, J. K. 1995. Alternatives for implementing fire policy. In: Brown, J. K. [and others], tech. coords. Proceedings: Symposium on fire in wilderness and park management; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 107-112.
- Agee, J. K. 1996. The influence of forest structure on fire behavior. In: Proceedings, 17th Annual Conference on Vegetation Management; January 16-18, 1996; Redding, CA: 52-68.
- Agee, J. K. 1997. The severe weather wildfire: too hot to handle? Northwest Science 71:253-256.
- Agee, J.K. 1998. The landscape ecology of western forest fire regimes. Northwest Science (special issue): 24-34.
- Agee, J. K. 1999. A coarse-filter strategy. Forum (Spring 1999): 15-19.
- Agee, J. K.; Flewelling, R. 1983. A fire cycle model based on climate for the Olympic Mountains, Washington. Fire and Forest Meteorology Conference 7: 32-37.
- Agee, J. K.; Huff, M. H. 1986. Structure and process goals for vegetation in wilderness areas. In: Lucas, R. C. 1986. Proceedings: National wilderness research conference: current research. Gen. Tech. Rep. INT-212. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 17-25.
- Agee, J. K.; Clark, D.; Broyles, R.; Rose, L. 1980. Wilderness fire management at Pinnacles National Monument. Fire Management Notes (Winter):10-12.
- Agee, J. K.; Finney, M. A.; deGouvenain, R. 1990. Forest fire history of Desolation Peak, Washington. Canadian Journal of Forest Research 20:350-356.
- Albini, F. A. 1976. Estimating wildfire behavior and effects. Gen. Tech. Rep. INT-30. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 92 p.
- Alexander, M. 1988. Help with making crown fire hazard assessments. In: Protecting people and homes from wildfire in the interior West: proceedings of the symposium and workshop. Gen. Tech. Rep. INT-251. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 147-156.
- Amman, G. 1991. Bark beetle-fire associations in the greater Yellowstone area. In: Nodvin, S. C.; Waldrop, T. A. (eds) Fire in the environment: ecological and cultural perspectives. Gen. Tech. Rep. SE-69. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeast Forest Experiment Station: 313-320.
- Amman, G. D.; Ryan, K. C. 1991. Insect infestation of fire-injured trees in the greater Yellowstone area. Res. Note INT-398. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.
- Andrews, P. 1986. BEHAVE: fire behavior prediction and fuel modeling system: BURN subsystem. Gen. Tech. Rep. INT-194. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.
- Arno, S. F. 1985. Ecological effects and management implications of Indian fires. In: Brown, J. K.; Kilgore, B. M.; Fischer, W. C.; Mutch, R. W. tech coords. Proceedings: Symposium and workshop on wilderness fire; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 81-86.
- Baisan, C. H.; Swetnam, T. L. 1990. Fire history on a desert mountain range: Rincon Mountain Wilderness, Arizona. Canadian Journal of Forest Research 20:1559-1569.
- Baker, G. A.; Rundel, P. W.; Parsons, D. J. 1982. Postfire recovery of chamise chaparral in Sequoia National Park, California. In: Proceedings: Symposium on dynamics and management of Mediterranean-type ecosystems. Gen. Tech. Rep. PSW-58: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 259-261.
- Baker, W. L. 1989. Effect of scale and spatial heterogeneity on fire-interval distributions. Canadian Journal of Forest Research 19:700-706.
- Baker, W. L. 1994. Restoration of landscape structure altered by fire suppression. Conservation Biology 8:763-769.
- Bancroft, L.; Nichols, T.; Parsons, D.; Graber, D.; Evison, B.; van Wagtenonk, J.W. 1985. Evolution of the natural fire management program at Sequoia and Kings Canyon national parks. In: Brown, J. K.; Kilgore, B. M.; Fischer, W. C.; Mutch, R. W. tech coords. Proceedings: Symposium and workshop on wilderness fire; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 174-181.
- Barrows, J. 1951. Fire behavior in northern Rocky Mountain forests. Station Pap. 29. Missoula, MT: U.S. Department of Agriculture, Forest Service, Northern Rocky Mountain Forest and Range Experiment Station. 102 p.
- Bessie, W. C.; Johnson, E. A. 1995. The relative importance of fuels and weather on fire behavior in subalpine forests. Ecology 76:747-762.
- Bevins, C.; Andrews, P. L. 1989. Fire growth simulation model. Final Rep. INT-89417-RJVA. Missoula, MT: Systems for Environmental Management. 5 p.
- Biswell, H. H. 1961. The big trees and fire. National Parks (April): 19-23.
- Biswell, H. H. 1967. Forest fire in perspective. Tall Timbers Fire Ecology Conference 7:43-63.
- Bonnicksen, T. M. 1985. Ecological information base for park and wilderness fire management planning. In: Brown, J. K.; Kilgore, B. M.; Fischer, W. C.; Mutch, R. W. tech coords. Proceedings: Symposium and workshop on wilderness fire; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 168-173.

- Bonnicksen, T. M.; Stone, E. C. 1982. Managing vegetation within U.S. national parks: a policy analysis. *Environmental Management* 6:101-102, 109-122.
- Botkin, D.B.; Janak, J. F.; Wallis, J. R. 1972. Some ecological consequences of a computer model of forest growth. *Journal of Ecology* 60:849-872.
- Botti, S.; Nichols, T. 1995. Availability of fire resources and funding for prescribed natural fire programs in the National Park Service. In: Brown, J. K. [and others], tech. coords. Proceedings: Symposium on fire in wilderness and park management; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 94-105.
- Brown, J. K. 1985. The "unnatural fuel buildup" issue. In: Brown, J. K.; Kilgore, B. M.; Fischer, W. C.; Mutch, R. W. tech coords. Proceedings: Symposium and workshop on wilderness fire; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 127-128.
- Brown, J. K.; Mutch, R. W.; Spoon, C. W.; Wakimoto, R. H. 1995a. Proceedings: Symposium on fire in wilderness and park management; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.
- Brown, J. K.; Arno, S. F.; Bradshaw, L. S.; Menakis, J. P. 1995b. Comparing the Selway-Bitterroot fire program with presettlement fires. In: Brown, J. K. [and others], tech. coords. Proceedings: Symposium on fire in wilderness and park management; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 48-54.
- Brown, A. A.; Davis, K. P. 1973. *Forest fire: control and use*. New York: McGraw-Hill.
- Burgan, R. E.; Rothermel, R. C. 1984. BEHAVE: Fire behavior prediction and fuel modeling subsystem: Fuel subsystem. Gen. Tech. Rep. INT-167. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.
- Butler, D. R.; Walsh, S. J.; Malanson, G. P. 1991. GIS applications to the indirect effects of forest fires in mountainous terrain. In: Nodvin, S. C.; Waldrop, T. A. (eds) *Fire in the environment: ecological and cultural perspectives*. Gen. Tech. Rep. SE-69. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeast Forest Experiment Station: 202-211.
- Carlton, D.; Wittala, M. (no date). RERAP. Portland, OR: U.S. Forest Service, Pacific Northwest Region.
- Chappell, C. B.; Agee, J.K. 1996. Fire severity and tree seeding establishment in *Abies magnifica* forests, southern Cascades, Oregon. *Ecological Applications* 6:628-640.
- Christensen, N. L. 1988. Succession and natural disturbance: paradigms, problems, and preservation of natural ecosystems. In: Agee, J. K.; Johnson, D. J. eds. *Ecosystem management for parks and wilderness*. Seattle, WA: University of Washington Press: 62-86.
- Christensen, N. L. 1991. Variable fire regimes on complex landscapes: ecological consequences, policy implications, and management strategies. In: Nodvin, S. C.; Waldrop, T. A. (eds) *Fire in the environment: ecological and cultural perspectives*. Gen. Tech. Rep. SE-69. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeast Forest Experiment Station: ix-xiii.
- Christensen, N. L.; Cotton, L.; Harvey, T.; Martin, R.; McBride, J.; Rundel, P.; Wakimoto, R. 1987. Review of fire management program for sequoia-mixed conifer forests of Yosemite, Sequoia and Kings Canyon national parks. Final Rep. San Francisco, CA: National Park Service, Western Region.
- Christensen, N. L.; Agee, J. K.; Brussard, P. F.; Hughes, J.; Knight, D. H.; Minshall, G. W.; Peek, J. M.; Pyne, S. J.; Swanson, F. J.; Thomas, J. W.; Wells, S.; Williams, S.; Wright, H. A. 1989. Interpreting the Yellowstone fires of 1988. *Bio-science* 39:678-685.
- Clark, J. S. 1988. Effect of climate change on fire regimes in northwestern Minnesota. *Nature* 334:233-235.
- Cohen, J. 1991. Some thoughts on prescribed natural fires. In: Nodvin, S. C.; Waldrop, T. A. (eds) *Fire in the environment: ecological and cultural perspectives*. Gen. Tech. Rep. SE-69. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeast Forest Experiment Station: 381-383.
- Cortner, H.; Zwolinski, M. J.; Carpenter, E. H.; Taylor, J. G. 1984. Public support for fire management policies. *Journal of Forestry* 82:359-361.
- Cotton, L.; McBride, J. 1987. Visual impacts of prescribed burning on mixed conifer and giant sequoia forests. In: Davis, J.B.; Martin, R. E. tech. coords. Proceedings of the symposium on wildland fire 2000. 1987, April 27-30; South Lake Tahoe, CA. Gen. Tech. Rep. PSW-101. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 32-37.
- Dale, V. H.; Hemstrom, M. 1984. CLIMACS: a computer model of forest stand development for western Oregon and Washington. Gen. Tech. Rep. PNW-327. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.
- Dale, V. H.; Noon, B.; Long, J. L.; Agee, J. K. 1999. Ecological sustainability is fundamental to manage the national forests and grasslands. *Bulletin Ecological Society of America*: 80,3: 207-209.
- Daniels, O. L. 1974. Test of a new land management concept: Fritz Creek 1973. *Western Wildlands* 1:23-26.
- Daniels, O. L. 1991. A forest supervisor's perspective on the prescribed natural fire. *Tall Timbers Fire Ecology Conference* 17:361-366.
- Daniels, O. L.; Mason, L. D. 1985. Management implications of the ignition source in park and wilderness fire management programs. Proceedings: Symposium and workshop on wilderness fire; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 95-97.
- Davis, J. B. 1984. Burning another empire. *Fire Management Notes* 45(4):12-17.
- DeBano, L. F.; Neary, D.G.; Ffolliott, P. F. 1998. *Fire's Effect on Ecosystems*. New York: McGraw-Hill.
- Despain, D. 1985. Ecological implications of ignition sources in park and wilderness fire management programs. In: Proceedings: Symposium and workshop on wilderness fire; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 93-94.
- Despain, D.; Romme, W.H. 1991. Ecology and management of high-intensity fires in Yellowstone National Park. *Tall Timbers Fire Ecology Conference* 17:43-58.
- Drury, W. H.; Nisbet, I. C. T. 1973. Succession. *Journal of the Arnold Arboretum* 54:331-368.
- Egler, F. E. 1954. Vegetation science concepts. I: Initial floristic composition - a factor in old field vegetation development. *Vegetatio* 4:412-417.
- Ewell, P. L. 1995. The Oakland-Berkeley Hills fire of 1991. In: Weise, D. R.; Martin, R. E. tech. coords. The Biswell symposium: fire issues and solutions in urban interface and wildland ecosystems. Gen. Tech. Rep. PSW-GTR-158. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 7-10.
- Finney, M. A. 1995. Fire growth modeling in the Sierra Nevada of California. In: Brown, J. K. [and others], tech. coords. Proceedings: Symposium on fire in wilderness and park management; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 189-191.
- Finney, M. A. 1998. FARSITE: fire area simulator-model development and evaluation. Gen. Tech. Rep. RMRS-RP-4. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Finney, M. A.; Sapsis, D.; Bahro, B. (in press). Use of FARSITE for simulating fire suppression and analyzing fuel treatment economics. In: Proceedings of fire in California ecosystems: integrating ecology, prevention, and management. 1997 November 17-20; San Diego.
- Fischer, W. C. 1980. Prescribed fire and bark beetle attack in ponderosa pine forests. *Fire Management Notes* 41 (Spring):10-12.
- Flannigan, M.D.; Van Wagner, C.E. 1991. Climate change and wildfire in Canada. *Canadian Journal of Forest Research* 21:66-72.
- Fonda, R. W.; Bliss, L. C. 1969. Forest vegetation of the montane and subalpine zones, Olympic Mountains, Washington. *Ecological Monographs* 39:271-301.
- Gara, R. I.; Littke, W. R.; Agee, J. K.; Geiszler, D. R.; Stuart, J. D.; Driver, C. H. 1985. Influences of fires, fungi, and bark beetles on

- development of a lodgepole pine forest in south-central Oregon. In: Baumgartner, D. M. and others (eds) Lodgepole pine: the species and its management. Pullman, WA: Washington State University Press: 153-162.
- Gleason, H. A. 1917. The structure and development of the plant association. Bulletin of the Torrey Botanical Club 43: 463-481.
- Greenlee, J. M. (ed) 1997. Proceedings, First conference on fire effects on rare and endangered species and habitats. Fairfield, WA: International Association of Wildland Fire.
- Grissino-Mayer, H.; Swetnam, T. L. 1995. Effects of habitat diversity on fire regimes in El Malpais National Monument, New Mexico. In: Brown, J. K. [and others], tech. coords. Proceedings: Symposium on fire in wilderness and park management; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 195-200.
- Gruell, G. E. 1985. Indian fires in the interior West: a widespread influence. In: Proceedings: Symposium and workshop on wilderness fire; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 68-74.
- Hartesveldt, R. 1964. Fire ecology of the giant sequoias: controlled fire may be one solution to survival of the species. Natural History 73:12-19.
- Heinselman, M.L. 1973. Fire in the virgin forests of the Boundary Waters Canoe Area. Quaternary Research 3:329-382.
- Henderson, J. A.; Peter, D. H.; Leshner, R. D.; Shaw, D. C. 1989. Forested plant associations of the Olympic National Forest. R6 ECOL Tech. Paper 001-88. Portland, OR: U.S. Department of Agriculture, Forest Service.
- Heyerdahl, E. K. 1997. Spatial and temporal variation in historic fire regimes of the Blue Mountains, Oregon and Washington: the influence of climate. Dissertation: University of Washington, Seattle.
- Hinckley, T. M.; Sprugel, D. G.; Batista, J. L. F.; Brooks, J. R.; Brubaker, L. B.; Compton, J.; Erickson, H. E.; Little, R. L.; Maguire, D. A.; McCarter, J. B.; McKay, S. J.; Pass, D.; Peterson, D. W.; Reed, J. P.; Tacey, W.; Wilkinson, L. E.; Whytemare, A. 1996. Use of the JABOWA family of individual-tree based models for exploration of forest responses to global climate change. Tech. Bull. 717. Research Triangle Park, NC: National Council of the Paper Industry for Air and Steam Improvement, Inc.
- Hunter, M. 1990. Wildlife, forests, and forestry. Engelwood Cliffs, NJ: Prentice-Hall.
- Johnson, E. A. 1992. Fire and vegetation dynamics: studies from the North American boreal forest. Cambridge: Cambridge University Press.
- Johnson, E. A.; Van Wagner, C. E. 1985. The theory and use of two fire history models. Canadian Journal of Forest Research 15:214-220.
- Keane, R. E.; Arno, S. F.; Brown, J. K. 1989. FIRESUM: an ecological process model for fire succession in western conifer forests. Gen. Tech. Rep. INT-266. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.
- Keane, R. E.; Arno, D. F.; Brown, J. K.; Tomback, D. F. 1990. Modelling stand dynamics in whitebark pine (*Pinus albicaulis*) forest. Ecological Modelling 51:73-95.
- Keane, R. E.; Morgan, P.; Running, S. W. 1996a. FIRE-BGC—a mechanistic ecological process model for simulating fire succession on coniferous forest landscapes of the northern Rocky Mountains. Res. Pap. INT-RP-484. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.
- Keane, R. E.; Ryan, K. C.; Running, S. W. 1996b. Simulating effects of fire on northern Rocky Mountain landscapes with the ecological process model FIRE-BGC. Tree Physiology 16:319-331.
- Keane, R. E.; Gartner, J.; Schmidt, K. M.; Long, D. G.; Menakis, J. P.; Finney, M. A. 1998. Development of input data layers for the FARSITE fire growth model for the Selway-Bitterroot Wilderness Complex, USA. Gen. Tech. Rep. RMRS-GTR-3. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Kendall, K. C.; Arno, S. F. 1990. Whitebark pine—an important but endangered wildlife resource. In: Proceedings of the symposium: Whitebark pine ecosystems: Ecology and management of a high mountain resource. March 29-31, 1989; Bozeman, MT. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 264-273.
- Kercher, J. R.; Axelrod, M. C. 1984. A process model of fire ecology and succession in a mixed-conifer forest. Ecology 65:1725-1742.
- Kilgore, B. M. 1971a. response of breeding bird populations to habitat changes in a giant sequoia forest. American Midland Naturalist 85:135-152.
- Kilgore, B. M. 1971b. The role of fire in managing red fir forests. Transactions of 36th North American wildlife and natural resources conference; 1971 March 7-10; Portland, OR Washington, DC: Wildlife Management Institute: 405-416.
- Kilgore, B. M. 1972. Fire's role in a sequoia forest. Naturalist 23:26-37.
- Kilgore, B. M. 1973. The ecological role of fire in Sierran conifer forests: its application to national park management. Quaternary Research 3:496-513.
- Kilgore, B. M. 1985. What is "natural" in wilderness fire management? In: Proceedings: Symposium and workshop on wilderness fire; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 57-67.
- Kilgore, B. M. 1986. The role of fire in wilderness: a state-of-knowledge review. In: Lucas, R. C. 1986 (ed). Proceedings: National wilderness research conference: issues, state-of-knowledge, future directions: 1985 July 23-26; Fort Collins, CO. Gen. Tech. Rep. INT-220. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 70-103.
- Kilgore, B. M.; Briggs, G. 1972. Restoring fire to high elevation forests in California. Journal of Forestry 70:266-271.
- Kilgore, B. M.; Nichols, T. 1995. National Park Service fire policies and programs. In: Brown, J. K. [and others], tech. coords. Proceedings: Symposium on fire in wilderness and park management; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 24-29.
- Kozlowski, T. T.; Ahlgren, C. E. 1974. Fire and ecosystems. New York: Academic Press.
- Kuhn, T.S. 1970. The structure of scientific revolutions. Chicago: University of Chicago Press.
- Leopold, A.S.; Cain, S. A.; Cottam, C. H.; Gabrielson, I. N.; Kimball, T. L. 1963. Wildlife management in the national parks. American Forests 69(4):32-35,61-63.
- Lertzman, K.; Fall, J.; Dorner, B. 1998. Three kinds of heterogeneity in fire regimes: at the crossroads of fire history and landscape ecology. Northwest Science 72 (Special Issue):4-23.
- Lewis, H. 1985. Why Indians burned: specific versus general reasons. In: . Proceedings: Symposium and workshop on wilderness fire; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 75-80.
- Lotan, J. E.; Kilgore, B.M.; Fischer, W. C.; Mutch, R. W. 1985. Proceedings: Symposium on fire in wilderness and park management; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.
- Lucas, R. C. 1986a. Proceedings: National wilderness research conference: issues, state-of-knowledge, future directions: 1985 July 23-26; Fort Collins, CO. Gen. Tech. Rep. INT-220. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.
- Lucas 1986b. Proceedings: National wilderness research conference: current research. Gen. Tech. Rep INT-212. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.
- Mills, T. 1985. Criteria for evaluating the economic efficiency of fire management programs in parks and wilderness. In: Proceedings: Symposium and workshop on wilderness fire; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 182-190.
- Mooney, H.A.; Bonnicksen, T. M.; Christensen, N.; Lotan, J.E.; Reiners, W.A. 1981. Fire regimes and ecosystem properties. Gen. Tech. Rep. WO-26. Washington, D.C.: U.S. Department of Agriculture, Forest Service.
- Mutch, L. S.; Swetnam, T. W. 1995. Effects of fire severity and climate on ring-width growth of giant sequoia after burning. In: Brown, J. K. [and others], tech. coords. Proceedings: Symposium on fire in wilderness and park management; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320. Ogden, UT: U.S.

- Department of Agriculture, Forest Service, Intermountain Research Station: 241-246.
- Mutch, R. 1974. "I thought forest fires were black". *Western Wildlands* 1: 16-23.
- National Park Service (NPS) 1990. *Western Region fire monitoring handbook*. San Francisco, CA: National Park Service, Western Region.
- Nodvin, S. C.; Waldrop, T. A. 1991. *Fire and the environment: ecological and cultural perspectives*. Gen. Tech. Rep. SE-69. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station.
- Odum, E. 1969. The strategy of ecosystem development. *Science* 164:262-270.
- Parsons, D. J. 1976. The role of fire in natural communities: an example from the southern Sierra Nevada. *Environmental Conservation* 3:91-99.
- Parsons, D. J. 1978. Fire and fuel accumulation in a giant sequoia forest. *Journal of Forestry* 76:104-105.
- Parsons, D. J.; Graber, D. M.; Agee, J. K.; van Wagtenonk, J. W. 1986. Natural fire management in national parks. *Environmental Management* 10:21-24.
- Parsons, D. J.; Landres, P. 1998. Restoring fire to wilderness: how are we doing? Pruden, T. L.; Brennan, L. A. (eds) *Fire in ecosystem management: shifting the paradigm from suppression to prescription*. Tall Timbers Fire Ecology Conference 20:366-373.
- Parsons, D. J.; Nichols, H. T. 1986. Management of giant sequoia in the national parks of the Sierra Nevada, California. In: Weatherspoon, C. P.; Iwamoto, Y. R.; Piirto, D. D., tech. coords. *Proceedings of the workshop on management of giant sequoia*; 1985 May 24-25; Reedley, CA. Gen. Tech. Rep. PSW-95. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 26-29.
- Peterson, D. L.; Ryan, K.C. 1986. Modeling post-fire conifer mortality for long-range planning. *Environmental Management* 10:797-808.
- Philpot, C.; Leonard, B. and others. 1989. *Report on fire management policy*. Washington, D.C.: Departments of Agriculture and Interior.
- Pyne, S. J. 1982. *Fire in America: a cultural history of wildland and rural fire*. Princeton, NJ: Princeton University Press.
- Pyne, S. J. 1989. The summer we let wildfire loose. *Natural History* (August):45-49.
- Pyne, S. J. 1995. *World fire: The culture of fire on earth*. Henry Holt. New York.
- Pyne, S. J.; Andrews, P. L.; Laven, R. 1996. *Introduction to wildland fire*. New York: John Wiley and Sons.
- Quigley, T.; Haynes, R. W.; Graham, R. T. (tech eds). 1996. *Integrated scientific assessment for ecosystem management in the Interior Columbia Basin*. Gen. Tech. Rep. PNW-GTR-382. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.
- Rauw, D. 1980. Interpreting the natural role of fire: implications for fire management policy. In: *Proceedings 6th Conference on Fire and Forest Meteorology*. Washington, D. C.: Society of American Foresters: 228-233.
- Rawson, R. P.; Billing, P. R.; Duncan, D. F. 1983. The 1982-83 forest fires in Victoria. *Australian Forestry* 46:163-172.
- Reeburg, P. 1995. The western region fire monitoring handbook. In: Brown, J. K. [and others], tech. coords. *Proceedings: Symposium on fire in wilderness and park management*; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 259-261.
- Reinhardt, E.; Keane, R.E.; Brown, J. K. 1997. FOFEM user's guide. Gen. Tech. Rep. INT-344. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.
- Robertson, W. 1962. Fire and vegetation in the Everglades. *Tall Timbers Fire Ecology Conference* 1:67-80.
- Romme, W. H. 1982. Fire and landscape diversity in subalpine forests of Yellowstone National Park. *Ecological Monographs* 52:199-221.
- Romme, W. H.; Despain, D. 1989. Historical perspective on the Yellowstone fires of 1988. *Bioscience* 39:695-699.
- Romme, W. H.; Turner, M. G. 1991. Implications of global climate change for biogeographic patterns in the greater Yellowstone ecosystem. *Conservation Biology* 5:373-386.
- Rothermel, R. C. 1972. A mathematical model for predicting fire spread in wildland fuels. Res. Pap. INT-115. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.
- Rothermel, R. 1991. Predicting size and behavior of crown fires in the northern Rocky Mountains. Res. Pap. INT-438. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.
- Rundel, P. W.; Parsons, D. J. 1979. Structural changes in chamise (*Adenostoma fasciculatum*) along a fire-induced age gradient. *Journal of Range Management* 32:462-466.
- Rundel, P. W.; Parsons, D. J. 1980. Nutrient changes in two chaparral shrubs along a fire-induced age gradient. *American Journal of Botany* 67:51-58.
- Running, S. W.; Gower, S. T. 1991. FOREST-BGC, a general model of forest ecosystem processes for regional applications. I. Dynamic carbon allocation and nitrogen budgets. *Tree Physiology* 9:147-160.
- Ryan, K.C.; Reinhardt, E. 1988. Predicting postfire mortality of seven western conifers. *Canadian Journal of Forest Research* 18:1291-1297.
- Scott, J.; Reinhardt, E. (in press). Assessing crown fire hazard by linking models of surface and crown fire behavior using Van Wagner's transition criteria. *International Journal of Wildland Fire*.
- Shinneman, D. J.; Baker, W. L. 1997. Non-equilibrium dynamics between catastrophic disturbance and old-growth forests in ponderosa pine landscapes of the Black Hills. *Conservation Biology* 11:1276-1288.
- Stankey, G. 1976. Wilderness fire policy: an investigation of visitor knowledge and beliefs. Res. Pap. INT-180. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 17 p.
- Stephenson, N. L. 1991. Ecology and management of giant sequoia groves. In: *Sierra Nevada Ecosystem Project: Final Report to Congress, Vol. II. Assessments and scientific basis for management options*. Davis: University of California, Centers for Water and Wildland Resources: 1431-1467.
- Stephenson, N.; Parsons, D. J.; Swetnam, T. W. 1991. Restoring natural fire to the sequoia-mixed conifer forest: should intense fire play a role? *Tall Timbers Fire Ecology Conference* 17:321-338.
- Stohlgren, T. J. 1993. Spatial patterns of giant sequoia (*Sequoiadendron giganteum*) in two sequoia groves in Sequoia National Park, California. *Canadian Journal of Forest Research* 23:120-132.
- Sweeney, J. N. 1985. Old burns limit size of fires. In: *Proceedings: Symposium and workshop on wilderness fire*; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 389.
- Swetnam, T. W. 1993. Fire history and climate change in giant sequoia groves. *Science* 262:885-889.
- Swetnam, T. W.; Wickman, B. E.; Paul, H. G.; Baisan, C. H. 1995. Historic patterns of western spruce budworm and Douglas-fir tussock moth outbreaks in the northern Blue Mountains, Oregon, since A.D. 1700. Res. Pap. PNW-RP-484. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.
- Swetnam, T. W.; Betancourt, J. L. 1990. Fire-Southern Oscillation relations in the southwestern United States. *Science* 249:1017-1020.
- Swezy, D. M.; Agee, J. K. 1991. Prescribed fire effects on fine root and tree mortality in old growth ponderosa pine. *Canadian Journal of Forest Research* 21:626-634.
- Tansley, A. G. 1924. The classification of vegetation and the concept of development. *Journal of Ecology* 8:118-149.
- Taylor, J. G.; Daniel, T. C. 1984. Prescribed fire: public education and perception. *Journal of Forestry* 82:361-365.
- Taylor, J. G.; Mutch, R. W. 1986. Fire in wilderness: public knowledge, acceptance, and perceptions. In: Lucas, R. C. 1986. *Proceedings: National wilderness research conference: current research*. Gen. Tech. Rep. INT-212. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 49-59.
- Thomas, D. 1989. A fire behavior analyst in Yellowstone Park. In: Walsh, T. (ed) *Wilderness and wildfire*. Misc. Pub. 50. Missoula, MT: Montana Forest and Range Experiment Station Wilderness Institute: 11-12.

- Thomas, T. L.; Agee, J. K. 1986. Prescribed fire effects on mixed-conifer forest structure at Crater Lake, Oregon. *Canadian Journal of Forest Research* 16:1082-1087.
- Tomback, D. 1982. Dispersal of whitebark pine seeds by Clark's nutcracker: a mutualism hypothesis. *Journal of Ecology* 51:451-467.
- Towle, E. 1985. Fire as forest wilderness policy. In: Frome, M. (ed) *Issues in wilderness management*. Boulder, CO: Westview Press.
- Turner, M. G. 1989. Landscape ecology: the effect of pattern on process. *Annual Review of Ecology and Systematics* 20:171-197.
- Turner, M.G.; Gardner, R. H.; Dale, V. H.; O'Neill, R. V. 1989. Predicting the spread of disturbance across heterogeneous landscapes. *Oikos* 55:121-129.
- Turner, M.G.; Romme, W.L. 1994. Landscape dynamics in crown fire ecosystems. *Landscape Ecology* 9:59-77.
- Urban, D. L.; Bonan, G. B.; Shugart, H. H. 1991. Spatial applications of gap models. *Forest Ecology and Management* 42:95-110.
- Van Wagner, C. E. 1977. Conditions for the start and spread of crown fire. *Canadian Journal of Forest Research* 7:23-34.
- Van Wagner, C. 1985. Does nature really care who starts the fire? In: Brown, J. K. [and others], tech. coords. *Proceedings: Symposium on fire in wilderness and park management; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 98-100.
- Van Wagner, C. E. 1987. Forest fire research – hindsight and foresight. In: Davis, J.B.; Martin, R. E. tech. coords. *Proceedings of the symposium on wildland fire 2000. 1987, April 27-30; South Lake Tahoe, CA. Gen. Tech. Rep. PSW-101*. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 115-120.
- van Wagtenonk, J. W. 1972. Fire and fuel relationships in mixed conifer ecosystems of Yosemite National Park. Dissertation. Berkeley, CA: University of California.
- van Wagtenonk, J. W. 1974. Refined burning prescriptions for Yosemite National Park. Occasional Paper 2. Washington, D.C.: U.S. Department of the Interior, National Park Service. 21 p.
- van Wagtenonk, J. W. 1978. Wilderness fire management in Yosemite National Park. In: Schofield, E. A. ed., *EARTHCARE: global protection of natural areas*. Boulder, CO: Westview Press: 324-335.
- van Wagtenonk, J. W. 1985. Fire suppression effects on fuels and succession in short-fire-interval wilderness ecosystems. In: *Proceedings: Symposium and workshop on wilderness fire; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 119-126.
- van Wagtenonk, J. W. 1991a. The evolution of National Park Service fire policy. In: Nodvin, S. C.; Waldrop, T. A. (eds) *Fire in the environment: ecological and cultural perspectives*. Gen. Tech. Rep. SE-69. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeast Forest Experiment Station: 328-332.
- van Wagtenonk, J. W. 1991b. GIS applications in fire management and research. In: Nodvin, S. C.; Waldrop, T. A. (eds) *Fire in the environment: ecological and cultural perspectives*. Gen. Tech. Rep. SE-69. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeast Forest Experiment Station: 212-214.
- van Wagtenonk, J. W. 1995. Large fires in wilderness areas. In: Brown, J. K. [and others], tech. coords. *Proceedings: Symposium on fire in wilderness and park management; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 113-116.
- van Wagtenonk, J. W. 1996. Use of a deterministic fire growth model to test fuel treatments. In: *Sierra Nevada Ecosystem Project: Final Report to Congress, Vol. II. Assessments and scientific basis for management options*. Davis: University of California, Centers for Water and Wildland Resources: 1155-1165.
- Wakimoto, R. 1989. Wilderness fire policy—"Let it what?". In: Walsh, T. (ed) *Wilderness and wildfire. Misc. Pub. 50*. Missoula, MT: Montana Forest and Range Experiment Station Wilderness Institute: 4-6.
- Weir, J. M. H.; Chapman, K. J.; Johnson, E. A. 1995. Wildland fire management and the fire regime in the southern Canadian Rockies. In: Brown, J. K. [and others], tech. coords. *Proceedings: Symposium on fire in wilderness and park management; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 275-280.
- White, P. S. 1979. Pattern, process, and natural disturbance in vegetation. *Botanical Review* 45:229-299.
- White, P. S.; Pickett, S. T. A. 1985. Natural disturbance and patch dynamics: an introduction. In: Pickett, S. T. A.; White, P. S. (eds) *The ecology of natural disturbance and patch dynamics*. New York: Academic Press.
- Wierzchowski, J. L.; White, C. A.; Pengelly, I. R. 1995. Evaluation of the Banff National Park planned ignition prescribed fire program. In: Brown, J. K. [and others], tech. coords. *Proceedings: Symposium on fire in wilderness and park management; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 281-283.
- Worff, W. A. 1985a. Wilderness management: a historical perspective on the implications of human-ignited fire. In: Brown, J. K.; Kilgore, B. M.; Fischer, W. C.; Mutch, R. W. tech coords. *Proceedings: Symposium and workshop on wilderness fire; 1983 November 15-18; Missoula, MT. Gen. Tech. Rep. INT-182*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 276-182.
- Worff, W. A. 1985b. Fire and the wilderness purpose. In: Frome, M. (ed) *Issues in wilderness management*. Boulder, CO: Westview Press.
- Wright, H.; Bailey, R. 1982. *Fire ecology of the United States and southern Canada*. New York: John Wiley and Sons.
- Wright, C. S. 1996. Fire history of the Teanaway River drainage, Washington. M.S. thesis, University of Washington, Seattle.
- Yamaguchi, D.K.; Atwater, B. F.; Bunker, D. E.; Benson, B. E.; Reid, M. S. 1997. Tree-ring dating the 1700 Cascadia earthquake. *Nature* 389:922.

Recreation Impacts and Management in Wilderness: A State-of-Knowledge Review

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Abstract—This paper reviews the body of literature on recreation resource impacts and their management in the United States, with a primary focus on research within designated wildernesses during the past 15 years since the previous review (Cole 1987b). Recreation impacts have become a salient issue among wilderness scientists, managers and advocates alike. Studies of recreation impacts, referred to as *recreation ecology*, have expanded and diversified. Research has shifted its focus more towards questions driven by wilderness and park planning frameworks such the Limits of Acceptable Change and the Visitor Experience and Resource Protection. This paper begins by providing an overview of recreation impacts and their significance in wilderness, followed by a review of research approaches and methods. Major findings from recent studies are summarized. The contribution of this knowledge base to management decisionmaking and practices is examined. The paper concludes with a discussion of major knowledge gaps and suggested areas for future research.

The passage of the Wilderness Act in 1964 and the creation of the National Wilderness Preservation System (NWPS) marked a milestone in nature conservation in the United States. The system has expanded from 54 units and 9 million acres at its inception to 624 wilderness areas and 104 million acres by 1998 (Landres and Meyer 1998).

The Wilderness Act recognizes the value of wilderness recreation and specifies that unconfined and undeveloped recreational opportunities are to be provided in wilderness areas as a legitimate type of use. Results from recent recreation trends studies show that wilderness visitation has experienced impressive growth during the past three decades (Cole 1996). Hiking, overnight camping, wildlife viewing, horseback riding and nature study remain popular activities, and participation in more specialized activities, such as caving and rock climbing, is increasing. In-depth discussion of wilderness recreational use and user trends is provided in another state-of-knowledge review (Watson, this volume).

Continued growth in recreational use in wilderness has tremendous environmental, economic and social implications. This paper focuses on the environmental challenges

wilderness managers face in addressing a large and expanding number of recreationists and their associated impacts. Sustaining current use and accommodating future growth in wilderness visitation while achieving an appropriate balance with resource protection presents a considerable challenge.

Scope and Definitions

Several definitions and limitations are provided here to clarify this discussion. The term *impact* is used to denote any undesirable visitor-related biophysical change of the wilderness resource. Social impacts are excluded from this review. The scope of this paper is generally limited to studies conducted in wildernesses designated by Congress. However, research studies from similar backcountry areas outside the NWPS are occasionally included for comparison. Active research in recreation impacts exists in other countries such as Australia, Britain, Canada and New Zealand, but this body of international literature deserves a separate review. Finally, this paper limits its scope to recreation impacts generated from within wilderness boundaries, although recreational use and development outside wilderness boundaries can pose an external threat to the integrity of wilderness resources (Cole and Landres 1996).

The Field of Recreation Ecology

Negative impacts on wilderness are an inevitable consequence of recreation. Even the most thoughtful visitors would leave footprints and unintentionally disturb wildlife. As recreation is a legitimate use in wilderness areas, the issue for managers is at what level do resource impacts become unacceptable based on wilderness management goals and mandates.

Recreation activities can cause impact to all resource elements in a wilderness ecosystem. Soil, vegetation, wildlife and water are four primary components that are affected (Table 1). Because various ecological components are inter-related, recreation impact on a single ecological element can eventually result in effects on multiple components (Hammit and Cole 1998). The scientific study of recreation impacts, also referred to as *recreation ecology*, is a research response to the knowledge gaps and information needs about ever-growing visitor impacts in wilderness as well as other protected areas.

Recreation ecology can be defined as the field of study that examines, assesses and monitors visitor impacts, typically to protected natural areas, and their relationships to influential factors (Hammit and Cole 1998; Liddle 1997; Marion 1998). Such knowledge can help managers identify and evaluate resource impacts, facilitating understanding of causes and

In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management; 1999 May 23–27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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Table 1—Common forms of recreation impacts in wilderness.

	Ecological component			
	Soil	Vegetation	Wildlife	Water
Direct effects	Soil compaction	Reduced height and vigor	Habitat alteration	Introduction of exotic species
	Loss of organic litter	Loss of ground vegetation cover	Loss of habitats	Increased turbidity
	Loss of mineral soil	Loss of fragile species	Introduction of exotic species	Increased nutrient inputs
		Loss of trees and shrubs	Wildlife harassment	Increased levels of pathogenic bacteria
		Tree trunk damage	Modification of wildlife behavior	Altered water quality
		Introduction of exotic species	Displacement from food, water and shelter	Reduced health of aquatic ecosystems
Indirect/ derivative effects	Reduced soil moisture	Composition change	Reduced health and fitness	Composition change
	Reduced soil pore space	Altered microclimate	Reduced reproduction rates	Excessive algal growth
	Accelerated soil erosion	Accelerated soil erosion	Increased mortality	
	Altered soil microbial activities		Composition change	

effects and improving insights regarding the prevention, mitigation and management of problems. In a broader sense, recreation ecology may be conceived as the study of ecological interrelationships between humans and the environment in recreation/tourism contexts (Leung and Marion 1996; Wagar 1964). Under this broader definition, recreation ecologists are also interested in how environmental attributes influence the availability and quality of recreation opportunities.

Recreation ecology began in the 1920s and '30s (Bates 1935; Meinecke 1928), although earlier observations of visitor impacts are available (Liddle 1997). However, substantial scientific work in this field did not occur until the late 1960s, when backcountry and wilderness visitor use in the United States increased sharply, along with associated resource impacts. A modest body of literature accumulated during the ensuing two decades and several conferences devoted specifically to recreation impacts were held (Bayfield and Barrow 1985; Ittner and others 1979; IUCN 1967). Since the mid-1980s, the study of recreation ecology has been expanding, diversifying and shifting its focus (Table 2).

Results of recreation ecology research in wilderness are disseminated in various forms, including scientific journals, conference proceedings and management reports. Some of the common journal outlets include *Biological Conservation*, *Environmental Conservation*, *Environmental Management*, *International Journal of Wilderness*, *Journal of Applied Ecology*, *Journal of Environmental Management* and *Journal of Soil and Water Conservation*. As findings and knowledge accumulated from these studies, monographs that synthesized the research literature and

management applications of recreation ecology began to appear (Edington and Edington 1986; Hammitt and Cole 1998; Knight and Gutzwiller 1995; Kuss and others 1990; Liddle 1997). Knowledge generated from research has also been applied to the management of wilderness resources and visitors, although many of these applications have not been documented in the published literature.

Cole (1987b) provided a succinct account of the historical development of recreation ecology, noting that there was only a small group of scientists who consistently conducted studies in this field. Fifteen years have passed since this

Table 2—The development and major events of recreation ecology research.^a

Approximate time period	Development/event(s)
1990s	Refinement of methods; new topics and perspectives
1980s	Integration with management frameworks
1970s	Period of active research
1960s	Period of rapidly increasing use and impact
1940-50s	First scientific studies in the United States
1930s	First experimental trampling studies in the United Kingdom
1920s	Early observations and descriptions of the problem

^aPartly based on Cole (1987b).

review, and Cole's statement remains valid. The size of the research community in this field is still not commensurate with the extent of the problems. Currently, the study of recreation impacts and their management attracts a growing yet still small number of scientists or students, even though wilderness and other resource managers increasingly require visitor impact assessment and management assistance.

Recreation Ecology Research in Wilderness

Generally, recreation ecology studies in wilderness have enjoyed better support from the USDA Forest Service, primarily at the interagency Aldo Leopold Wilderness Research Institute (formerly Wilderness Research Unit of the Intermountain Research Station). As a result, the majority of recreation ecology studies have been conducted in wilderness areas managed by the Forest Service. Less research has been conducted in USDI National Park Service-managed wilderness areas, with some notable exceptions, such as Shenandoah and Yosemite National Parks. Very little research has been conducted in wildernesses managed by USDI Bureau of Land Management and USDI Fish and Wildlife Service.

David Cole, Forest Service, has produced a substantial number of publications and has been influential in the building of a recreation ecology knowledge base. Jeffrey Marion, Virginia Tech Cooperative Park Studies Unit (USGS Patuxent Wildlife Research Center), has conducted numerous recreation ecology studies in national parks, with a primary focus on refining impact assessment, monitoring and management techniques. A smaller institutional research effort is supported by the National Outdoor Leadership School (NOLS), led by Christopher Monz. Recreation ecology studies are also conducted by faculty members and graduate students at several academic institutions such as Clemson University, Colorado State University, North Carolina State University, University of Idaho, University of Montana and Virginia Tech.

The Significance of Recreation Impacts

Why should we care about recreation impacts? Recreation impacts are significant because they reflect success in meeting two primary legal mandates: resource protection and recreation provision. Derived from the Wilderness Act, these mandates state that wilderness areas "shall be administered for the use and enjoyment of the American people in such manner as will leave them unimpaired for future use and enjoyment as wilderness, and so as to provide for the protection of these areas [and] the preservation of their wilderness character..." (Public Law 88-577, 1964). The Wilderness Act thus identifies two concerns relative to recreation impacts: (1) protection of the integrity of wilderness environments, and (2) protection of the quality of recreational experiences. A minimal system of trails and campsites is generally viewed as essential to support recreational use of wilderness. Wilderness managers must

therefore be willing to accept some degree of resource degradation associated with the creation, maintenance, and use of these recreation facilities. However, excessive resource degradation of facilities and the proliferation of user-created trails or unnecessary campsites are viewed as unacceptable.

The managerial significance of recreation impacts is also reflected in the substantial costs incurred by managing agencies to construct, maintain and rehabilitate trails and campsites, and to operate visitor management programs. While some of these costs reflect provisions for recreational use, many are directed at avoiding or minimizing recreation impacts. For example, a trail both facilitates wilderness travel and concentrates recreation traffic and impact along a single narrow tread designed and maintained to minimize resource impacts.

Resource Protection

How and to what extent recreation impacts affect the integrity of wilderness environments and natural processes have not been thoroughly examined. We do know that many wilderness areas have extensive networks of trails and campsites which are frequently in poor condition (Marion and others 1993; Washburne and Cole 1983). Cole (1990a) suggests that impacts which seriously disrupt ecosystem function and that either occur over very large areas or affect rare ecosystems are most significant. In particular, long-term or irreversible changes are problematic.

Several studies show that recreation impacts relatively small proportions of wilderness areas. For example, campsite monitoring at the heavily visited Great Smoky Mountains National Park (83% of which is recommended for and managed as wilderness) located and assessed 327 backcountry campsites and shelters with an aggregate disturbed area of 550,824 ft² (Marion and Leung 1997). The Park's 930 miles of trails contribute an additional 9,820,800 ft² of recreation-related disturbance, assuming a conservative average trail width of two feet. While these values may seem large, they represent only .05 percent of the Park's total acreage. Campsite monitoring surveys of six less visited wilderness areas in Virginia's Jefferson National Forest revealed camping had disturbed only .0007 to .015 percent of the wilderness (Leung and Marion 1995). Vegetation disturbance resulting from use of areas adjacent to campsites and trails would likely only double or triple these areal estimates.

While recreation impacts directly affect small percentages of wilderness areas, the effects are usually distributed unevenly due in part to visitor use patterns (Lucas 1990b), with intensive disturbance in some places and less intensive disturbance in surrounding areas. However, even localized impact can harm rare or endangered species, damage sensitive resources or diminish ecosystem health. For example, the collection and burning of firewood in desert ecosystems and at high elevations, where wood production is low, can disrupt nutrient cycling critical to plants that depend upon organic matter and nutrients contained in woody debris (Fenn and others 1976). Furthermore, certain forms of impact (such as soil loss) and certain environments (such as alpine meadows) have extremely low resource recovery rates, requiring long periods to recover from even limited degradation (Liddle 1997).

Visitor impacts may also extend far beyond localized use areas (Cole 1990a). Hunting and fishing directly alter the abundance, distribution and demographics of wildlife and can lead to changes in the relative abundance and composition of nongame fauna and flora (Knight and Cole 1991). The introduction and stocking of fish, particularly introduced species, alter aquatic food webs and have been cited as a contributing cause to the decline of native species (Liddle 1997). Similarly, the introduction of exotic plant species in wilderness is widespread, and some naturalized species are able to alter plant dynamics over large areas (Marion and others 1986). Other examples include stream sedimentation from trail and campsite erosion, which reduces the quality of aquatic habitats for insect and fish populations.

The mere presence of visitors may harm wildlife by displacing them from essential habitats or disrupting their raising of young (Knight and Cole 1995; Liddle 1997). Trail networks and campsites may cause a landscape fragmentation effect similar to that of roads, possibly interfering with movement of some animal species (Noss and Cooperrider 1994).

Impacts to Visitors

Recent studies suggest that perceived impacts can degrade the quality of visitor experiences (Roggenbuck and others 1993; Vaske and others 1982). Perceptions are based on how visitors believe impacts affect the overall attributes of the setting like scenic appeal or solitude, and whether or not the impacts are considered to be undesirable (Lucas 1979; Whittaker and Shelby 1988). Visitors appear to be more sensitive to impacts caused by inappropriate behavior, such as litter and tree damage, and to particularly obtrusive examples of physical impacts, such as badly exposed tree roots.

Surveys of wilderness visitors reveal considerable variability in visitor responses to recreation impacts. While several earlier studies found that visitor satisfaction was not diminished by trail and campsite impacts (Knudson and Curry 1981; Lucas 1979), Roggenbuck and others (1993) reported that littering and human damage to campsite trees were among the most highly rated indicators affecting the quality of wilderness experiences. Similarly, wilderness visitors rated ground vegetation loss and bare ground on campsites as two important determinants of their satisfaction (Hollenhorst and Gardner 1994).

The mere presence of trails and campsites, particularly those in degraded condition, also remind visitors of those that preceded them. The proliferation and high densities of trails and campsites in popular locations give wilderness a "soiled" or "used" appearance, in contrast to the ideal of a pristine wilderness. Particularly in remote areas, the discovery of even a single trail or campsite can diminish opportunities for solitude.

Impacts associated with a specific type of use may intensify perceived crowding and conflict between different visitors or groups (Vaske and others 1982). For example, horse manure or excessive muddiness on trails or trash at hunting camps might provoke negative impressions about horseback riders among other wilderness users. Such negative reactions could polarize user groups and lead to tensions with land managers.

Finally, recreation impacts such as trail rutting and excessive muddiness can provoke visitor dissatisfaction by increasing the difficulty of hiking and making it an unpleasant experience. Such impacts may also jeopardize visitor or packstock safety and increase agency liability.

Research Methods

Since the previous review (Cole 1987b), there has been a steady increase in the diversity and sophistication of research methods employed to investigate recreation resource impacts in wilderness. Research methods range from simple qualitative descriptions of impact conditions to controlled laboratory experiments with elaborate experimental designs. Some studies involved intensive and sophisticated measurements but included only a limited number of sample sites. Other studies encompassed a large number of sample sites distributed over a large landscape but often involved rapid field observations and measurements. Studies of various approaches and designs generally complement each other in developing a thorough understanding of recreation impacts. These studies, if well designed and executed, can yield useful data for wilderness managers. The choice of methods is essentially based on the research questions asked, types of data needed, character of study area, the training of investigators and logistical constraints.

Major Research Questions and Themes

1. What types of recreation impact exist?

Previous studies have documented the obvious and direct forms of recreation impact, including the area of disturbance, tree damage, soil exposure, soil erosion, vegetation loss, trash, human waste and wildlife disturbance. Among these, soil and vegetation attributes are most frequently measured (Hammit and Cole 1998). Less attention has been paid to less visible environmental qualities, such as bacteriological water quality, soil microbial communities and wildlife physiology. However, the number of studies on these ecological components has been increasing in recent years (Knight and Gutzwiller 1995; Zabinski and Gannon 1997).

Indirect or secondary effects of recreational use, such as increased predation rates on wildlife displaced by recreation visitation, have seldom been examined. In addition, the types of recreation impacts examined have been restricted in spatial, temporal and ecological scales (Cole and Landres 1996). Few studies have investigated ecosystem or landscape-level effects. As the popularity of non-conventional types of recreational activity and equipment increases, new forms of recreation impact are likely, which will require further research, assessment and monitoring. Caving, rock climbing, llamas as pack animals, and use of hiking poles are some more common examples.

2. What is the magnitude and significance of recreation impacts?

Knowledge of the magnitude of impacts is needed to evaluate their ecological and social significance and acceptability, and to prioritize management and maintenance needs. The magnitude of recreation impacts is often judged by two components: the intensity of impact and the spatial qualities

of impact (Clark and Stankey 1979; Cole 1994). The assessment of impact intensity has received more attention than the spatial component (Cole 1989c). Examples of spatial qualities include spatial extent, distribution and association of impacts. Spatial extent is perhaps the most examined spatial quality, although recent studies have begun to investigate the distribution of impacts in space (Cole 1993a; Leung and Marion 1998; McEwen and others 1996).

As mentioned earlier, a number of studies have examined the social significance of recreation impacts (Knudson and Curry 1981; Marion and Lime 1986; Roggenbuck and others 1993; Shelby and Shindler 1992; Shelby and others 1988). Two important issues—perception and acceptability of impacts to visitors and managers—are beyond the scope of this paper.

3. What is the relationship between amount of use and intensity of impact?

Research addressing this question was highlighted by the concept of carrying capacity and its application to recreation and park management. One objective of this large body of research has been to determine a threshold level of use beyond which recreation impacts will intensify. Unfortunately, these studies often concluded that the use-impact relationship is both complex and situational, depending on a diverse array of environmental and social factors. Recognizing limitations of the traditional carrying capacity model, recent work has been redirected at determining appropriate indicators and standards that reflect explicit levels of acceptable impacts. A detailed discussion on recreation carrying capacity is provided in another state-of-knowledge review (Manning and Lime, this volume).

4. What factors contribute to the problem?

Although amount of use is the most studied factor influencing recreation impacts, other use-related and environmental factors interact to determine the intensity and extent of impacts (Hammit and Cole 1998; Leung and Marion 1996). Visitor and site management actions can moderate many of these factors and thus influence the quality of impacts (Marion 1995).

5. Have conditions worsened or improved over time?

Recent studies have examined trends of recreation impacts over time. The increasing availability of long-term monitoring data sets permits such analyses. Examples include trail monitoring (Cole 1991), campsite monitoring (Cole 1993a; Cole and Hall 1992) and a 30-year trampling/trail study in Glacier National Park (Hartley 1999).

6. How effective are visitor and site management actions?

As wilderness managers implement various visitor and site management actions to reduce or contain resource impacts, they need to know which actions have the greatest chance of success (Hammit and Cole 1998). An example is the national Leave No Trace (LNT) outdoor skills and ethics program. Little research has been conducted to evaluate the effectiveness of recommended LNT practices in reducing the intensity and extent of impact.

7. How can research and impact assessment methods be improved?

Methodological improvements address the accuracy and precision of different methods, as well as the need to make procedures more efficient. The possibility of reducing the number of indicators for campsite assessment and monitoring has been addressed (Gettinger and others

1998; Leung and Marion 1999b), as has the choice of sampling interval for trail assessment and monitoring (Leung and Marion 1999c).

Research Approaches and Designs

A substantial number of recreation ecology studies during the past three decades were associated with the carrying capacity framework (Sumner 1942; Wagar 1964). Research approaches and methods were developed for evaluating the relationship between amount of use and intensity of impact. Another group of studies has evaluated relationships between environmental attributes and the quality of recreation impacts. For instance, a significant portion of trail research was devoted to environmental influence on trail degradation, including soil compaction, trail widening and soil erosion (Leung and Marion 1996). Experimental studies on trampling effects have also been conducted to evaluate the relative resistance and resilience of various vegetation types (Cole 1988; Cole 1993b; Cole 1995b; Cole 1995c; Marion and Cole 1996). Most recently, with the increasing adoption and implementation of the *Limits of Acceptable Change* (LAC) framework (Stankey and others 1985), the *Visitor Impact Management* framework (Graefe and others 1990) and the *Visitor Experience and Resource Protection* (VERP) framework (National Park Service 1997a; National Park Service 1997b), recreation ecology studies have begun to focus on the selection of indicators, standards and monitoring protocols to support these management planning processes (Belnap 1998).

Cole (1987b) discussed the following four major study designs in recreation ecology studies (Table 3). The ability of these designs to isolate cause and effect varies.

1. Descriptive surveys of recreation sites.
2. Comparisons of used and unused sites.
3. Before-and-after natural experiments.
4. Before-and-after simulated experiments.

Trampling and wildlife impact studies tend to adopt before-and-after experimental designs with controls, while trail and campsite condition assessments often adopt the first two designs with few exceptions (Cole 1995a). A large number of recent studies were still conducted within a short time-frame, although more long-term assessment and monitoring studies on recreation impacts have emerged.

In addition to these four types of research design, a few conceptual and simulation studies have been published (Cole 1992; Leung and Marion 1999c). Such studies are likely to increase with continued advancements and expanding application of geographic information systems (GIS) and statistical software programs.

Research Methods and Techniques

Research methods for four specific topics are discussed in this subsection. These topics, which include trampling studies, trail impacts, campsite impacts, and indicators and indices, are highlighted because they constitute a large portion of the recreation ecology literature.

Trampling Research—Trampling studies are often regarded as basic research in recreation ecology (Liddle 1997).

Table 3—Four common study designs employed in recreation ecology research with recent examples.^a

Study design	Description	Recent example(s)
Descriptive surveys	Estimates or measurements are taken on recreation sites to assess current resource conditions	<i>Trails and Campsites</i> : Cole and others (1997); Rochefort and others (this volume)
Comparison of used and unused sites	Measurements are taken on recreation sites and nearby undisturbed sites (control) and compared to infer amount of impact	<i>Trails</i> : Hall and Kuss (1989) <i>Campsites</i> : Marion and Leung (1997); Monz (1998); Zabinski and Gannon (1997)
Before-and-after natural experiments	Measurements are taken before and after (1) commencing or ceasing use of sites, or (2) applying management action(s) to sites to infer amount of impact due to the change	<i>Trails</i> : Doucette and Kimball (1990) <i>Campsites</i> : Marion (1995); Spildie and others (this volume)
Before-and-after simulated experiments	Measurements are taken before and after treatments (including known type, frequency and intensity of use) are applied, often with random assignment, to infer amount of impact due to the treatment	<i>Trampling</i> : Cole (1993b, 1995d); Cole and Spildie (1998); Hartley (1999) <i>Trails</i> : DeLuca and others (1998) <i>Campsites</i> : Cole (1995a)

^aPartly based on Cole (1987b).

As such, experimental designs usually employ varying trampling intensities, randomly assigned to replicated experiment plots or lanes. Known intensities or frequencies of trampling are applied by artificial or human trampers.

Most trampling studies have been directed at the relationship between amount of use and intensity of impact and the different susceptibilities of plant species or vegetation types. A few studies have assessed the effects of different types of trampers, such as human and horses. Recent trampling studies have included new use-related variables such as shoe type and trampling weights (Cole 1995d) and emerging types of use such as llamas (Cole and Spildie 1998).

The designs of these trampling studies varied significantly across different studies, limiting valid comparisons (Bayfield and Aitken 1992; Kuss 1986a). In response to the need for standardized procedures, trampling experiment protocols and guidelines have been proposed (Cole and Bayfield 1993).

Methods for Studying Trail Impacts—Early research on trail impacts focused on impact severity and environmental factors affecting trail degradation (Leung and Marion 1996). Very few data sets exist on temporal change of trail conditions, with an exception of a 11-year trail assessment conducted in the Selway-Bitterroot Wilderness of Montana (Cole 1991). A variety of trail assessment and monitoring techniques have been developed (Cole 1983), which can be classified into three approaches (Table 4). These techniques, many of which have been applied to wilderness, include condition class assessments (Cole and others 1997), evaluation of aerial photos (Coleman 1977; Price 1983) and quantitative measurements and experiments (Bratton and others 1979; DeLuca and others 1998; Hall and Kuss 1989). Improving some of these methods has been the subject of several

recent studies. In the Eastern U.S., a problem-assessment method was developed and applied to Great Smoky Mountains National Park (Leung and Marion 1999a; Marion 1994a). The sampling issue of trail assessment methods has also been examined (Leung and Marion 1999c).

In Montana, the influence of use type on trail erosion was examined using trampling and rainfall simulation experiments (DeLuca and others 1998). Intrusion experiments were also conducted in several studies by Gutzwiller and his colleagues to examine disturbance of birds by walkers on existing trails or trailless experiment sites (Gutzwiller and Anderson 1999; Gutzwiller and others 1998; Gutzwiller and others 1994; Riffell and others 1996).

Methods for Studying Camping Impacts—Due to activity concentration and duration of stay, campsites receive the highest level of visitor impacts, particularly those related to inappropriate behavior. Campsite impact assessment approaches range from condition class (Frissell 1978) and photographic approaches (Magill 1989) to more intensive quantitative measurements (Table 5). These procedures provide managers with objective data on campsite conditions, both at a general level (reconnaissance approach) and for individual resource indicators (multiple-indicator approach). Replicating procedures allow monitoring of changes in campsite conditions, which can be used to document trends in site conditions and to evaluate the effectiveness of management actions.

Interrelationships between campsite impacts and use-related or environmental factors often require the application of more complex research designs. An interrelated set of recreation ecology studies within backcountry zones of three Eastern national parks provides an example (Cole and Marion 1988; Marion and Cole 1989; Marion and Cole 1996).

Table 4—A summary of different trail impact assessment and monitoring approaches and designs.

Item	Reconnaissance approach		Sampling-based approach		Census-based approach	
	Condition class	Photo appraisal	Point sampling	Point-quadrat sampling	Sectional evaluation	Problem assessment
Implementation	Descriptive classes are defined and assigned to trails/segments	Trails are identified and evaluated from aerial photos	Measurements are performed at a series of points along a trail that is determined by a sampling scheme	Measurements are performed within quadrats at a series of points that is determined by a sampling scheme	Trail is divided into sections; evaluation is made for each section	Impact problems are defined, followed by complete census of these problems
Unit of observation	Segment/trail	Trail/regional	Site (point)	Site (quadrat)	Segment	Dimension of impact problem
Typical data type(s)	Nominal/ordinal	Interval/ratio	Interval/ratio	Interval/ratio	Ordinal/percentage	Interval/ratio
Major utility	Prompt assessment of trail conditions	Detect proliferation of trail networks; detect new trails	Quantitative data for statistical analysis; adaptable to management frameworks	Quantitative data for statistical analysis; adaptable to management frameworks	Prompt assessment of trail conditions and their spatial variations	Data on the frequency, extent, and distribution of impacts; adaptable to management frameworks
Limiting factor(s)	Singular qualitative measure; conflicting criteria within a condition class	Availability; resolution of aerial photos; photo interpretation skills	Relocation of sampling points; measurement error; field time	Relocation of sampling points; measurement error; field time	Definition of section; scale dependence of results	Quantitative definition of impact problems; interrater variability
Examples	Cole and others (1997)	Coleman (1977); Price (1983)	Cole (1991)	Hall and Kuss (1989)	Bratton and others (1979)	Marion (1994a); Leung and Marion (1999a)

Multiple-indicator measurements taken on campsites and paired control sites over five years were recorded and analyzed to evaluate the effect of: (1) different amounts and types of use, (2) different environmental settings, (3) temporal variation in vegetation and soil conditions, (4) initial degradation following campsite creation, and (5) initial recovery following campsite closure.

In the past 15 years, refinement of campsite impact assessment procedures for monitoring has received more emphasis. This work has been driven by management needs for longitudinal data to support management planning frameworks and decisionmaking. Refinement has occurred through numerous applications of these procedures in the Western (Cole 1993a; Gettinger and others 1998), Central (McEwen and others 1996; Williams and Marion 1997; Farrell and Marion 1997) and Eastern U.S. (Cole and Marion 1988; Leung and Marion 1995; Marion 1991; Marion 1994b; Marion and Leung 1997; Marion and Snow 1990; Williams and Marion 1995). Attempts have been made to standardize campsite assessment procedures (Marion 1991). There have also been refinements of assessment and analytical procedures and adaptation of assessment procedures to different environment types (Gettinger and others 1998; Leung and Marion 1999b; Monz 1998).

Impact Indicators and Indices—To a large extent the increased emphasis on indicators and indices over the past 15 years was a direct result of the adoption and implementation of standards-based management frameworks such as LAC and VERP. Judicious selection and periodic monitoring of indicators are critical components in these management frameworks.

An indicator may be broadly defined as an important quality that indicates resource change due to recreational use. Watson and Cole (1992) and Merigliano (1990) provided reviews and examples of indicators adopted or proposed in the wilderness management literature. Examples include amount of bare ground on a campsite, number of cut trees, incision depth of a trail and flush distance of an avian species.

In contrast, an index is generally referred to as a mathematical combination of two or more indicators (Westman 1985). They are constructed to simplify and facilitate the communication and evaluation of results. These impact indices may be classified into four groups. First, indices of impact intensity are constructed to represent the severity of environmental damage. Two examples are floristic dissimilarity and cover alteration (Cole 1978; Cole 1993b). Shannon-Wiener species diversity index (H) and community similarity index,

Table 5—A summary of different campsite impact assessment and monitoring approaches and designs.

Item	Reconnaissance approach		Multiple-indicator approach	
	Condition class	Photo appraisal	Ratings	Quantitative measurement
Implementation	Descriptive classes are defined and assigned to each campsite	Site photo is taken and evaluated for each campsite	Assessment at ordinal scale is made on each selected indicator on a campsite	Measurement is taken for each selected indicator on a campsite
Typical data type(s)	Nominal/ordinal	Interval/ratio	Ordinal	Interval/ratio
Major utility	Prompt characterization of campsite conditions	Visualize campsite conditions; relocation	Efficient field work; minimal training required	Accurate and precise; permit quantitative analysis; allow aggregate measures; adaptable to management frameworks
Limiting factor(s)	Singular measure; conflicting criteria within a condition class	Scale and quality of aerial photos; photo interpretation skills	Composite ratings may not be mathematically appropriate	Field time; staff training; accuracy and precision
Examples	Frissell (1978); Marion (1995)	Magill (1989)	McEwen and others (1996)	Marion (1991); Marion and Cole (1996)

two indices commonly used in the ecological literature have also been employed (Hall 1989). Indices of spatial qualities may also be constructed to represent the spatial extent and distribution of impacts. Examples include the index of trail area (Cole and others 1997), the campsite expansion index (Gettinger and others 1998), Gini coefficients and linear nearest neighbor index (Leung and Marion 1998). The third group of indices provides a summary of resource condition of a site (Marion 1991). Area of vegetation loss (Cole 1989a), summary impact index (Cole and Hall 1992; McEwen and others 1996) and the impact index (Stohlgren and Parsons 1992) are some examples of summary indices. The final group of indices are designed to represent environmental sensitivity to impacts. Examples include the resistance and resilience indices (Cole 1995b; Cole 1995c) and the durability index (Cole 1993b).

Research Results

Since the last review more than a decade ago (Cole 1987b), there has been substantial progress in knowledge and understanding of recreation impacts and in practices of impact management. Study locations have expanded, and research topics and methods have been diversified. Many studies have focused on vegetation and soil parameters, and most have investigated impacts on campsites and trails. However, there has been more work on wildlife impacts, impact assessment and monitoring techniques and the effectiveness of management actions.

Much of this section is organized by two primary locations where recreation impacts occur—trails and campsites, with emphasis placed on studies conducted between 1986 and 1999. Earlier studies are reviewed by Cole (1987b). More extensive reviews are presented in Hammitt and Cole (1998), Kuss and others (1990) and Liddle (1997). A comprehensive bibliographic database of the recreation ecology literature is being developed as an update of the previous compilation (Cole and Schreiner 1981). This searchable database will be accessible online through the Aldo Leopold Wilderness Research Institute web site (<http://www.wilderness.net/Leopold/default.htm>).

Trail-Related Impacts

Soil and Vegetation Impacts—Trail construction and use can have substantial impacts to soil and vegetation, including soil compaction, erosion, muddiness, loss of vegetative groundcover and changes in species composition. Most recent research on soil and vegetation related trail impacts has been conducted outside wilderness and in other countries (Figure 1). This body of literature is beyond the scope of this paper but has been reviewed by Hammitt and Cole (1998), Kuss and others (1990) and Liddle (1997). A few studies were conducted in wilderness or similar backcountry areas. For example, Hall and Kuss (1989) investigated vegetation change along backcountry trails in Shenandoah National Park, Virginia. They found that groundcover and species diversity increased closer to trails, a finding they

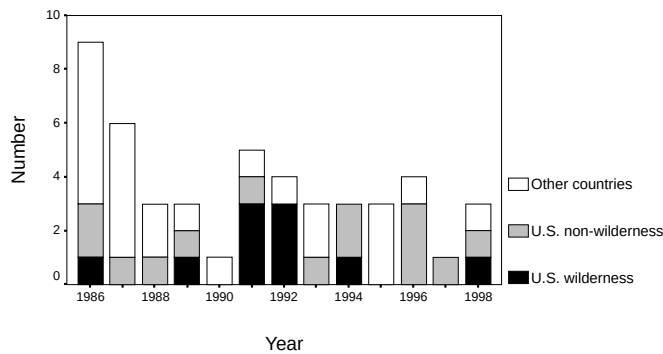


Figure 1—The numbers of publications on trail impacts between 1986 and 1998 (based on the literature that was available to the authors when this paper was prepared).

attributed to environmental alterations along trail corridors (Hall and Kuss 1989).

Trail impacts are influenced by a diverse array of use-related and environmental factors. Many studies identified environmental factors to be more important in determining the levels and rates of trail incision and associated soil erosion than use-related factors (Leung and Marion 1996). Environmental information may not be useful to predict trail impact problems in some cases, however (Burde and Renfro 1986). For example, trail widening is often associated with amount of use than site attributes (Cole 1991).

Trail impact assessments in Great Smoky Mountains National Park found that heavily used trails had significantly more soil erosion and tree root exposure, while trails receiving a high proportion of horse use were significantly wider, muddier and had more multiple treads (Leung and Marion 1999a; Marion 1994a). Trails located on ridgetops and upper slopes exhibited the greatest erosion, probably due to higher precipitation rates, more open forest canopies and reduced root mass from woody vegetation. Ridgeline trails also often directly ascend slopes, hindering the removal of water from treads of embedded trails. Problems with tread muddiness were most common in valley bottom positions, where treads commonly become embedded in moist organic soils. The number of tread drainage features (for example, water bars or drainage dips) was not correlated with these impacts, suggesting that increased trail maintenance is not a substitute for good trail positioning and layout. A recommended solution to both problems was trail relocation to valley walls with side-hill construction methods.

Introduction of Exotic Species—Cole (1987b) noted the paucity of research on recreation as a means of introducing exotic plant species into wilderness. Several studies have recently investigated this issue in greater detail. In Glacier National Park, Tyser and Worley (1992) found that trail corridors were an effective conduit for introducing exotic species such as Canadian bluegrass (*Poa compressa*), Kentucky bluegrass (*Poa pratensis*) and common selfheal (*Prunella vulgaris*) to fescue (*Festuca*) grasslands. Exotic species richness remained at high levels 330 ft from the sampled backcountry trails (Tyser and Worley 1992). In Rocky Mountain National Park, exotic species richness was found to be negatively correlated with distance from the

trailhead (Benninger-Truax and others 1992). In contrast, Marcus and others (1998) reported a less serious problem with exotic species in the Selway-Bitterroot Wilderness, Montana. They found that spotted knapweed was present only along limited portions of 5 sampled trails and on 6 of 30 surveyed campsites. Over 95% of spotted knapweed along the trails occurred within 0.31 mile of the trailhead and within 15 ft of the trail. (Marcus and others 1998).

Trail Effects on Wildlife—A number of recent empirical studies examined wildlife disturbance caused by recreational use of trails. The first group of studies investigated community composition and wildlife behavior in relation to trails. The existence of a trail network can act as a barrier or attraction to different wildlife species. In northwestern Montana, grizzly bears were found to avoid roads and trails (Kasworm and Monley 1990). In a Colorado recreational area, Miller and others (1998) found that generalist bird species were more abundant near trails, while specialist species were less common. Higher nest predation rates were also recorded near trails (Miller and others 1998). Visitors hiking on trails may disturb wildlife, displacing them from trail corridors during times of heavy use (temporal displacement) or permanently (spatial displacement). Knight and Cole (1995) reviewed research that documented highly variable wildlife responses to the presence of visitors, depending on the visitors' behaviors, the context of the disturbance and the wildlife's learned responses.

The second group of studies utilized an experimental approach to examine human disturbance related to trail use. In the Medicine Bow National Forest, Wyoming, Gutzwiller and others (1998) identified variations in avian response to an experimental walker. Intrusion tolerance was found to be lower when birds were in smaller groups, for more conspicuous species, and for species that are active closer to the ground (Gutzwiller and others 1998).

Results from these two groups of trail-wildlife studies suggest that trails and their visitors have the potential to generate undesirable impacts on wildlife from population to ecosystem levels, with significant implications for biodiversity conservation (Cole and Knight 1990).

Trail Impact Assessment and Monitoring—Trail impact assessment studies have been conducted in both Eastern and Western environments over the past decade. In Great Smoky Mountains National Park, a census-based problem assessment method identified the locations, extent, and frequency of selected trail impact problems (Leung and Marion 1999a; Marion 1994a). Subsequent work at this Park and an ongoing study in Shenandoah National Park compare the problem-oriented survey approach to the more traditional point sampling approach (Leung and others 1997). The point sampling method provides a lineal sequence of values typically assessed at a fixed interval along the trail, summarized with descriptive statistics (such as range, mean, median). The problem assessment method characterizes trail conditions by providing statistics such as number and location of occurrences, feet/mile, percent of trail length and aggregate distance for predefined trail impact problems. Preliminary observations suggest that the higher utility of this type of data for managers may be offset by reduced precision, a result of inherent subjectivity in defining and assessing where impact problems begin and end along a trail.

Cole replicated his earlier trail assessment (Cole 1983) in the Selway-Bitterroot Wilderness. Over an 11-year period, the monitored trail systems remained relatively stable, with cross-sectional area measurements revealing virtually no net erosion or deposition on tread surfaces. Individual sections did change markedly, primarily influenced by trail location and design. Tread width increased an average of 9.8 inches over a nine-year period, but bare width did not change significantly. In Rocky Mountain National Park, Summer reported that the degree of soil erosion and deposition was primarily a function of active geomorphological processes interacting with climatic factors (Summer 1986). Steep, upper-slope trail positions were most erodible. Intermediate positions experienced both erosion and deposition; and level terrain was most stable, though trail widening was problematic. Intensive runoff from natural events was cited as a more significant cause of erosion than visitor use.

Camping-Related Impacts

Campsites are primary destinations for many wilderness visitors and receive high levels of use. In contrast to trail studies, most campsite studies were conducted in the U.S., and many were conducted in designated wildernesses (Figure 2). Earlier studies on campsite impacts have been reviewed by Cole (1987b). Recent research has focused on: (1) understanding previously ignored topics of impacts (Zabinski and Gannon 1997), (2) examining the effectiveness of site restoration techniques (Spildie and others, this volume), (3) improving assessment and monitoring procedures (Cole 1989d; Leung and Marion 1999b; Marion 1991), and (4) adapting procedures to new environments and recreation settings (Monz 1998).

Soil and Vegetation Impacts—Camping activities can generate substantial and usually localized soil and vegetation changes (McEwen and Cole 1997). Most studies have found high levels of groundcover loss and soil exposure even with modest use (Cole 1986). For example, in Prince William Sound of Alaska, low-use campsites lost 93% of their vegetation cover on gravel sites and 81% on organic soil sites (Monz 1998). An experimental camping study conducted by Cole (1995a) found that one night of camping activity caused significant groundcover loss in all four vegetation types

examined. In more heavily used wilderness areas, such as Shining Rock Wilderness in North Carolina, frequent camping use often results in extensive land disturbance and vegetation damage (Saunders 1986).

Little research has been conducted on recreation impacts to soil microbial communities and underground processes (Cole and Landres 1996). Zabinski and Gannon (1997) examined this issue and reported less microbial activity in the upper layer (0-2.4 in) of soil on campsites than on their undisturbed controls, although there was no significant difference in the lower soil layer (2.4-6.8 in). The percentage of total carbon sources utilized by soil microbes was also significantly less in disturbed camping areas than in undisturbed control sites (Zabinski and Gannon 1997).

While camping impacts are usually spatially concentrated, some forms are more extensive. Taylor investigated 30 campsites in Yellowstone National Park and found that tree sapling density on campsites was only one-eighth that on control sites, which were located 160 ft from camp (Taylor 1997). Such decreases in tree saplings due to recreational use have a significant implication on tree regeneration and future forest structure.

Using a modeling approach, Cole (1992) examined the relative influence of use-related and environmental factors in determining the total amount of campsite impact. He demonstrated that degree of activity concentration is the most important factor. Several studies have documented the effectiveness of site locations and management actions that increase spatial concentration of use. In Great Smoky Mountains National Park, campsites at mid-slope topographic positions tend to be smaller than those on valley bottom or ridgetop positions, attributable to the site expansion resistance offered by sloping terrain (Leung and Marion 1999b). In the Chisos Mountains of Big Bend National Park, Texas, placement of campsite posts and logs to mark indistinct campsite borders have helped concentrate visitor activities within core use areas (Williams and Marion 1997). Median campsite size for these designated sites was only 650 ft². Similarly, the placement of many Isle Royale National Park campsites in sloping terrain, coupled with design and construction practices that create small flat camping benches, reduced median campsite size to 550 ft² (Farrell and Marion 1997). Camping shelters were even more effective in concentrating camping activities, with a median area of disturbance of 377 ft².

Other environmental factors, including elevation, aspect and plant community type, have also been investigated. Analyses of the influence of elevation on campsite conditions in Shenandoah and Great Smoky Mountains National Parks found no significant relationships with campsite size, vegetation loss or exposed soil (Williams and Marion 1995; Marion and Leung 1997). Campsites in Shenandoah National Park with a northerly aspect had more onsite vegetation cover and less than one-third the areal loss of vegetation cover than those with other aspects; no patterns were found in similar analyses at Great Smoky Mountains National Park. Analyses of forest cover type at Shenandoah National Park found that the chestnut oak and northern red oak forest types generally had the smallest and least altered campsites (Williams and Marion 1995). Campsites in the hemlock type were largest and had the least onsite vegetation cover at Great Smoky Mountains National Park (Marion

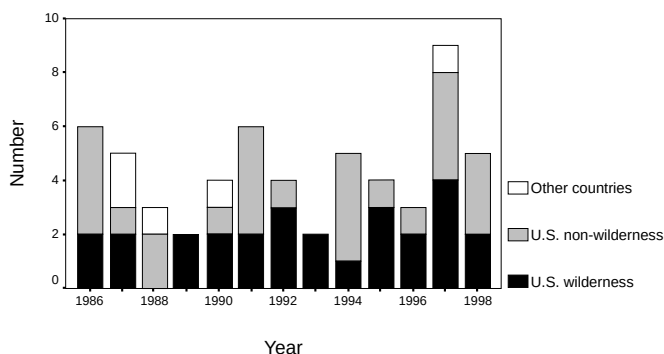


Figure 2—The numbers of publications on campsite impacts between 1986 and 1998 (based on the literature that was available to the authors when this paper was prepared).

and Leung 1997). Hemlocks have particularly dense canopies that support limited ground vegetation, so expansion potential is often high while trampling resistance is low. Evaluations of forest canopy densities consistently reveal a positive relationship between decreasing canopy density and increasing onsite vegetation groundcover (Marion 1994b; Marion and Leung 1997; Williams and Marion 1995). This finding is attributed to the higher trampling resistance and resilience of shade-intolerant grasses and herbs.

Very little recent work has examined use-related factors. An experimental camping study by Cole (1995a) found that one night of camping reduced relative vegetation height by 60% or more. Relative vegetation cover was reduced to as low as 66% following only one night of camping in four vegetation types. The impact associated with three additional nights of camping was less substantial, further reducing relative cover to only 50%. Results from this study generally corroborate those of earlier studies (Cole 1987b) that describe a curvilinear use-impact relationship.

McEwen and others (1996) investigated differences in impact from two types of use on campsites in four south-central U.S. wildernesses. Sites used by horse groups and hikers were more highly impacted than sites used only by hikers. Specifically, horse-hiker sites were larger and had more exposed soil and more tree damage than hiker-only sites.

Camping-Related Wildlife Impacts—Visitors spend considerable time on campsites, and their activities can disrupt normal wildlife activities, attract animals or alter wildlife habitat through vegetation and soil impacts. Wildlife that avoid areas with campsites can be displaced from vital riparian vegetation and water sources, a particularly critical impact in desert environments (Hammit and Cole 1998). Intentional or unintentional wildlife feeding is also common at campsites, leading to attraction behavior and unhealthy food dependencies. Species that frequent campsites in search of food include birds, mice, rats, ground and red squirrels, skunks, raccoons, foxes and bears. Consistent human feeding can lead to increases in small animal populations, which then crash suddenly at the end of the use season. Bears that obtain food pose a serious safety threat to visitors, and many must be relocated or killed (Merrill 1978).

Campsite Impact Assessment and Monitoring—Campsite impact assessment and monitoring programs are generally more common than trail assessments, and a large number have been conducted in the past decade. The campsite monitoring program in Kings Canyon and Sequoia National Parks of California is one of the earliest and best documented of its kind (Parsons 1986; Parsons and Stohlgren 1987; Stohlgren and Parsons 1986; Stohlgren and Parsons 1992; van Wagtendonk and Parsons 1996). Over 8,000 sites had been assessed as of 1990 (Fodor 1990). Published accounts of assessment programs are also available for wildernesses and national parks in Arizona (Cole and Hall 1992), Montana (Cole 1993a; Cole and Hall 1992), Oregon (Cole and Hall 1992; Cole and others 1997), Washington (Cole and others 1997; Gettinger and others 1998; Rochefort and Swinney, this volume; Scott 1998), Michigan (Farrell and Marion 1997), North Carolina/Tennessee (Leung and Marion 1999b; Marion and Leung 1997; Marion and Leung 1998), Virginia (Williams and Marion 1995), Texas (Williams and Marion 1997), and Illinois/Missouri/Arkansas (McEwen and others 1996).

Studies of trends in campsites (Cole and Hall 1992) monitored for 5 to 11 years in three Western backcountry areas found that campsites both improved and degraded over time. Campsite size, mineral soil exposure and tree damage were some of the impacts that increased (Cole and Hall 1992). In three Western wildernesses, Cole (1993a) found that the number of campsites increased 53% to 123% over 12 to 16 years. Campsite proliferation contributed more to net increase in the total amount of impact than change in the condition of existing campsites (Cole 1993a).

Trampling Research

Trampling, either by humans or recreational stock, is the fundamental impact force applied to trails and campsites, directly affecting vegetation and soil within trampled zones. Although often localized, trampling may have widespread effects. The extirpation of Scarbrous black sedge (*Carex atratiformis*), northern singlespike sedge (*Carex scirpoidea*) and other alpine plant species in the New England region (Zika 1991) and the decline of endangered desert fish populations in Zion National Park of Utah (Shakarjian and Stanford 1999) have been attributed to human trampling. Research on trampling and traffic effects on soil and vegetation have recently been compiled and reviewed (Yorks and others 1997).

Several trampling experiments were conducted in wilderness and backcountry areas in the past decade. Cole continued his earlier work (as reviewed by Cole 1987b) on six forest and grassland vegetation types in the Bob Marshall Wilderness complex (Cole 1987a; Cole 1988). He expanded his studies to 16 vegetation types in four Western and Eastern states (Cole 1993b; Cole 1995b; Cole 1995c). Using standardized experimental procedures, he compared vegetation types by their differential responses to foot trampling. The relationship between trampling intensity and vegetation damage was curvilinear in most cases, corroborating previous research (Figure 3). Resistant vegetation types, such as sedges (*Carex* spp.), were found to be able to absorb 25 to 30 times as much trampling as the least resistant type, ferns (*Dryopteris* spp.) (Cole 1993b). Morphological characteristics were the primary factor influencing plant resistance to trampling. Grasses and sedges have flexible stems growing in mats or tufts. More fragile were woody plants and taller herbs.

The resilience of plants, their ability to recover following trampling disturbance, varied substantially by habitat, with higher recovery in the most productive environments—those with higher soil fertility and moisture. For example, recovery rates are high in riparian areas in the Eastern states (Cole and Marion 1988; Marion and Cole 1996). In contrast, trampling impacts in less resilient environments, such as alpine and subalpine environments, require a long time to recover (Hartley 1999; Stohlgren and Parsons 1986). Plant characteristics, notably the position of the plants' perennating bud and physiological characteristics such as reproductive capacity and growth rates, also influence resilience (Cole 1988; Kuss 1986b).

In the Wind River Range of Wyoming, trampling response of five native plant species was examined (Monz and others 1994). Increased trampling intensities were associated with substantial increases in soil compaction and decreases in species richness at forest understory sites. Little effect was

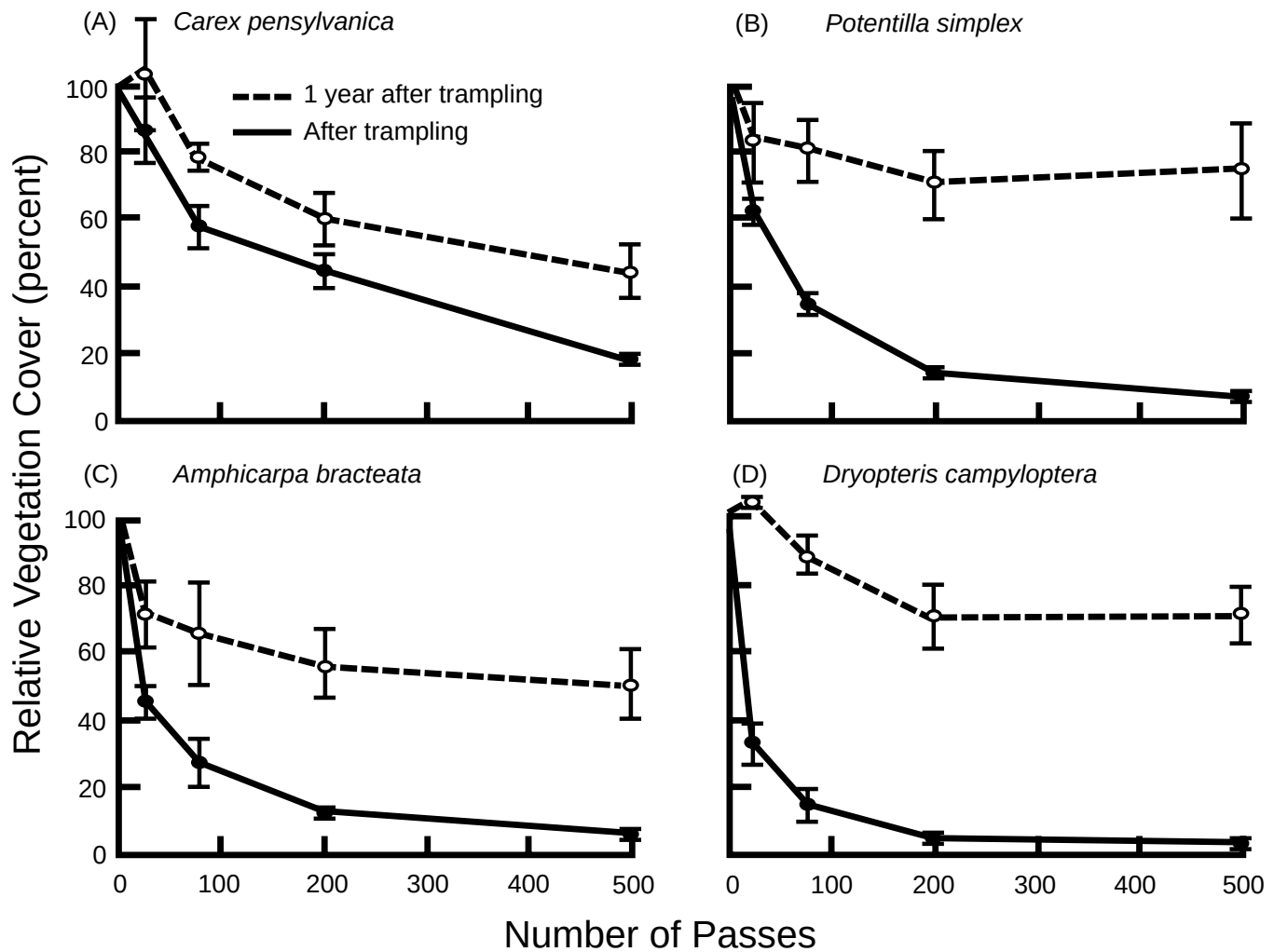


Figure 3—Relationships between trampling intensity and relative groundcover in four vegetation types in the Great Smoky Mountains, North Carolina. Bars denote one standard error (Source: Cole 1993b).

found on subalpine meadows. More recently, Monz and others (1996) examined trampling and increased temperature on moist and dry arctic tundra. Moist tundra was found to be highly susceptible to trampling disturbance, though warmer temperatures resulted in decreased leaf nitrogen, increased percent cover and increased number of leaves in mountain-aven (*Dryas octopetala*) (Monz and others 1996).

Hartley conducted a long-term study of trampling effects and recovery in the subalpine meadows of Glacier National Park, Montana (Hartley 1999). Thirty years after trampling was first applied in 1967, he reported significantly shorter inflorescence heights of fleabane (*Erigeron*) and significantly lower densities of both yellow avalanche-lily (*Erythronium* spp.) and fleabane (*Erigeron* spp.) within trampled plots than in control plots. As has been reported elsewhere (Kuss and Hall 1991; Weaver and Dale 1990), recovery of trampling impacts can be exceptionally slow in less resilient environments.

Cole investigated trampling effects on cryptogamic soil crusts in Grand Canyon National Park (Cole 1990b). He found that cryptogamic soil crusts, which are ecologically important features in arid ecosystems, are fragile and

extremely susceptible to trampling impact. Crust structure damage was caused by only 15 trampling passes. Complete loss of crust cover occurred after 250 passes (Cole 1990b). de Govenain (1996) examined indirect effects of soil trampling on plant growth in the northern Cascade Mountains, Washington. He reported significantly higher soil water content and temperature on trampled sites, which may have influenced long-term plant succession in the study area (de Govenain 1996).

Cole conducted trampling experiments to evaluate two recommended LNT practices: removing boots and the use of a geotextile ground cloth in camp. His results showed that these two practices have small short-term benefits but no long-term benefits (Cole 1997).

Recent increases in popularity of llamas and other non-traditional pack stock have generated research interest in their relative trampling effects (McClaran and Cole 1993). DeLuca and others (1998) compared the effects of llamas with horses and hikers on soil erosion and found that horse traffic produced significantly higher sediment yield from established forest trails in Montana than either llama or hiker traffic, which did not significantly differ from each other.

Cole and Spildie (1998) also found greater trampling impact on vegetation by horses than by llamas or humans.

Other Types of Recreation Impact Research

Effectiveness of Management Actions—There have been few but increasing numbers of trail and campsite studies that investigate the effectiveness of impact management strategies and actions. The placement of scree walls along trail boundaries, for instance, was reported to be effective in containing hikers and associated trampling impacts within trail treads (Doucette and Kimball 1990). Marion (1995) provides a detailed case study of management success in Delaware Water Gap National Recreation Area in Pennsylvania. Two major management actions were the designation of campsites and the provision of anchored fire grates. Together with supporting actions, these management efforts effectively reduced aggregate camping-induced land disturbance by more than 50 percent between 1986 and 1991, even with modest increases in visitation (Marion 1995).

A management program recently adopted in Idaho's Selway-Bitterroot Wilderness also demonstrates the effectiveness of a spatial containment strategy (Spildie and others, this volume). A coordinated set of management actions, including: (1) designation of stock containment areas, (2) closure of some sites to stock use or all use, and (3) intensive site restoration. In five years, the areal extent of recreation disturbance was reduced by 37 percent, and bare soil was reduced more than 40 percent. Designated camping policies and site restoration actions were also found to be effective in the Boundary Waters Canoe Areas Wilderness, Minnesota (Marion and Sober 1987).

Attempts to restore impacted sites have been less effective, however. In Yosemite National Park of California, efforts to restore bare core areas on degraded high-elevation campsites by transplanting vegetation met with only modest success. Three years after program initiation, species richness and percent plant cover increased only slightly and the survival rate of transplants was low (Moritsch and Muir 1993). A 1998 study of these same campsites found that plant re-establishment was substantial on campsites with higher soil moisture, while recovery on dry sites was low (Eagan and Newman 1999). Some success with soil amendments and planting techniques as a means of speeding recovery rates was recently reported from the Eagle Cap Wilderness (Cole, this volume).

Impact Indicators for Management Frameworks—As input to management planning frameworks such as LAC and VERP, a diverse array of resource and impact indicators and their utility have been reviewed (Merigliano 1990; Watson and Cole 1992). Belnap (1998) investigated steps for selecting resource indicators in Arches National Park, Utah, as part of the Park's VERP planning and implementation process. Based on a list of selection criteria and a ranking system, she selected eight resource and impact indicators to define the health of this arid ecosystem. Indicators were assigned to two categories, one requiring measurements every year, and another requiring measurements every five years (Belnap 1998).

Setting standards for recreation impacts is another emerging issue with little research and management attention. A recent survey of trail managers found that condition standards for backcountry trails were either lacking or poorly defined in the parks studied (Burde and others 1998). With increasing adoption of management frameworks in wilderness and backcountry areas, research in this area is much needed.

Packstock Grazing Impact—The impact of packstock grazing and recovery processes were the subject of two pack horse grazing studies in subalpine meadows within the Lee Metcalf Wilderness of Montana (Olson-Rutz and others 1996a; Olson-Rutz and others 1996b). The grazing behavior of horses was quantified and related to the intensity and extent of impact. Results indicated that increased grazing duration was associated with reduced plant heights, and that grass heights appeared to be reduced more than forb heights (Olson-Rutz and others 1996a). One year after the pack horse grazing, more bare ground and less litter and vegetative cover were recorded, attributed to reduced stem numbers (Olson-Rutz and others 1996b). Research on packstock grazing impact on meadows is currently being conducted in Yosemite National Park (van Wagtenonk and others, this volume).

Climbing—Rock climbing is rapidly growing in popularity. Potential climbing-related impacts, including trail creation and use in steep approach areas, cleaning of vegetation and lichens from cliff faces, and use of protective hardware such as expansion bolts, have received little research attention until recently (Attarian and Pyke 2000). Earlier studies focused primarily on the proliferation of social trails and trampling of climbers in the access zone at the base of cliffs (Genetti and Zenone 1987). More recent studies have turned their attention to the cliff plant and wildlife communities on the vertical climbing zone. In Joshua Tree National Park of California, cliffs used intensively for climbing were found to have the lowest richness of cliff plant communities, and the number of individual plants and plant cover decreased with increased level of use (Camp and Knight 1998). Other studies in nonwilderness areas also found significant impact on vegetation and microflora (Nuzzo 1995; Nuzzo 1996).

Human Waste—The problem of improper human waste disposal is a perennial concern among wilderness managers (Cilimburg and others 2000). In Mount Rainier National Park of Washington, up to 10,000 climbers visit the summit of Mount Rainier each year, raising the possibility of fecal contamination in high-elevation areas such as the Muir Snowfield. An initial investigation was conducted recently to determine if surface water runoff from the snowfield was contaminated by fecal microorganisms such as fecal coliforms, fecal streptococci, fecal enterococci and *E. coli* (Ells 1997). Results indicated no significant evidence of contamination. Cilimburg and others (2000) provide a comprehensive review of the human waste disposal problem and management options.

Management Responses and Related Research

The identification and selection of effective management techniques requires knowledge of the impacts that are

occurring, their underlying causes and the role of various influential factors. The research described in the preceding section should be integrated with current monitoring data and management expertise in a careful problem analysis prior to the identification and selection of management strategies and actions.

Management Needs and Constraints

Faced with a limited wilderness resource base and increasing recreational demands, managers must decide how much and what kinds of recreation use are acceptable, recognizing that any visitation generates some degree of resource impairment. They must explicitly define when visitation-related environmental change becomes an unacceptable impact, requiring management intervention. Research and monitoring can inform such decisions, but managers must make them, preferably in consultation with the public.

The Wilderness Act (P.L. 88-577) defines wilderness as “undeveloped” lands “without permanent improvements” which “has outstanding opportunities for solitude or a primitive and unconfined type of recreation,” and where “the imprint of man’s work is substantially unnoticeable.” Furthermore, it states that “except as necessary to meet minimum requirements for the administration of the area...there shall be no...motorized equipment...and no structure or installation within any such area.” In light of this mandate, managing agencies have generally adopted what has become known as the *minimum tool rule* to guide their wilderness management actions (Hendee and others 1990). This rule directs managers to apply only the minimum tools, equipment, device, force, regulations or practice that will accomplish the desired result.

This guidance is frequently interpreted as a need to first select and attempt indirect management actions, such as Leave No Trace educational practices or improved trail and site design and maintenance before more direct controls such as regulations. However, if indirect methods fail to resolve resource protection problems, managers must be prepared to apply more restrictive measures. It has been argued that managers must not hesitate to employ direct controls, even as initial actions, when long-term or irreversible resource degradation is occurring (Dustin and McAvoy 1982).

Decisions about the use of site hardening and facility development actions in wilderness are particularly difficult. A constructed and maintained trail is a permanent wilderness facility designed both to facilitate wilderness travel and protect resources. Such facilities can involve vegetation disturbance, soil excavation and deposition, and the potential disruption of surface water movement. However, a properly managed trail system limits the areal extent and severity of recreation impacts by concentrating traffic on resistant tread surfaces. The absence of formal trails in popular locations would lead to a proliferation of poorly located and heavily impacted visitor-created trails. Similarly, although less common in wilderness, designated campsites can be located, constructed and maintained to substantially reduce the areal extent and severity of camping impacts. The Wilderness Act clearly permits managers to employ

such facilities, although their use must be justified as the minimum means for managing sustainable visitation.

Management Strategies and Tactics

Recreation impact problems may be addressed through an array of management strategies and tactics (Anderson and others 1998; Brown and others 1987; Cole and others 1987; Hammitt and Cole 1998; Hendee and others 1990; Leung and Marion 1999d). The following discussion follows the strategies and tactics described by Cole and others (1987) (Table 6).

Management interventions seek to avoid or minimize recreation impacts by manipulating either use-related or environmental factors. Use-related factors, particularly the redistribution or limitation of visitor use, have received more research and management attention. However, research has increasingly demonstrated the importance of environmental factors, such as focusing use in environmentally resistant locations or increasing resource resistance through the use of facilities like trails and campsites (Cole 1990a). The modification of visitor behavior through educational and regulatory actions is another frequently applied strategy.

Modification of Use-Related Factors—Managers can control or influence amount of use, density of use, type of use, and user behavior. The type of visitor action contributing to the management problem is often an important consideration (Cole 1990a). For example, impacts from visitors knowingly engaging in illegal actions require a law enforcement response. Careless, unskilled or uninformed actions are often most appropriately addressed through visitor contacts and educational responses (Lucas 1982). Unavoidable impacts are commonly reduced by relocating visitation to resistant surfaces or by limiting use.

1. Amount of Use: Amount of use is perhaps the most studied use-related factor in recreation ecology. Earlier studies have consistently found a nonlinear asymptotic relationship between amount of use and amount of impact (Cole 1987b). Most forms of camping impact occur rapidly with initial and low levels of use (up to 10 nights/year), then begin to level off as near-maximum impact levels are reached at moderate to high use levels. This use-impact relationship has been corroborated by recent trampling studies for most impact parameters with a few exceptions (such as exposure of mineral soil) (Cole 1987a; Cole 1988; Cole 1990b; Cole 1993b; Cole 1995b; Cole 1995c; Cole and Trull 1992; Kuss and Hall 1991).

The curvilinear use-impact relationship reduces the potential effectiveness of use limitation for reducing recreation impacts (Strategies I & II, Table 6). Substantial use reductions would be necessary to achieve even modest improvements in resource condition on heavily impacted trails and campsites. However, use reductions can lead to pronounced improvements at lower use levels, where use and impact are more strongly related (although slow recovery rates prevent rapid improvements) (Cole 1995a). Also, limitations on the number of groups, particularly during times of peak use (Strategy IV), can reduce the total area of camping disturbance by shrinking the number of campsites needed. For example, a popular travel zone may receive over twice the

Table 6—Strategies and tactics for managing recreation impacts to resources or visitor experiences.

I.	Reduce use of the entire area
	• Limit number of visitors in the entire area • Limit length of stay in the entire area • Encourage use of other areas • Require certain skills and/or equipment • Charge a flat visitor fee • Make access more difficult throughout the entire area
II.	Reduce use of problem areas
	• Inform potential visitors of the disadvantages of problem areas and/or advantages of alternative areas • Discourage or prohibit use of problem areas • Limit number of visitors in problem areas • Encourage or require a length-of-stay limit in problem areas • Make access to problem areas more difficult and/or improve access to alternative areas • Eliminate facilities or attractions in problem areas and/or improve facilities or attractions in alternative areas • Encourage off-trail travel • Establish differential skill and/or equipment requirements • Charge differential visitor fees
III.	Modify the location of use within problem areas
	• Discourage or prohibit camping and/or stock use on certain campsites and/or locations • Encourage or permit camping and/or stock use only on certain campsites and/or locations • Locate facilities on durable sites • Concentrate use on sites through facility design and/or information • Discourage or prohibit off-trail travel • Segregate different types of visitors
IV.	Modify the timing of use
	• Encourage use outside of peak use periods • Discourage or prohibit use when impact potential is high • Charge fees during periods of high use and/or high-impact potential
V.	Modify type of use and visitor behavior
	• Discourage or prohibit particularly damaging practices and/or equipment • Encourage or require certain behavior, skills and/or equipment • Teach a wilderness ethic • Encourage or require a party size and/or stock limit • Discourage or prohibit stock • Discourage or prohibit pets • Discourage or prohibit overnight use
VI.	Modify visitor expectations
	• Inform visitors about appropriate uses • Inform visitors about conditions they may encounter
VII.	Increase the resistance of the resource
	• Shield the site from impact • Strengthen the site
VIII.	Maintain or rehabilitate the resource
	• Remove problems • Maintain or rehabilitate impacted locations

Source: Cole and others (1987).

visitation on peak use weekends than it does during more typical high use periods. Use can also be limited during times when resources are more vulnerable to impact, by restricting horse traffic when trails are particularly wet, for

example. Tactics for rationing use are reviewed in Anderson and others (1998), and Cole and others (1987).

2. Density of Use: How much visitation is concentrated spatially affects both the areal extent and severity of resource impacts (Marion and Cole 1996). Educational programs and regulations may be used to shape visitation density, generally through one of two strategies: visitor dispersal, which spreads use sufficiently to avoid or minimize long-term impacts, and visitor containment, which concentrates use to limit the areal extent of impact (Cole 1981; Cole 1992; Leung and Marion 1999d). Containment, as evidenced by the development and maintenance of formal trail systems, has a long tradition of use in wilderness. Its application to camping management is less common, but a variety of options are now in use (Marion, Roggenbuck and Manning 1993). In contrast, dispersal is rarely applied to reduce hiking impacts except for remote low-use areas. Its application to camping management is more common, although many factors thwart the success of this strategy.

When camping is unregulated, visitors are free to choose any existing campsite or create new ones. This policy can result in many poorly located campsites (Cole 1993a; Leung and Marion, this volume; McEwen and others 1996). For example, wilderness campsites in the Jefferson National Forest of Virginia were frequently located on trampling-susceptible herbaceous groundcover in areas that readily permit site expansion and proliferation (Leung and Marion, this volume). Campsites were also located close to trails and other campsites, enhancing the potential for visitor conflicts and reducing solitude for both campers and hikers.

A successful application of dispersal and containment strategies can reduce camping impacts. Consider three campsites that receive intermediate amounts of use (10-20 nights/year) under an unregulated camping policy (Figure 4). Aggregate resource impact for these sites would be three times the "a" amount of impact. Under the purest form of dispersed camping, these sites would be closed and their use distributed across 45 pristine sites, each receiving only one night of use/year. Most vegetation types can sustain such light camping with no permanent impact visible the following year. More resistant surfaces, like grassy groundcover, sand, gravel and rock, can accommodate many more nights of use without permanent impact. The

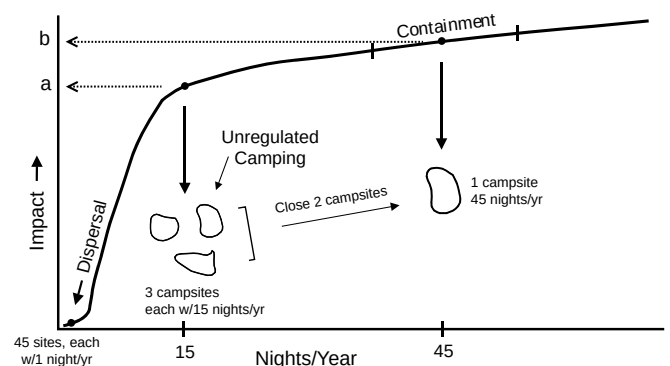


Figure 4—A generalized use-impact curve illustrating the intended locations of typical or average campsites under dispersal and containment strategies.

low camping densities under a dispersal strategy also resolve problems with crowding and conflicts.

In contrast, a containment strategy could be implemented by closing two of the three original sites and distributing their use to the third. Due to the curvilinear use-impact relationship, impact on this third site would increase only marginally, from “a” to “b” (Figure 4). Aggregate impact would decline substantially, from three sites with an “a” level of impact to one site with a “b” level of impact. Application of this strategy was largely responsible for a 50 percent reduction in the total area of disturbance from river camping at Delaware Water Gap National Recreation Area (Marion 1995). Furthermore, in addition to favoring resistant sites, site selection criteria emphasized the closure of sites within dense clusters, addressing crowding and conflict problems by maximizing intersite distances.

While these strategies may seem straightforward, additional issues often complicate their implementation. Achieving the level of camping dispersal necessary to prevent impacts has proven exceptionally difficult. In most vegetation types more than a few nights of camping will quickly create lasting impacts—that is, permanent campsites (Cole 1995a). Mountainous topography, dense vegetation, and availability of water frequently limit the number of potential camping locations, and few of these contain resistant surfaces (Williams and Marion 1995). Furthermore, most visitors prefer camping on established sites close to trails, water and popular features (Lucas 1990a). Generally, a dispersed camping strategy will be effective only in areas that receive low levels of use, have numerous potential camping locations that are resistant and/or resilient, and where visitors are willing to learn and apply Leave No Trace camping practices (Cole 1981; Leung and Marion 1999d). Managers at Denali National Park and Preserve of Alaska have developed one of the most successful dispersed camping programs, although visitor use numbers are also highly restricted.

A successful containment strategy requires concentrating camping activities on the smallest number of sites needed to accommodate the intended level of use (Leung and Marion 1999d). Reserved, designated site camping permits the smallest number of campsites and aggregate impact. However, fixed itineraries are difficult to follow in wilderness and entail a substantial loss of visitor freedom (Stewart 1989). Designated site camping without a reservation system allows greater flexibility. Visitor use surveys can provide information for matching campsite numbers and locations to visitor use patterns, or entry point quotas can restrict use based on available campsite numbers (Lime and Buchman 1974). To avoid excessively large inventories of campsites, use surveys should be conducted during average high use periods rather than peak use periods. In comparison to areas with site reservation systems, somewhat larger numbers of campsites are necessary to avoid the “musical chairs” dilemma of too many visitor groups and too few campsites. An educational approach, asking visitors to camp only on well-established campsites, may also be used (Cole and Benedict 1983).

Some wilderness and backcountry areas have adopted multi-strategy camping policies (Leung and Marion 1999d). New backcountry camping management policies at Shenandoah National Park provide an example (National Park Service 1998). A few areas containing sensitive cultural

and natural resources or that accommodate high day use will be closed to camping. In high-use areas, visitors will be required to camp on a limited number of designated campsites on a first-come, first-served basis. In remaining areas, visitors will be asked to camp on well-established campsites, a limited number of which will be selected by managers for resistance and ability to promote solitude. Dispersed camping on pristine sites will be permitted when all available campsites are used. While more complex, such combined strategies offer substantial flexibility in balancing wilderness resource protection and recreation provision objectives.

3. Type of Use: Types of uses that result in greater or disproportionate impacts are often subject to special regulations or educational programs (Strategy V). For example, visitors with horses have been restricted to a subset of more resistant trails and campsites specifically selected and maintained to sustain such use. While large groups create larger campsites than small groups, splitting them up may require more campsites and an equivalent amount of aggregate impact (Cole 1987b; Cole and Marion 1988). Matching group size with site size is therefore a significant management challenge. Further research on the relationship between party size and resource impact is needed.

4. User Behavior: Many impacts are avoidable, often caused by uninformed or careless behavior (Lucas 1982). Managers can educate and regulate visitors to avoid or reduce visitor behavior that contributes to avoidable impacts (Strategy V). The most common avoidable resource impacts include littering, cutting switchbacks, creating new trails and campsites, trail widening and campsite expansion, moving or building new fire sites, improper disposal of human and food waste, wildlife and cultural resource disturbance and cutting trees or tree limbs. Management efforts can also target many unavoidable impacts, such as vegetation disturbance and soil compaction caused by foot traffic. A variety of low-impact hiking and camping practices have been described to address these impacts (Cole 1989b; Hampton and Cole 1995), along with alternative education techniques for conveying such practices to visitors (Doucette and Cole 1993).

The four federal wilderness management agencies in partnership with the National Outdoor Leadership School have founded and actively promote a national *Leave No Trace* program that teaches outdoor ethics and low impact hiking and camping practices (Hammit and Cole 1998; Marion and Brame 1996). *Leave No Trace* training courses, publications and a comprehensive web site (<http://www.LNT.org>) are now reaching millions of potential wilderness visitors. Agency wilderness-specific educational contacts, signs and materials reinforce this effort and target specific problems.

Although more restrictive to visitor freedom and experiences, regulations offer another option for altering visitor behavior to reduce impacts (Lucas 1982). For example, regulations requiring proper food storage or fines for visitors who feed wildlife can help return wildlife to natural diets. Generally, regulations should only be used when indirect options are likely to be ineffective (Lucas 1990b). Interventions may employ both educational and regulatory responses. For example, excessive tree damage may be addressed by instructing campers to use stoves or to build small fires using dead down wood that can be broken by hand. A

regulation prohibiting axes, hatchets and saws removes the unnecessary tools most commonly used to damage trees.

Modification of Environmental Factors—Managers can also influence or control the location of visitor use in wilderness (Strategy III) and manage the trails and campsites that sustain that use (Strategies VII and VIII). For example, trails may be designed to avoid areas prone to muddiness, fragile vegetation types and steep slopes or erodible soils. Camping may be encouraged in durable vegetation types. Trail and campsite impacts can be reduced through careful site selection, design, construction and maintenance.

1. **Environmental Resistance:** Previous research has demonstrated considerable variability in the trampling resistance of different vegetative growth forms and plant communities (Cole 1987b; Kuss 1986b; Liddle 1991). Resistant plant communities and environments may be targeted for camping, while fragile communities may be avoided or identified for closures to camping. Examples of resistant plant communities include dry open forests and meadows with substantial grass or sedge cover, dense forests with little or no vegetation cover and sand, gravel and bedrock substrates.

Soils also vary in their resistance to compaction and erosion. Moist soils with little organic matter and a wide range of particle sizes (such as loams) are the most prone to compaction, while soils with a narrow range of particle sizes, particularly those high in silt and fine sands, are most prone to erosion (Hammitt and Cole 1998; Kuss and others 1990). Both soil compaction and erosion are accelerated by the absence of vegetation and organic litter, and slope is a critical determinant of erosion potential.

Wilderness managers can do little to modify environmental resistance. However, the construction and use of trails and campsites frequently opens forest canopies, allowing greater sunlight penetration and enhancing the survival and spread of shade-intolerant, trampling-resistant grasses, sedges and herbs. Seeding and transplanting resistant vegetation, using locally obtained sources of native plant materials, have been done in some wildernesses, and there is guidance for site restoration methods (Hanbey 1992). Although most commonly applied to closed campsites, many of these techniques have been employed by managers of the Boundary Waters Canoe Area Wilderness to reduce the size of open campsites (Marion and Sober 1987).

2. **Environmental Resilience:** Knowledge of the relative resiliency (ability to recover) of different vegetation and soil types may also be used to direct camping to areas that will recover quickly after trampling disturbance. However, impact rates are far greater than recovery rates, so off-season resource recovery is generally minimal and rest-rotation schemes to minimize impact are not warranted (Cole and Ranz 1983; Marion and Cole 1996). Environmental resilience can be an important consideration in low-use areas where dispersed hiking and camping are promoted (Cole 1995c). In more popular areas, the concentration of visitor activities is often sufficient to permanently remove most of the vegetation cover on trails and campsites. However, highly resilient vegetation still helps to restrict the size and further expansion of disturbance in these areas.

3. **Site Management:** Wilderness trails and campsites have rarely been planned and developed after careful evaluation of their expected ability to sustain use with minimal impact. Most wilderness managers simply inherit an inventory of trails dating back to earlier uses as Indian and settler travel ways, fire fighting roads and trails, logging roads and informal visitor-created trails. Similarly, most campsites, even those formally designated, were originally visitor-created. Examples abound of poorly located trails and campsites that are severely degraded. However, knowledge is now available to direct visitors to trails and campsites able to sustain heavy recreational traffic with far less resource impact than many existing recreation facilities. When necessary, site development that includes primitive facilities and sound maintenance can also contribute substantially to the avoidance and minimization of recreation impacts in wilderness.

Site Selection and Development—Knowledge of the environmental resistance and resilience of vegetation and soil types can be applied to select new and relocated trails and campsites (Hammitt and Cole 1998). Management options include educating visitors to improve site selection, marking resistant sites to encourage their use and designating resistant sites (Leung and Marion 1999b). Topography and other environmental attributes such as rockiness and vegetation density can also be considered to select locations that minimize impact severity and area of disturbance. In the Chisos Mountains of Big Bend National Park, managers have carefully selected and designated campsites to resist site expansion and promote solitude. The mean site size for these campsites is only 686 ft² (Williams and Marion 1997).

Managers at Isle Royale National Park have constructed campsites in sloping terrain, using standard cut-and-fill practices to create small benches for tenting and cooking areas (Farrell and Marion 1997). Camping posts and embedded logs or rocks are used in flat terrain to identify intended use areas and discourage site expansion. Managers can spatially arrange the sites to promote solitude and to minimize trail development to water sources and shared facilities like bear bag hanging devices and toilets (Hammitt and Cole 1998; Leung and Marion 1999d).

Site Maintenance—Trail maintenance programs exist in most wilderness areas, and many excellent manuals have been developed to guide this work (Birchard and Proudman 2000; Demrow and Salisbury 1998; Hesselbarth and Vachowski 1996). Active trail maintenance reduces impacts by providing a durable tread able to accommodate the intended traffic while minimizing problems with tread muddiness, erosion, widening and multiple tread development.

Much of the expertise gained in maintaining trails can be extended to maintaining campsites, although the appropriateness of such work in wilderness has been questioned (Cole 1990a). Maintenance work can reduce campsite sizes to the minimum necessary, prevent erosion and reduce campfire-related impacts (Hammitt and Cole 1998; Marion and Sober 1987). For example, excessive site size may be addressed by subtly improving tenting locations in core use areas (creating smooth, gently sloped areas) and ruining tenting locations in peripheral use areas. Site ruination work commonly includes “ice-berging” large rocks (burial except for sharp protruding tips), creating an irregular

tenting surface by digging shallow scrapes and mounding soil and renaturalizing areas with large logs, organic debris and vegetative transplants. Such work should use native materials and be carefully blended to match natural conditions (Marion and Sober 1987). However, more artificial work may be justified in high-use areas or on particularly troublesome sites. Such work includes embedding rocks or logs to visually identify intended campsite boundaries or placing a camping post to attract and spatially concentrate visitor activities.

Site Facilities—Site facilities are not always visitor conveniences, and many serve important safety and resource protection functions (Cole 1990a). Bridges along trails are often built to safely transport trail users across deep or dangerous currents. Bridges also protect sensitive riparian areas from vegetation damage and soil erosion on steep slopes. Placement of small, firmly anchored steel fire rings can be used to identify preferred or legal campsites, spatially concentrate visitor activities to reduce site size and limit resource impacts by focusing fire-related activities at only one spot (Marion 1995). Pit toilets can resolve problems with improperly disposed human waste, particularly on high-use campsites where the volume of waste poses a threat to human health. Impacts from recreational stock can be concentrated by placement of stock restraint facilities.

Site Closures—Camping closures represent a final resource protection strategy, generally most appropriate for protecting sensitive environments, rare flora and fauna or fragile historic sites (Cole 1990a; Hammitt and Cole 1998). Camping closures around popular features such as waterfalls, cliffs, ponds and lakes may be appropriate to separate overnight campers from intensive day use. Closures of popular highly impacted campsites are often ineffective and inappropriate. Little recovery will occur unless all use is removed, and new campsites with greater aggregate impact are frequently created in nearby areas (Cole and Ranz 1983). Generally, such closures are warranted only when use is shifted from impact-susceptible locations to impact-resistant locations, although social considerations may also provide justification (Cole and Ranz 1983; Trafimow and Borrie 1999).

Impact Management Decisionmaking

Management of recreation impacts directly affects the quality of recreation resources and visitor experiences. For example, restricting camping to designated campsites may reduce campsite numbers and aggregate impact, but it also imposes a direct management “presence” and control on visitor freedom to travel and select campsites. Achieving an appropriate balance between the dual management objectives of resource protection and recreation provision frequently requires decisions that trade off recreation experience quality with natural resource quality. Such decisions are difficult and often controversial and must be defensible in both the court of public opinion and law.

A decision framework is simply a standard process that provides structure to decisionmaking for planning or management purposes (Hendee and Koch 1990). Historically, managers have relied on informal decisionmaking when

addressing visitor impact issues. Common problems with this approach include a failure to explicitly describe intended resource or social conditions, evaluate the acceptability of existing conditions, conduct a thorough problem analysis or consider a comprehensive array of management alternatives (McCool and Cole 1997). Subsequent decisions may be indefensible and ineffective at restoring desired resource conditions.

The expanding popularity of wilderness recreation, greater public scrutiny of management decisionmaking and widening demands for participatory public land management are placing greater demands on land managers to further develop and communicate the processes by which decisions are made (Krumpe and McCool 1997). Formal decisionmaking frameworks have been developed and applied to guide both planning and operational decisions. These frameworks offer a defensible process for defining desired future resource conditions for visitor impact management, identifying impact indicators and assessing impact acceptability, conducting problem analyses, and evaluating and selecting preferred management actions.

Types of Frameworks

Formal frameworks may be simple or complex, as long as they identify and describe the steps by which decisions are made. Management constraints, such as limitations in funding, staffing and time, must be considered carefully in selecting the most appropriate framework. Recently, the most widely applied frameworks include *Limits of Acceptable Change* (LAC) (Stankey and others 1985) and *Visitor Experience and Resource Protection* (VERP) (National Park Service 1997a; National Park Service 1997b). These frameworks transform wilderness mandates into prescriptive objectives that can be implemented and evaluated with standards defining the limits of acceptable conditions for selected resource and social indicators (Figure 5). Monitoring permits periodic comparisons of conditions to standards. If standards are exceeded, a problem analysis evaluates causal factors to aid in selecting appropriate and effective management intervention(s). These models provide dynamic decision processes; future monitoring evaluates the success of implemented actions, so managers can select and implement additional actions if unacceptable conditions persist. Comprehensive reviews of these frameworks and their application to wilderness are provided in two state-of-knowledge reviews (Krumpe, this volume; Manning and Lime, this volume).

Decision frameworks require objective monitoring to characterize resource conditions for comparison to management objectives and/or indicator standards and to evaluate the success of implemented actions. Monitoring may be informal, such as staff observations or simple inventories, or formal, involving the application of standardized qualitative or quantitative procedures (Cole 1983; Cole 1989d; Marion 1991). Formal visitor impact monitoring programs employing quantitative ratings or measures are required for frameworks that use indicators and standards. Quantitative monitoring data can also be used to document trends in resource conditions, providing a permanent record of conditions that transcend changes in wilderness staff. Monitoring data may reveal subtle trends, alerting managers and allowing time

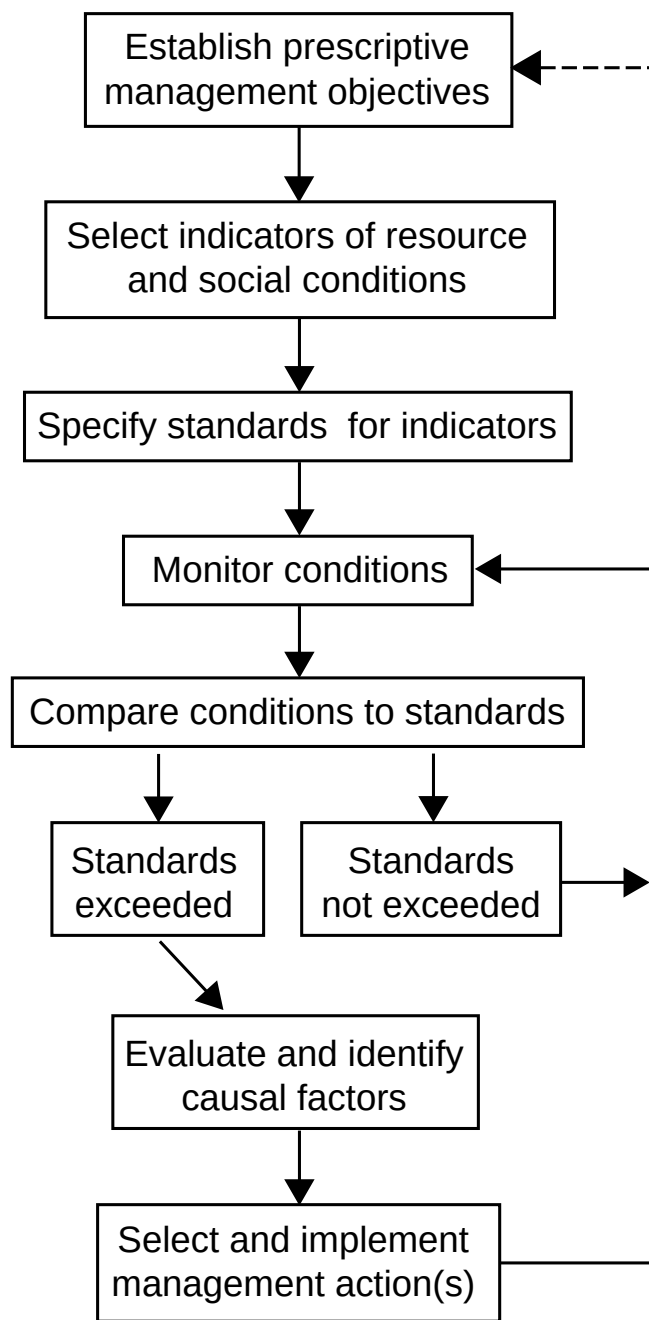


Figure 5—Diagram illustrating contemporary management planning frameworks such as LAC, VIM and VERP.

for the implementation of corrective actions that will avoid severe or irreversible impacts.

Monitoring data may also help gauge the effectiveness of management interventions implemented to correct deteriorating or unacceptable resource conditions. For example, analysis of campsite monitoring data at Delaware Water Gap National Recreation Area revealed the success of several site and visitor management actions implemented following the initial monitoring survey (Marion 1995).

Monitoring data may assist in identifying the underlying causes of impacts and help managers select effective

management strategies and actions. For example, campsite monitoring data at Shenandoah National Park were used to develop campsite selection criteria based on vegetation type, topography and aspect (Williams and Marion 1995). Park staff are applying these criteria to rank existing campsites and potential campsite locations to shift camping to more durable locations.

Other uses of monitoring information include the formulation and justification of budget requests and resource or visitor management actions (Marion 1995). For example, monitoring data documenting a decline in trail conditions over time might suggest the need for increased trail maintenance funding. Similarly, data showing an increasing trend in tree damage following educational efforts might justify a ban on axes and saws. Finally, monitoring data may be used to assign limited agency funding or staffing within different wildernesses or regions of a single wilderness.

Knowledge Gaps and Future Directions

Recreation ecology is essential to the professional management of wilderness resources and recreational experiences. Managers frequently turn to scientific knowledge for the information needed to make informed decisions. The inadequate knowledge base of recreation resource impacts has meant that managers must act in the absence of scientific information, taking actions that are increasingly being challenged by the public.

Basic Processes and Factors

Cole and Landres (1996) reviewed various threats to wilderness ecosystems, including criteria for evaluating their significance. They highlighted gaps in knowledge about the pollution of water bodies and alteration of their biota due to the introduction of fish, disruption of natural conditions due to fishing, hunting and the introduction and translocation of game animals, belowground processes, including biotic-biotic interactions, and of nonconsumptive visitor impacts to wildlife. Many of these impacts, particularly at larger spatial and temporal scales, are so poorly understood that effective impact indicators cannot be identified, and monitoring programs cannot be initiated (Cole and Landres 1996).

Long-Term Consequences and Significance of Impact

More longitudinal research and monitoring studies are needed to document and evaluate the long-term consequences of wilderness visitation (Cole and Landres 1996; Hartley 1999). Managers are increasingly adopting containment strategies for limiting visitor impacts, concentrating and reducing the areal extent of traffic. A primary question is whether such locations will be able to sustain such intensive visitation and what ecological consequences this policy will produce. A more thorough examination of the managerial, ecological and social significance of recreation resource impacts is also needed.

Design, Accuracy, and Precision Issues in Impact Assessment and Monitoring

Increasing application of management decision frameworks that employ indicators and standards requires more objective resource monitoring protocols and programs. Few investigations of the accuracy and precision of existing impact assessment and monitoring methodologies have been conducted. Results suggest considerable subjectivity in assessment procedures for some indicators. Additional investigations are needed to characterize and find new ways to reduce measurement error so that monitoring data reflect real changes in resource conditions. Further work on employing the results of precision investigations to define confidence intervals for management decisionmaking is also needed (see Williams and Marion 1995). Working at odds with this issue is the need for efficient and flexible monitoring protocols; otherwise managing agencies cannot adopt or sustain them over time.

Management Effectiveness

Most recreation ecology investigations have focused directly on relationships between use-related and environmental factors and fail to consider management interventions that seek to manipulate these factors. The effectiveness of management actions in avoiding or minimizing visitor impacts represents a significant and largely untapped research topic of considerable importance to managers. Examples include evaluations of improved campsite or trail design and construction, containment and dispersal impact management strategies, visitor management practices such as group size limits and Leave No Trace educational efforts, use of facilities such as fire grates, and campsite and trail maintenance efforts. Very little is known about the relative effectiveness of these and other management strategies and tactics, or the role of supporting actions.

New Locations, Activities, and Technologies

Early investigations of recreation impacts often focused on large and remote wilderness areas in the western U.S. Recently research has expanded to Midwestern and Eastern states, as well as high-use wilderness destinations (Cole and others 1997). More research is needed in high-use areas to assess the magnitude of impacts and evaluate the effectiveness of management actions in more intensively visited locations.

Impacts from off-trail hiking or dispersed activities around campsites have seldom been documented. One example is the potential ecological effects of off-site trampling and wood removal related to campfire wood collection.

As new recreation pursuits and new types of recreation equipment are gaining popularity in wilderness, there will be needs for corresponding research. One example is the use of hiking poles, which have become a common hiking and backpacking aid. Initial observations seem to suggest that poles with long sharp tips could loosen soil aggregates, possibly leading to increased muddiness and erosion by water or wind. However, no research that we are aware of

has been conducted to determine potential impacts induced by hiking poles. More empirical research is also needed for examining the impacts caused by expanding or new activities such as climbing, caving and the use of llamas.

The rapid advancement of computer and other technologies offers great potential for recreation ecology investigations, but few benefits have been realized. Promising technologies include global positioning system (GPS), geographic information systems (GIS), image capture technology and the Internet. With a greater accuracy and direct transferability of data to computer systems, GPS has been used for mapping the location of wilderness campsites and trails (Leung and Marion 1995; Monz 1998) and recently experimented on backcountry trails. The use of GIS is expanding, with a growing number of applications from spatial mapping and display of visitor distribution patterns (Wing and Shelby 1999) to spatial planning to predict potential human-wildlife conflict zones (Harris and others 1995). Image capture technology has been applied to simulate different scenarios of campsite impacts (Nassauer 1990). The Internet and World Wide Web offer an unprecedented opportunity to disseminate research results of recreation ecology studies and low-impact recreation practices. Although the applications are currently limited, use of these technologies will soon be common in all aspects of wilderness recreation research, including recreation ecology studies.

Staffing and Funding

Little progress has been made in the previous 15 years to develop and expand permanent recreation ecology research programs. The Aldo Leopold Wilderness Research Institute, established in 1993 by the USDA Forest Service, is the only national research group dedicated to developing the knowledge needed to improve the management of wilderness and other natural areas. Only one scientist at the Institute conducts research on recreation impacts in wilderness. Similarly, only one scientist in the U.S. Department of the Interior focuses on recreation impacts, in spite of that agency's considerable land and recreation management responsibilities - including National Park Service units, U.S. Fish and Wildlife Refuges and Bureau of Land Management areas. Academia and a nonprofit organization, the National Outdoor Leadership School, also each employ one scientist in the recreation ecology field of study, contributing to a national total of four scientists.

Funding is also extremely limited, with the Leopold Institute the only organization having a permanent base of annual research funding. This funding may be used to address system-wide or regional information needs of a basic or applied nature. However, even this support is generally insufficient for studies other than those of the Institute's recreation ecologist. Other funding is derived primarily from national forests and parks and is tied to specific management information needs. The most common needs over the past 15 years have been the development and initial application of visitor impact assessment and monitoring protocols.

Enhanced support for permanent federal land management sponsored centers of recreation ecology research are needed. Increased funding, particularly for basic research focused on the improvement of fundamental recreation

ecology knowledge and methodological development, is required to move this field of study to an advanced level of understanding. An increased number of scientists, representing a greater array of disciplines, are also essential to build the critical mass of researchers necessary to substantially advance knowledge. For example, there has never been a recreation ecologist with a career-level focus on visitor impacts to wildlife.

Concluding Remarks

Wilderness managers continue to be confronted by significant visitor impact problems throughout the 624-unit, 104-million-acre National Wilderness Preservation System. Visitor impacts threaten to compromise wilderness management mandates for preserving and sustaining high quality natural environments and recreational experiences. A principal goal for managing wilderness visitation is to avoid impacts that are avoidable and to minimize those that are not. To achieve this goal, wilderness managers must effectively educate and regulate visitors and manage wilderness resources.

While the areal extent of visitor impacts remains small, there is growing recognition and appreciation of their ecological, social and managerial significance. Recreation ecology has begun to document many of the impacts occurring to vegetation, soils, wildlife and water resources. Studies are also beginning to describe the extent and rates of change of these impacts, where they are occurring and their relationships to causal and noncausal factors. However, considerable gaps in our knowledge continue, and existing research staffing and funding severely limit the attainment of further knowledge.

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References

- Anderson, Dorothy H.; Lime, David W.; Wang, Theresa L. 1998. Maintaining the Quality of Park Resources and Visitor Experiences: A Handbook for Managers. TC-777. St. Paul, MN: University of Minnesota, Department of Forest Resources, Cooperative Park Studie Unit. 134p.
- Attarian, Aram; Pyke, Kath. 2000. Climbing and Natural Resources Management: An Annotated Bibliography. Raleigh, NC: North Carolina State University, Department of Parks, Recreation and Tourism Management. 59p.
- Bates, G. H. 1935. The vegetation of footpaths, sidewalks, cattracks and gateways. *Journal of Ecology*. 23: 468-487.
- Bayfield, Neil G.; Aitken, Robert 1992. Managing the Impacts of Recreation on Vegetation and Soils: A Review of Techniques. Banchory: Institute of Terrestrial Ecology. 100p.
- Bayfield, Neil G.; Barrow, Graham C. 1985. The Ecological Impacts of Outdoor Recreation on Mountain Areas in Europe and North America; Cumbria, UK. RERG Report No. 9. Wye, UK: Recreation Ecology Research Group. 203p.
- Belnap, Jayne 1998. Choosing indicators of natural resource condition: A case study in Arches National Park, Utah, USA. *Environmental Management*. 22(4): 635-642.
- Benninger-Truax, Mary; Vankat, John L.; Schaefer, Robert L. 1992. Trail corridors as habitat and conduits for movement of plant species in Rocky Mountain National Park, Colorado, USA. *Landscape Ecology*. 6(4): 269-278.
- Birchard, William, Jr.; Proudman, Robert. 2000. Appalachian trail design, construction, and maintenance (2nd ed.). Harpers Ferry, WV: Appalachian Trail Conference. 237p.
- Bratton, Susan P.; Hickler, Matthew G.; Graves, James H. 1979. Trail erosion patterns in Great Smoky Mountains National Park. *Environmental Management*. 3(5): 431-445.
- Brown, Perry J.; McCool, Stephen F.; Manfredi, Michael J. 1987. Evolving concepts and tools for recreation user management in wilderness: A state-of-knowledge review. In: Lucas, Robert C., comp. *Proceedings-National Wilderness Research Conference: Issues, State-of-Knowledge, Future Directions*; Fort Collins, CO. Gen Tech Rep INT-220. Ogden, UT: USDA Forest Service, Intermountain Research Station: 320-346.
- Burde, John H.; Conway, Terry; Ervin, Denise 1998. Backcountry trails standards in Eastern wilderness and natural areas. In: Kulhavy, David L. and Legg, Michael H., eds. *Wilderness and Natural Areas in Eastern North America: Research, Management and Planning*. Nacogdoches, TX: Stephen F. Austin State University, Center for Applied Studies in Forestry: 133-137.
- Burde, John H.; Renfro, James R. 1986. Use impacts on the Appalachian Trail. In: Lucas, Robert C., comp. *Proceedings-National Wilderness Research Conference: Current Research*; Fort Collins, CO. General Technical Report INT-212. Ogden, UT: USDA Forest Service, Intermountain Research Station: 138-143.
- Camp, Richard J.; Knight, Richard L. 1998. Effects of rock climbing on cliff plant communities at Joshua Tree National Park, California. *Conservation Biology*. 12(6): 1302-1306.
- Cilimborg, A.; Monz, Christopher A.; Kehoe, S. K. 2000. Wildland recreation and human waste: A review of problems, practices and concerns. *Environmental Management*. 25(6):587-598.
- Clark, Roger N.; Stankey, George H. 1979. Determining the acceptability of recreational impacts: An application of the Outdoor Recreation Opportunity Spectrum. In: Ittner, Ruth; Potter, Dale R.; Agee, James K.; Anschell, Susan, eds. *Recreational Impact on Wildlands: Conference Proceedings*; Seattle, WA. USFS#R-6-001-1979. Seattle, WA: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station and USDI National Park Service: 32-42.
- Cole, David N. 1978. Estimating the susceptibility of wildland vegetation to trailside alteration. *Journal of Applied Ecology*. 15: 281-286.
- Cole, David N. 1981. Managing ecological impacts at wilderness campsites: An evaluation of techniques. *Journal of Forestry*. 79(2): 86-89.
- Cole, David N. 1983. Assessing and Monitoring Backcountry Trail Conditions. Research Paper INT-303. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 10p.
- Cole, David N. 1986. Ecological Changes on Campsites in the Eagle Cap Wilderness, 1979-1984. Research Paper INT-368. Ogden, UT: USDA Forest Service, Intermountain Research Station. 15p.
- Cole, David N. 1987a. Effects of three seasons of experimental trampling on five montane forest communities and a grassland in western Montana, USA. *Biological Conservation*. 40: 219-244.
- Cole, David N. 1987b. Research on soil and vegetation in wilderness: A state-of-knowledge review. In: Lucas, Robert C., comp. *Proceedings-National Wilderness Research Conference: Issues, State-of-Knowledge, Future Directions*; Fort Collins, CO. General Technical Report INT-220. Ogden, UT: USDA Forest Service, Intermountain Research Station: 135-177.
- Cole, David N. 1988. Disturbance and Recovery of Trampled Montane Grassland and Forests in Montana. Research Paper INT-389. Ogden, UT: USDA Forest Service, Intermountain Research Station. 37p.
- Cole, David N. 1989a. Area of Vegetation Loss: A New Index of Campsite Impact. Research Note INT-389. Ogden, UT: USDA Forest Service, Intermountain Research Station. 4p.
- Cole, David N. 1989b. Low-Impact Recreational Practices for Wilderness and Backcountry. General Technical Report INT-265. Ogden, UT: USDA Forest Service, Intermountain Research Station. 131p.

- Cole, David N. 1989c. Recreation ecology: What we know, what geographers can contribute. *Professional Geographer*. 41(2): 143-148.
- Cole, David N. 1989d. Wilderness Campsite Monitoring Methods: A Sourcebook. General Technical Report INT-259. Ogden, UT: USDA Forest Service, Intermountain Research Station. 57p.
- Cole, David N. 1990a. Ecological impacts of wilderness recreation and their management. In: Hendee, John C.; Stankey, George H., and Lucas, Robert C. *Wilderness Management* (2nd Ed.). Golden, CO: North American Press: 425-466.
- Cole, David N. 1990b. Trampling disturbance and recovery of cryptogamic soil crusts in Grand Canyon National Park. *Great Basin Naturalist*. 50(4): 321-325.
- Cole, David N. 1991. Changes on Trails in the Selway-Bitterroot Wilderness, Montana, 1978-1989. Research Paper INT-212. Ogden, UT: USDA Forest Service, Intermountain Research Station. 5p.
- Cole, David N. 1992. Modeling wilderness campsites: Factors that influence amount of impact. *Environmental Management*. 16(2): 255-264.
- Cole, David N. 1993a. Campsites in Three Western Wildernesses: Proliferation and Changes in Condition Over 12 to 16 Years. Research Paper INT-463. Ogden, UT: USDA Forest Service, Intermountain Research Station. 15p.
- Cole, David N. 1993b. Trampling Effects on Mountain Vegetation in Washington, Colorado, New Hampshire, and North Carolina. Research Paper INT-464. Ogden, UT: USDA Forest Service, Intermountain Research Station. 56p.
- Cole, David N. 1994. Backcountry impact management: Lessons from research. *Trends*. 31(3): 10-14.
- Cole, David N. 1995a. Disturbance of natural vegetation by camping: Experimental applications of low-level stress. *Environmental Management*. 19(3): 405-416.
- Cole, David N. 1995b. Experimental trampling of vegetation. I. Relationship between trampling intensity and vegetation response. *Journal of Applied Ecology*. 32: 203-214.
- Cole, David N. 1995c. Experimental trampling of vegetation. II. Predictors of resistance and resilience. *Journal of Applied Ecology*. 32: 215-224.
- Cole, David N. 1995d. Recreation Trampling Experiments: Effects of Trampler Weight and Shoe Type. Research Note INT-RN-425. Ogden, UT: USDA Forest Service, Intermountain Research Station. 4p.
- Cole, David N. 1996. Wilderness Recreation Use Trends, 1965 Through 1994. Research Paper INT-RP-488. Ogden, UT: USDA Forest Service, Intermountain Research Station. 10p.
- Cole, David N. 1997. Experimental Evaluations of Two Leave-No-Trace Techniques: Removing Boots and Using Geotextile Groundcloths (Scrim). Research Paper INT-RP-497. Ogden, UT: USDA Forest Service, Intermountain Research Station. 7p.
- Cole, David N. 2000. Soil amendments and planting techniques: campsite restoration in the Eagle Cap Wilderness, Oregon. In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. *Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management*; 1999 May 23-27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Cole, David N.; Bayfield, Neil G. 1993. Recreational trampling of vegetation: Standard experimental procedures. *Biological Conservation*. 63: 209-215.
- Cole, David; Benedict, Jim 1983. Wilderness campsite selection - What should users be told? *Park Science*. 3(4): 5-7.
- Cole, David N.; Hall, Troy E. 1992. Trends in Campsite Condition: Eagle Cap Wilderness, Bob Marshall Wilderness, and Grand Canyon National Park. Research Paper INT-453. Ogden, UT: USDA Forest Service, Intermountain Research Station. 40p.
- Cole, David N.; Knight, Richard L. 1990. Impacts of recreation on biodiversity in wilderness. In: *Wilderness Area: Their Impacts - Proceedings of a Symposium*; Logan, UT. Logan, UT: Utah State University: 33-40.
- Cole, David N.; Landres, Peter B. 1996. Threats to wilderness ecosystems: Impacts and research needs. *Ecological Applications*. 6(1): 168-184.
- Cole, David N.; Marion, Jeffrey L. 1988. Recreation impacts in some riparian forests of the eastern United States. *Environmental Management*. 12(1): 99-107.
- Cole, David N.; Ranz, Beth 1983. Temporary campsite closures in the Selway-Bitterroot Wilderness. *Journal of Forestry*. 81(11): 729-732.
- Cole, David N.; Schreiner, Edward G. S. 1981. Impacts of Backcountry Recreation: Site Management and Rehabilitation - An Annotated Bibliography. General Technical Report INT-121. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 58p.
- Cole, David N.; Spildie, David R. 1998. Hiker, horse, and llama trampling effects on native vegetation in Montana, USA. *Journal of Environmental Management*. 53(1): 61-71.
- Cole, David N.; Trull, Susan J. 1992. Quantifying vegetation response to recreational disturbance in the North Cascades, Washington. *Northwest Science*. 66(4): 229-236.
- Cole, David N.; Petersen, Margaret E.; Lucas, Robert C. 1987. Managing Wilderness Recreation Use: Common Problems and Potential Solutions. General Technical Report INT-230. Ogden, UT: USDA Forest Service, Intermountain Research Station. 60p.
- Cole, David N.; Watson, Alan E.; Hall, Troy E. and others 1997. High-Use Destinations in Wilderness: Social and Biophysical Impacts, Visitor Responses, and Management Options. Research Paper INT-RP-496. Ogden, UT: USDA Forest Service, Intermountain Research Station. 30p.
- Coleman, Rosalind A. 1977. Simple techniques for monitoring footpath erosion in mountain areas of North-West England. *Environmental Conservation*. 4(2): 145-148.
- de Gouvenain, R. C. 1996. Indirect impacts of soil trampling on tree growth and plant succession in the North Cascade Mountains of Washington. *Biological Conservation*. 75: 279-287.
- DeLuca, T. H.; Patterson, W. A. IV; Freimund, Wayne A. and others 1998. Influence of llamas, horses, and hikers on soil erosion from established recreation trails in western Montana, USA. *Environmental Management*. 22(2): 255-262.
- Demrow, Carl; Salisbury, David 1998. *The Complete Guide to Trail Building and Maintenance* (3rd Ed.). Boston, MA: Appalachian Mountain Club Books. 256p.
- Doucette, Joseph E.; Cole, David N. 1993. Wilderness Visitor Education: Information About Alternative Techniques. General Technical Report INT-295. Ogden, UT: USDA Forest Service, Intermountain Research Station. 37p.
- Doucette, Joseph E.; Kimball, K. D. 1990. Passive trail management in northeastern alpine zones: A case study. In: More, Thomas A.; Donnelly, Maureen P.; Graefe, Alan R.; Vaske, Jerry J., eds. *Proceedings of the 1990 Northeastern Recreation Research Symposium*; Saratoga Springs, NY. General Technical Report NE-145. Radnor, PA: USDA Forest Service, Northeastern Forest Experiment Station: 195-201.
- Dustin, Daniel L.; McAvoy, Leo H. 1982. The decline and fall of quality recreation opportunities and environments? *Environmental Ethics*. 4(1): 49-57.
- Eagan, Sean; Newman, Peter 1999. Plant community re-establishment in former high-elevation campsites in Yosemite National Park. In: Harmon, David, ed. *On the Frontiers of Conservation: Proceedings of the 10th Conference on Research and Resource Management in Parks and on Public Lands*; Asheville, NC. Hancock, MI: The George Wright Society: 239-244.
- Edington, John M.; Edington, M. Ann 1986. *Ecology, Recreation and Tourism*. Cambridge, UK: Cambridge University Press. 200p.
- Ells, M. D. 1997. Impact of human waste disposal on surface water runoff, The Muir Snowfield, Mount Rainier. *Journal of Environmental Health*. 59(8): 6-12.
- Farrell, Tracy A.; Marion, Jeffrey L. 1997. An Evaluation of Camping Impacts and Their Management at Isle Royale National Park. Final Management Report. Blacksburg, VA: U.S. Geological Survey, Virginia Tech Cooperative Park Studies Unit. 98p.
- Fenn, Dennis B.; Gogue, G. Jay; Burge, Raymond E. 1976. Effects of Campfire on Soil Properties. *Ecological Services Bulletin* No. 5. Washington, DC: USDI National Park Service. 16p.
- Fodor, Paul A. 1990. Backcountry campsite management and monitoring in Sequoia and Kings Canyon National Parks. In: Lime, David W., ed. *Managing America's Enduring Wilderness Resource*; Minneapolis, MN. St. Paul, MN: University of Minnesota, Agricultural Experiment Station and Extension Service: 188-191.
- Frissell, Sidney S. 1978. Judging recreation impacts on wilderness campsites. *Journal of Forestry*. 76(8): 481-483.

- Genetti, C. M.; Zenone, P. G. 1987. The Effects of Rock Climbers on the Environment at Pinnacles National Monument, Monterey and San Benito Counties, California. Technical Report No. 27. Davis, CA: USDI National Park Service, Cooperative Park Studies Unit, University of California at Davis. 68p.
- Gettinger, Dean S.; Krumpke, Edwin E.; Wright, Richard G. 1998. Recreational Impacts to Wilderness Campsites at North Cascades National Park. Natural Resources Report NPS/CCSOU/NRTR-98/14. Moscow, ID: USGS Biological Resources Division, University of Idaho Wildlife Management Institute. 109p.
- Graefe, Alan R.; Kuss, Fred R.; Vaske, Jerry J. 1990. Visitor Impact Management: The Planning Framework. Washington, DC: National Parks and Conservation Association. 105p.
- Gutzwiller, Kevin J.; Anderson, Stanley H. 1999. Spatial extent of human-intrusion effects on subalpine bird distributions. *The Condor*. 101: 378-389.
- Gutzwiller, Kevin J.; Marcum, Heidi A.; Harvey, Henry B. and others 1998. Bird tolerance to human intrusion in Wyoming montane forests. *The Condor*. 100: 519-527.
- Gutzwiller, Kevin J.; Wiedenmann, Richard T.; Clements, Krista L. and others 1994. Effects of human intrusion on song occurrence and singing consistency in subalpine birds. *The Auk*. 111(1): 28-37.
- Hall, Christine N. 1989. Using Impact Indices and Baseline Vegetation Data to Assess the Condition of an Eastern Wilderness: A Case Study of the Dolly Sods. College Park, MD: The University of Maryland. PhD Dissertation. 315p.
- Hall, Christine N.; Kuss, Fred R. 1989. Vegetation alteration along trails in Shenandoah National Park, Virginia. *Biological Conservation*. 48: 211-227.
- Hammitt, William E.; Cole, David N. 1998. *Wildland Recreation: Ecology and Management* (2nd Ed.). New York: John Wiley and Sons. 361p.
- Hampton, Bruce; Cole, David 1995. *Soft Path: How to Enjoy the Wilderness Without Harming It* (Revised and Updated). Harrisburg, PA: Stackpole Books. 222p.
- Hanbey, Russell 1992. *On-Site Restoration Methods for Mountainous Areas of the West*. Missoula, MT: USDA Forest Service, Intermountain Research Station.
- Harris, Lisa K.; Gimblett, H. Randy; Shaw, William W. 1995. Multiple use management: Using a GIS model to understand conflicts between recreationists and sensitive wildlife. *Society and Natural Resources*. 8: 559-572.
- Hartley, Ernest 1999. Visitor impacts at Logan Pass, Glacier National Park: A thirty-year vegetation study. In: Harmon, David, ed. *On the Frontiers of Conservation: Proceedings of the 10th Conference on Research and Management in National Parks and on Public Lands*; Asheville, NC. Hancock, MI: The George Wright Society: 297-305.
- Hendee, John C.; Koch, Russel W. 1990. Wilderness management planning. In: Hendee, John C.; Stankey, George H., and Lucas, Robert C. *Wilderness Management* (2nd Ed.). Golden, CO: North American Press: 195-214.
- Hendee, John C.; Stankey, George H.; Lucas, Robert C. 1990. *Wilderness Management* (2nd Ed.). Golden, CO: North American Press. 546p.
- Hesselbarth, Woody; Vachowski, Brian 1996. *Trail Construction and Maintenance Notebook*. 9623-2833-MTDC. Missoula, MT: USDA Forest Service, Technology and Development Program. 139p.
- Hollenhorst, Steven; Gardner, L. 1994. The indicator performance estimate approach to determining acceptable wilderness conditions. *Environmental Management*. 18(6): 901-906.
- Ittner, Ruth; Potter, Dale R.; Agee, James K. and others 1979. *Recreational Impact on Wildlands: Conference Proceedings*; Seattle, WA. R-6-001-1979. Seattle, WA: USDA Forest Service and USDI National Park Service. 333p.
- IUCN 1967. *Towards a New Relationship of Man and Nature in Temperate Lands. Part I: Ecological Impact of Recreation and Tourism upon Temperate Environments*; Lucerne, Switzerland. IUCN Publications New Series No. 7. Morges, Switzerland: IUCN-The World Conservation Union. 287p.
- Kasworm, W. F.; Monley, T. L. 1990. Road and trail influences on grizzly bears and black bears in northwest Montana. In: Darling, L. M.; Archibald, W. R., eds. *Bears: Their Biology and Management: Proceedings of the 8th International Conference*; Victoria, British Columbia: International Association for Bear Research and Management: 79-84.
- Knight, Richard L.; Cole, David N. 1991. Effects of recreational activity on wildlife in wildlands. *Transactions of the North American Wildlife and Natural Resource Conference*. 56: 238-247.
- Knight, Richard L.; Cole, David N. 1995. Wildlife responses to recreationists. In: Knight, Richard L. and Gutzwiller, Kevin J., eds. *Wildlife and Recreationists: Coexistence Through Management and Research*. Washington, DC: Island Press: 51-70.
- Knight, Richard L.; Gutzwiller, Kevin J. 1995. *Wildlife and Recreationists: Coexistence Through Management and Research*. Washington, DC: Island Press. 373p.
- Knudson, Douglas M.; Curry, Elizabeth B. 1981. Campers' perceptions of site deterioration and crowding. *Journal of Forestry*. 79(2): 92-94.
- Krumpe, Edwin E. 2000. Wilderness planning: a state-of-knowledge review. In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. *Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management*; 1999 May 23–27; Missoula, MT. *Proceedings RMRS-P-15-VOL-5*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Krumpe, Edwin E.; McCool, Stephen F. 1997. Role of public involvement in the Limits of Acceptable Change wilderness management system. In: McCool, Stephen F.; Cole, David N., comps. *Proceedings—Limits of Acceptable Change and Related Planning Processes: Progress and Future Directions General Technical Report INT-GTR-371*. Ogden, UT: USDA Forest Service, Intermountain Research Station: 16-20.
- Kuss, Fred R. 1986a. Impact ecology knowledge is basic. In: Lucas, Robert C., comp. *Proceedings-National Wilderness Research Conference: Current Research*; Fort Collins, CO. General Technical Report INT-212. Ogden, UT: USDA Forest Service, Intermountain Research Station: 92-93.
- Kuss, Fred R. 1986b. A review of major factors influencing plant responses to recreation impacts. *Environmental Management*. 10(5): 637-650.
- Kuss, Fred R.; Hall, Christine N. 1991. Ground flora trampling studies: Five years after closure. *Environmental Management*. 15(5): 715-727.
- Kuss, Fred R.; Graefe, Alan R.; Vaske, Jerry J. 1990. *Visitor Impact Management: A Review of Research*. Washington, DC: National Parks and Conservation Association. 256p.
- Landres, Peter; Meyer, Shannon 1998. *National Wilderness Preservation System database. General Technical Report RMRS-GTR-18*. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station. 97p.
- Leung, Yu-Fai; Marion, Jeffrey L. 1995. A Survey of Campsite Conditions in Eleven Wilderness Areas of the Jefferson National Forest. Blacksburg, VA: USDI National Biological Service, Virginia Tech Cooperative Park Studies Unit. 79p.
- Leung, Yu-Fai; Marion, Jeffrey L. 1996. Trail degradation as influenced by environmental factors: A state-of-knowledge review. *Journal of Soil and Water Conservation*. 51(2): 130-136.
- Leung, Yu-Fai; Marion, Jeffrey L. 1998. Evaluating spatial qualities of visitor impacts to recreation resources: An index approach. *Journal of Applied Recreation Research*. 23(4): 367-389.
- Leung, Yu-Fai; Marion, Jeffrey L. 1999a. Assessing trail conditions in protected areas: An application of a problem-assessment method in Great Smoky Mountains National Park, USA. *Environmental Conservation*. 26(4).
- Leung, Yu-Fai; Marion, Jeffrey L. 1999b. Characterizing backcountry camping impacts in Great Smoky Mountains National Park, USA. *Journal of Environmental Management*. 57(3): 193-203.
- Leung, Yu-Fai; Marion, Jeffrey L. 1999c. The influence of sampling interval on the accuracy of trail impact assessment. *Landscape and Urban Planning*. 43(4): 167-179.
- Leung, Yu-Fai; Marion, Jeffrey L. 1999d. Spatial strategies for managing visitor impacts in national parks. *Journal of Park and Recreation Administration*. 17(4): 20-38.
- Leung, Yu-Fai; Marion, Jeffrey L. 2000. Wilderness campsite conditions under an unregulated camping policy: An Eastern example. In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. *Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management*; 1999 May 23–27; Missoula, MT. *Proceedings RMRS-P-15-VOL-5*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

- Leung, Yu-Fai; Marion, Jeffrey L.; Ferguson, Janet Y. 1997. Methods for assessing and monitoring backcountry trail conditions: An empirical comparison. In: Harmon, David, ed. *Making Protection Work: Proceedings of the 9th George Wright Society Conference on Research and Resource Management in Parks and on Public Lands*; Albuquerque, NM. Hancock, MI: The George Wright Society: 406-414.
- Liddle, Michael J. 1991. Recreation ecology: Effects of trampling on plants and corals. *Trends in Ecology, Evolution and Systematics*. 6(1): 13-17.
- Liddle, Michael J. 1997. *Recreation Ecology: The Ecological Impact of Outdoor Recreation and Ecotourism*. London: Chapman and Hall. 664p.
- Lime, David W.; Buchman, R. G. 1974. Putting wilderness permit information to work. *Journal of Forestry*. 72: 622-626.
- Lucas, Robert C. 1979. Perceptions of non-motorized recreational impacts: A review of research findings. In: Ittner, Ruth; Potter, Dale R.; Agee, James K.; Anschell, Susan, eds. *Recreational Impact on Wildlands: Conference Proceedings*; Seattle, WA. R-6-001-1979. Seattle, WA: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station and USDI National Park Service: 24-31.
- Lucas, Robert C. 1982. Recreation regulations-When are they needed? *Journal of Forestry*. 80(3): 148-151.
- Lucas, Robert C. 1990a. How Wilderness Visitors Choose Entry Points and Campsites. Research Paper INT-428. Ogden, UT: USDA Forest Service, Intermountain Research Station. 12p.
- Lucas, Robert C. 1990b. Wilderness use and users: Trends and projections. In: Hendee, John C.; Stankey, George H., and Lucas, Robert C. *Wilderness Management* (2nd Ed.). Golden, CO: North American Press: 355-398.
- Magill, Arthur W. 1989. Monitoring Environmental Change with Color Slides. General Technical Report PSW-117. Berkeley, CA: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station. 55p.
- Manning, Robert E.; Lime, David W. 2000. Recreation experiences and management: A state-of-knowledge review. In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. *Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management*; 1999 May 23-27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Marcus, W. Andrew; Milner, Gary; Maxwell, Bruce 1998. Spotted knapweed distribution in stock camps and trails of the Selway-Bitterroot Wilderness. *Great Basin Naturalist*. 58(2): 156-166.
- Marion, Jeffrey L. 1991. Developing a Natural Resource Inventory and Monitoring Program for Visitor Impacts on Recreation Sites: A Procedural Manual. Natural Resources Report NPS/NRVT/NRR-91/06. Denver, CO: USDI National Park Service, Natural Resources Publication Office. 59p.
- Marion, Jeffrey L. 1994a. An assessment of trail conditions in Great Smoky Mountains National Park. Research/Resources Management Report. Atlanta, GA: USDI National Park Service, Southeast Region. 155p.
- Marion, Jeffrey L. 1994b. Changes in campsite condition: results from campsite monitoring at Delaware Water Gap National Recreation Area. Technical Report NPS/MARDEWA/NRTR-94/063. Philadelphia, PA: USDI National Park Service, Mid-Atlantic Region. 83p.
- Marion, Jeffrey L. 1995. Capabilities and management utility of recreation impact monitoring programs. *Environmental Management*. 19(5): 763-771.
- Marion, Jeffrey L. 1998. Recreation ecology research findings: Implications for wilderness and park managers. In: Kirchner, H., ed. *Proceedings of the National Outdoor Ethics Conference*; St. Louis, MO. Gaithersburg, MD: Izaak Walton League of America: 188-196.
- Marion, Jeffrey L.; Brame, Susan C. 1996. Leave No Trace outdoor skills and ethics: An educational solution for reducing visitor impacts. *Park Science*. 16(3): 24-26.
- Marion, Jeffrey L.; Cole, David N. 1989. Evaluating recreation impacts: A multi-faceted research design. *Park Science*. 9(2): 23-24.
- Marion, Jeffrey L.; Cole, David N. 1996. Spatial and temporal variation in soil and vegetation impacts on campsites. *Ecological Applications*. 6(2): 520-530.
- Marion, Jeffrey L.; Leung, Yu-Fai 1997. An assessment of campsite conditions in Great Smoky Mountains National Park. Research/Resources Management Report. Atlanta, GA: USDI National Park Service, Southeast Regional Office. 135p.
- Marion, Jeffrey L.; Leung, Yu-Fai 1998. Campsite survey implications for managing designated campsites at Great Smoky Mountains National Park. In: Kulhavy, David L. and Legg, Michael H., eds. *Wilderness and Natural Areas in Eastern North America: Research, Management and Planning*. Nacogdoches, TX: Stephen F. Austin State University, Center for Applied Studies in Forestry: 146-155.
- Marion, Jeffrey L.; Lime, David W. 1986. Recreational resource impacts: Visitor perceptions and management responses. In: Kulhavy, David L.; Conner, Richard N., eds. *Wilderness and Natural Areas in the Eastern United States: A Management Challenge*. Nacogdoches, TX: Stephen F. Austin State University, School of Forestry: 229-235.
- Marion, Jeffrey L.; Snow, Skip 1990. Developing a campsite impact monitoring system for Everglades National Park: A case study. In: Lime, David W., ed. *Managing America's Enduring Wilderness Resource*; Minneapolis, MN. St. Paul, MN: University of Minnesota, Agricultural Experiment Station and Extension Service: 192-198.
- Marion, Jeffrey L.; Sober, Toivo 1987. Environmental impact management in the Boundary Waters Canoe Area Wilderness. *Northern Journal of Applied Forestry*. 4(1): 7-10.
- Marion, Jeffrey L.; Cole, David N.; Bratton, Susan P. 1986. Exotic vegetation in wilderness areas. In: Lucas, Robert C., comp. *Proceedings-National Wilderness Research Conference: Current Research*; Fort Collins, CO. General Technical Report INT-212. Ogden, UT: USDA Forest Service, Intermountain Research Station: 114-120.
- Marion, Jeffrey L.; Roggenbuck, Joseph W.; Manning, Robert E. 1993. Problems and Practices in Backcountry Recreation Management: A Survey of National Park Service Managers. Natural Resources Report NPS/NRVT/NRR-93/12. Denver, CO: USDI National Park Service, Natural Resources Publication Office. 64p.
- McClaran, Mitchel P.; Cole, David N. 1993. Packstock in Wilderness: Use, Impacts, Monitoring, and Management. General Technical Report INT-301. Ogden, UT: USDA Forest Service, Intermountain Research Station. 33p.
- McCool, Stephen F.; Cole, David N., comps. 1997. *Proceedings—Limits of Acceptable Change and Related Planning Processes: Progress and Future Directions*; Lubrecht Exp. Forest, MT. General Technical Report INT-GTR-371. Ogden, UT: USDA Forest Service, Intermountain Research Station. 84p.
- McEwen, Douglas; Cole, David N. 1997. Campsite impact in wilderness areas. *Parks and Recreation*. 32(2): 24-32.
- McEwen, Douglas; Cole, David N.; Simon, Mark 1996. Campsite Impacts in Four Wildernesses in the South-Central United States. Research Paper INT-RP-490. Ogden, UT: USDA Forest Service, Intermountain Research Station. 12p.
- Meinecke, Emilio 1928. A Report on the Effect of Excessive Tourist Travel on the California redwood parks. Sacramento, CA: California State Printing Office. 20p.
- Merigliano, Linda L. 1990. Indicators to monitor wilderness conditions. In: Lime, David W., ed. *Managing America's Enduring Wilderness Resource*; Minneapolis, MN. St. Paul, MN: University of Minnesota, Agricultural Experiment Station and Extension Service: 205-209.
- Merrill, Evelyn H. 1978. Bear depredations at backcountry campgrounds in Glacier National Park. *Wildlife Society Bulletin*. 6(3): 123-127.
- Miller, Scott G.; Knight, Richard L.; Miller, Clint K. 1998. Influence of recreational trails on breeding bird communities. *Ecological Applications*. 8(1): 162-169.
- Monz, Christopher A. 1998. Monitoring recreation resource impacts in two coastal areas of western North America: An initial assessment. In: Watson, Alan E.; Alphet, Gregory H.; Hendee, John C., comps. *Personal, Societal and Ecological Values of Wilderness: Sixth World Wilderness Congress Proceedings on Research, Management and Allocation*, Vol. 1. RMRS-P-4. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station: 117-122.
- Monz, Christopher A.; Cole, David N.; Johnson, L. A. and others 1994. Response of five native plant communities to trampling in

- the Wind River Range, Wyoming, USA. *Bulletin of the Ecological Society of America*. 75(2): 158.
- Monz, Christopher A.; Meier, G. A.; Buckley, R. C. and others 1996. Responses of moist and dry arctic tundra to trampling and warmer temperatures. *Bulletin of Ecological Society of America*. 77(3): 311.
- Moritsch, Barbara J.; Muir, Patricia S. 1993. Subalpine revegetation in Yosemite National Park, California: Change in vegetation after three years. *Natural Areas Journal*. 13(3): 155-163.
- Nassauer, Joan I. 1990. Using image capture technology to generate wilderness management solutions. In: Lime, David W., ed. *Managing America's Enduring Wilderness Resource*; Minneapolis, MN. St. Paul, MN: University of Minnesota, Agricultural Experiment Station and Extension Service: 553-562.
- National Park Service 1997a. A Summary of the Visitor Experience and Resource Protection (VERP) Framework. Publication No. NPS D-1214. Denver, CO: NPS Denver Service Center. 35p.
- National Park Service 1997b. The Visitor Experience and Resource Protection (VERP) Framework: A Handbook for Planners and Managers. Publication No. NPS D-1215. Denver, CO: NPS Denver Service Center. 103p.
- National Park Service 1998. Shenandoah National Park: Backcountry and Wilderness Management Plan. Luray, VA: Shenandoah National Park.
- Noss, Reed F.; Cooperrider, Allen Y. 1994. *Saving Nature's Legacy: Protecting and Restoring Biodiversity*. Washington, DC: Island Press. 416p.
- Nuzzo, Victoria A. 1995. Effects of rock climbing on cliff goldenrod (*Solidago sciaphila* Steele) in Northwest Illinois. *American Midland Naturalist*. 133(2): 229-241.
- Nuzzo, Victoria A. 1996. Structure of cliff vegetation on exposed cliffs and the effect of rock climbing. *Canadian Journal of Botany*. 74(4): 607-617.
- Olson-Rutz, K. M.; Marlow, C. B.; Hansen, K. and others 1996a. Packhorse grazing behavior and immediate impact on a timberline meadow. *Journal of Range Management*. 49(6): 546-550.
- Olson-Rutz, K. M.; Marlow, C. B.; Hansen, K. and others 1996b. Recovery of a high elevation plant community after packhorse grazing. *Journal of Range Management*. 49(6): 541-545.
- Parsons, David J. 1986. Campsite impact data as a basis for determining wilderness use capacities. In: Lucas, Robert C., comp. *Proceedings-National Wilderness Research Conference: Current Research*; Fort Collins, CO. General Technical Report INT-212. Ogden, UT: USDA Forest Service, Intermountain Research Station: 449-455.
- Parsons, David J.; Stohlgren, Thomas J. 1987. Impacts of Visitor Use on Backcountry Campsites in Sequoia and Kings Canyon National Parks, California. Technical Report No. 25. Davis, CA: USDI National Park Service, Cooperative Park Studies Unit, University of California at Davis. 79p.
- Price, Martin F. 1983. Management planning in the Sunshine Area of Canada's Banff National Park. *Parks*. 7(4): 6-10.
- Riffell, S. K.; Gutzwiller, Kevin J.; Anderson, Stanley H. 1996. Does repeated human intrusion cause cumulative declines in avian richness and abundance? *Ecological Applications*. 6(2): 492-505.
- Rochefort, Regina; Swinney, Darin D. 2000. Human impact surveys in Mount Rainier National Park: Past, present, and future. In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. *Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management*; 1999 May 23–27; Missoula, MT. *Proceedings RMRS-P-15-VOL-5*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Roggenbuck, Joseph W.; Williams, Daniel R.; Watson, Alan E. 1993. Defining acceptable conditions in wilderness. *Environmental Management*. 17(2): 187-197.
- Saunders, Paul R. 1986. Shining Rock Wilderness: impacts of dispersed use. In: Kulhavy, David L.; Conner, Richard N., eds. *Wilderness and natural areas in the Eastern United States: a management challenge*. Nacogdoches, TX: Stephen F. Austin State University, School of Forestry: 260-264.
- Scott, Ruth L. 1998. Wilderness management and restoration in high use areas of Olympic National Park, Washington, U.S.A. In: Watson, Alan E.; Alphet, Gregory H.; Hendee, John C., comps. *Personal, Societal and Ecological Values of Wilderness: Sixth World Wilderness Congress Proceedings on Research, Management and Allocation*, Vol. 1 RMRS-P-4. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station: 144-147.
- Shakarjian, Mikel J.; Stanford, Jack A. 1999. Impacts to river biota studied in Zion Narrows. *Park Science*. 19(1): 12.
- Shelby, Bo; Shindler, Bruce 1992. Interest group standards for ecological impacts at wilderness campsites. *Leisure Sciences*. 14(1): 17-27.
- Shelby, Bo; Vaske, Jerry J.; Harris, Richard 1988. User standards for ecological impacts at wilderness campsites. *Journal of Leisure Research*. 20(3): 245-256.
- Spildie, David R.; Cole, David N.; Walker, Sarah C. 2000. Effectiveness of a confinement strategy in reducing packstock impacts at campsites in the Selway-Bitterroot Wilderness, Idaho. In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. *Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management*; 1999 May 23–27; Missoula, MT. *Proceedings RMRS-P-15-VOL-5*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Stankey, George H.; Cole, David N.; Lucas, Robert C. and others 1985. The Limit of Acceptable Change (LAC) System for Wilderness Planning. General Technical Report INT-176. Ogden, UT: USDA Forest Service, Intermountain Research Station. 37p.
- Stewart, William P. 1989. Fixed itinerary systems in backcountry management. *Journal of Environmental Management*. 29: 163-171.
- Stohlgren, Thomas J.; Parsons, David J. 1986. Vegetation and soil recovery in wilderness campsites closed to visitor use. *Environmental Management*. 10(3): 375-380.
- Stohlgren, Thomas J.; Parsons, David J. 1992. Evaluating wilderness recreational opportunities: Application of an impact matrix. *Environmental Management*. 16(3): 397-403.
- Summer, Rebecca M. 1986. Geomorphic impacts of horse traffic on montane landforms. *Journal of Soil and Water Conservation*. 41(1): 126-128.
- Sumner, E. Lowell 1942. The biology of wilderness protection. *Sierra Club Bulletin*. 27(4): 14-22.
- Taylor, James Y. 1997. Leave only footprints? How backcountry campsite use affects forest structure. *Yellowstone Science*. 5(1): 14-17.
- Trafimow, David; Borrie, William 1999. Influencing future behavior by priming past behavior: A test in the context of Petrified Forest National Park. *Leisure Sciences*. 21: 31-42.
- Tyser, Robin W.; Worley, Christopher A. 1992. Alien flora in grasslands adjacent to road and trail corridors in Glacier National Park, Montana (U.S.A.). *Conservation Biology*. 6(2): 253-262.
- van Wagtendonk, Jan W.; Parsons, David J. 1996. Wilderness research and management in the Sierra Nevada national parks. In: Halvorson, William L. and Davis, Gary E., eds. *Science and Ecosystem Management in the National Parks*. Tucson, AZ: University of Arizona Press: 281-294.
- van Wagtendonk, Jan; McClaran, Mitchel P.; Cole, David N. and others. 2000. Meadow response to packstock grazing in the Yosemite wilderness: Integrating research and management. In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. *Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management*; 1999 May 23–27; Missoula, MT. *Proceedings RMRS-P-15-VOL-5*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Vaske, Jerry J.; Graefe, Alan R.; Dempster, A. 1982. Social and environmental influences on perceived crowding. In: Boteler, Franklin E., ed. *Proceedings: Third Annual Conference of the Wilderness Psychology Group*; Morgantown, WV. Morgantown, WV: West Virginia University, Division of Forestry: 211-227.
- Wagar, J. Alan 1964. *The Carrying Capacity of Wild Lands for Recreation*. Forest Science Monograph 7. Washington, DC: Society of American Foresters. 24p.
- Washburne, Randel F.; Cole, David N. 1983. Problems and Practices in Wilderness Management: A Survey of Managers. Research Paper INT-304. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 56p.
- Watson, Alan E. 2000. Wilderness use in the Year 2000: Societal changes that influence human relationships with wilderness. In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. *Wilderness science in a time of change*

- conference—Volume 5: Wilderness ecosystems, threats, and management; 1999 May 23–27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Watson, Alan; Cole, David 1992. LAC indicators: An evaluation of progress and list of proposed indicators. In: Merigliano, Linda L., ed. *Ideas for Limits of Acceptable Change Process (Book 2)*. Washington, DC: USDA Forest Service, Recreation Staff: 65-84.
- Weaver, Tad; Dale, Donn 1990. Effects of trampling on the understories of whitebark pine forests. In: Schmidt, W. C.; McDonald, K. J., eds. *Proceedings - Symposium on Whitebark Pine Ecosystems: Ecology and Management of a High-Mountain Resource*; Bozeman, MT. General Technical Report INT-270. Ogden, UT: USDA Forest Service, Intermountain Research Station: 375.
- Westman, Walter F. 1985. *Ecology, Impact Assessment, and Environmental Planning*. New York: John Wiley and Sons. 544p.
- Whittaker, Douglas; Shelby, Bo 1988. Types of norms for recreation impacts: Extending the social norms concept. *Journal of Leisure Research*. 20(4): 261-273.
- Williams, Peter B.; Marion, Jeffrey L. 1995. *Assessing Campsite Conditions for Limits of Acceptable Change Management in Shenandoah National Park*. Technical Report NPS/MARSHEN/NRTR-95/071. Blacksburg, VA: USDI National Biological Service, Virginia Tech Cooperative Park Studies Unit. 138p.
- Williams, Peter B.; Marion, Jeffrey L. 1997. *Assessment of Backcountry Campsite Conditions in Big Bend National Park*. Research/Resources Management Report. Blacksburg, VA: U.S. Geological Survey, Virginia Tech Cooperative Park Studies Unit. 112p.
- Wing, Michael; Shelby, Bo 1999. Using GIS to integrate information on forest recreation. *Journal of Forestry*. 97(1): 12-16.
- Yorks, Terrence P.; West, Neil E.; Mueller, Richard J. and others 1997. Tolerant of traffic by vegetation: Life form conclusions and summary extracts from a comprehensive data base. *Environmental Management*. 21(1): 121-131.
- Zabinski, Catherine A.; Gannon, James E. 1997. Effects of recreational impacts on soil microbial communities. *Environmental Management*. 21(2): 233-238.
- Zika, Peter F. 1991. The role of recreation in the extirpation of alpine plant species in the northeastern United States. In: *Yosemite Centennial Symposium Proceedings - Natural Areas and Yosemite: Prospects for the Future*; Yosemite NP, CA. NPS D-374. Denver, CO: USDI National Park Service, Denver Service Center: 554-559.

Improving Livestock Management in Wilderness

Mitchel P. McClaran

Abstract—Recreation livestock (horses, mules, llamas, and goats) use accounts for 11% of all wilderness visits, and production livestock (cattle and sheep) use occurs in 37% of all wilderness areas. Recreation use is expected to increase at the same rate as total wilderness use, but production use will change little. Managers should recognize that the relationship between the severity of impacts and the intensity of livestock use can be linear or curvilinear because different management approaches will be effective for each type of relationship. Improved livestock management will occur with greater coordination of knowledge and staff in range and wilderness management.

Recreation and production livestock in wilderness are authorized under different provisions in the Wilderness Act, and distinct tools and criteria are prescribed to manage their impacts. Recreation livestock use conforms to the recreational mission of the Act and is subject to full discretionary interpretation by agencies to manage that use within levels consistent with a goal of maintaining the wilderness character of an area. In contrast, production livestock use is one of five uses (mining, aircraft & motorboats, control of fire, disease & insects, water resources facilities, and livestock grazing) that were granted special status to continue in wilderness if they existed prior to designation. Further, some provisions for production livestock management explicitly ignore the wilderness character goal of an area, leaving little room for discretionary interpretation by the agencies.

Each type of livestock use produces impacts from defoliation, trampling, concentration of animal waste, reduction of wildlife, conflicts with other users, and as vectors for the spread of noxious species. However, the expression of these impacts can be quite different between recreation and production livestock use. Both the relationships among the intensity, timing and type of use, and their spatial arrangement and impacts differ between types of livestock. Recognizing these similarities and differences, can assist in improving livestock management in wilderness. To this end, this state-of-knowledge review of livestock use in wilderness will (1) describe the extent of occurrence and managers' concerns about the two livestock uses, (2) compare the nature and management implications of impacts caused by the types of livestock, (3) compare the legal framework and administrative tools applied to management of recreation

and production livestock in wilderness, and (4) outline the challenges to management, and the research and development that can improve livestock management in wilderness.

Recreation livestock in wilderness are the horses mules, burros, llamas, and goats that are used as beasts-of-burden to carry people or their belongings. Production livestock in wilderness are sheep and cattle raised for meat, wool, and leather products. Packstock is a common synonym for recreation livestock (for example, McClaran 1989), and domestic livestock is a synonym for livestock grazing on federal lands (36 Code of Federal Regulations 222.1 (1998); 43 Code of Federal Regulations 4100.5 (1998)). This paper uses the terms recreation livestock and production livestock because the shared noun stresses the similarities of the impacts and management principles for these two groups of animals, and "production" is used rather than domestic because all these animals are domesticated. Finally, the term recreation livestock is consistent with regulations promulgated by the USDA Forest Service and USDI Bureau of Land Management (36 Code of Federal Regulations 293.3 (1998); 43 Code of Federal Regulations 8560.1 (1998)).

Wilderness areas that receive either of these two types of livestock use are most common west of the Mississippi River including Alaska and Hawaii (McClaran and Cole 1993; Washburne and Cole 1983). Recreation livestock use occurs most frequently in areas administered by the Forest Service (FS) and Bureau of Land Management (BLM), to a lesser degree in USDI National Park Service (NPS) areas, and rarely in USDI Fish and Wildlife Service (FWS) areas. In general, production livestock use in wilderness follows a very similar pattern of occurrence among agencies, except for a lower frequency in NPS areas.

Beyond the practical matters of where, when, and how livestock use occurs in wilderness, it has played a role in the evolution of important wilderness concepts and in the articulation of congressional guidelines for wilderness management. Leopold's (1921) original definition of wilderness proposed that the minimum size of wilderness be large enough to allow for a two-week recreation livestock trip. The first studies of recreation impacts in wilderness focused on recreation livestock in Sequoia and Kings Canyon National Parks in the 1940s, where Sumner (1942) devised the precursor to the concept of recreational carrying capacity as a tool to set use limits that would not impair "the essential qualities of the area." Production livestock grazing was included as an accepted use in the first wilderness established in 1924, and that status has not faced serious challenge (McClaran 1990). To some, the inclusion of production livestock grazing privileges was a simple exchange for rancher acceptance of wilderness designation (Roth 1984). To the contrary, Leopold (1921) suggested that benefits would accrue to both recreationists and ranchers because livestock production under frontier conditions

In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management; 1999 May 23–27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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would hold some fascination to visitors, and the exclusion of roads and the ensuing “settlers and hordes of visitors” would benefit ranchers. Further, Wallace Stegner (1961) felt the presence of production livestock would “emphasize a man’s feeling of belonging in the natural world.” Despite these early opinions, it is the detailed congressional guidelines for the administration of the facilities and motorized vehicle use associated with production livestock in wilderness (McClaran 1990) that most influence production livestock and management in wilderness. These directives allow agencies to make very little distinction between production livestock management inside and outside of wilderness, which is in sharp contrast to the broad discretion available for recreation livestock management.

Extent of Use and Managers’ Concerns

Recreation Livestock

In 1990, recreation livestock use occurred in about 55% of FS and BLM areas, 35% of NPS areas, 7% of FWS areas, and about 50% of all wilderness areas (McClaran and Cole 1993). Overnight visits with recreation livestock accounted for about 20% of total overnight visits to all wilderness areas; 36% of visits in the Rocky Mountain region and 15% of visits in the Pacific region used recreation livestock. Among agencies, the proportion of overnight visits with recreation livestock range from nearly 30% for FS areas to 1% for BLM areas. In general, about 30% of recreation livestock use was by commercial enterprises such as outfitters, pack stations and other concessioners, about 60% of recreation livestock use was by private parties, and about 10% was by the agencies for administrative purposes such as trail maintenance and ranger patrols. Commercial and administrative uses were proportionately higher in NPS areas and in the Western states. Private use was proportionately greatest in BLM areas and in the Southeast and Midwest regions.

In both 1980 and 1990, the proportion of all wilderness area visitation (overnight and day use) by people with recreation livestock was 11% (McClaran and Cole 1993; Washburne and Cole 1983). This contrasts with the change from 1960 to 1980, when it recreation livestock use declined from the dominant to a secondary use behind backpacking. This occurred even though absolute levels of livestock use were steady or increased during that period (Lucas 1985; McClaran 1989; McClaran and Cole 1993). The recent stability in proportion of recreation livestock use is important because it occurred when wilderness visitation continued to increase in all but a few high-use wilderness areas (Cole 1996). This suggests that there has been a steady and comparable increase in demand for a wilderness experience by backpackers and groups using recreation livestock. The popularity of wilderness visits using recreation livestock is illustrated by feature articles in the travel sections of prominent newspapers (Tannen 1999). Furthermore, a majority of wilderness managers expect this increase in recreation livestock use to continue in the near future (Watson and others 1998).

Use of llamas and goats began in earnest during the 1980s in some wilderness areas, and by 1990, about half of all areas

had received some amount of use by these alternative types of recreation livestock (McClaran and Cole 1993). Between 1985 and 1990, llamas use had occurred in over half of all wilderness areas, seven wilderness areas reported more than 10 visits with llamas, and llama use accounted for more than 20% of recreation livestock use in four areas. Only 5% of all wilderness areas reported goat use between 1985 and 1990. Llama and goat use was most frequent in the Pacific region and in areas administered by the FS or NPS. Future use levels of these alternative recreation livestock will depend on the cost of obtaining and maintaining animals, creating less impact than traditional recreation livestock (Cole and Spildie 1998; DeLuca and others 1998; and Watson and others 1998), changing visitor preference from riding animals to leading animals, and developing the practice of combining animals such as riding horses and pack llamas into single strings of animals to transport both people and supplies.

In 1980, recreational livestock impacts to trails, campsites and lakeshores were considered to be a problem by about a third of all wilderness managers: nearly 30% reported impacts to trails, 45% reported impacts to campsites and about 30% reported impacts to lakeshores (Washburne and Cole 1983). These problems were most common in FS areas, and in the Rocky Mountain and Pacific regions where use was greatest. In 1989, about 60% of FS managers reported at least some moderate level of impact and about 15% reported at least a great level of impact, to trails by recreation livestock (General Accounting Office 1989). By 1990, about 45% of all managers felt that ecological impacts from recreational livestock were not adequately controlled by existing regulations (McClaran and Cole 1993). There was relatively little variation among agencies and regions in the perception of inadequacy of regulations. Although these are not exactly comparable measures of concern, they suggest a growing concern about recreational livestock impacts that are consistent with the increased amount of use during the 1980s.

Production Livestock

In 1980, about 30% of all wilderness areas had some amount of production livestock use, and that proportion rose to 35% in 1987, probably as a result of newly designated wilderness areas that had preexisting production livestock use (McClaran 1990; Reed and others 1989; Washburne and Cole 1983). In the Rocky Mountain and Pacific regions, at least half of the FS and BLM areas experienced some level of production livestock use, and, surprisingly, about 10% of NPS areas received some production livestock use (Washburne and Cole 1983). In the FS areas, sheep use was about three times more common than cattle use (General Accounting Office 1989), but cattle were more common on BLM areas. Production livestock use in wilderness has been stable over the past decade. It is extremely rare for use to commence in previously unused areas, and the termination of use is also rare (General Accounting Office 1989; McClaran 1991), but there has been a modest trend of reduced numbers of animals in these grazed areas (McClaran 1991; Reed and others 1989). This stable amount of production livestock use is in sharp contrast to the increasing use of recreation livestock, which suggests a need to extend the management expertise and attention from production livestock (range

management personnel) to recreation livestock use (recreation management personnel).

Among managers in wilderness areas with production livestock use in 1980, 10% felt that the existence of these animals contributed to management problems; and these perceptions were most common in FS and BLM areas in the Rocky Mountain and Pacific regions (Washburne and Cole 1983). Furthermore, illegal production livestock use was on a list of most significant problems in wilderness areas by five or more wilderness managers (Washburne and Cole 1983). In 1989, about 25% of FS wilderness managers stated that overgrazing by production livestock created moderate or greater impacts (General Accounting Office 1989).

Natural resource managers in Montana rated threats from production livestock impacts fifth behind fire suppression, atmospheric pollutants, recreation, and adjacent land use (Cole 1994). Cole and Landres (1996) suggested that threats to wilderness from the impacts of production livestock, fire suppression, nonnative species, and adjacent land uses were more extensive than threats from the impacts of mining, atmospheric pollutants, recreation, and water projects. Livestock impacts were considered most severe in aquatic ecosystems and riparian vegetation, especially in arid regions where these ecosystems were most rare.

Even though these measures of production livestock use and concerns are not perfectly matched with those for recreation livestock, the stable amount of production livestock use is in sharp contrast to the increasing use of recreation livestock. Managers also seem to have a more intense and pervasive concern about recreation livestock impacts than production livestock impacts. It is possible that the differences in perceived impacts reflect the managers' greater concern about impacts to recreation areas such as trails and campsites compared to the more dispersed impacts from production livestock. Nonetheless, the different trends in the amount of use between these types of livestock should stimulate increased attention to recreation livestock management. Assistance from existing personnel with production livestock management expertise can help meet this refocused attention.

Livestock Impacts and Implications for Management

Recreation and production livestock impacts to wilderness share the same agents of defoliation, trampling, concentration of animal waste products, reduction of wildlife, conflicts with other users, and vectors for noxious species. The severity of these impacts can vary in relation to the intensity, timing, and type of livestock use. The structure of these relationships between use and impact is sometimes curvilinear.

Recognizing curvilinear relationships where the severity of impact varies between each additional level of use can be valuable for resource managers (fig. 1). For example, a convex curvilinear relationship is where impacts are greatest with the initial increments of use. A convex curvilinear relationship shows that preventing the initial impacts will minimize impacts more than reducing use levels when use is already high. In contrast, the concave curvilinear relationship indicates that the greatest change in impact severity

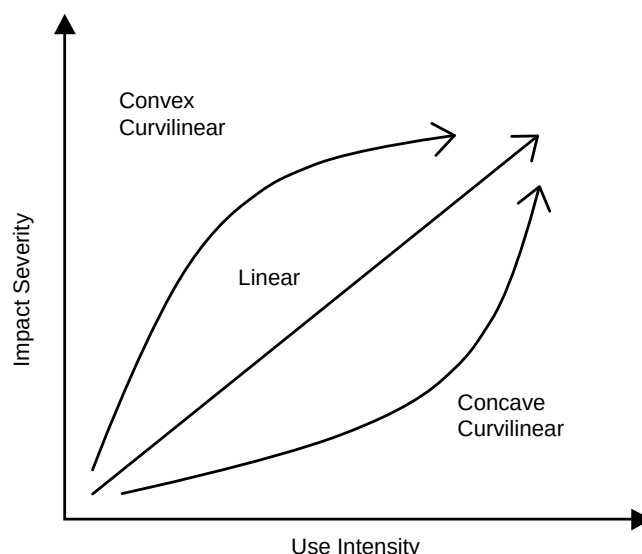


Figure 1—Comparison of linear, convex curvilinear, and concave curvilinear responses of impact severity to increasing levels of use intensity. The linear response shows equal amounts of impact with each additional increment of use. The convex curvilinear response shows unequal amounts of impact for each increment of use, and the greatest levels of impact occur with the initial amounts of use intensity. The concave curvilinear response shows unequal amounts of impact for each increment of use, and the greatest levels of impact occur with increasingly larger amounts of use intensity.

occurs when intensity of use is already high. Recognizing a concave curvilinear relationship is important when use levels are increasing toward a threshold where the next increment of use will create more severe impacts than all the previous increments of increased use.

The spatial distribution of impacts can be very different between these classes of livestock. Compared to production livestock, recreation livestock use is more concentrated along trails and in camps, and is more common in areas with little forage production. The patterns of impact severity can have important implications for the primacy of different livestock management tools. For example, if impact severity is most sensitive to intensity of use, managers should focus efforts to control the length of time and number of animals allowed in an area.

Defoliation

Defoliation of vegetation occurs when animals eat or otherwise remove plant biomass. Defoliation initially reduces leaf area, root activity, and the rate of photosynthesis; more lasting impacts are reductions in regrowth potential and the inability to persist among less heavily defoliated plants. The likelihood of defoliation is dependent on the density of livestock, availability of plant biomass, and diet preference of livestock.

In general, the intensity of defoliation limits the rate and extent of regrowth because nearly all the energy used for regrowth is generated by the remaining ungrazed leaf tissue (see Briske 1996; Briske and Richards 1995). The timing of defoliation limits regrowth if meristems (locations of cell

differentiation and elongation) are removed or environmental conditions (temperature, light, and nutrients) are limiting. Community-level impacts result from differential selection of plant species by herbivores and differential resistance to defoliation among plant species. Total plant cover decreases with increasing defoliation pressure. This pattern of reduced cover is more quickly apparent in preferred forage species and species with meristems that are easily defoliated because they are elevated or they all develop at one time (synchronized development). Eventually, reduced growth of these selected or less resistant species will result in changes in the vegetation composition.

Impacts to plant productivity appear to be more sensitive to changes in grazing intensity (animals/area/time or percent utilization of available plant biomass) than the timing of grazing. Van Poolen and Lacey's (1979) meta-analysis of 32 grazing studies in the western U.S., showed that reducing grazing intensity produced a greater response in increased plant production than implementing a seasonal rotation grazing system.

The structure of the relationship between change in species composition and grazing intensity differs from that for the relationship between change in plant productivity and grazing intensity. Based on a meta-analysis of over 250 grazing impact studies from around the world, Milchunas and Lauenroth (1993) found that changes in species composition, measured as departure from ungrazed comparisons, were linearly related to grazing intensity, site productivity, and the length of exposure (evolutionary time-frame) to grazing pressures. Changes in biomass productivity, measured as departure from ungrazed comparisons were convex curvilinearly related to grazing intensity, and linearly related to total plant productivity in the area.

The convex curvilinear relationship between impacts and grazing intensity is partly a function of decreasing intake of forage (defoliations) by animals as continued defoliation reduces the amount of available forage. This pattern is consistent among large herbivores (Huston and Pinchak 1991). For example, defoliation by recreation livestock increased, in a convex curvilinear manner as grazing time increased; decreases in plant cover after eight hours of use on a picket pin were less than double the impact after only four hours (Olson-Rutz and others 1996a,b).

Management guidelines for allowable intensity of grazing pressure (utilization levels) have become more conservative, but the emphasis on intensity over timing has been consistent. Guidelines in the early 1910s were set to prevent utilization from exceeding 75-90% of current year production; by the 1940s, they were reduced to 75%, and current guidelines are 30-45% and occasionally as high as 50% (Holechek and others 1998; Sampson 1952). There is also a trend to move from grazing intensity standards based on utilization of current-year growth to standards based on remaining plant biomass. The rationale for this shift is that regrowth is the result of the amount of leaf area remaining after defoliation, and remaining biomass is more easily and accurately measured (Heady and Child 1994). The median utilization values in the studies included in the meta-analysis by Milchunas and Lauenroth (1993) were about 45%.

Management implications for these patterns of plant response to defoliation start with the proposition that there has been a short length of exposure (evolutionary

time-frame) to grazing pressures in western U.S. wilderness areas (Milchunas and others 1988), and that these areas can be classified as low productivity sites. Given this, we should expect grazing intensity to have greater influence on plant productivity than on species composition. In wilderness areas, one can expect the impacts from production livestock to be more widely dispersed than recreation livestock, which do not venture far from camps. Actions to manage the defoliation impacts from recreation livestock can rely more on controlling the location of grazing and requiring pack-in feed compared to the management of production livestock, which will rely more on control of animal numbers. Because of a possible convex curvilinear relationship, the initial defoliation increments created by moving recreation livestock to new areas will create many new areas where there are significant impacts to productivity.

Trampling

Trampling of vegetation and soil occurs when the hooves of livestock strike the vegetation and soil during travel, grazing, or confinement, and when animals lie or roll on the ground. In general, the severity of these impacts exhibit a convex curvilinear pattern, where the initial trampling produces more severe impacts than later trampling (Cole 1995a,b; Cole and Spildie 1998; DeLuca and others 1998).

The trampling impact to vegetation (cover and height) and to soil erosion is between two and 10 times greater from horse travel (along trails or not) than from hikers or llamas doing the same amount of travel (Cole and Spildie 1998; DeLuca and others 1998; Weaver and Dale 1978; Wilson and Seney 1994). These greater impacts from horses are probably the result of both more weight per surface area contacting the ground and the metal shoes on their hooves. Apparently, these traits create a greater shearing potential, which increases the likelihood of direct plant damage and soil erosion by both compaction and loosening of soil particles. It is likely that these same patterns hold for trampling impacts in camps, because soil compaction and reduced plant cover are positively associated with camps used by recreation livestock compared with those used by hikers (Cole 1983). The severity of impacts to vegetation, for both horse and llama, varies in relation to the life form of the vegetation: graminoid (grass and grass-like plants) vegetation is the most resistant and resilient, erect forb (nongrass and nongrass-like herbaceous plants) vegetation is least resistant, and woody shrub vegetation is the least resilient (Cole 1995a,b; Cole and Spildie 1998). As a result, the convex curvilinear pattern is less pronounced in graminoid vegetation, but it is most pronounced in forb-dominated vegetation with high species diversity (Cole 1995a,b).

Production livestock trampling in corrals and locations near drinking water and forage supplements (salt, minerals, and protein) provides the closest analogy to recreation livestock trampling impact to trails and campsites. Although a direct comparison has not been reported, one would expect the trampling impact from production cattle to be less than horses and mules because they have no metal shoes, but greater than llamas because they have greater weight per surface area contacting the soil. The impact from production

sheep should be slightly greater than llamas because the hooves of sheep are less padded than llamas.

The concentration of trampling impacts is less common in grazed areas than along travel routes and in camps because animals are more likely to venture widely for forage than follow the same path. Given the choice, animals try to avoid wet areas (Platts and Nelson 1985), and the more resistant and resilient graminoid vegetation is more common in grazed areas than forb or shrub vegetation. In general, when cattle are grazing, the severity of impacts to vegetation and soil are positively associated with soil wetness and negatively associated with the length of growing season (for example, Clary 1995). The worst scenario for trampling impacts from grazing cattle is areas with wet soils and short growing seasons because compaction is more severe and the time for recovery of vegetation is shorter. In the earliest range management guidelines, Jardine and Anderson (1919) warned of trampling impacts that happen if use occurs on soils that are too wet to support the animal's weight. One must assume that this impact pattern was well accepted by managers because later guidelines (for example, Heady and Child 1994; Stoddart and Smith 1955) largely ignore the trampling impacts of early season use and focus instead on the impacts of defoliation.

Cattle appear to avoid trampling bunchgrass vegetation while grazing, and this behavior is expressed even at a high animal density, albeit at lower avoidance rates (Balph and Malechek 1985; Balph and others 1989).

Damage to trees and tree death are trampling impacts unique to recreation livestock. They result from animals being tied to trees in camps and popular day use areas (Cole 1989c). This type of tree damage is cumulative; for example, tree damage increased over a 12-year period, even when absolute use of these camps declined during that time (Cole 1993).

The implications of these patterns of trampling impact are different for high use areas and grazed areas. In high-use areas (trails, camps, corrals, water, and supplemental feed sites) where use is concentrated, the type of animal, the type of vegetation, and the history of use are the primary influences on the severity of trampling impact. Managers should attempt to prevent unintentional use in areas previously undisturbed by horses and mules, particularly where the vegetation is dominated by forbs and low shrubs. This is most critical when considering the relocation of use facilities like trails, camps, and corrals. Furthermore, active measures that prevent tying to trees should be applied in all areas, independent of use levels. For grazed areas, where use is more dispersed, intensity and season of use are most critical. Managers should attempt to minimize grazing in areas that are wet and have a short growing season.

Concentration of Animal Wastes

Fecal and urine wastes from livestock have important influences on water quality, soil nutrient status, defoliation patterns, and insect and odor concentrations. The severity of the impacts from wastes appeared to be related first to the distribution of animals, second to their concentration, and less importantly to the aridity of the area and type of animal.

Fecal coliform (FC) contamination in surface waters is used to indicate the likely presence of such pathogens as *Salmonella* and *Giardia* (Tiedemann and others 1987). FC contamination is most likely if feces are deposited directly in surface waters, but this is relatively rare (around 5%) for free-roaming animals (Gary and others 1983; Larson and others 1988). However, the likelihood of contamination increases exponentially as the proportion of animal use within a few meters of surface waters increases, because bacteria are carried to water as runoff during precipitation events. For example, increased FC contamination was more strongly influenced by the cattle use of meadows near streams than the stocking density in the entire pasture (Tiedemann and others 1987). These patterns result from a logarithmic decline in the FC concentration with distance and age of feces: significant contaminations are largely restricted to a one meter radius of feces (Buckhouse and Gifford 1976). Although FC concentration in cattle feces remains high after 30 days, it is several orders of magnitude less than the concentration found at one to two days (Kress and Gifford 1984; Thelin and Gifford 1983). Because drying strongly reduces the probability of contamination, contamination will be more likely in mesic than arid areas, and from cattle feces because they are more moist than feces from horses and sheep.

Urine deposits create patches of high nitrogen concentration in soil and plants, because urine contains the majority of nitrogen in animal wastes (Archer and Smeins 1991), even though the majority of this nitrogen is volatilized (Woodmansee 1978). This high concentration of nitrogen is followed by increased intensity of defoliation by grazing animals in the growing season subsequent to urination (Jamarillo and Detling 1992).

The relationships between waste concentration and increased number of insects and intensity of odor are not clear. They probably have a convex curvilinear structure, where the initial amounts of waste generate more of an increase in insects and odors than similar additions of wastes would generate if wastes were already very abundant.

The implications of these patterns of impact from animal wastes center on the type of livestock and the ability to prevent animal use near bodies of water. Impacts from recreation livestock may be more easily controlled if camps and holding areas are away from streams, and because horse and mule feces are drier than cattle feces. Further, the significance of dry feces in reducing contamination implies that activities to break up fecal mounds will hasten drying and reduce the probability of contamination, particularly in areas near water. Reducing production livestock use near streams may be more difficult than recreation livestock because their use is more dispersed, but efforts to fence riparian areas and develop drinking water sources away from streams can be effective. Finally, a convex curvilinear structure to insect and odor problems suggests that efforts to concentrate fecal deposits in existing areas that are far from streams should take precedence over moving use to new areas.

Reduction of Wildlife

In wilderness, livestock can reduce the abundance and occurrence of wildlife species, directly through displacement and transmission of disease, and indirectly through habitat change and reduction of forage.

Displacement has been described primarily where the presence of livestock can alter the location and movement of large mammals. Displacement by cattle and sheep is most common (see Krausman 1996). Observations of displacement by horses is limited to pronghorn antelope and wild horses (Miller 1983); no studies have described displacement by llamas. Theoretically, the seasonal or yearly ungrazed pastures in multi-pasture rotational grazing systems for production livestock rather than the alternative, continuously grazed pastures, should provide preferred areas for these wild ungulates. However, observations of wildlife preference in areas managed under rotational grazing systems have recorded mixed results. Mule deer (Peek and Krausman 1996) and white-tailed deer (Teer 1996) appear to favor these grazing systems over continuously grazed areas, but there were inconsistent results in the studies for elk (Wisdom and Thomas 1996).

Bighorn sheep are the most sensitive species to diseases transmitted by livestock, and pronghorn antelope also exhibit sufficient susceptibility to warrant concern (Jessup and Boyce 1996). Pneumonia transmission from production sheep to bighorn sheep has been repeatedly documented, and transmission by cattle is suspected. Llamas are known carriers of paratuberculosis (Jessup and Boyce 1996), but there appear to be no known cases of transmission to wild animals. For these diseases, the only effective management is complete isolation of livestock from bighorn sheep.

Defoliation and trampling by livestock can create immediate and more long-lasting changes in vegetation structure and composition that can indirectly influence the habitat quality for wildlife species. Immediate changes include reduction of herbaceous plant abundance, most particularly plant height, and these changes can greatly alter the abundance of upland birds (Knopf 1996). The significance of these short-term changes are primarily a function of the season of use because the habitat requirement of many upland birds is not year-long and the vegetation will regrow. Aquatic life, particularly cold-water fish, are also susceptible to these short-term changes in plant height because tall vegetation shades the water, modifying temperatures, and contributing detritus that supports insect that are prey (Platts and Nelson 1985). Current livestock management recommendations prescribe minimum plant heights that should remain in areas grazed by livestock (Clary 1995; Knopf 1996). Longer-term changes in structure and composition caused by livestock use include the increase of woody species and reduction of herbaceous species, and a general loss of plant cover (Archer and Smeins 1991). Livestock grazing intensity has more influence on the severity of these long-term changes than season of grazing.

Finally, the impacts of insufficient forage for wildlife species is largely a function of the intensity of livestock use. The dietary overlap between livestock and wildlife species will largely determine the relative susceptibility of wildlife species. Cattle and horse diets generally overlap most with elk, and bison; while sheep (and presumably llamas) diets generally overlap most with deer, bighorn sheep, and pronghorn antelope (Vallentine 1990). However, feral horse diets can be quite similar to pronghorn antelopes (McInnis and Vavra 1987).

These general patterns of impacts to wildlife suggest that displacement by production livestock will be greater than by

recreation livestock because the former are more widely dispersed. Therefore, seasonal rotation of livestock among pastures may be beneficial because the availability of ungrazed areas will reduced displacement problems. Production livestock, especially sheep, can transmit disease to wildlife, and the prevention of transmission by any livestock species should be taken seriously. Controlling the timing of both production and recreation livestock use will best address short-term impacts to the vegetation structure of wildlife habitat, whereas, controlling livestock numbers is more critical in minimizing long-term changes in habitat. Finally, controlling the number and type of livestock will best address the problems of reduced forage for wildlife.

Conflicts With Other Users

Conflicts between livestock and other wilderness users come in two forms: conflicts with firmly held attitudes of appropriateness that can be considered a predisposition to conflict, and conflicts with activities encountered during a visit to wilderness that can be considered situational conflicts (Ivy and others. 1992). In general, the severity of both types of conflicts with hikers is greater with production than recreation livestock.

About 40% of hikers in five FS wilderness areas were predisposed to conflict with production livestock in wilderness (Johnson and others 1997), and the severity of that conflict is greater for visitors that reside in urban versus rural areas (Mitchell and others 1996). Furthermore, that conflict is greater for production livestock in wilderness than in nonwilderness camping areas (Mitchell and Wallace 1998). The severity of conflicts with production livestock declines as the hikers' expectation of encountering livestock increases (Johnson and others 1997), but the structure (linear or curvilinear) of this relationship is unknown. Situational conflicts center on encountering manure and livestock-related structures such as fences (Johnson and others 1997). The quality of the wilderness visit was diminished for about two-thirds of hikers when they encountered cattle and sheep, compared to nearly 75% of visitors when encountering fences, and about 50% of visitors when encountering recreation livestock or any type of visitor. Observing these animals near water and camps was the most sensitive encounter for hikers. Finally, perception of overgrazing or excessive defoliation of plant biomass was the foremost indicator of the visitor perceiving improper livestock management by the FS.

A predisposition to believe that horses are inappropriate in wilderness was the most consistent contribution to severe hiker conflict with recreation livestock in wilderness, but most hikers did not express conflicts (Watson and others 1993). Manure on trails, large group size, and litter were the most irritating recreation livestock situations encountered by hikers, but hikers were most sensitive to a general impression of overcrowding from all visitor types. At least one study suggests that the severity of these conflicts is inversely related to the intensity of recreation livestock use (Stankey 1979): conflicts were less severe in areas with higher amounts of livestock use. The level of wilderness manager's acceptance of recreation livestock increased with their level of experience using these animals in wilderness (Moore and McClaran 1991). Predisposition appears to play

a role in greater acceptance of encounters with llama than horses and mules (Blahna and others 1995). Interestingly, hikers are more accepting of llamas than horse and mule users, and acceptance by wilderness managers is between that of hikers and traditional livestock users (Blahna and others 1995; Watson and others 1998). Transmission of disease to wild animals and encouraging the introduction of nonnative plants by llamas were the situations of greatest concern to all wilderness users (Watson and others 1998).

The management implications of these patterns of users conflicts focus on two areas: predisposition to conflict and on-site reduction of negative encounters. Several authors suggest that increasing hikers' awareness of encountering production or recreation livestock may help relieve conflicts by discouraging visits by those with the greatest predisposition against these uses (Blahna and others 1995; Johnson and others 1997; Watson and others 1993). Zoning wilderness areas to separate recreation livestock from sensitive visitors has also been discussed (Blahna and others 1995; Watson and others 1993). Onsite reduction of animals and manure near water and camps may help relieve the severity of situational conflicts, but the use of fences to achieve these goals may be counterproductive because 75% of hikers reported that fences detracted from the quality of their visit. Reducing the level of defoliation by livestock may reduce conflicts. Finally, encouraging greater familiarity with a wilderness experience that includes recreation livestock may encourage greater acceptance of these uses and their unique impacts, but it may not reverse strongly held predispositions.

Vectors for Noxious Species

Livestock have long been labeled as vectors that encourage the spread of noxious (unwanted native and nonnative species), either directly through the spread of seed or indirectly through disturbance (Elton 1958). In wilderness areas, it is common to find a greater abundance of nonnative plants in recreation livestock camps than backpacker camps, a difference that has increased over time (Cole and Hall 1992). To control this problem, managers must understand whether actions that prevent seed introduction will be more effective than efforts to prevent disturbance.

Contrary to most suggestions (see review in D'Antonio and Vistousek 1992), livestock grazing is not a prerequisite for the presence or abundance of nonnative plant species. Recent work in the western United States suggests that the abundance of nonnative plant species is no different in grazed and ungrazed areas (Lacy 1989; Stohlgren and others 1999), and that the rate of spread is not associated with the intensity of livestock grazing (McClaran and Anable 1992). The pattern is different for camps: recreation livestock camps have a higher abundance of nonnative species than hiker camps (Cole and Hall 1992). However, although the extent and intensity of disturbance are greater in livestock camps, the role of livestock use as a vector for the transport of these plants must also be considered. Recent calls for weed-free feed certification led Cash and others (1998) to assess the viability of weed seed following the feed pelleting process. They found seed viability dropped dramatically with increasing hammering and drying applications in the pelleting process, but no treatment completely eliminated viable weed seeds.

In relation to the spread of animal species, increasing abundance of the brown-headed cowbird, a native brood parasite, with livestock movements can be detrimental to some bird species. Furthermore, the relationship between cowbird abundance and livestock is probably convex curvilinear. Because of their association with bison, cowbirds were probably common throughout the Rocky Mountain region and the Great Plains (Chance and Cruz 1998), but the spread of cowbirds is a historical event in the Pacific and Great Basin regions where bison were absent (Rothstein 1994).

Because the spread of nonnative plants is loosely correlated with the presence of livestock, management efforts should focus on preventing livestock from transporting seed into wilderness areas by requiring the most aggressive weed-free feed and animal handling. Pelleted feed may not be aggressive enough, and more attention should be paid to quarantining animals for one or two days before admission into wilderness to prevent transport of ingested seed. Impacts from cowbirds are more likely from production livestock than recreation livestock because the former are more widely dispersed.

Legal and Administrative Frameworks for Recreation Livestock Management in Wilderness

The legal framework for livestock management defines the discretion given to agencies by Congress and the formal regulations that the agencies have developed to meet the directives from Congress. The discretion available to develop unique management programs is much greater for recreation livestock than for production livestock.

Administration of livestock use includes the establishment of impact standards and monitoring, the application of management tools, and the assignment of personnel to these responsibilities. There is more variation in the administration of recreation than production livestock, and this is expected considering the smaller amount of discretion for managing the latter.

Legal and Regulatory Framework

The legal framework for recreation livestock use and management in wilderness is given in the Wilderness Act (16 United States Code 1133b (1998)). Recreation is a public purpose that agencies will provide in wilderness, while at the same time the agencies are responsible for preserving the wilderness character of an area so that "...its community of life [is] untrammelled by man...without permanent improvements... and] the imprint of man's work [is] substantially unnoticeable." (16 United States Code 1131c (1998)). While these two purposes are instantly at odds when agencies permit potentially destructive recreation use, the conflict is even more pronounced for recreation livestock use because it has a greater potential for destruction than hiking. However, the use of recreation livestock can become a practical matter because the prohibition of motor vehicles and mechanical transport (16 United States Code 1133c

(1998)) makes recreation livestock the only non-pedestrian means of transporting people, equipment and supplies on land that fully complies, without exception, with the intent of Congress. (Simple exceptions to this prohibition are emergencies of human health and safety, administrative use, and pre-existing uses grandfathered in statute or congressional guidelines.) In combination, these three elements of the Wilderness Act—visitation, preservation, and transportation—define the latitude and the tension of recreation livestock administration in wilderness.

Overarching directives to BLM, FS and FWS agents stress that recreation use is subordinate to the maintenance of wilderness conditions (36 Code of Federal Regulations 293.2 (1998); 43 Code of Federal Regulations 8560.0-6 (1998); 50 Code of Federal Regulations 35.2 (1998)). In contrast, recreation management regulatory directives for NPS agents do not specify any unusual management for wilderness areas (36 Code of Federal Regulations 2.16 (1998)).

BLM agents are given very detailed regulatory directives for wilderness recreation management compared to the other agencies. BLM agents are directed to use (1) the principles of nondegradation to establish recreation use capacity, (2) the minimum management tool to establish use facilities, (3) the principle of wilderness dependence to resolve conflicts between different recreation uses, and (4) indirect methods to reduce recreation impacts such as trail design, and information and education, rather than direct methods such as regulating the use of saddle horses and or packstock, managing areas strictly for foot or horse use, requiring permits for entry, limiting party size or number of parties during overnight visits, limiting number of users, and restrictions to stock grazing or canoe/boat beaching in popular areas (46 Federal Register 47183-47188 (1981)). Specific to recreation livestock, agents are authorized to (1) issue permits for commercial users, and, (2) require users to carry native feed or pellets, and (3) hobble rather than tether horses (46 Federal Register 47196-47197 (1981)). FS agents are authorized to prohibit entry and grazing (36 Code of Federal Regulations 261.57(1998)) in wilderness areas, and specifically they may limit grazing of recreation livestock (36 Code of Federal Regulations 293.3 (1998)). FWS agents are authorized to limit number of visitors, season of use, kind and location of use, and require permits for access (50 Code of Federal Regulations 35.6(1998)), but there is no specific directive pertaining to recreation livestock management. NPS agents are directed to prohibit loose-herding and use outside of trails or other designated areas; and to enforce any other prohibition established by park superintendents (36 Code of Federal Regulations 2.16 (1998)).

Administrative Framework

Impact Standards and Monitoring—The Limits of Acceptable Change process (Stankey and others 1985) and general wilderness management philosophy (Hendee and others 1990) suggest that the development of impact standards should be part of the wilderness management planning process, where users and administrators interact to set the level of acceptable impacts (standards) and where these standards will be applied. Surprisingly, only 35% of all wilderness areas used public participation to develop impact standards for recreation livestock use in 1990, whereas

professional judgement was used in about 61% of areas (McClaran and Cole 1993). Tradition-based standards were the second most common approach (used in 40% of the areas), while existing standards for production livestock and research-based standards were the least common (27% and 22% of areas) approaches to establishing impact standards.

The greater reliance on professional judgement and tradition may be a function of very limited research about recreation livestock impacts and management (Cole 1989a; General Accounting Office 1989), but the low frequency of public participation is antithetical to the principles and legal requirements of public land management. Furthermore, the absence of public involvement is likely to result in conflict between managers and users, and among users. It is unfortunate that more use was not made of production livestock standards because their long history of development and administration could provide a useful perspective when considering impact standards for recreation livestock.

In 1990, about two-thirds of wilderness areas monitored recreation livestock impacts in at least some camps and about one-half monitored impacts to trails, but less than 30% monitored impacts to grazed areas or other visitors (McClaran and Cole 1993). This surprisingly low frequency of monitoring, especially in grazed areas, may be explained by findings that managers have insufficient resources to monitor impacts (General Accounting Office 1989), but it may also reflect insufficient development and training in monitoring methods for grazed areas. Monitoring methods for trails and campsites are outlined in Cole (1989b), and grazed areas methods are outlined in the FS and BLM range management handbooks and in many other sources (see Bonham 1989; McClaran and Cole 1993; Muir and McClaran 1997). The missed opportunities from the infrequent use of traditional range management resources about monitoring impacts, mirrors the infrequent use of production livestock standards to set impact standards. Monitoring efforts that concentrate on measurements of intensity, such as utilization or standing biomass, may be the most robust parameters to measure because intensity of use is important in the severity of many livestock impacts. Simple plant height measures can provide reliable estimates of defoliation intensity.

Management Tools—Each agency that is responsible for recreation livestock management in wilderness promotes different management tools, and, moreover, the application of these tools ranges from strictly enforced regulations to guidelines used to promote voluntary behavior. Some have argued that the use of guidelines conforms to a minimum management tool that should be used to least infringe on the visitor's experience (Hendee and others 1990; Lucas 1982, 1983), whereas others suggest that regulations can be viewed as a more equitable approach to visitor management (Dustin and McAvoy 1984).

In practice, the application of regulations versus guidelines for managing recreation livestock is different among the agencies. About 60% of all wilderness areas with overnight recreation livestock use had some form of use regulation in 1990 (McClaran and Cole 1993). NPS agents were most likely (91%) to rely on regulations and BLM agents were least likely (13%). The NPS's greater inclination to regulate reflects an impact prevention philosophy, while the less frequent application of regulations by the FS reflects a philosophy of

reducing impacts only after they reach unacceptable levels (see discussion in McClaran and Cole 1993). In addition to these differences among agencies, there appears to be a greater propensity to rely on new regulations to solve problems if some regulations are already in place. On closer examination of responses to the 1990 survey (McClaran and Cole 1993), I found that only 34% of managers in areas without any regulations perceived a need for more regulations to correct excessive impacts compared to 56% of managers in areas with at least one existing regulation.

Less than 5% of wilderness areas with recreation livestock controlled amount of use by regulating total animal numbers or number of groups using livestock in 1990 (McClaran and Cole 1993). The use of regulations was very rare except in NPS areas, where they were present in about 30% of those areas. The infrequent attention to intensity or amount of use neglects the very important influence that intensity has on the severity of impacts from plant defoliation to vegetation and wildlife habitat; it also overlooks the importance of perceptions of overgrazing by other visitors. In addition, this absence of regulations addressing total use stands in contrast to the general concern among managers about overgrazing.

Timing of use was regulated in 5% of all wilderness areas with recreation livestock use in 1990 (McClaran and Cole 1993), a proportion that was unchanged from 1980 (Washburne and Cole 1983). In general, guidelines to control season of use are more popular than regulations, however, no NPS or FWS areas employed guidelines and no BLM area attempted to control timing of use. The infrequent control of timing of use ignores the important influence that time of use can have on the severity of trampling impacts when soils are most wet. Greater attention to controlling the timing of use may address managers' concerns about recreation livestock impacts to trails and camps.

Management tools that address the location of impacts recognize that impact severity is often site-specific. For example, the severity of user conflicts is related to encounters along trails and in camps, and trampling impacts are most severe in areas that have not previously received use and chronically wet areas. These location-oriented tools include efforts to alter the location of use with behavioral rules, site-specific rules, and the construction of facilities that attract use to less sensitive areas.

In 1990, management to alter user behavior were most likely to address length-of-stay and group size limits (about 40% of areas), and most other controls were applied in less than 20% of areas. In comparison, guidelines were applied in 50-65% of areas to reduce off-trail use, prevent tying to trees, and encourage the use of pack-in feed (McClaran and Cole 1993). The situational (on-site) conflicts among users can be managed with length-of-stay limits in areas with popular campsites and grazing areas, or by prohibiting loose-herding (no ropes tied between animals) in areas with an abundance of hiker use. However, these tools will not address the predispositional conflicts among users. Efforts to prohibit off-trail travel would prevent the formation of new trails, and limits on group size might reduce the probability that existing campsites would expand to accommodate larger groups. Similarly, preventing the tying of stock to trees and encouraging the use of hitchlines (ropes tied between trees on which livestock are tied) can prevent the initial and most severe trampling impacts to trees.

In 1990, the use of pack-in feed was encouraged in nearly two-thirds of all areas and required in 15% (McClaran and Cole 1993). In general, this behavior is encouraged to manage the impacts of grazing in areas that have especially sensitive conditions such as water-logged soils or rare plants, conflicts with other users, or areas that want to limit the amount of grazing. Pack-in feed can be used to maintain animal vigor during a long journey. However, a point of diminishing returns develops when the benefits of reduced impact per animal from pack-in feed is diminished by the greater number of animals that are needed to carry more feed. While encouraging the use of pack-in feed is a worthy practice, there must be some provision to ensure that these feeds do not hasten the spread of nonnative plant species. Therefore, it is essential that all pack-in feed, whether required or recommended, be certified weed-free or processed into pellets to reduce the transport of weed seeds. One to two day quarantine measures before admission to wilderness might be tested as well, and should be considered even without the use of pack-in feed.

In 1990, site-specific rules were most commonly applied as regulations concerning the location of campsites: about 40% of all areas with overnight use required camping a minimum distance (either 100 or 200 feet) from water and nearly 10% limited camping to specifically designated "stock camps." In those few (about 5%) wilderness areas using season of use or total use regulations, about 80% applied them on a site-specific basis (McClaran and Cole 1993). These site-specific controls certainly address animal waste contamination of surface waters and control the location of impacts in high-use areas.

Providing facilities as an indirect approach to modifying user behavior, was implemented in up to 20% of wilderness areas. Facilities included hitchracks, pastures/corrals, drift fences, or water developments were most common in NPS and FS areas (McClaran and Cole 1993). Because these facilities are very effective at attracting use, they can increase the severity of impacts in these areas, but they will reduce impacts to areas without these facilities. These facilities should be evaluated for both the overall reduction in impacts they provide to other areas, and for their compliance with minimum tool directives.

In summary, the application of management tools largely ignores direct controls of intensity and timing of use; and instead focuses on altering use behavior, and, secondarily, on the location of use. This style of management largely ignores the important influence that intensity and timing of use have on the severity of impacts by recreation livestock. One must wonder if regulations on intensity and timing were envisioned by the 45% of managers reporting a need for more regulations in 1990 (McClaran and Cole 1993). The greater application of tools that address behavior and location can prevent initial impacts in new areas and avoid undue regulations in lightly used areas.

Personnel—In the vast majority of wilderness areas, recreation livestock use is administered by recreation or wilderness staff rather than range management personnel. Most recreation and wilderness personnel have little technical training in establishing impact standards or monitoring impacts to soils, vegetation, wildlife or other users. Fortunately, in some areas, wilderness management planning is conducted by interdisciplinary teams that include members

with more expertise in these areas. However, the implementation of these plans usually resorts to the wilderness and recreation staff, and in the majority of cases, personal judgement forms the basis for recreation livestock impacts standards in wilderness. Administrators should foster a greater involvement from range management staff in the development of these standards and the implementation of monitoring programs.

Legal and Administrative Frameworks for Production Livestock Management in Wilderness

Legal and Regulatory Framework

The legal and regulatory directives for production livestock and their management in wilderness are far more specific than for recreation livestock. These directives maintain production livestock use if it was present before wilderness designation, and they stipulate very different impact standards and management tools for increases in livestock use compared to maintenance of existing use levels (McClaran 1990).

Growing tension between Congress and the FS about the administration of production livestock in wilderness climaxed in 1980, when wilderness designation legislation was proposed for wilderness areas with significant amounts of grazing (Roth 1984). In an effort to ensure designations and to standardize livestock administration to conform with congressional intent, a set of grazing guidelines were forged by a group composed of House of Representative Committee members, FS staff, wilderness advocates, and livestock industry representatives. Although these guidelines are not an official amendment to the Wilderness Act, they have been cited as management criteria in every wilderness designation statute since 1980 for FS and BLM areas with any pre-existing grazing, and they have been incorporated into regulatory language (46 Federal Register 47194 (1981)) and agency handbooks (McClaran 1990). McClaran (1990) documents this trend from 1980 through the Arizona Desert Wilderness Act of 1990. Since 1989, the congressional grazing guidelines were explicitly cited in all four statutes designating areas with pre-existing grazing. The four statutes are: Arizona Desert Wilderness Act of 1990 (104 Statutes at Large 4469 Sec. 101.f); Nevada Wilderness Protection Act of 1990 (104 Statutes at Large 1784 Sec. 6.a); Colorado Wilderness Act of 1994 (107 Statutes at Large 756 Sec. 3.2.b); and the California Desert Conservation Act of 1994 (108 Statutes at Large 4471 Sec. 103.c.).

These five congressional grazing guidelines appeared in House Report No. 617 (96th Congress, 1st Session, prepared for the Colorado Wilderness Act of 1980, 94 Statutes at Large 3265 and is codified at 16 United States Code 1133-other provisions). The guidelines address the administration and management of animal numbers, facilities, and mechanized equipment aspects of production livestock use in wilderness (McClaran 1990):

1. Wilderness designation will not be a criteria for reducing animal numbers, and increasing animal numbers is

permitted only if wilderness values are not adversely impacted.

2. Using motorized equipment and vehicles to continue to maintain livestock management structures and facilities will follow a rule of practical necessity and reasonableness.

3. Using natural materials is not required when repairing or constructing livestock management structures and facilities, unless it does not result in unreasonable additional cost.

4. Construction of new facilities should be primarily for resource protection and management, not for increasing the amount of livestock use, but replacement of existing facilities is permitted.

5. Using motorized vehicles will be permitted for emergency access to sick animals and emergency placement of supplemental feed.

Administrative Framework

In essence, the congressional guidelines created two sets of standards for the administration of production livestock use in wilderness; one that applied to use that existed prior to wilderness designation and a second set for any additional animals, facilities, or equipment use after wilderness designation. The impact criteria and management tools for maintaining existing use are the same as those applied outside of wilderness; but for additional use, the impact criteria and management tools are based on preventing impacts to wilderness values.

Whether in wilderness or outside wilderness, livestock administration is organized by grazing allotments, where impact standards, monitoring, and management tools are prescribed in an allotment management plan (36 Code of Federal Regulations 222.2 (1998); 43 Code of Federal Regulations 4120.2 (1998); 46 Federal Register 47195 (1981)). These allotment management plans (AMP) conform to the multiple-use provisions established in the relevant FS Forest Plan or BLM Resource Management Plan (36 Code of Federal Regulations 222.2 (1998), 43 Code of Federal Regulations 4100.0-8 (1998), so the types of natural resources and the mix of multiple-uses in each allotment will result in different standards and tools for each grazing allotment. Since the early 1990s, the scheduled revisions of AMPs have conformed with requirements of the National Environmental Policy Act (83 Statutes at Large 852) which include formal public participation procedures. The average AMP is revised either every 10-15 years, or more frequently if ownership of the grazing permit changes or there are new resource conflicts among the multiple uses. All FS AMPs are scheduled for revision using National Environmental Act procedures between 1995-2000 under the Recission Act of 1995 (109 Statutes at Large 212).

Impact Standards and Monitoring—Impact standards used for pre-existing livestock use typically prescribe acceptable levels of forage utilization, changes in vegetation composition, and soil erosion. These standards are set to minimize resource deterioration while integrating livestock use with the other ongoing uses and values in an area. In general, utilization standards range from 30-50% use of current season production of biomass, and this is typically measured for the dominant forage species in the area. More

conservative utilization standards are applied to support other uses, such as wildlife habitat requirements, or to stimulate changes in vegetation composition. Vegetation composition standards have traditionally been based on the potential natural vegetation in the area before Anglo-American settlement, but there is an increasing trend to establish standards that match the “desired” composition determined in the FS Forest Plan, BLM Resource Management Plan and AMP public participation process because some uses may result in composition that is different from the potential or pristine vegetation (West and Smith 1997). Soil erosion standards are the least articulated of the three, but the general goal is to prevent erosion rates from exceeding natural levels.

In contrast, impact standards are based on preserving the wilderness character of the area when there is a proposal to increase livestock use above the level existing before wilderness designation. For example, standards applied by the BLM (46 Federal Register 47184 (1981)) include minimizing the detection of human work on the land, maximizing potential natural vegetation composition, and minimizing erosion.

It is possible that the difference between wilderness and nonwilderness standards will diminish with the recent implementation of a new impact standard process by the BLM and a possible new direction for standards in the FS. The BLM is beginning the process of applying new Standards and Guidelines (43 Code of Federal Regulations 4180 (1998)) that are likely to stress ecological conditions more than previous standards. The FS solicited a recent review by a group of scientists that concluded with recommendations refocusing attention toward ecological sustainability and less emphasis on multiple use (Johnson and others 1999).

Monitoring grazing allotments to assess the level of impacts with respect to the impact standards is not performed as frequently as one would hope, and “problem” allotments typically are monitored most frequently. Utilization is the impact standard that is most commonly monitored, but it is rarely measured every year. Utilization is typically estimated with a standard height-weight conversion for dominant forage species, or clipping biomass in paired grazed-ungrazed plots (Bonham 1989). Vegetation composition and soil erosion receive cursory attention during efforts to monitor utilization, and the vegetation composition on many allotments has not been formally measured for at least 10 years and often more than 20 years.

Management Tools—The management tools used on any allotment are articulated in the AMP and the lease agreement between the agency and the livestock operator. These documents include a grazing schedule that details the amount, season, type of grazing animal, and location. The amount of use is measured in animal unit months (AUM), the amount of forage needed to support a mature cow with small calf for a month (approximately 800-1000 lbs of forage). Theoretically, AUMs can be converted among different types of animals (for example 1 AUM = 5 sheep grazing for a month), but differences in metabolism and diet will distort the accuracy of these conversions (Holechek et al. 1998). Season of use describes the start and finish of grazing in a given year, and location of use describes where the grazing will take place. Taken as a whole, the AMP describes a grazing schedule that can be as simple as one herd of animals in one location (pasture) for a set period of time, or

as complicated as several herds of animals moving among many different locations where the length of stay is determined by amount of utilization rather than a set calendar date. These more complicated arrangements are referred to as rotational grazing systems, and they are developed to foster improvement in vegetation composition and/or animal performance. Fences, water developments, herding, and diet supplements (salt, minerals, and protein are most common) are used to control the location of animals.

Certainly, the potential for controlling the severity of production livestock impacts are in place with the availability of these diverse management tools and the planning requirements for AMPs. These tools are capable of addressing the intensity, timing, and location of use. However, it is not easy to know how frequently they are being applied or how effectively they are working. It is important to recognize that while congressional grazing guidelines for livestock administration in wilderness provide the opportunity to construct new facilities to control resource damage (see abbreviated guideline #4); such as fences to exclude livestock from surface water areas, wet areas, or sensitive vegetation (Cole and Landres 1996), new fences are very likely to be a major source of conflict for hikers.

Personnel—Most production livestock managers have university-level training in range management or similar disciplines. This training includes monitoring methods, plant and soil identification, and livestock management. Furthermore, most new AMPs are developed using an interdisciplinary team that includes wildlife and recreation specialists. However, the number of range management personnel has declined over the past 15 years in most FS and BLM units, and this may explain the infrequency of monitoring on allotments.

Management Challenges and Research and Development Needs

Management Challenges

The challenge to wilderness managers is to control and reduce the livestock impacts that 25-45% of managers find unacceptable, and to accomplish this in the face of increasing recreation livestock use and constrained options for management of production livestock that are defined in the congressional grazing guidelines. The probability of meeting these challenges could be improved with the following changes in livestock administration: (1) develop defensible impact standards, (2) implement reliable and frequently applied monitoring programs, (3) apply needed management tools, and (4) increase the number of personnel working in wilderness that have been trained in range management.

The development of defensible impact standards will require a combination of public input, research findings, use of accepted production livestock standards, and continual validation from repeated monitoring. Resolving differences among users will be difficult, given the high degree of predisposition of hikers against any livestock other than llamas, and the predisposition of traditional recreation livestock users against llamas. Planning tools

like Limits of Acceptable Change will be challenged when developing impact standards for production livestock use that must conform to the nonwilderness standards in the congressional grazing guidelines. This challenge will be especially great when many visitors demand less livestock or livestock removal based on impacts to wilderness traits because such criteria are not permitted by the congressional grazing guidelines. Research findings are not plentiful for recreation livestock (Cole 1989c), but production livestock standards can provide a starting point to form the acceptable impact standards.

Increasing the level of monitoring will be essential to keep abreast of the impacts associated with increasing use levels and to help develop defensible impact standards. Using monitoring to both assess impacts and revise management tools and standards is a form of adaptive management, in which monitoring informs managers of conditions and then stimulates continual revision of management. It is critical that managers recognize the utility of monitoring for these dual purposes. It is not always obvious that monitoring can provide valuable information for development of impact standards by documenting trends in impact severity. For example, long-term monitoring can describe how often any hypothetical impact standard has been exceeded, and if that occurrence has been increasing. Monitoring will help evaluate the effectiveness of different management tools by describing the difference in resource conditions (and user attitudes) before and after the new tools were applied.

Recreation livestock management will need to increase the application of management tools that control the intensity and timing of use, especially as use increases. Some may argue for greater uniformity among agencies in the use of guidelines versus regulations, because visitors are inconvenienced and managers are frustrated when wilderness travel crosses jurisdictions that use different tools (guidelines or regulations) and apply different standards (for example, 20% versus 35% utilization). However, there are some lessons to be learned from this interagency variation that merit perpetuation of these differences. The variation in approaches provide a means to evaluate the effectiveness of different approaches to ensure that erroneous management decisions are not made throughout the wilderness preservation system.

For production livestock, the application of management tools such as fences to control site-specific impacts will face increasing resistance from visitors, even though the congressional grazing guidelines allow for these tools if the main purpose is resource protection. Increased efforts to provide information materials to visitors about the location of fences, the need for fences, and the directives in the congressional guidelines may help increase acceptance of these tools.

More trained personnel will be needed to monitor and manage the impacts from the expected increases in recreation livestock use. One way to meet this challenge would be more cooperation and coordination between wilderness and range management personnel. Range management staff should be encouraged to provide more assistance in recreation livestock monitoring and management, while wilderness staff should be encouraged to provide assistance with production livestock management in wilderness. Any differences in impact standards should not hinder this coordination because monitoring techniques can be the

same and only the standards will differ. While there is certainly merit to Cole's (1989a) plea for more range scientists to address wilderness management situations, it seems equally obvious that wilderness management could benefit from better use of the relevant information in the range management discipline. Some means for facilitating this exchange if information include: handbooks, workshops, and wilderness range management courses offered at land grant universities.

Research and Development

Both basic research and research leading to the development of effective livestock management will help improve livestock management in wilderness. Excellent basic research on the resistance and resilience of vegetation in relation to horse and llama trampling help managers to be more diligent about limiting off-trail and travel in wet areas. Building on this basic research, development of techniques to form horse with llama teams could combine the demand for horses for riding with the opportunity to minimize impacts with llamas. Furthermore, the growing popularity of llamas is justification to expand recent trampling impact studies, and examine the diet and intake rates of these animals.

The increasing recreation livestock use and the perpetuation of production livestock use means that conflicts rooted in predisposition against livestock will not disappear. In fact, given the increase of wilderness users from urban areas, these conflicts will probably increase. Therefore, information and planning tools need to be developed to reduce these conflicts by spatial separation. This separation may be a voluntary behavior induced by information describing the location of livestock in wilderness, or it may come from prescribed behavior required by the designation of livestock-free areas.

Research describing the results of the various management tools being applied throughout the Wilderness Preservation System will help managers understand the variety of available tools and their effectiveness. This type of research is no substitute for the strong inference possible when controls and treatments are replicated in an experimental design. Nonetheless, greater communication of management failures and successes, when joined with the few experiments, can help managers see the possible and understand the impossible.

Finally, research describing the rate of recovery (resilience) when use is reduced will help complete the information managers now have about recovery after use is terminated. This information is critical because the termination of use is rarely an option compared to use reduction. Cole and Hall (1992) described the recovery of vegetation in recreation livestock camps when use was terminated as well as when use was reduced. They noted that while some impacts diminished when use was reduced, damage to trees from the tying of stock is cumulative and actually continued to increase even with reduced use. They also noted that when use was terminated, the rate of recovery was more rapid in mesic than arid areas. Expansion of this type of research will help managers predict the probability and rate of response to use reduction. Specifically, this research would address how much and how rapidly recovery will occur with each increment of

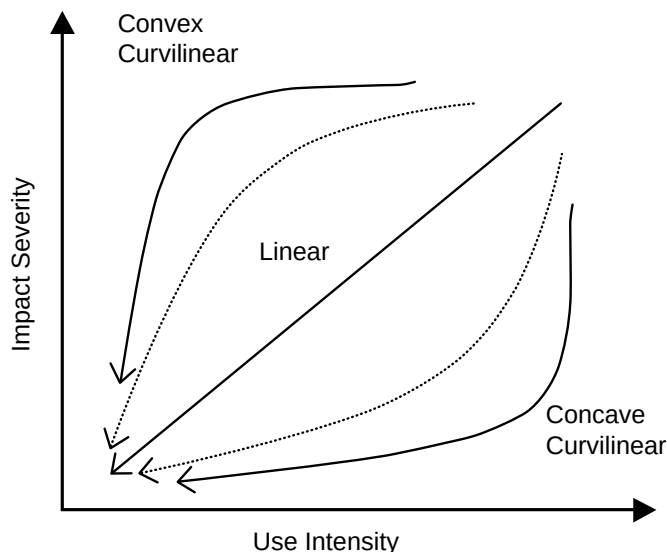


Figure 2—The rate of recovery following the reduction in the intensity of use can vary from linear, curvilinear to extreme curvilinear relationships. The linear relationship suggests that each unit of reduction of use will result in equal amounts of reductions in the severity of impacts. The modest convex curvilinear relationship (dashed curve) suggests that the rate of recovery (resilience) will be greatest when use is reduced towards a minimum intensity. The extreme convex curvilinear relationship suggests (solid curve) that recovery will be very minimal with reductions of intensity until use reaches very low intensities. Similarly, the modest concave curvilinear relationship (dashed curve) suggests that the rate of recovery will be greatest with the initial reductions of use, and the extreme concave curvilinear relationship (solid curve) suggests that nearly all recovery will occur with the initial reduction in use.

use reduction (fig. 2). The benefits of this information will be greatest for impacts most strongly controlled by use intensity, such as defoliation in grazed areas, and less useful for trails, where the initial impacts and timing are more important.

References

- Archer, S. R.; Smeins, F. E. 1991. Ecosystem-level processes. In: Heitschmidt, R.; Stuth, J. W., eds. *Grazing management: an ecological perspective*. Portland OR: Timber Press: 109-139.
- Balch, D. F.; Balch, M. H.; Malechek, J. C. 1989. Cues cattle use to avoid stepping on crested wheatgrass tussocks. *Journal of Range Management*. 47:64-69.
- Balch, D. F.; Malechek, J. C. 1985. Cattle trampling of crested wheatgrass under short-duration grazing. *Journal of Range Management*. 38:226-227.
- Blahna, D. J.; Smith, K. S.; Anderson, J. A. 1995. Backcountry llama packing: visitor perceptions of acceptability and conflict. *Journal of Leisure Sciences*. 17:185-204.
- Bonham, C.D. 1989. *Measurements for terrestrial vegetation*. New York, NY: John Wiley. 338 p.
- Briske, D. D. 1996. Strategies of grass survival in grazed systems: a functional interpretation. In: Hodgson, J.; Illius, A. W., eds. *The ecology and management of grazing systems*. New York, NY: CAB International: 37-67.
- Briske, D. D.; Richards, J. H. 1995. Plant responses to defoliation: a physiologic, morphologic, and demographic evaluation. In: Bedunah D. J.; Sosebee, R.E., eds. *Wildland plants: physiological ecology and developmental morphology*. Denver, CO: Society for Range Management: 635-710.
- Buckhouse, J. C.; Gifford, G. F. 1976. Water quality implications of cattle grazing on a semiarid watershed in southeastern Utah. *Journal of Range Management*. 29:109-113.
- Cash, S. D.; Zamora, D. L.; Lenssen, A.W. 1998. Viability of weed seeds in feed pellet processing. *Journal of Range Management*. 51:181-185.
- Chance, J. F.; Cruz, A. 1998. Range of the brown-headed cowbird in Colorado, past and present. *Great Basin Naturalist*. 58:245-249.
- Clary, W. P. 1995. Vegetation and soil response to grazing simulation on riparian meadows. *Journal of Range Management*. 48:18-25.
- Cole, D. N. 1983. Campsite conditions in the Bob Marshall Wilderness, Montana. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. Research Paper INT-312. 18 p.
- Cole, D. N. 1989a. Needed research on domestic and recreational livestock in wilderness. *Journal of Range Management*. 42:84-86.
- Cole, D. N. 1989b. Wilderness campsite monitoring methods: a sourcebook. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. General Technical Report INT-259. 57 p.
- Cole, D. N. 1989c. Low-impact recreational practices for wilderness and backcountry. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. General Technical Report INT-265. 131 p.
- Cole, D. N. 1994. The wilderness threats matrix: a framework for assessing impacts. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. Research Paper INT-475. 14 p.
- Cole, D. N. 1995a. Experimental trampling of vegetation. I. Relationship between trampling intensity and vegetation response. *Journal of Applied Ecology*. 32:203-214.
- Cole, D. N. 1995b. Experimental trampling of vegetation. II. Predictors of resistance and resilience. *Journal of Applied Ecology*. 32:215-224.
- Cole, D. N. 1996. Wilderness recreation in the United States-trends in use, users, and impacts. *International Journal of Wilderness*. 2:14-18.
- Cole, D. N.; Hall, T. E. 1992. Trends in campsite condition: Eagle Cap Wilderness, Bob Marshall Wilderness, and Grand Canyon National Park. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. Research Paper INT-453. 40 p.
- Cole, D. N.; Landres, P. B. 1996. Threats to wilderness ecosystems: impacts and research needs. *Ecological Applications*. 6:168-184.
- Cole, D. N.; Spidle, D. R. 1998. Hiker, horse and llama trampling on native vegetation in Montana, USA. *Journal of Environmental Management*. 53:61-71.
- D'Antonio, C. M.; Virousek, P. M. 1992. Biological invasion by exotic grasses, the grass/fire cycle and global change. *Annual Review of Ecology and Systematics*. 23:63-87.
- DeLuca, T. H.; Peterson IV, W.A.; Freimund, W. A.; Cole, D. N. 1998. Influence of llamas, horses, and hikers on soil erosion from established recreation trails in western Montana, USA. *Environmental Management*. 22:255-262.
- Dustin, D. L.; McAvoy, L. H. 1984. The limitations of the traffic light. *Journal of Park and Recreational Administration*. 2:28-32.
- Elton, C. S. 1958. *The ecology of invasions by animals and plants*. New York, NY: 1977 printing by Chapman and Hall. 181 p.
- Gary, H. L.; Johnson, S. R.; Ponce, S. L. 1983. Cattle grazing impact on surface water quality in a Colorado Front Range stream. *Journal of Soil and Water Conservation*. 38:124-128.
- General Accounting Office. 1989. *Wilderness preservation*. Washington, DC: Government Printing Office. GAO/RCED 89-202. 91 p.
- Heady, H. F.; Child, R. D. 1994. *Rangeland ecology and management*. 2nd edition. Boulder, CO: Westview Press: 519 p.
- Hendee, J. C.; Stankey, G. H.; Lucas, R. C. 1990. *Wilderness management*. 2nd ed. Golden, CO: Fulcrum Publishing: 546 p.
- Holechek, J.; Pieper, R. D.; Herbel, C. H. 1998. *Range management: principles and practices*. 3rd ed. Upper Saddle, NJ: Simon and Shuster. 542 p.
- Huston, J. E.; Pinchak, W. E. 1991. Range animal nutrition. In: Heitschmidt, R.; Stuth, J. W., eds. *Grazing management: an ecological perspective*. Portland, OR: Timber Press: 27-63.
- Ivy, M. I.; Stewart, W. P.; Lue, C. 1992. Exploring the role of tolerance in recreation conflict. *Journal of Social Science*. 24: 348-360.

- Jamarillo, V. J.; Detling, J.K. 1992. Small-scale heterogeneity in a semi-arid North American grassland. II. Cattle grazing of simulated urine patches. *Journal of Applied Ecology*. 37:430-438.
- Jardine, J. T.; Anderson, M. 1919. Range management on the National Forests. Washington, DC: U.S. Department of Agriculture. Bulletin 790. 98 p.
- Jessup, D. A.; Boyce, W. M. 1996. Diseases of wild ungulates and livestock. In: Krausman, P. R., ed. *Rangeland wildlife*. Denver, CO: Society for Range Management: 395-413.
- Johnson, L. C.; Wallace, G. N.; Mitchell, J. E. 1997. Visitor perceptions of livestock grazing in five wilderness areas: a preliminary assessment. *International Journal of Wilderness*. 3(2):14-20.
- Johnson, N.; Agee, J.; Beschta, R.; Dale, V.; Hardesty, L.; Long, J.; Nielsen, L.; Noon, B.; Sedjo, R.; Shannon, M.; Trosper, R.; Wilkinson, C.; Wondelleck, J. 1999. Sustaining the people's lands: recommendations for stewardship of the National Forests and Grasslands into the next century. *Journal of Forestry*. 97(5):6-12.
- Knopf, F. L. 1996. Perspectives on grazing nongame bird habitats. In: Krausman, P. R., ed. *Rangeland wildlife*. Denver, CO: Society for Range Management: 51-58.
- Krausman, P. R. ed. 1996. *Rangeland wildlife*. Denver, CO: Society for Range Management. 440 p.
- Kress, M.; Gifford, G. F. 1984. Fecal coliform release from cattle fecal deposits. *Water Resource Bulletin*. 20:61-66.
- Lacey, C. A. 1989. Knapweed management: a decade of change. In: Fay, P. K.; Lacey, J. R. eds. *Proceedings of the knapweed symposium*. Bozeman, MT: Montana State University Cooperative Extension Service Bulletin 45: 1-6.
- Larson, R.E.; Buckhouse, J. C.; Moore, J. A.; Miner, J. R. 1988. Rangeland cattle and manure placement: a link to water quality. *Proceedings of the Oregon Academy of Sciences*. 24:7-15.
- Leopold, A. 1921. The wilderness and its place in recreational policy. *Journal of Forestry*. 19:718-721.
- Lucas, R. C. 1982. Recreation regulations—when are they needed? *Journal of Forestry*. 80:148-151.
- Lucas, R. C. 1983. The role of regulations in wilderness management. *Western Wildlands*. 9(2):6-10.
- Lucas, R. C. 1985. Visitor characteristics, attitudes, and use patterns in the Bob Marshall Wilderness Complex, 1970-1982. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. Research Paper INT-345. 32 p.
- McClaran, M. P. 1989. Recreational packstock management in Sequoia and Kings Canyon National Parks. *Rangelands*. 11:3-8.
- McClaran, M. P. 1990. Livestock in wilderness: a review and forecast. *Environmental Law*. 20:857-889.
- McClaran, M. P. 1991. Forest Service and livestock permittee behavior in relation to wilderness designation. *Journal of Range Management*. 44:483-486.
- McClaran, M. P.; Anable, M.E. 1992. Spread of introduced Lehmann lovegrass along a grazing intensity gradient. *Journal of Applied Ecology*. 29:92-98.
- McClaran, M. P.; Cole, D.N. 1993. Packstock in wilderness: use, impacts, monitoring, and management. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. General Technical Report INT-301. 33 p.
- McInnis, M. L.; Vavra, M. 1987. Dietary relationship among feral horse, cattle, and pronghorn in southeastern Oregon. *Journal of Range Management*. 40:60-66.
- Milchunas, D. G.; Sala, O.E.; Lauenroth, W.K. 1988. A generalized model of the effects of grazing by large herbivores on grassland community structure. *American Naturalist*. 132:375-384.
- Milchunas, D. G.; Lauenroth, W.K. 1993. Quantitative effects of grazing on vegetation and soils over a global range of environments. *Ecological Monographs*. 63:327-366.
- Miller, R. 1983. Habitat use of feral horses and cattle in Wyoming's Red Desert. *Journal of Range Management*. 36:195-199.
- Mitchell, J. E.; Wallace, G.N. 1998. Managing grazing and recreation across boundaries in the Big Cimmarron watershed. In: Knight, R. L.; Landres, P. B. eds. *Stewardship across boundaries*. Washington, DC: Island Press: 217-236.
- Mitchell, J. E.; Wallace, G. N.; Wells, M. D. 1996. Visitor perceptions about cattle grazing on national forest land. *Journal of Range Management*. 49:81-86.
- Moore, S. D.; McClaran, M.P. 1991. Symbolic dimensions of the packstock debate. *Journal of Leisure Sciences*. 13:221-237.
- Muir, S.; McClaran, M.P. 1997. Rangeland inventory, monitoring, and evaluation. <http://ag.arizona.edu/OALS/agnic/knowledge/chapters5>.
- Olson-Rutz, K. M.; Marlow, C. B.; Hansen, K.; Gagnon, L. C.; Rossi, R. J. 1996a. Recovery of a high elevation plant community after packhorse grazing. *Journal of Range Management*. 49:541-545.
- Olson-Rutz, K. M.; Marlow, C. B.; Hansen, K.; Gagnon, L. C.; Rossi, R. J. 1996b. Packhorse grazing behavior and immediate impact on a timberline meadow. *Journal of Range Management*. 49: 546-550.
- Peek, J. M.; Krausman, P. R. 1996. Grazing and mule deer. In: Krausman, P. R., ed. *Rangeland wildlife*. Denver, CO: Society for Range Management: 183-192.
- Platts, W. S.; Nelson, R. L. 1985. Streamside and upland vegetation use by cattle. *Rangelands*. 7:5-7.
- Reed, P.; Haas, G.; Beum, F.; Sherrick, L. 1989. Non-recreational uses of the National Wilderness Preservation System: a 1988 telephone survey. In: Freilich, H. R. compiler. *Wilderness Benchmark 1988: Proceedings of the National Wilderness Colloquium*. Asheville, NC: U.S. Department of Agriculture, Forest Service, Forest Experiment Station. General Technical Report SE-51: 220-228.
- Roth, D. 1984. The National Forests and the campaign for wilderness legislation. *Journal of Forest History*. 28:112-125.
- Rothstein, S. I. 1994. The cowbird's invasion of the far west: history, cause, and consequences experienced by host species. *Studies in Avian Biology*. 15:301-315.
- Sampson, A. W. 1952. *Range management: principles and practices*. New York, NY: John Wiley. 570 p.
- Stankey, G. H. 1979. A comparison of carrying capacity perceptions among visitors in two wilderness areas. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. Research Paper INT-242. 34 p.
- Stankey, G. H.; Cole, D. N.; Lucas, R. C.; Petersen, M. E.; Frissell, S. S. 1985. The limits of acceptable changes (LAC) system for wilderness planning. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. General Technical Report INT-176. 37 p.
- Stegner, W. 1961. The wilderness idea. In: Brower, D. ed. *Wilderness: America's living heritage*. San Francisco, CA: Sierra Club: 97-102.
- Stoddart, L. A.; Smith, A. D. 1955. *Range management*. 2nd ed. New York, NY: McGraw-Hill. 433 p.
- Stohlgren, T. J.; Schell, L. D.; Heuvel, B. V. 1999. How grazing and soil quality affect native and exotic plant diversity in Rocky Mountain grasslands. *Ecological Applications*. 9:45-64.
- Sumner, E. L. 1942. The biology of wilderness protection. *Sierra Club Bulletin*. 27(4):14-22.
- Tannen, M. 1999. High on the trail in Wyoming. *The New York Times*. July 11, 1999, TR 17, 26.
- Teer, J. G. 1996. The white-tailed deer: natural history and management. In: Krausman, P. R. ed. *Rangeland wildlife*. Denver, CO: Society for Range Management, Denver: 193-210.
- Thelin, R.; Gifford, G. F. 1983. Fecal coliform release from fecal material of cattle. *Journal of Environmental Quality*. 12:57-63.
- Tiedemann, A. R.; Higgins, D. A.; Quigley, T. M.; Sanderson, H. R.; Marx, D. B. 1987. Responses of fecal coliform in streamwater to four grazing strategies. *Journal of Range Management*. 40: 322-329.
- Vallentine, J. F. 1990. *Grazing management*. San Diego, CA: Academic Press, 533 p.
- Van Poolen, H. W.; Lacey, J. R. 1979. Herbage response to grazing systems and stocking intensities. *Journal of Range Management*. 32:250-253.
- Washburne, R. F.; Cole, D. N. 1983. Problems and practices in wilderness management: a survey of managers. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. Research Paper INT-304. 56 p.
- Watson, A. E.; Christensen, N. A.; Blahna, D. J.; Archibald, K. S. 1998. Comparing manager and visitor perceptions of llama use in wilderness. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station Research Paper. RMRS-RP-10. 7 p.

- Watson, A. E., Niccolucci, M. J.; Williams, D. R. 1993. Hikers and recreational stock users: predicting and managing recreation conflicts in three wildernesses. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. Research Paper INT-468. 35 p.
- Weaver, T.; Dale, D. 1978. Trampling effects of hikers, motorcycles and horses in meadows and forests. *Journal of Applied Ecology*. 15:451-457.
- West, N. E.; Smith, E. L. 1997. Improving the monitoring of rangelands. *Rangelands*. 19(6):9-14.
- Wilson, J. P.; Seney, J. P. 1994. Erosional impact of hikers, horses, motorcycles, and off-road bicycles on mountain trails in Montana. *Mountain Research and Development*. 14:77-88.
- Wisdom, M. J.; Thomas, J. W. 1996. Elk. In: Krausman, P. R., ed. *Rangeland wildlife*. Denver, CO: Society for Range Management: 157-182.
- Woodmansee, R. G. 1978. Additions and losses of nitrogen in grassland ecosystems. *Bioscience*. 28:448-453.

Improving Management of Nonnative Invasive Plants in Wilderness and Other Natural Areas

John M. Randall

Abstract—Nonnative invasive plants invade wilderness and other natural areas throughout North America and invasive organisms as a group are now considered the second worst threat to biodiversity, behind only habitat loss and fragmentation. In the past 10-20 years there have been upsurges in interest in the ecology of plant invasions among researchers and in concern about how to prevent and control them among land managers. Much research has focused on how to identify and predict which species are most likely to be invasive and which habitats or areas are most likely to be invaded and some progress has been made. A number of studies clearly demonstrate that plant invasions can alter ecosystem processes, displace native species, promote nonnative animals, fungi or microbes and alter the genetic make up of native species populations through hybridization. Some invasions can be prevented or controlled and efforts continue to refine and improve current techniques. Improved prevention and management of invasive plants will require development and use of adaptive management strategies, tools to help managers set weed control priorities, techniques for using remote sensing technologies to map weed infestations, improved control methods and increased attention to preventing new invasions and quickly detecting and eradicating those that do occur.

Nonnative invasive plants have dramatically changed North America's ecological landscape. They are most notorious for invading island ecosystems and sites subjected to human or natural disturbances, but they also invade large mainland wildernesses and natural areas that appear to have suffered no other disturbance in recent decades. Nonnative plants were recognized as a problem and an interesting topic of study by the mid-1800s, but interest among ecologists picked up markedly following publication of Elton's (1958) book "The Ecology of Invasions by Animals and Plants." A great deal of interest and work has been directed at discovering what, if anything, makes some species more invasive than others and what, if anything, makes some habitats and systems more susceptible to invasion than others. Answers to these questions remain elusive, but there have been significant new findings in the past few years. This has rekindled hopes that we may yet gain enough understanding of these phenomena to make more reliable predictions, which could be used to help prevent new invasions.

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There has also been interest and concern about invasive weeds among managers of wilderness and other natural areas since at least the mid-1800s and both have risen sharply in the past 10-15 years. This concern has grown in part because we have learned more about the impacts invasive weeds can have. Some alter the ecosystems and communities they infest, using resources that would have been consumed by native species and altering wildlife habitats in ways that make these places unfit for native animals. Some invasive species like the tamarisks (*Tamarix* spp.), cheat grass (*Bromus tectorum*), Scotch broom (*Cytisus scoparius*) and European beachgrass (*Ammophila arenaria*) completely alter natural ecosystem functions and processes, such as fire patterns, nutrient cycling, soil stability and hydrological regimes. In so doing, they 'change the rules of the game' of survival and growth, placing many native species at a gross disadvantage. Even when they don't noticeably affect ecosystem processes, invasive plants outcompete and displace native plants, which in turn displaces native animals. Some invasive plants also hybridize with native species and with time could eliminate purely native strains. For example, in some tidal creeks around the San Francisco Bay, it is now impossible to find 'pure' native strains of California Cord grass (*Spartina foliosa*) – every plant has at least some genes from the invasive Atlantic cord grass (*S. alterniflora*) (Ayres and others in press).

Invasive species are now widely recognized as threats to native biological diversity second only to direct habitat loss and fragmentation (Pimm and Gilpin 1989; Scott and Wilcove 1998). In fact, when biological invasion is considered as a single phenomenon, it is clear that, to date, it has had greater impacts on the biota worldwide than more notorious aspects of global environmental change such as rising CO₂ concentrations, climate change and decreasing stratospheric ozone levels (Vitousek and others 1996). What's worse is that invasive organisms continue to spread on their own and do not degrade with time, unlike pollutants; once introduced, they can spread from site to site, region to region, without further human assistance.

Fortunately, many plant invasions into wildlands can be halted or slowed, and, in certain situations, even badly infested areas can be restored to relatively healthy communities dominated by native species (for example, see Barrows 1993; Pickart and Sawyer 1998; Randall and others 1997). Because control and restoration efforts can limit or reverse the severe damage caused by invasive plants, these activities are now widely regarded as necessary in many natural areas. This need has driven a great deal of research and demonstration work aimed at developing better techniques to kill or suppress unwanted weeds without harming desirable native plants and animals. One technique that has received a great deal of attention is classical biological

control—the release of host-specific natural enemies (pathogens, parasites and herbivores) from the native range of the weed into the invaded environment. Although sometimes the only practical method available for controlling invasive weeds across vast areas, this technique can backfire if the biocontrol agent is less host-specific than expected and begins feeding on and reducing populations of desirable native species. Fortunately, some recent work urges greater caution in the selection of biocontrol agents and suggests concrete ways to accomplish this (Louda and others 1997; McEvoy and Coombs 1999). Unfortunately, we have far too little quantitative information about the impacts of biocontrol agents or of other weed control techniques on the native species, communities and systems we are trying to protect. This information is of utmost concern since controlling the weed(s) is only a means to our ultimate goal of protecting or restoring the natives.

The need to use limited resources efficiently to prevent and control invasive weed problems has driven land managers to set priorities and adopt adaptive management approaches for weed management.

Definitions of Terms

Nonnative plants are those species beyond their natural range or natural zone of potential dispersal, including all domesticated and feral species and all hybrids involving at least one nonnative parent species. Other terms that are often used as synonyms for nonnative include alien, exotic, introduced, adventive, nonindigenous, nonaboriginal and naturalized. With rare exceptions, conservation programs are dedicated to the preservation of native species and communities. The addition of nonnative species rarely contributes positively to this, unless they alter the environment in ways that favor native species, as some grazers and biological control agents do.

Natural ranges should not be confused with political or administrative boundaries. Bush lupine (*Lupinus arboreus*), for example, may be thought of as a California native, but its original, native range is only the central and southern coasts of the state. It is a nonnative along the north coast, where it was intentionally planted outside its natural range (Miller 1988). All hybrids between introduced or domesticated species and native species are also nonnative.

Invasive species are those that spread into areas where they are not native (Rejmánek 1995). Not all nonnative plants are invasive; in fact, only a minority of introduced species have escaped cultivation, and only a minority of those that have escaped are invasive in wildlands.

The terms **pest plant** and **weed** may be used interchangeably to refer to species, populations and individual plants that are unwanted because they interfere with management goals and objectives. Plants regarded as pests in some wildlands may not be troublesome elsewhere. For example, the Empress tree (*Paulownia tomentosa*) is a pest in deciduous forests of the eastern U.S., particularly in the southern Appalachians, but it is not known to escape from cultivation in California, although it is often used as an ornamental landscape tree there. Some species that are troublesome in agricultural or urban areas rarely, if ever, become weeds of wildlands. The term environmental weeds is used by many Australians (Groves 1991; Humphries and

others 1991) to refer to wildland weeds, but few North America land managers or researchers use this term.

Research on Invasive Weed Ecology and Control: What Have We Learned and How Has It Helped Us Manage Wilderness and Other Natural Areas?

Early Recognition of the Issue in Natural Areas and Increasing Recognition of Its Importance

Invasions by nonnative species have been recognized as an important topic of study for natural history and ecology for nearly 150 years. Charles Darwin (1859) commented on the phenomenon of nonnative plants invading new areas and put forth hypotheses about what might predispose certain areas to be prone to invasion and what might predispose certain species to be invasive. Here in North America, the impacts of invasive nonnative weeds on the native biota of designated natural areas were recognized at least as early as 1865 by Frederick Law Olmsted. He filed a report on the newly set-aside Yosemite Valley, noting that unless actions were taken, its vegetation would likely be pushed out by common weeds from Europe. The report pointed out that this had already happened “in large districts of the Atlantic States.” Botanists and other students of natural history noted the establishment of nonnative species across the continent in published papers. By the 1930s, natural area managers in Yosemite and scattered parks and preserves around the nation began controlling invading nonnative species that were recognized as agricultural pests (Randall 1991). Invasive species impacts were brought into the mainstream of ecology in the late 1950s with the publication of Charles Elton’s book, *The Ecology of Invasions by Animals and Plants* (1958). Concern and interest among both land managers and researchers has grown since then, particularly since the mid-1980s.

Research on ‘Invasiveness’—What Characteristics Enable Certain Species to Invade New Areas?

Many people have wondered if certain traits distinguish species that become invasive from those that don’t. Despite a great deal of study, no single answer presents itself, and researchers have been surprised by the success of some species and the failure of others. It has proven even more difficult to find traits that distinguish between the subset of successful invading species that become pests from those that appear to have little impact. Work on this topic continues, in part because of the hope that answers may enable us to predict which of the many nonnative species not yet established are most likely to invade and become pests if given the chance.

Recent work points to several factors that may help predict which species are likely to be invasive. In two studies, the best predictor was whether a species was invasive somewhere

else (Panetta 1993; Reichard and Hamilton 1997). For example, if a species native to Spain is invasive in Australia, it is likely to be invasive in California and South Africa as well. Rejmánek and Richardson (1996) analyzed characteristics of 20 species of pines and found that species which produce many small seeds and which begin reproducing within their first few years are most likely to be invasive. When they extended the analysis to a group of flowering trees, these same characteristics usually discriminated between invasive and noninvasive species. This study and several others also found that plants with animal-dispersed seeds, like bush honeysuckles or privets, are much more likely to be invasive in forested communities (Reichard 1997; Reichard and Hamilton 1997). It has also been suggested that species capable of reproducing both by seed and vegetative growth have a better chance to spread in a new land (Reichard 1997).

Self-compatible species, with individuals that can fertilize themselves, have been thought more likely to invade since just one plant of this type could start an invasion (Baker 1965). Many self-incompatible species are successful invaders, however, including some that are dioecious (male and female flowers on separate plants). It is also thought that plants dependent on one or a few other species for pollination, fruit dispersal or the uptake of nutrients from the soil are less likely to invade new areas unless these organisms are introduced at the same time. As a group, figs may be relatively poor invaders because, with few exceptions, each species is pollinated by a distinctive species of wasp, which is in turn dependent on that species of fig. On the other hand, the edible fig's pollinator was introduced intentionally to promote fruit production, and now the species is invasive in parts of California (Randall in press). Other plant invasions may also be promoted by introduced animals. For example, honeybees boost seed production of invaders whose flowers they favor (Barthell and others in prep). In Hawaii, feral pigs promote the spread of banana poka (*Passiflora mollissima*) and other species by feeding voraciously on their fruits and distributing them in their scat, often in areas they have disturbed by rooting in the soil for more food.

It has also been suggested that species with relatively small DNA contents in their cell nuclei are more likely to be invasive in disturbed habitats (Rejmánek 1996). Plants that germinate and grow rapidly can quickly occupy such areas and exclude other plants following a disturbance. It turns out that under given conditions, cells with low DNA contents can usually divide and multiply more quickly, and consequently these plants grow more rapidly than species with higher cellular DNA content.

A species is most likely to invade an area with a climate similar to that in its native range, but some nonnative species now thrive in novel conditions. An analysis of the distribution of nonnative herbs of the sunflower and grass families in North America indicated that species with a larger native range in Europe and Asia are more likely to become established and to have a larger range here than species with small native ranges (Rejmánek 1995). Species with large native ranges may be well adapted to a variety of climate and soil conditions and so more likely to find suitable habitat in a new area. Some of this ability to cope with different conditions can be due to genetic differences among individuals of a species or 'genetic plasticity.' Some of it may

also be due to phenotypic plasticity, the ability of any given individual of some species to cope with a variety of conditions. Another factor that may contribute to whether a plant will be likely to invade a site is whether it is closely related (e.g. in the same genus) to any native species. Plants without close relatives appear more likely to become established (Rejmánek 1996).

A species may be more likely to establish if many individuals are introduced at once or if they are introduced repeatedly. It is presumed that introductions of more individuals ensure that they will be able to find one another to mate and produce offspring and that there will be more genetic variability in the population, enabling it to cope with a wider variety of conditions. If sites where the species can successfully germinate and grow are limited in number, the chance that at least one seed scattered at random will land on an appropriate site increases as the number of seeds scattered increases. Chance may be important in other ways. For example, species that happen to be introduced at the beginning of a drought may be doomed to fail, although they might easily establish following a return to normal rainfall.

There is often a time lag of many decades or more between the first introduction of a plant and its rapid spread. As far as we know, Atlantic cord grass was present in small patches in a few spots on the Pacific coast for 50 years or more before it appeared to spread. In fact, some species that rarely spread today may turn out to be troublesome 40, 50 or more years from now. This makes it all the more pressing that we find some way of determining which species are most likely to become invasive so that we can control them now, while their populations are still small and manageable.

What Makes Certain Sites More or Less Prone to Invasion?

Another question, which has long intrigued ecologists, is why some areas appear more prone to invasion than others. Again, many hypotheses have been advanced, but we have few solid answers. It is not even clear which areas have suffered the most invasions since this may differ depending on the types of organism considered and which species are regarded as firmly established as opposed to rarely escaping from gardens or persisting around old homesites. In fact, a given area may be highly susceptible to invasion by one type of organism and highly resistant to another, while the situation might be reversed in other areas.

It is recognized that areas where vegetation and soil have been disturbed by humans or their domestic animals are more susceptible to invasion. In North America, disturbed sites are often invaded by plants native to the Mediterranean region and the fertile crescent of the Old World, where they had millennia to adapt to agricultural disturbances. Changes in streamflows, the frequency of wildfires or other environmental factors caused by dam building, firefighting and other human activities may also hinder survival of native plants and promote invasion by nonnatives. Nonetheless, reserves and protected areas are not safe from nonnative species invasions, at least in part because natural disturbances ranging from gopher mounds to hurricane damage can and do strike even the most pristine sites.

It is also safe to say that remote islands in temperate and tropical areas appear to be highly susceptible to invasions by nonnative plants and animals. For, example, nearly half (49%) of the flowering plant species found in the wild in Hawaii are nonnative (Wagner and others 1990). Most remote islands had no large native herbivores, so pigs, cattle, sheep and other grazers introduced by humans found the native plants completely unprotected by spines or chemical deterrents. Introduced grazers often denuded large areas of native vegetation, leaving them open for colonization by introduced species adapted to grazing. Islands, peninsulas such as southern Florida and other areas with low numbers of native species or without any representatives of distinctive groups appear to be more prone to invasions. For example, there are no rapidly growing woody vines native to the Hawaiian Islands, where several introduced species have become pests. Some researchers theorize that where such gaps exist, certain resources are used inefficiently if at all, resulting in 'open niches.' Nonnative species that are preadapted to exploit these resources are thus highly likely to invade such areas. Other researchers reject the concept of 'empty niches,' saying they are impossible to identify in advance and that when new species move in, they do not slip into unoccupied slots but instead use resources that would have been used by the organisms present initially, and rearrange the community.

It has also been hypothesized that areas with low numbers of native species—whether on islands or continents—are more susceptible to nonnative species invasions than species-rich areas (Elton 1958; MacArthur and Wilson 1967; McNaughton 1983). Recent experimental work in a tallgrass prairie site by Tilman (1997) supported this hypothesis, showing that small plots (1m²) with relatively few native species were more prone to invasion than plots with greater native species richness. Observations by Stohlgren and others (1998; 1999) in mixed-grass prairie and in Rocky Mountain meadow and parkland sites indicated that relationships between native species abundance and invasibility are scale dependant. Most alarmingly, they found that at landscape and biome scales, areas with higher native species richness and cover support higher numbers of exotic species too. They also found evidence that relatively resource-rich areas, and in particular riparian areas, support greater numbers of invading species and hence appear to be more prone to invasion.

History too, likely plays a large role in determining the susceptibility of a site to invasion too. Sites like busy seaports, railroad terminals or military supply depots are exposed to more introductions. People from some cultures are more likely to intentionally introduce plants from their homelands when they migrate to new regions. In fact, colonization of much of the Americas, Australia and other areas of the world by western European peoples and the plants and animals from their homelands may go hand in hand, the successes of one species further promoting the successes of the others. European colonists were followed, sometimes even preceded, by animals and plants they were familiar with and knew how to exploit, and the plants and animals benefited in turn when the people cleared native vegetation and plowed the soil.

Impacts—A Few Excellent Studies on Ecosystem and Community Impacts

Nonnative plant invasions can have a variety of effects on wildlands, including alteration of ecosystem processes, displacement of native species, support of nonnative animals, fungi or microbes, and hybridization with native species and subsequent alteration of gene pools. Some invaders move into wilderness and other areas of national parks, preserves and natural areas, where they reduce or eliminate the species and communities these sites were set aside to protect. Rare species appear to be particularly vulnerable to the changes wrought by nonnative invaders. For example, the California Natural Heritage Database (1996) indicates 181 of the state's rare plant species are experiencing threats from invasive weeds. Habitats for rare animals such as the San Clemente Sage Sparrow and the Palos Verde Blue butterfly are also being invaded and displaced by weedy species. Hobbs and Mooney (1998) point out that invasive species have already brought about local extinctions and drastic population declines for many once-common species that are likely to lead to the final endpoint of species extinction.

Although we have great volumes of anecdotal information about impacts of invasive weeds, we have too little quantitative information about these impacts and even less that has been experimentally demonstrated. Symptomatic of this were arguments by Anderson (1995) and Hager and McCoy (1998) that the negative impacts of purple loosestrife (*Lythrum salicaria*) have not been conclusively demonstrated, and thus the efforts and resources devoted to control this species may have been misplaced.

We do, however, know a great deal about the impacts of certain invasive weeds and about the variety of impacts invasive weeds can have.

Ecosystem Effects

The invasive species that cause the greatest damage are those that alter ecosystem processes such as nutrient cycling, the intensity and frequency of fire, hydrological cycles, sediment deposition and erosion (D'Antonio and Vitousek 1992; Vitousek 1986; Vitousek and Walker 1989; Vitousek and others 1987; Whisenant 1990). Cheat grass (*Bromus tectorum* L.) is a well-studied example of an invader that has altered ecosystem processes. This annual grass has invaded millions of acres of rangeland in the Great Basin, leading to widespread increases in frequency of fires from once every 60 to 110 years to once every 3 to 5 years (Billings 1990; Whisenant 1990). Native shrubs do not recover well from the more frequent fires and have been eliminated or reduced to minor components in many of these areas (Mack 1981).

Some invaders alter soil chemistry, making it difficult for native species to survive and reproduce. For example, iceplant (*Mesembryanthemum crystallinum*) accumulates large quantities of salt, which it releases after it dies. The increased salinity prevents native vegetation from reestablishing (Kloot 1983; Vivrette and Muller 1977). Scotch broom (*Cytisus scoparius*) and gorse (*Ulex europaea*) can increase the availability of nitrogen in soil. Although this

increases soil fertility and overall plant growth, it probably gives a competitive advantage to nonnative species that thrive in nitrogen rich soil. Researchers have found that the nitrogen-fixing firetree (*Myrica faya*) increases soil fertility and consequently alters succession in Hawaii, (Vitousek and Walker 1989).

Wetland and riparian area invaders alter hydrology and sedimentation rates. Tamarisks (*Tamarix chinensis*; *T. ramosissima*; *T. pentandra*, *T. parviflora*) invade wetland and riparian areas in the southern and central California and throughout the American Southwest and are believed responsible for lowering water tables at some sites (Horton 1977). This may reduce or eliminate surface water habitats that native plants and animals need to survive (Brotherson and Field 1987; Neill 1983). For example, tamarisk invaded Eagle Borax Spring in Death Valley in the 1930s or 1940s. By the late 1960s, this large marsh had dried up, and had no visible surface water. When managers removed tamarisk from the site, surface water reappeared, and the spring and its associated plants and animals recovered (Neill 1983). Tamarisk infestations also can trap more sediments than stands of native vegetation and thus alter the shape, carrying capacity and flooding cycle of rivers, streams and washes (Blackburn and others 1982). Interestingly, the only species of *Tamarix* that is established in the southwestern U.S., but not generally regarded as invasive (athel; *T. aphylla*), is regarded as a major riparian area invader in arid central Australia (Griffin and others 1989).

Other wetland and riparian invaders and a variety of beach and dune invaders dramatically alter rates of sedimentation and erosion. One example is saltmarsh cordgrass (*Spartina alterniflora*), which is native to the U.S. Atlantic and Gulf coasts but was introduced to the Pacific coast where it invades intertidal habitats. Sedimentation rates may increase dramatically in infested areas, while nearby mudflats deprived of sediment erode and become open water areas (Sayce 1990). The net result is a sharp reduction in the area of the open intertidal areas where many migrant and resident waterfowl feed.

Coastal dunes along the Pacific coast from central California to British Columbia have been invaded and altered by European beachgrass (*Ammophila arenaria*). Dunes in infested areas are generally steeper and oriented roughly parallel to the coast rather than nearly perpendicular to it, as they are in areas dominated by *Leymus mollis*, *L. pacificus*, and other natives (Barbour and Johnson 1988). These weeds eliminate habitats for rare native species, such as Antioch Dunes evening-primrose (*Oenothera deltoides* ssp. *howellii*) and Menzies' wallflower (*Erysimum menziesii* ssp. *menziesii*). Species richness on foredunes dominated by European beachgrass may be just half of that on adjacent dunes dominated by *Leymus* species (Barbour and others 1976). These changes in the shape and orientation of the dunes also alter the hydrology and microclimate of the swales and other adjacent habitats, affecting species in these areas.

Some upland habitat invaders also alter erosion rates. For example, runoff and sediment yield under simulated rainfall were 56% and 192% higher on plots in western Montana dominated by spotted knapweed (*Centaurea maculosa*) than on plots dominated by native bunchgrasses (Lacey and others 1989). This species is now established in northern California and the southern Peninsular range and was

recently found on an inholding within Yosemite National Park (Hrusa 1998, personal communication).

Habitat Dominance and Displacement of Native Species

Invaders that move into and dominate habitats without obviously altering ecosystem properties can nevertheless cause grave damage. They may outcompete native species, suppress native species recruitment and thus alter community structure, degrade or eliminate habitat for native animals or provide food and cover for undesirable nonnative animals. Edible fig invades riparian forests in California's Central Valley and surrounding foothills and can become a canopy dominant. Invasive vines are troublesome in forested areas across the continent. In California, for example, Cape ivy (*Delairea odorata*) infests riparian forests along the coast from San Diego north to the Oregon border (Elliott 1994).

Nonnative subcanopy trees and shrubs invade forest understories, particularly in the Sierra Nevada and California's coast ranges. Scotch broom (*Cytisus scoparius*), French broom (*Genista monspessulana*) and Gorse (*Ulex europaea*) are especially troublesome invaders of forests and adjacent openings and coastal grasslands (Bossard 1991; Mountjoy 1979). Herbaceous species can colonize and dominate grasslands or the ground layer in forests. Eupatory (*Ageratina adenophora*) invades and dominates riparian forests along California's southern and central coast. Impacts of these ground layer invaders have not been well studied, but it is suspected that they displace native herbs and perhaps prevent recruitment of trees.

Annual grasses and forbs native to the Mediterranean region have replaced most of California's native grasslands. Invasion by these species was so rapid and complete that we do not know what the dominant native species were on the vast areas of bunchgrass lands in the Central Valley and other valleys and foothills around the state. The invasion process continues today, as medusa head (*Taeniatherum caput-medusae*) and yellow star thistle (*Centaurea solstitialis*) spread to sites already dominated by other nonnatives. Yellow starthistle is an annual that produces large numbers of seeds and grows rapidly as a seedling (Prather and Callihan 1991). It is favored by soil disturbance but invades areas that show no sign of being disturbed by humans or livestock for years and has colonized several relatively pristine preserves in California, Oregon and Idaho (Randall 1996).

Invasive, nonnative weeds can also prevent reestablishment of native species following natural or human-caused disturbance, altering natural succession. Ryegrass (*Lolium multiflorum*), used to seed burned areas in southern California, interferes with herb establishment (Keeley and others 1981) and, at least in the short term, with chaparral recovery (Gautier 1982; Schultz and others 1955; Zedler and others 1983).

Hybridization With Native Species

Some nonnatives plants hybridize with natives and could, in time, effectively eliminate native genotypes. The nonnative *Spartina alterniflora* hybridizes with the native

S. foliosa where they occur together. Recent studies found few or no individual plants without nonnative genes in some *Spartina* populations in some salt marshes around the San Francisco Bay (Ayres and others in press).

Promotion of Nonnative Animals

Many nonnative plants facilitate invasions by nonnative animals and vice versa. *Myrica faya* invasions of volcanic soils in Hawaii promote populations of nonnative earthworms, which increase rates of nitrogen burial and accentuate the impacts these nitrogen-fixing trees have on soil nutrient cycles (Aplet 1990). *Myrica faya* is in turn aided by the nonnative bird Japanese white-eye (*Zosterops japonica* Temminck), perhaps the most active of the many native and nonnative species that consume its fruits and disperse its seeds to intact forest (Vitousek and Walker 1989).

Control and Restoration Methods Continue to Develop

The past 10-20 years have seen a surge in efforts to develop better methods to control invasive weeds and restore native vegetation in natural areas. A great deal of work of this sort has been reported in journals like the *Natural Areas Journal*, *Restoration & Management Notes*, and *Restoration Ecology*. Some has even been published in journals traditionally focused more on agricultural lands and rangelands such as *Weed Science*, *Weed Technology*, and *Rangelands*. Unfortunately, even more probably remains in unpublished reports, which are unlikely to be read by those who could profit most from them, or worse, was never written up in any fashion.

A variety of weed control methods is available: manual, mechanical, encouraging competition from native plants, grazing, biocontrol, herbicides, prescribed fire, flooding and other, more novel, techniques. Each method has pluses and minuses, and research and field experience have both shown it is often best to use a combination of methods. Much study has been devoted to the use of non-chemical methods of weed control due to fears that herbicides will kill desirable species or otherwise pollute and damage the environment. Unfortunately, most manual and mechanical methods, such as hand pulling, the use of mulches and plastic sheeting are often too costly, in terms of both labor and money, to be used against large infestations. However, Pickart and Sawyer (1998) reported that a 4 ha infestation of European beachgrass (*Ammophila arenaria*) on the Lanphere Dunes area of Humboldt Bay National Wildlife Refuge was cleared using hand-labor to repeatedly pull up this deep-rooted grass. This successful effort cost \$86,700/ha in 1997 dollars, and the authors indicate that studies to develop techniques that will reduce these costs continue.

Biological control can be an extremely selective control tool, and more and more biocontrol projects targeting invasive weeds of natural areas have begun in recent years. Within the past 10 years, new biological control agents have been released against several natural area weeds, including purple loosestrife (*Lythrum salicaria*), melaleuca (*Melaleuca quinquenervia*), yellow starthistle (*Centaurea solstitialis*) and leafy spurge (*Euphorbia esula*), and one insect was

released against weedy tamarisks (*Tamarix* spp.) in 1999. Research and exploration for biocontrol agents has begun for several other natural area weeds, including garlic mustard (*Alliaria petiolata*), Cape ivy (*Delaria odorata*) and the native species Phragmites (*Phragmites australis*).

Unfortunately, there is some risk that the agents might attack desirable species. Concern about the specificity of control, or lack thereof, of biocontrol agents has prevented natural area managers from embracing their use more wholeheartedly. Howarth (1991) notes that no plant species are believed to have been driven to extinction by biological control agents and suggests this may be due to the greater care and stricter guidelines for introductions of herbivores than for insect predators and pathogens. Indeed, until two years ago it was frequently stated that "classical" biological control of weeds had a proven safety record and that none of the approximately 300 insects introduced to control weeds had ever become a pest itself (DeLoach 1991; Groves 1989; LaRoche 1994). Then, Louda and others (1997) reported that the biocontrol agent *Rhinocyllus conicus* had been found attacking several native thistles, including the Platte thistle (*Cirsium canescens*) in such numbers that it was clearly capable of reducing populations of these desirable, nontarget natives.

Herbicides can be effective against many of the weeds that invade wilderness and other natural areas, but they can also kill or damage desirable native species. A great deal of effort has gone into developing application techniques or timing herbicide applications so that only targeted weeds will be killed. Examples include using cut-stump and basal bark methods of herbicide application on tree and shrub weeds like *Rhamnus catharticus* and *Ailanthus altissima*, and applying herbicides at a time of year when weeds like Japanese honeysuckle are green and photosynthesizing, but most native plants in the area are not.

Few Studies Quantify Impacts of Control Efforts on the Native Species and Ecosystem Process We Are Managing for

Unfortunately, relatively few studies have followed the impacts of wildland weed control on the recovery of the native species and ecosystem process managers sought to promote. Most have focused on whether the targeted weed was killed or suppressed. A noteworthy exception to this has been the extensive work by McEvoy and colleagues (1990; 1991; 1993a,b; Diehl and McEvoy 1990; James and others 1992) following impacts of the tansy ragwort (*Senecio jacobaea*) biocontrol program in western Oregon not only on the target weed, but also on native species abundance and diversity. Earlier research following the impacts of the Klamathweed (*Hypericum perforatum*) biocontrol program in the Pacific states also provided useful information on the recovery of native species (Huffaker and Kennet 1959). Similarly, Rice and colleagues' (1997) studied the impacts of herbicidal control of spotted knapweed (*Centaurea maculosa*) on the diversity and abundance of native species in western Montana grasslands and early seral forests. Fortunately, there are more studies of this sort underway, for example a five-year study of the impacts of large-scale herbicidal fennel (*Foeniculum vulgare*) control on the native plants, insects

and herptiles on Santa Cruz Island, CA. However, land managers need to keep urging researchers to focus even more attention on the impacts of weed control efforts on the native species they seek to promote. In this regard, we can follow the lead of the agricultural community, where most weed control research is clearly focused on the ultimate goal in that realm - increasing crop production.

What Do We Still Need to Do and Know to Better Manage Invasive Wildland Weeds?

Despite a strong upsurge in awareness and actions to control invasive wildland weeds over the past decade, the problem continues to get worse. There are so many species of nonnative plants established in most natural areas that wildland managers will never have enough resources to control or contain them all. Therefore, there is a need for the development of weed management strategies that will efficiently and effectively address the most pressing problems. To implement these strategies, land managers will need better ways to prioritize their invasive weed problems. And to do this they will need better information on the ecological impacts of different invasive weeds, which ones can cause significant damage and which ones are relatively harmless, even if conspicuous. They need more information on the likely impacts of control on the weeds and the native plants and animals they want to protect. They need to know how to detect and map weeds over the large landscapes that they manage. They also need to know what steps they can take to prevent or slow invasions by new species and how to most quickly detect and contain new invaders. And they need good decision systems to help them synthesize all of this information and set logical priorities. Fortunately, work has begun on many of these fronts.

Adaptive Weed Management

Many land managers have begun using an 'adaptive strategy' for weed management. This is based on the precepts of adaptive management widely publicized and refined by Holling and Walters (Holling 1978; Walters 1986; Walters and Holling 1990). Randall and Robison (in prep) describe this as: 1) establishing management goals and objectives for the site; 2) identifying species that block you from reaching these goals and assigning them priorities based on the severity of their impacts; 3) selecting methods for controlling harmful species or otherwise diminishing their impacts and, if necessary, reordering priorities based on likely impacts of control on target and nontarget species; 4) developing and implement weed control plans based on steps 1-3; 5) monitoring the results of management actions; and 6) evaluating this information in light of the overall goals and objectives for the site and using this information to modify and improve control priorities, methods and plans, starting the cycle again. While use of this type of strategy is becoming more common, it is still too early to tell whether it will significantly improve weed management on the ground.

Setting Management Priorities

An important step in any comprehensive weed management program is setting priorities. This is often difficult because there are usually many invasive species and many invaded areas in a given wildland, and it can be difficult to collect and synthesize all the information necessary to set priorities. Hiebert and Stubbendieck (1993) developed and continue to improve upon a simple system (Hiebert 1997) designed to help land managers prioritizing invasive in logical step-by step fashion. This system is now available on the internet: <http://www.ripon.edu/faculty/beresk/aliens/>. But it will become more useful as information about the impacts of various species and of various control programs improves.

Quantifying Impacts of Weeds and Weed Control on Wildlands

A relatively small number of studies have clearly documented how certain weed species degrade the natural areas that they invade. Documented impacts include alteration of ecosystem functions like nutrient cycling, intensity and/or frequency of wildfires and hydrology, outcompeting and displacing native plants and animals and hybridizing with native species. Unfortunately, the impacts of many invasive species have not yet been clearly demonstrated. Experimental documentation of how well weed control programs work to restore native species and communities is even harder to find. These questions and information needs provide exciting challenges and opportunities for collaboration between weed scientists, conservation biologists and ecologists.

Mapping Wildland Weeds

Setting weed management priorities and assessing the impacts of control actions can be extremely difficult without accurate information on where the weeds are and whether their populations are spreading or contracting over time. Maps can fill this information gap but can be expensive and time-consuming to create, especially when the site is large. Several research groups have had some success accurately mapping selected wildland weeds, including leafy spurge, tamarisk, yellow hawkweed and yellow starthistle using images taken from airplanes (Birdsall and others 1997; Carson and others 1995; Everitt and others 1995, 1996; Lass and others 1996.). Progress has also been reported with the use of Global Positioning Systems and geostatistics to accurately map weed infestations (Child and de Waal 1997; Donald 1994; Webster and Cardina 1997). These techniques could significantly improve the coordination and success of wildland weed management in many areas, but their use is unlikely to become widespread until they become more affordable.

Improving Control Methods

There is, of course, also a great need for better control techniques. Methods that will kill or suppress only the

targeted species while leaving all other species unharmed would be ideal, but we will likely have to settle for less in many cases. One of the greatest differences between management of weeds on wildlands and agricultural lands is this desire for extreme specificity of control techniques. This means that we need to place great importance on how various control techniques affect other nontarget species. Even biocontrol, in some cases the most specific tool available, should be studied more carefully for nontarget impacts.

Can Native Insects and Pathogens Control Some Weeds?

There is some hope that some nonnative weeds will eventually be brought under control by native insects and pathogens that adapt to feeding on them. It has been hypothesized that some species introduced to new areas do not become invasive, or at least do not attain pest status, because they are attacked and kept in check by pathogens, parasites and predators (including herbivores) native to the new area. In most cases where this phenomenon is known or suspected, the introduced species never escaped control or became a pest. It is possible, however, that native species might not begin to feed on a new invader for decades or centuries, long after it has become established and abundant in the new land. In fact, many land managers hold out hopes that some of the weeds that plague them will someday be turned on by native herbivores and pathogens. One of the very few instances where this appears to have happened involves the native weevil (*Euhrychosis lecontei*), which is known to feed on the nonnative Eurasian watermilfoil (*Myriophyllum spicatum*; Sheldon and Creed 1995, Creed and Sheldon 1995). Significant impacts of this feeding were first noted only in this decade, and control of Eurasian watermilfoil attacked by the weevil remains irregular—satisfactory in some years, barely noticeable in others. The weevil may also cause watermilfoil to crash by early August, but the insect itself then becomes inactive, and watermilfoil may resurge dramatically by September.

The circumstances that allow this kind of “host-switching” of native species onto nonnative pests may occur only rarely. Nonetheless, it might prove extremely useful to learn more about what these circumstances are and whether there are ways to promote them. It would also be useful to know whether we might expect the likelihood of such host-shifting to increase with time and, if so, over what time-scale.

Preventing New Invasions

Basic research on invasiveness and invasibility can provide some help. The better our ability to predict which species are most likely to invade and become pests, the easier it will be to work with the nursery industry and other groups interested in importing new plant species to screen out at least a few of the likely bad actors. Greater knowledge of what makes a site prone to invasion may help managers set priorities for inventory and management activities. Unfortunately, we might get the most information about invasiveness and invasibility from experiments that are too dangerous and unethical to contemplate seriously: studies

in which new species were intentionally released and observed as they spread or died out over time.

References

- Aplet, G. 1990. Alteration of earthworm community biomass by the alien *Myrica faya* in Hawaii. *Oecologia* 82: 414-416.
- Anderson, M.G. 1995. Interactions between *Lythrum salicaria* and native organisms: a critical review. *Environmental Management* 19(2):225-231.
- Ayres, D.R., Garcia-Rossi, D., Davis, H.G. and Strong, D.R. In press. Extent and degree of hybridization between exotic (*Spartina alterniflora*) and native (*S. foliosa*) cordgrass (*Poaceae*) in California, USA determined by RAPDs Molecular Ecology.
- Baker, H.G. 1965. Characteristics and modes of origin of weeds. pp. 147-172 In Baker, H.G. and Stebbins, G.L. (eds.) *The Genetics of Colonizing Species*. Academic Press, New York.
- Barbour, M., DeJong, T. M. and Johnson, A. F. 1976. Synecology of beach vegetation along the Pacific Coast of the United States of America: a first approximation. *Journal of Biogeography* 3:55-69.
- Barbour, M. and Johnson, A. F. 1988. Beach and dune. pp. 223-261 in Barbour, M. and J. Major (eds.) *Terrestrial Vegetation of California*, new expanded edition. California Native Plant Society Special Publication No. 9.
- Barrows, C. W. 1993. Tamarisk control II. A success story. *Restoration and Management Notes* 11(1):35-38.
- Barthell, J.F., Randall, J.M., Thorp, R.W. and Wenner, A.M. In Prep. Yellow starthistle (*Centaurea solstitialis*), and feral honeybees (*Apis mellifera*): a case of invaders assisting invaders.
- Billings, W.D. 1990. Grasslands. In Woodwell, G.M. editor. *The Earth in transition: patterns and processes of biotic impoverishment*. Cambridge University Press, New York.
- Birdsall, J.L., Quimby, P.C., Jr., Rees, N.E., Svejcar, T.J. and Sowell, B.F. 1997. Image analysis of leafy spurge (*Euphorbia esula*) cover. *Weed Technology* 11: 798-803.
- Blackburn, W., R., Knight, W. and Schuster, J.L. 1982. Saltcedar influence on sedimentation in the Brazos River. *Journal of Soil and Water Conservation* 37:298-301.
- Blossey, B., Skinner, L. and Taylor, J. In press. *Lythrum salicaria* management: hypothesis testing, Type I and Type II errors and the precautionary principle.
- Bossard, C. C. 1991. The role of habitat disturbance, seed predation and ant dispersal on establishment of the exotic shrub *Cytisus scoparius* in California. *American Midland Naturalist* 126:1-13.
- Brotherson, J.D. and Field, D. 1987. Tamarix: impacts of a successful weed. *Rangelands* 9:10-19.
- California Natural Heritage Database. 1996. Natural Heritage Division, California Department of Fish and Game, Sacramento, CA 95814.
- Carson, H.W., Lass, L.W. and Callihan, R.H. 1995. Detection of yellow hawkweed (*Hieracium pratensis*) with high resolution multispectral digital imagery. *Weed Technology* 9: 477-483.
- Child, L. and de Waal, L. 1997. The use of GIS in the management of *Fallopia japonica* in the urban environment. pp. 207-220 in J.H. Brock, M. Wade, P. Pysek and D. Green (editors) *Plant invaders: studies from North America and Europe*. Backhuys Publishers, Leiden, The Netherlands.
- Creed, R.P. and Sheldon, S.P. 1995. Weevils and watermilfoil: did a North American herbivore cause the decline of an exotic plant? *Ecological Applications* 5: 1331-1321.
- Cronk, Q.C.B. and J.L. Fuller 1995. *Plant Invaders: the threat to natural ecosystems*. Chapman and Hall, London. 241 pp.
- D'Antonio, C.M. and Vitousek, P.M. 1992. Biological invasions by exotic grasses, the grass/fire cycle and global change. *Annual Review of Ecology and Systematics* 23: 63-87.
- Darwin, C. 1859. *The origin of species*. Reprinted by Penguin Books, London.
- DeLoach, C.J. 1991. Past successes and current prospects in biological control of weeds in the United States and Canada. *Natural Areas Journal* 11:129-142.
- Diehl, J.W. and McEvoy, P.B. 1990. Impact of the cinnabar moth (*Tyria jacobaeae*) on *Senecio triangularis*, a nontarget native plant in Oregon. *Proceedings of the VIII International Symposium on Biological Control of Weeds*: 119-126

- Donald, W.W. 1994. Geostatistics for mapping weeds, with a Canada thistle (*Cirsium arvense*) patch as a case study. *Weed Science* 42: 648-657.
- Elliot, W. 1994. German ivy engulfing riparian forests and heading for the uplands. *CalEPPC News*. 2(1):9.
- Elton, C.S. 1958. The ecology of invasions by animals and plants. Methuen, London.
- Everitt, J.H., Anderson, G.L., Escobar, D.E. Davis, M.R., Spencer, N.R. and Andrascik, R.J. 1995. Use of remote sensing for detecting and mapping leafy spurge (*Euphorbia esula*) *Weed Technology* 9: 599-609.
- Everitt, J.H., Escobar, D.E., Alaniz, M.A., Davis, M.R. and Richerson, J.V. 1996. Using spatial information technologies to map Chinese tamarisk (*Tamarix chinensis*) infestations. *Weed Science* 44: 194-201.
- Gautier, C.R. 1982. The effects of ryegrass on erosion and natural vegetation recovery after fire. In *Dynamics and management of Mediterranean-type systems*. General Technical Report PSW-58. USDA Forest Service, Pacific SW Forest and Range Experiment Station, Berkeley, California.
- Griffin, G. F., Stafford Smith, D. M., Morton, S. R. Allen, G. E. and Masters, K. A. 1989. Status and implications of the invasion of tamarisk (*Tamarix aphylla*) on the Finke River, Northern Territory, Australia. *Journal of Environmental Management* 29:297-315.
- Groves, R.H. 1989. Ecological control of invasive terrestrial plants. pp. 437-461 In J.A. Drake, H.A. Mooney, F. Di Castri, R.H. Groves, F.J. Kruger, M. Rejmanek and M. Williamson (eds.) *Biological invasions: a global perspective*. Wiley, Chichester.
- Groves, R. H. 1991. Status of environmental weed control in Australia. *Plant Protection Quarterly* 6:95-98.
- Hager, H.A. and McCoy, K.D. 1998. The implications of accepting untested hypotheses: a review of the effects of purple loosestrife (*Lythrum salicaria*) in North America. *Biodiversity and Conservation* 7: 1069-1079.
- Hiebert, R.D. 1997. Prioritizing invasive plants and planning for management. pp. 195-212 In J.O. Luken and J. W. Thieret (eds.) *Assessment and Management of Plant Invasions*. Springer-Verlag, New York.
- Hiebert, R. D. and Stubbendieck, J. 1993. Handbook for ranking exotic plants for management and control. Natural Resources Report NPS/NRMWRO?NRR-93/08. National Park Service, Natural Resources Publications Office, Denver.
- Hobbs, R.J. and Mooney, H.A. 1998. Broadening the extinction debate: population deletions and additions in California and Western Australia. *Conservation Biology* 12 (2): 271-283.
- Holling, C.S. (ed.) 1978. *Adaptive environmental assessment and management*. J. Wiley, London.
- Horton, J. S. 1977. The development and perpetuation of the permanent tamarisk type in the phreatophyte zone of the Southwest. pp. 124-127 in R.R. Johnson and D.A. Jones eds. *Importance, preservation and management of riparian habitat: a symposium*. General Technical Report RM-43. U.S. Forest Service, Fort Collins, Colorado.
- Howarth, F.G. 1991. Environmental impacts of classical biological control. *Annual Reviews of Entomology* 36:485-509.
- Hrusa, G.F. 1998. Senior Plant Taxonomist, California Department of Food and Agriculture. Personal communication.
- Huffaker, C. B. and Kennett, C.E. 1959. A ten-year study of vegetational changes associated with control of Klamath weed. *Journal of Range Management* 12:69-82.
- Humphries, S. E., Groves, R. H. and Mitchell, D. S. (eds.) 1991. *Plant invasions: the incidence of environmental weeds in Australia*. Kowari 2. Australian National Parks Wildlife Service, Canberra.
- James, R.R., McEvoy, P.B. and Cox, C.S. 1992. Combining the cinnabar moth (*Tyria jacobaeae*) and the ragwort flea beetle (*Longitarsus jacobaeae*) for control of ragwort (*Senecio jacobaeae*): an experimental analysis. *Journal of Applied Ecology*. 29: 3, 589-596.
- Julien, M.H. (ed.) 1987. *Biological Control of Weeds: A world catalogue of agents and their target weeds*, CAB International Institute of Biological Control, Ascot, UK.
- Keeley, S.C., Keeley, J.E., Hutchinson, S.M. and Johnson, A.W. 1981. Postfire succession of the herbaceous flora in southern California chaparral. *Ecology* 62:1608-1621.
- Kloot, P.M. 1983. The role of common iceplant (*Mesembryanthemum crystallinum*) in the deterioration of mesic pastures. *Australian Journal of Ecology* 8:301-6.
- Lacey, J.R., Marlow, C.B. and Lane, J.R. 1989 Influence of spotted knapweed (*Centaurea maculosa*) on surface runoff and sediment yield. *Weed Technology* 3:627-631.
- LaRoche, F.B. (ed.) 1994. *Melaleuca management plan for Florida: recommendations from the Melaleuca Task Force.*, second edition. Exotic Pest Plant Council, West Palm Beach, Florida.
- Lass, L.W., Carson, H.W. and Callihan, R.H. 1996. Detection of yellow starthistle (*Centaurea solstitialis*) and common St. Johnswort (*Hypericum perforatum*) with multispectral digital imagery. *Weed Technology* 10: 466-474.
- Louda, S.M. 1998. Ecology of interactions needed in biological control practice and policy. *Bulletin of the British Ecological Society* 29(4):8-11.
- Louda, S.M., Kendall, D. Conner, J. and Simberloff, D. 1997. Ecological effect of an insect introduced for the biological control of weeds. *Science* 277:1088-1090.
- Mack, R.N. 1981. Invasion of *Bromus tectorum* into western North America: an ecological chronicle. *Agroecosystems* 7:145-165.
- Maddox, D.M., Mayfield, A. and Poritz, N.H. 1985. Distribution of yellow starthistle (*Centaurea solstitialis*) and Russian knapweed (*Centaurea repens*). *Weed Science* 33:315-327.
- MacArthur, R.H. and Wilson, E.O. 1967. *The theory of island biogeography*. Princeton University Press, Princeton, NJ.
- McEvoy, P. B., Cox, C. S. and Coombs, E. 1991. Successful biological control of ragwort *Senecio jacobaeae*, by introduced insects in Oregon. *Ecological Applications* 1:430-442.
- McEvoy, P. B. and Coombs, E.M. 1999. Biological control of plant invaders: regional patterns, field experiments, and structured population models. *Ecological Applications* 9(2):387-401.
- McEvoy, P. B., Cox, C. S., James, R. R. and Rudd, N. T. 1990. Ecological mechanisms underlying successful biological control: field experiments with ragwort, *Senecio jacobaeae*. pp. 55-66 in E. S. Delfosse, ed. *Proc. of the VII Int. Symp. on Biological Control of Weeds*, 6-11 March 1988, Rome, Italy. Ministero dell'Agricoltura e delle Foreste, Rome/CSIRO, Melbourne Australia.
- McEvoy, P. B. and Rudd, N.T. 1993. Effects of vegetation disturbances on insect biological control of tansy ragwort, *Senecio jacobaeae*. *Ecological Applications* 3:682-698.
- McEvoy, P. B., Rudd, N. T., Cox, C. S. and Huso, M. 1993. Disturbance, competition and herbivory effects on ragwort, *Senecio jacobaeae*, populations. *Ecological Monographs* 63:55-75.
- McNaughton, S.J. 1983. Serengeti grassland ecology: the role of composite environmental factors and contingency in community organization. *Ecological Monographs* 53:291-320.
- Miller, L. 1988. How bush lupine came to Humboldt Bay. *Fremontia* 16(3):6-7.
- Mountjoy, J.H. 1979. Broom—a threat to native plants. *Fremontia* 6:11-15.
- Neill, W. 1983. The tamarisk invasion of desert riparian areas. Educ. Bull. #83-84 Publ. of the Education Foundation of the Desert Protective Council. Spring Valley, CA. 4 p.
- Panetta, F.D. 1993. A system of assessing proposed plant introductions for weed potential. *Plant Protection Quarterly* 8:10-14.
- Pickart, A.J. and Sawyer, J.O. 1998. Ecology and restoration of northern California coastal dunes. *California Native Plant Society*, Sacramento.
- Pimm, S. L. and Gilpin, M.E. 1989. Theoretical issues in conservation biology. pp. 287-305 in J. Roughgarden, R. M. May, and S. A. Levin, eds. *Perspectives in Ecological Theory*. Princeton Univ. Press, Princeton, NJ.
- Prather, T. S. and Callihan, R.H. 1991. Interactions between a yellow starthistle community and a pubescent wheat grass community. *J. Range Manage.* 44:443-447.
- Randall, J.M. 1991. The Ecology of an Invasive biennial, *Cirsium vulgare*, in California. Ph.D. dissertation, University of California, Davis.
- Randall, J.M. 1996. Assessment of the invasive weed problem on preserves across the United States. *Endangered Species Update* 12: 4-6.
- Randall, J.M. In press. *Ficus carica* L. In C.C. Bossard, J.M. Randall and M. Hoshovsky (eds.) *Invasive plants of California's wildlands*. University of California Press, Berkeley.

- Randall, J.M., Lewis, R.R. III and Jensen, D.B. 1997. Ecological restoration and nonindigenous species invasions in Florida. pp. 205-219 *In* D. Simberloff, D. C. Schmitz and T. C. Brown (eds.) *Strangers in paradise: impact and management of nonindigenous species in Florida*. Island Press, Washington, D.C.
- Randall, J.M. and Robison, R.A. In preparation. Planning weed management in natural areas.
- Reichard, S.H. 1997. Prevention of invasive plant introductions on national and local levels. pp. 215-227 *in* J.O. Luken and J.W. Thieret (eds.) *Assessment and Management of Plant Invasions*. Springer, New York.
- Reichard, S.H. and Hamilton, C.W. 1997. Predicting invasions of woody plant introductions into North America. *Conservation Biology* 11:193-203.
- Rejmánek, M. 1995. What makes a species invasive? pp. 3-13 *in* P. Pysek, K. Prach, M. Rejmánek and P.M. Wade, eds. *Plant Invasions*. SPB Academic Publishing, The Hague, The Netherlands.
- Rejmánek, M. 1996. A theory of seed plant invasiveness: the first sketch. *Biological Conservation* 78:171-181.
- Rejmánek, M., and Richardson, D.M. 1996. What attributes make some plant species more invasive? *Ecology* 77:1655-1661.
- Rice, P. M., Toney, J.C., Bedunah, D. J. and Carlson, C. E. 1997. Plant community diversity and growth form responses to herbicide applications for control of *Centaurea maculosa*. *Journal of Applied Ecology* 34:1397-1412.
- Sayce, K. 1990. Species displacement by *Spartina* in the Pacific Northwest. pp. 26-27 *in* T. F. Mumford, Jr., P. Peyton, J. R. Sayce, and S. Harbell (eds.) *Spartina workshop record*. Seattle, November 14-15 1990.
- Schultz, A.M. , Launchbaugh, J.L. and Biswell, H.H. 1955. Relationship between grass density and brush seedling survival. *Ecology* 36:226-238.
- Scott, J.M. and Wilcove, D.S. 1998. Improving the future for endangered species. *Bioscience* 48 (8): 579-580.
- Sheldon, S.P. and Creed, R.P., Jr. 1995. Use of a native insect as a biological control for an introduced weed. *Ecological Applications* 5: 1122-1132.
- Stohlgren, T.J. Binkley, D., Chong, G.W., Kalkhan, M.A., Schell, L.D., Bull, K.A., Otsuki, Y., Newman, G., Bashkin, M. and Son, Y. 1999. Exotic plant species invade hot spots of native diversity. *Ecological Monographs* 69(1): 25-46.
- Stohlgren, T.J. Bull, K.A., Otsuki, Y., Villa, C.A. and Lee, M. 1998. Riparian zones as havens for exotic plant species in the central grasslands. *Plant Ecology* 138: 113-125.
- Tilman, D. 1997. Community invasibility, recruitment limitation, and grassland biodiversity. *Ecology* 78(1):81-92.
- USDA and USDI 1992. Guidelines for Coordinated Management of Noxious Weeds in the Greater Yellowstone Area.
- Vivrette, N.J. and Muller, C.H. 1977. Mechanism of invasion and dominance of coastal grassland by *Mesembryanthemum crystallinum*. *Ecological Monographs* 47:301-318.
- Vitousek, P.M. 1986. Biological invasions and ecosystem properties: can species make a difference? pp. 163-176 *in* H. A. Mooney and J. A. Drake eds. *Ecology of Biological Invasions of North America and Hawaii*. Springer-Verlag, New York.
- Vitousek, P.M., Walker, L. R., Whiteaker, L. D., Mueller-Dombois, D. and Matson, P. A. 1987. Biological invasion by *Myrica faya* alters ecosystem development in Hawaii. *Science* 238:802-804.
- Vitousek, P.M., D'Antonio, C.M., Loope, L.L. and Westbrooks, R. 1996. Biological invasions as global environmental change. *American Scientist* 84: 468-478.
- Vitousek, P.M. and Walker, L.R. 1989. Biological invasion by *Myrica faya* in Hawaii: plant demography, nitrogen fixation, ecosystem effects. *Ecological Monographs* 59:247-265.
- Wagner, W.L., Herbst, D.R. and Sohmer, S.H. 1990. *Manual of the Flowering Plants of Hawaii*, Volume 1. University of Hawaii Press and Bishop Museum Press, Honolulu.
- Walters, C.J. 1986. *Adaptive management of renewable resources*. McGraw- Hill, New York.
- Walters, C.J. and Holling, C.S. 1990. Large-scale management experiments and learning by doing. *Ecology* 71:2060-2068.
- Webster, T.M. and Cardina, J. 1997. Accuracy of a global positioning system (GPS) for weed mapping. *Weed Technology* 11: 782-786.
- Whisenant, S.G. 1990. Changing fire frequencies on Idaho's Snake River Plains: ecological and management implications. pp. 4-10 *in* E.D. McArthur, E.V. Romney, S.D. Smith, and P.T. Tueller, eds. *Proceedings - Symposium on Cheatgrass Invasion, Shrub Die-off, and Other Aspects of Shrub Biology and Management*. Las Vegas, NV. USDA Forest Service Intermountain Research Station General Technical Report INT-276.
- Zedler, P.H., Gautier, C.R. and McMaster, G.S. 1983. Vegetation change in response to extreme events: the effects of a short interval between fires in California chaparral and coastal scrub. *Ecology* 64:809-818.

Protecting Wilderness Air Quality in the United States

K. A. Tonnessen

Abstract—Federal land managers are responsible for protecting air quality-related values (AQRVs) in parks and wilderness areas from air pollution damage or impairment. Few, if any, class 1 areas are unaffected by regional and global pollutants, such as visibility-reducing particles, ozone and deposition of sulfur (S), nitrogen (N) and toxics. This paper lays out the basic definitions and research findings that managers need to protect natural resources and scenic vistas. A detailed case study is presented that traces the development of scientific knowledge of the effects of S and N on wilderness resources. Gaps in our understanding of deposition and its effects, and managers' need for monitoring, modeling and data synthesis tools are discussed, with recommendations on how to use science and technology to protect AQRVs in wilderness areas and parks.

External threats to wilderness areas come in many forms. One of the most pervasive stresses is air pollution from local, regional and global emission sources. Federal land managers (FLMs) were initially concerned about the effects of local air pollution on surface waters, native vegetation, soils, wildlife and cultural resources. These threats included sulfur dioxide (SO₂), nitrogen oxides (NO_x), fluorides, lead (Pb) and soot from power plants, industries and urban areas. The United States has made considerable strides since the passage of the Clean Air Act in 1970 to clean up local sources of pollution. However, with the advent of "tall stacks" on large point sources, there is now more opportunity for long-distance transport of pollution to parks and wilderness areas. The greatest air pollution threat to natural resources and scenic vistas in remote wilderness areas currently is from regional and global pollutants.

The focus of this discussion will be on regional pollution issues: visibility, ozone and deposition of sulfur (S) and nitrogen (N) compounds (also known as "acid rain"). Other air pollutants of concern in wilderness areas will be defined, but not explored in any depth. The detailed case study of deposition includes information on (1) history of deposition research and monitoring, (2) what we know, (3) gaps in our knowledge, (4) how managers have used the data, (5) current needs of managers, and (6) research, monitoring and assessment strategies for FLMs.

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Definitions and Overview

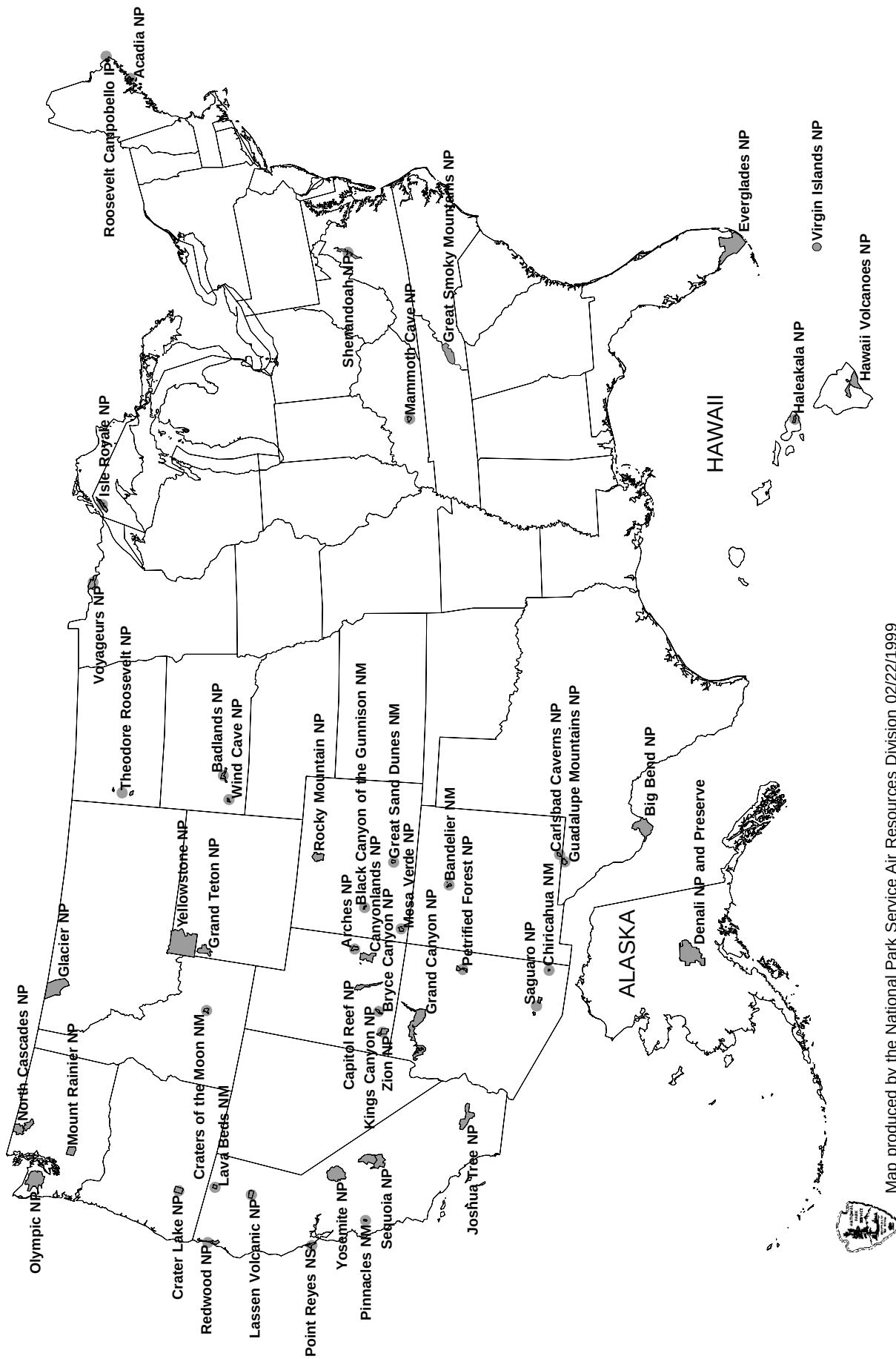
Basics of Class 1 Air Quality

Class 1 Areas—Wilderness areas over 5,000 acres in size, and national parks greater than 6,000 acres were singled out for special protection from air pollution under the Clean Air Act Amendments (CAAA) of 1977. There were 158 units in 1977 that received this level of protection. They are managed by the following Federal Land Managers (FLMs): USDA-Forest Service (USFS) (88 wilderness areas); DOI-National Park Service (NPS) (48 national parks and 1 international park); and DOI-U.S. Fish and Wildlife Service (FWS) (21 wilderness areas). Figure 1 shows the distribution of NPS protected areas. It is possible to add class 1 areas through a process known as redesignation. Five Native American lands that have been "redesignated" class 1.

Federal Land Managers—For the purposes of this discussion, the agencies that have stewardship over public lands designated as class 1 are known as federal land managers (FLMs). These include DOI-National Park Service, DOI-U.S. Fish and Wildlife Service, and USDA-Forest Service. FLMs that will not be specifically discussed in this paper are the DOI-Bureau of Land Management (BLM), which manages one class 1 wilderness area, and the Native American tribes, which can redesignate their lands as class 1. The three FLMs with the largest number of class 1 parks and wilderness areas have joined forces as part of the Federal Land Managers Air Quality-Related Values Work Group (FLAG), in an effort to coordinate activities in protecting air quality-related values (AQRVs) from air pollution. This group has recently issued a draft report that outlines the major air quality concerns and starts the process of setting thresholds and critical loads to protect sensitive resources (FLAG 1999).

Legal Responsibilities—The array of legislative requirements to protect parks and wilderness areas from air pollution are listed in the FLAG report (1999). These include the FLMs' Organic Acts, park and wilderness enabling legislation, Wilderness Act and Clean Air Act and its amendments. The National Environmental Policy Act requires that air quality be considered in environmental impact statements (EISs) for significant federal actions. Details of these mandates are included in Bunyak (1993).

Methods used by FLMs in an effort to control air pollution effects in class 1 areas include: (1) new source review of proposed air pollution sources within 100 km of the wilderness boundary, (2) request for Best Available Retrofit Technology (BART) to be installed on large power plants to remedy visibility impairment, (3) participation in regional air quality groups to implement the regional haze regulations (i.e., Western Regional Air Partnership), (4) providing



Map produced by the National Park Service Air Resources Division 02/22/1999

Figure 1—National Park Service Class 1 areas.

research and monitoring data to the Environmental Protection Agency (EPA) in the review of National Ambient Air Quality Standards (NAAQS), (5) providing comments on environmental impact statements (EISs) for development that will affect class 1 areas, (6) providing data and comments on State Implementation Plans (SIPs), and (7) participation in bioregional assessments, such as the Sierra Nevada Ecosystem Project (SNEP) and the Southern Appalachian Mountains Initiative (SAMI 1999).

Criteria Air Pollutants—These air pollutants include sulfur dioxide (SO_2), nitrogen oxides (NO_x), ozone (O_3), particulate matter (PM-10) and lead (Pb) and were specifically identified by the EPA as harmful to human health and welfare. The EPA has set specific control levels for these pollutants, known as National Ambient Air Quality Standards (NAAQS), based on the concentrations in ambient air. For a discussion of current trends in these pollutants, see U.S. EPA (1998); and for a tutorial on urban air pollution, including its chemistry and physics, see Seinfeld (1989). In 1997, the EPA revised the NAAQS for ozone and introduced a new NAAQS for PM-2.5, fine particles less than 2.5 microns in diameter, known to affect human lung function, visibility and deposition of acidic materials. However, these new NAAQS were recently called into question in a court decision (May 1999); additional litigation will determine if they are reinstated.

The values for criteria air pollutants and NAAQS are based on protecting the sensitive people in the population. Sensitive scenic values and natural resources in parks and wilderness areas can be more sensitive to injury due to air pollution than the standards set by EPA (as in the case of ozone effects on sensitive tree species, such as Ponderosa pine (*Pinus ponderosa*) and black cherry (*Prunus serotina*)). Also, the form of the pollution that affects natural areas is often different from the form of the criteria pollutants. The major regional air pollutants discussed in this paper include: (1) fine particles (less than 2.5 microns), which affect visibility and scenic resources (Malm 1992), (2) ozone, which affects forest health (U.S. EPA 1996a; 1996b), and (3) deposition of sulfur and nitrogen, which has a myriad of effects: acidification of soils and freshwaters, eutrophication of estuaries and near-coastal marine systems and alteration of ecosystem processes and nutrient cycling by altering soil biogeochemistry (NAPAP 1998).

Regional Air Pollutants—Regional air pollutants that affect scenic and natural resources in class 1 areas include: fine particles, ozone, deposition of nitrogen and sulfur, and toxic air contaminants, especially mercury (Hg).

1. Fine particles: This class of pollutant is also known as visibility-reducing particles, or PM-2.5, and includes both primary and secondary particles with a diameter of less than 2.5 microns. The primary particles come from diesel exhaust, smelter emissions, forest fires and windblown dust. Secondary particles are the result of atmospheric transformations of SO_2 , NO_x , ammonia (NH_3) and organic compounds. The chemical composition of the fine particles include, generally, sulfate and nitrate particles, organics and carbon (soot). These particles are most effective at absorbing light. These same particles are the most likely to enter the human lung and cause health effects in sensitive human populations. For this reason, the EPA recently set a new NAAQS for PM-2.5 (U.S. EPA 1998).

Since visibility and scenic vistas are important air quality-related values, the FLMs, in concert with the EPA, states and industries, created the Interagency Monitoring of Protected Visual Environments (IMPROVE) monitoring network. As part of the newly promulgated regional haze regulations (April 22, 1999), the total number of monitors in parks and wilderness areas will increase to 110, to be installed by early 2000.

A fully complemented IMPROVE site employs three types of monitors: photographic, optical and aerosol. Photographic monitoring documents the condition of a scenic vista in a park several times a day using a 35-mm camera. Optical monitoring directly measures the light extinction coefficient with transmissometers or the light scattering coefficient with nephelometers. The light extinction coefficient is a measure of the attenuation of light per unit distance caused by the scattering and absorption of gases and particles in the atmosphere. The scattering coefficient has a similar definition, except absorption is not included. Aerosol monitoring includes the collection of fine (PM-2.5) and coarse (PM-10) particles on different types of filters, which are analyzed for mass, chemical constituents, organics, elemental carbon and optical absorption. The concentrations of aerosol constituents are used to estimate their contributions to the light extinction coefficient, and allow for the plotting of "reconstructed extinction." For more information on the network and results of the analyses, see Eldred and Cahill (1994), Malm (1992) and Sisler and others (1996).

A sample of data collected at class 1 parks from 1991-1997 is included in Figure 2. This bar graph depicts the reconstructed extinction at 11 parks included in the park index site network, Park Research and Intensive Monitoring of Ecosystems Network (PRIMENet). The data are expressed as inverse megameters (Mm^{-1}), with Denali National Park having the lowest concentration of fine particles and extinction values that correspond to a 186-km standard visual range. At this "clean site," most of the light extinction is explained by atmospheric light scattering by gas molecules, known as Rayleigh scattering. In contrast, park sites in the eastern U.S., especially Great Smoky Mountains and Shenandoah National Parks, show large extinctions associated with sulfate aerosol. The next largest contributors to visibility degradation at all the sites are organic carbon and soot, attributed to biomass burning and urban emissions.

The Clean Air Act Amendments (CAAA) of 1977 provide special protection for visibility in class 1 areas. There are two emission control programs specifically concerned with visibility in national parks and wilderness areas: the Prevention of Significant Deterioration (PSD) program (directed mainly at new sources) and the visibility protection program, which allows for control of existing sources of pollution (National Research Council 1993). The first major action under the CAAA provisions was the certification of visibility impairment in all NPS class 1 areas, including the Grand Canyon, by the Department of the Interior, assistant secretary for Fish, Wildlife and Parks in 1985 (Shaver and Malm 1996). After a series of intensive studies to determine the contribution of the Navaho Generating Station (NGS) to winter haze in Grand Canyon National Park, Canyonlands National Park, and Glen Canyon National Recreation Area, the EPA issued a proposed regulation to require a 70% reduction in NGS SO_2 emissions, to be achieved through the

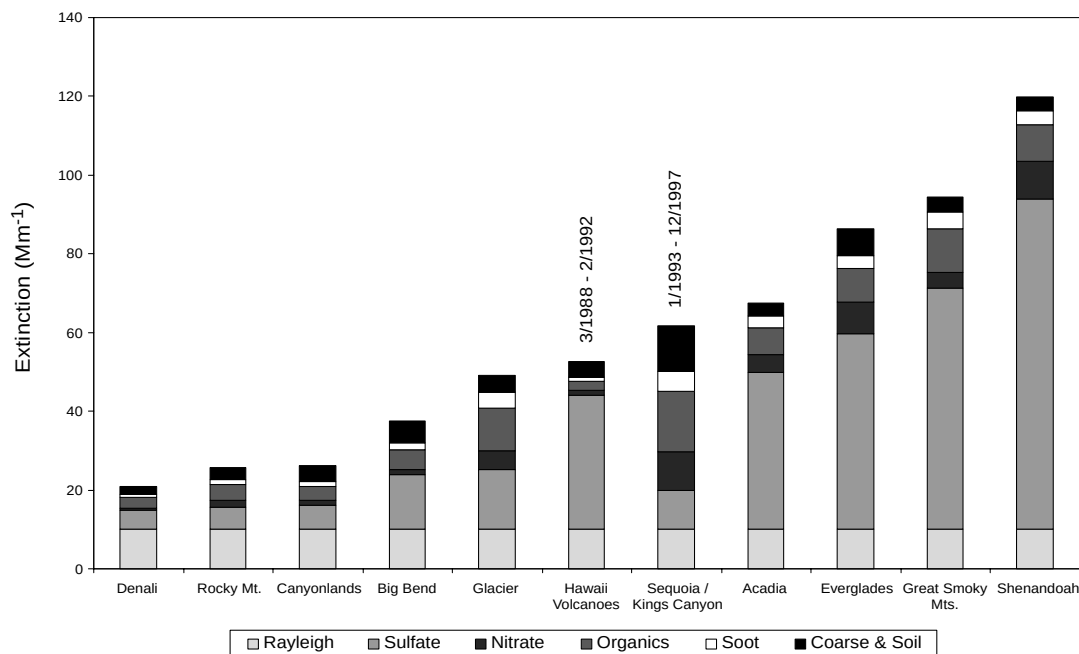


Figure 2—Average reconstructed extinction at PRIMENet sites, 1991-1997.

installation of scrubbers. Negotiations among industry, environmental groups and the EPA resulted in the recommendation of a 90% SO₂ reduction, with an initial delay in installation of the control equipment. This recommendation was adopted in the final regulation, announced by President Bush at the Grand Canyon in September 1991.

Federal land managers have tried this strategy to control large coal-fired power plants located upwind of class 1 areas. The USFS certified visibility impairment to the Mount Zirkel Wilderness Area in Colorado, due to SO₂ emissions from the Craig and Hayden power plants. A lawsuit by the Sierra Club, prompted by numerous violations of the opacity standard at Hayden, resulted in an agreement by the utility to install SO₂ and NO_x control equipment. SO₂ emissions from the Centralia power plant in Washington State were linked to visibility degradation at Mount Rainier National Park and several USFS wilderness areas in the Cascades. Through a “collaborative decisionmaking” process among all affected parties, there was an agreement to install scrubbers on this, the largest source of SO₂ in the West after NGS. In each case, special studies of visibility and other AQRVs were organized to allow for “attribution” to specific sources. This costly and time-consuming process led to the requirement in the CAAA of 1990 for the creation of a Grand Canyon Visibility Transport Commission (GCVTC 1996), which came up with recommendations to EPA on how to protect visibility in class 1 areas of the Colorado Plateau.

Many of these recommendations were included in the regional haze regulations announced by Al Gore on Earth Day 1999. This comprehensive approach to reductions in regional haze acknowledges the impairment of visibility at class 1 areas in all 50 states; its long-term goal is to return visibility conditions in the parks and wildernesses to “natural background.” These regulations call for states to form regional groups to come up with pollution reduction strategies,

which are likely to include the use of Best Available Retrofit Technology (BART) for existing point sources of pollution. One such regional group, the successor to GCVTC, is an association of Western states, now known as Western Regional Air Partnership (WRAP). For information on the new regional haze rules, see the EPA website: <http://www.epa.gov/ttn/oarpg>.

2. Tropospheric or ground-level ozone: This is also a criteria pollutant, formed by the reaction of NO_x and volatile organic compounds (VOCs) in the presence of sunlight. Ozone is a strong oxidizing agent that affects human lung function and damages vegetation by entering through the stomates and causing cell death. This pollutant is transported to class 1 areas in proximity to urban areas, especially on the East and West Coasts. Ozone injury to native vegetation has been documented in parks and wilderness areas in California (Miller and others 1996) and in the Southeast (Chappelka and Samuelson 1998). The sensitive indicator plants include Ponderosa and Jeffrey pine in the West and hardwoods, such as black cherry and white ash, in the East. A number of understory plants, such as milkweed, asters and blackberry have shown visual injury symptoms due to ozone during controlled-fumigation experiments and in the field (Neufeld and others 1995). The response of vegetation to ozone exposure varies with other environmental conditions. For instance, during drought periods, plant stomates remain closed, cutting down on the uptake of ozone.

Ozone levels are typically reported in terms of the primary NAAQS set by EPA. The standard to protect human health was set at a one-hour average of 120 parts per billion (ppb); the new standard promulgated in 1997 is an eight-hour average of 80 ppb, considerably more restrictive. The setting of this new standard means that a number of class 1 areas may exceed the health standard for ozone. Note: the

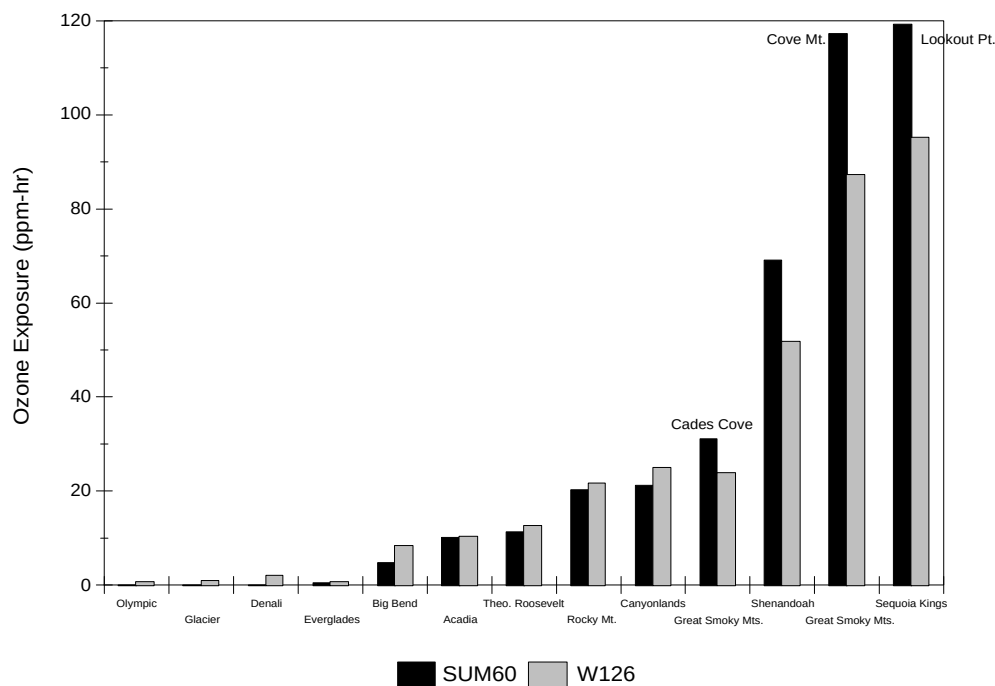


Figure 3—Ozone exposures at PRIMENet sites for the 1997 growing season.

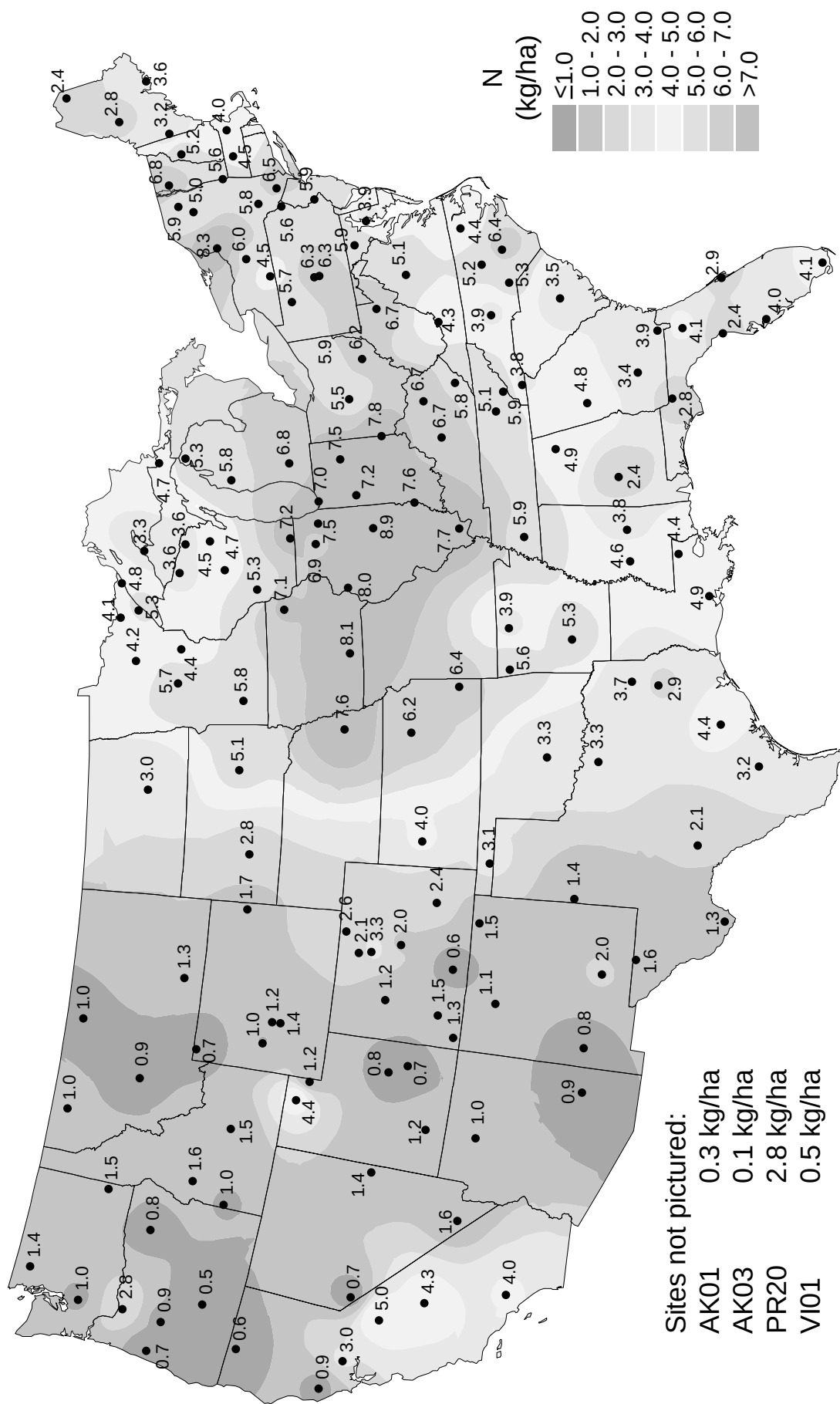
standard has been challenged in court. Vegetation responds differently to ozone exposure, so scientists have come up with two integrating statistics to describe ozone levels during the growing season (U.S. EPA 1996a, 1996b). Figure 3 shows the calculated SUM60 and W126 indices for 12 PRIMENet parks for the 1997 growing season (May-September). The SUM60 is a sum of all hourly ozone concentrations equal to or exceeding 60 ppb; the W126 is the sum of all hourly ozone concentrations, weighted by a function that gives greater emphasis to concentrations above 60 ppb (Lefohn and others 1992). Lookout Point, in Sequoia National Park (CA) and Cove Mountain, in Great Smoky Mountains National Park (TN/NC), recorded the highest ozone exposures in 1997. These parks typically have sensitive vegetation that show ozone injury by the end of the growing season. The contrast between Cove Mountain and Cades Cove in Great Smoky Mountains National Park points out the influence of elevation on total ozone exposure. At most, if not all, sites monitored for ozone in mountain parks, the highest levels of ozone are measured at the higher elevations.

3. Deposition of sulfur and nitrogen compounds: Deposition includes chemical constituents that accumulate on surfaces, delivered via rain, snow, mist, fog, clouds and dry-deposited gases and particles. The most commonly measured form of deposition is wetfall, usually rain and snow, measured by the National Trends Network/National Atmospheric Deposition Program (NTN/NADP). NTN/NADP is a national network of monitors where wetfall is measured weekly, with samples sent to a Central Analytical Lab, in Champaign, IL, for analysis of chemical constituents, including pH (H ion), major anions (including sulfate and nitrate) and major cations (including calcium, magnesium, sodium and ammonium) (Lynch and others 1995). NTN/NADP includes more than 220 sites, primarily in rural

areas, with many sites located in or adjacent to class 1 areas. The list of class 1 monitoring sites is included in the FLAG (1999) report.

Analytical products from the network include isopleths maps of chemical concentrations and wet deposition collected during each calendar year (NADP 1999). These maps, such as the one shown in Figure 4, allow for regional assessment of pollutant loading. The map shows 1997 deposition of nitrogen in rain and snow.

Another way of presenting the data is to compare volume-weighted, average concentrations of selected constituents across sites (Figure 5). This plot of nitrate and sulfate in precipitation averaged over the period of 1984-1997 shows the relative loading of these two pollutant species across a number of NPS class 1 areas. The lowest mean concentrations of nitrate in rain were recorded at Olympic National Park (WA) and Denali National Park (AK). Many of the parks throughout the country show similar nitrate concentrations (10-15 ueq/l), with Sequoia-Kings Canyon National Parks (CA) and Rocky Mountain National Park (CO) (Beaver Meadow site) having more nitrate than sulfate in rain. The spatial patterns in rainfall sulfate concentrations at these parks reflect the influence of sulfur emissions in the eastern U.S. and the U.S./Mexico border region. Two of the national parks with the highest sulfate concentrations, Great Smoky Mountains and Shenandoah National Parks, are also the parks that have adverse impacts to their natural resources as a result of acidic deposition. In Shenandoah National Park streams are experiencing both chronic and episodic acidification (Bulger and others 1998), and there are documented effects on fisheries in the park. In Great Smoky Mountains National Park nitrate is leaking out of watershed soils into streamwater, causing episodic acidification. There is also evidence that soil water is acidified by



National Atmospheric Deposition Program/National Trends Network
<http://nadp.sws.uiuc.edu>

Figure 4—Estimated inorganic nitrogen deposition, NTN/NADP, 1997.

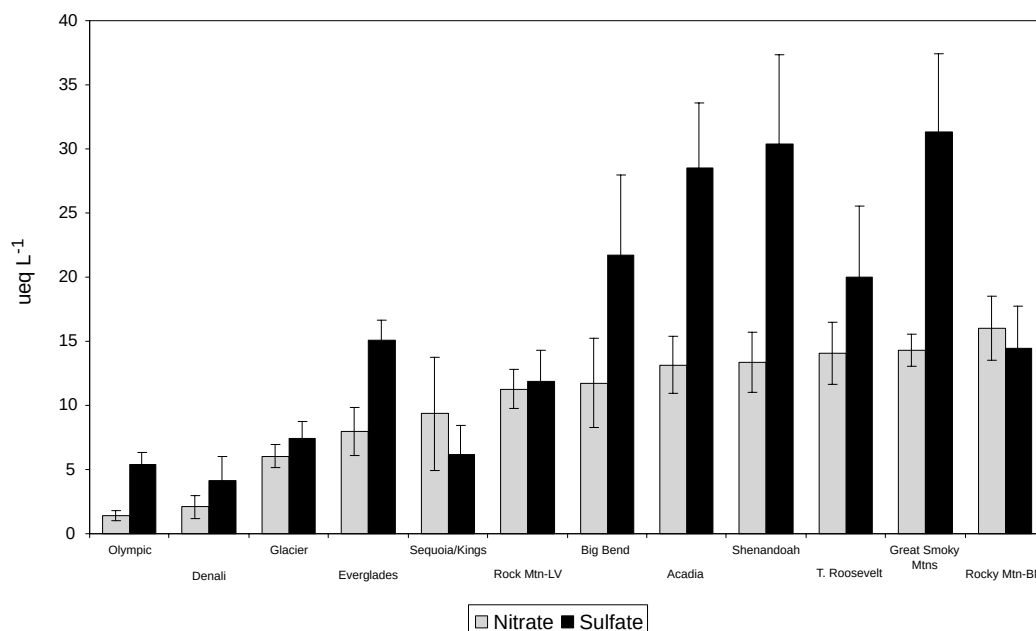


Figure 5—Average annual concentrations of nitrate and sulfate in precipitation, 1984-1997.

deposition, as evidenced by the indicator of terrestrial health, the calcium to aluminum ratio (Johnson and others 1991; van Miegroet and others 1992).

The data summarized in Figure 5 focus on nitrate and sulfate because these are the two chemical constituents that contribute to acid loading and are “acid anions,” which can leach nutrients such as calcium and magnesium from the soils (Lawrence and Huntington 1999), and contribute to acidification of freshwaters characterized by low buffering capacity or acid-neutralizing capacity (ANC). Nitrate can also act as a fertilizer, especially in waters where phosphorus is abundant, as in the case of many estuaries along the Atlantic and Gulf Coasts (U.S. EPA 1994, 1997).

From an ecosystem perspective, it is important to determine the total amount (or loading) of these chemicals to sensitive ecosystems in protected areas. The NTN/NADP data summaries include estimates of deposition of nitrogen and sulfur in wet deposition, based on the amount of rain or snow that fell at that point. For many class 1 areas, especially in mountainous terrain, the greatest loading comes in the form of seasonal snow (Elder and others 1991). There are sampling problems in snow collection using NTN/NADP buckets at high-elevation sites (Williams and others 1998). In some Western mountain ranges, chemical loading in seasonal snowpacks is estimated at maximum accumulation, which allows for measurement of both wet and dry deposition during the snow-covered period (Heuer and others 2000; McGurk and others 1989).

There are protected areas where rain and snow are small contributors to total chemical deposition from the atmosphere. Many of these sites are now included as part of a national dry deposition network, known as the Clean Air Status and Trends Network (CASTNet), with a number of partner agencies, including EPA-Office of Air and the National Park Service (Lear and Frank 1998). National Parks class 1 and 2 areas that have a dry deposition filter pack in

or adjacent to them include: Big Bend National Park (TX); Canyonlands National Park (UT); Chiricahua National Monument (AZ); Death Valley National Park (CA); Everglades National Park (FL); Glacier National Park (MT); Grand Canyon National Park (AZ); Great Smoky Mountains National Park (TN/NC); Hawaii Volcanoes National Park; Joshua Tree National Park (CA); Mesa Verde National Park (CO); Mount Rainier National Park (WA); North Cascades National Park (WA); Olympic National Park (WA); Pinnacles National Monument (CA); Rocky Mountain National Park (CO); Sequoia-Kings Canyon National Parks (CA); Shenandoah National Park (VA); Voyageurs National Park (MN); Yellowstone National Park (WY); Yosemite National Park (CA); Acadia National Park (ME); Denali National Park (AK); Virgin Islands National Park; Chiricahua Wilderness Area (AZ); and Lye Brook Wilderness Area (NH).

CASTNet sites include a three-stage filter pack that collects particles and gases, including nitric acid, particulate nitrate, sulfur dioxide and particulate sulfate. These sites typically include a continuous ozone monitor and meteorological instruments that collect data needed to run the models used to estimate deposition from the ambient measurements. Both an Eastern park (Shenandoah National Park (VA)) and a Western park (Sequoia-Kings Canyon National Parks (CA)) have the highest concentrations of nitrogen species in ambient air. By summer 1999, the CASTNet website will include dry deposition estimates derived for network sites using the NOAA “big leaf” model.

4. Toxic air contaminants are defined in the CAAA of 1990, which identifies 188 substances that need to be controlled to protect human health. However, the regulatory approach used to control emissions of these substances (also called persistent toxic substances) is based on technology controls of emissions from the major sources of these pollutants, such as power plants, industrial facilities, incinerators, and smelters. The toxic air contaminants that have

the most relevance to class 1 area resources are mercury, dioxin, chlordane and PCBs (polychlorinated biphenyls). These are substances that travel long distances from sources and bioaccumulate in fish and other wildlife. Thirty states have consumption advisories for specific waterbodies to warn consumers about Hg-contaminated fish and shellfish (U.S. EPA 1994, 1997).

The toxic air contaminant that has received the most attention from FLMs and state managers of fish and game is mercury (Hg). This toxic metal accumulates in fish and wildlife tissue and is a potent neurotoxin. Hg has many natural and man-made sources and has a complicated geochemical cycle. It is emitted from large point sources such as electrical-generating plants, chlor-alkali plants and waste incinerators. But is also emitted during forest fires, and from degassing of soils. High concentrations of Hg have been measured in sediments and fish tissue in certain remote parts of the high Arctic (Landers and others 1998). In recognition of its importance, federal and state agencies, Canadian agencies, universities and industry partners set up the Mercury Deposition Network (MDN) in 1996, as a sub-network of the National Atmospheric Deposition Program, to measure the annual concentration and deposition of Hg in wetfall (Sweet and others 1998). It is important to note that due to its high volatility, the predominant form in the atmosphere is gaseous Hg. This form of Hg can be transported long distances, and has a low solubility in water,

and is therefore not efficiently scavenged by rainfall (Brosset and Lord 1991).

Figure 6 shows the distribution of Hg deposition among the 30 MDN sites. A number of FLM areas are included in the network: Everglades National Park (FL), Acadia National Park (ME), Congaree Swamp NM (SC), Okefenokee National Wildlife Refuge (GA), Chassahowitzka National Wildlife Refuge (FL) and Mount Zirkel Wilderness Area (CO). Data for annual deposition ($\mu\text{g}/\text{m}^2$) in 1997 show the highest loading for Everglades National Park.

One class of toxics that is of current concern to natural resource managers is endocrine-disrupting compounds (EDCs). These are complex, organic compounds that “mimic” estrogens and can affect reproductive systems in wildlife and humans (Colborn and Clement 1992). These compounds include dioxin, DDT, DDE and other pesticides. Recent studies indicate that these compounds have wide distribution in the environment and are scavenged by snow in high-elevation regions in the mid-latitudes (Blais and others 1998).

Routine monitoring for toxic substances is limited. The EPA is setting up a national dioxin monitoring network, called National Dioxin Air Monitoring Network (NDAMN). Some class 1 areas, such as Big Bend National Park (TX), Everglades National Park (FL); Craters of the Moon National Monument (ID); and Grand Canyon National Park (AZ), have been proposed as network sites because they meet the siting criteria outlined in the EPA’s Dioxin Exposure

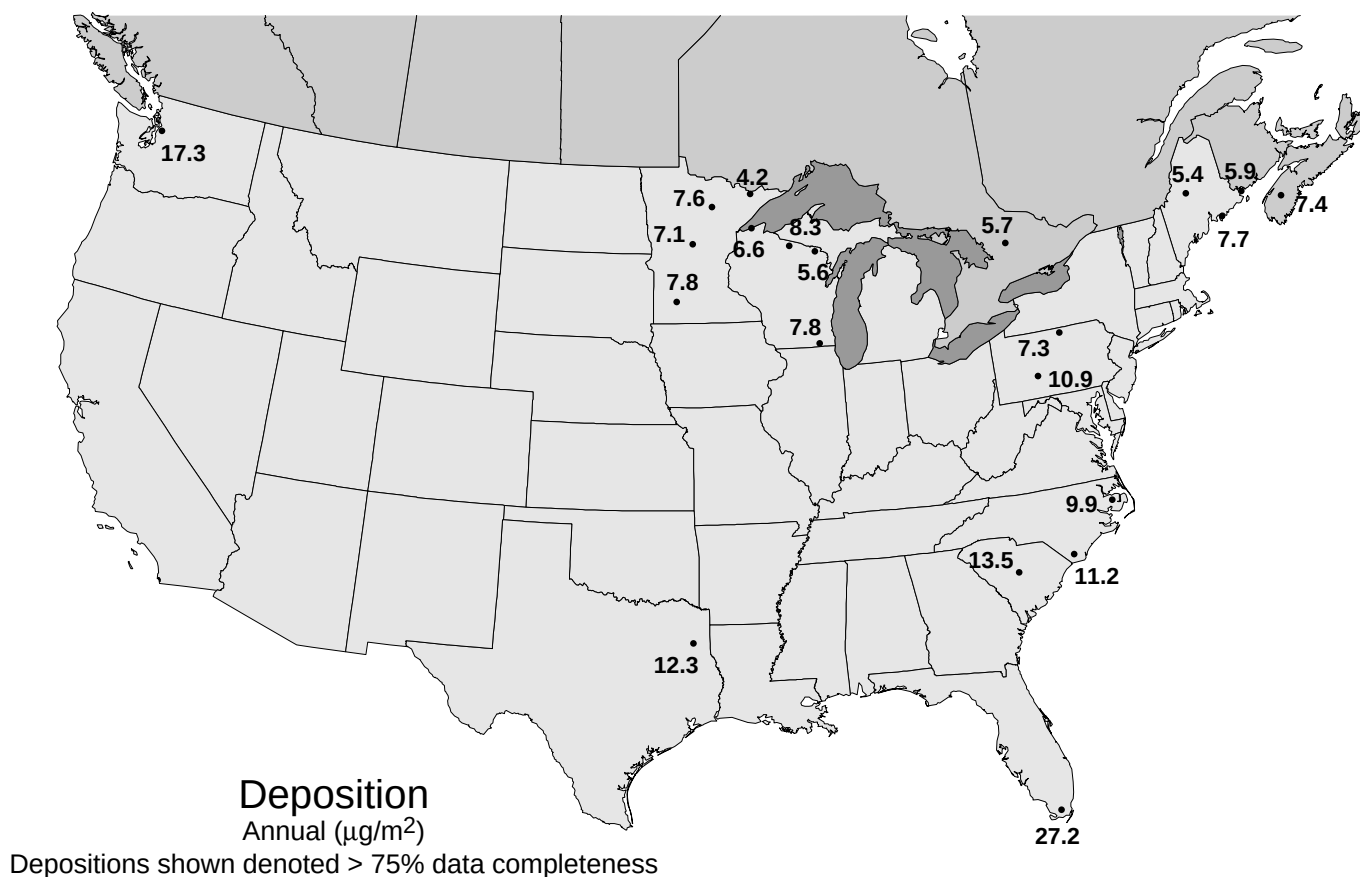


Figure 6—Annual deposition of mercury, 1997.

Initiative. The NDAMN sampler is the PUF (poly-urethane foam), which collects particle and vapor-phase pesticides.

The only long-term toxics monitoring network is sponsored by the EPA in the Great Lakes region. The Integrated Atmospheric Deposition Network (IADN) includes one site on each of the Great Lakes on both sides on the border. Sleeping Bear Dunes National Lakeshore (MI) is the IADN site on Lake Michigan. This network tracks both inorganic and organic pollutant trends and is associated with the Great Lakes Water Quality Agreement signed by the U.S. and Canada. The Commission on Environmental Cooperation (CEC), created by the NAFTA "side agreement" on environment, is planning a trilateral air monitoring network to measure toxic air contaminants in Canada, the U.S. and Mexico (CEC 1998).

In summer 1998, the NPS and the EPA collaborated in a contaminant screening study to collect and analyze organic and inorganic pollutants in various media, including water, sediment, fish and vegetation in 12 class 1 areas. The project is part of the index site network, Park Research and Intensive Monitoring of Ecosystems. Data from this "screening" study are expected in summer 2000.

Global Air Pollutants—These air pollutants fall into two classes: ozone-depleting compounds (ODCs) and greenhouse gases, including carbon dioxide, methane and nitrous oxides. These air pollutants tend to be long-lived in the atmosphere and have the ability to travel globally in both the troposphere and the stratosphere (upper layer of the atmosphere).

Ozone-depleting compounds include chlorofluorocarbons (CFCs) and freons. They are used in refrigeration and as solvents. These substances are transported to the stratosphere, where they chemically destroy the protective ozone that filters out UV light (WMO 1994). In 1985, the scientific community discovered the stratospheric ozone "hole" over the Antarctic, which resulted in more damaging UV-B reaching the surface of the earth. Ozone thinning has been detected throughout the globe, with seasonal depressions in this protective shield being most severe at the poles (Madronich 1993). In 1987, the major industrial nations signed the Montreal Protocol on Substances that Deplete the Ozone Layer, which calls for a phase-out of CFCs. Because of the long lifetimes of CFCs in the upper atmosphere, it is not known when the ozone thinning will be reversed. In the mid-latitudes of the U.S., UV-B levels have increased 4-5% over the past 10 years (U.S. EPA 1998).

Effects of UV-B on biological systems include: increases in human skin cancers and cataracts, damage to phytoplankton and reduction in growth of fish, molluscs and crustacea, damage to DNA and photosynthesis in plants and possible effects on animals, including benthic invertebrates and amphibians (Tevani 1992; Williamson and Zagarese 1994).

Because they are located relatively distant from local pollution sources, 14 class 1 parks were selected by the EPA as UV monitoring sites. These parks are part of a larger index site network known as Park Research and Intensive Monitoring of Ecosystems Network (PRIMENet) (see map of sites in figure 7). Each site is equipped with a Brewer



Figure 7—Map of NPS/EPA PRIMENet sites.

spectrophotometer, an instrument designed to measure different wavelengths of light, with a focus on the ultraviolet spectra (UV-B radiation is in the 300-320 nm range of light). These instruments track the sun as they monitor the variation in solar irradiance throughout the day; they also record other data, such as total column ozone and ambient concentration of gases. These data are then used to calculate the “dose” of UV at the surface of the earth. Because of the influence of sun angle, clouds and other forms of air pollution, the seasonal variation in UV-B detected at the surface is large, as shown in the annual data. Therefore, it will take many years of monitoring to detect trends in the incidence of UV-B.

The PRIMENet sites complement a larger Brewer network in the U.S. that includes seven monitors located in cities. These monitoring devices have also been deployed in Canada and on other continents, to allow for a global assessment of the status of the stratospheric ozone layer (Wilson and others 1992).

The major pollutant gas contributing to global warming (85 % of total) is carbon dioxide (CO₂), produced during the combustion of fossil fuels. Methane (CH₄) is the second largest contributor to greenhouse gas emissions. This compound is emitted from agricultural lands, landfills and natural wetlands. There is scientific consensus among the scientists who drafted the report of the Intergovernmental Panel on Climate Change (IPCC) in 1995 that “climate change is likely to have wide-ranging and mostly adverse impacts on human health, with a significant loss of life.” Satellite observations indicate that growing seasons in the high latitudes may have increased by 12 days from 1981-1991 (Myneni and others 1997). Snowcover in the northern hemisphere appears to have retreated by 10% between 1972-1992, likely affecting boreal and arctic ecosystems (Groisman and others 1994).

The U.S. is being called on under the Kyoto Protocol (1997) to reduce greenhouse gas emissions to 7% below the base year of 1990. The U.S. “Climate Change Action Program,” devised in 1993, resulted in a 15-million ton reduction in greenhouse gases during 1997 (U.S. EPA 1998).

Emission Sources—Power plants: Nationally, power plants account for the majority of SO_x and CO₂ emissions and significant amounts of NO_x and Hg (U.S. EPA 1998). Power plants burning fossil fuels contribute an estimated 67% of SO_x, 28% of NO_x, 35% of CO₂ and 33% of mercury (although there is considerably uncertainty in the emission inventories, especially for Hg). Most of the point source SO_x and NO_x is emitted from coal-burning power plants built before 1980. Other major point sources of these criteria pollutants include smelters, refineries and industrial facilities. Under the Clean Air Act and its amendments, new facilities are required to install clean technology; under the CAAA of 1990, identified sources are scheduled to install retrofit technology or use cleaner fuels to achieve targeted reductions. However, there is a class of power plants that was “grandfathered” under the CAAA, those in operation before the mid-1980s. Many of these plants, especially in the eastern U.S., are operating past their 30-year projected life span and, therefore, are the major sources of acid deposition precursor emissions.

Mobile sources: Fuel combustion in the transportation sector is the largest contributor to NO_x emissions; stationary

combustion sources account for most of the remaining emissions. In the period of 1988-1997, there was a 1% decrease in NO_x emissions in the U.S. (U.S. EPA 1998). Two recent developments in the regulatory arena are likely to control growth or reduce NO_x emissions. In 1998, the EPA called on the 22 Eastern states to revise their State Implementation Plans (SIPs) to reduce NO_x emissions in the summer to achieve reductions in ozone. This control strategy resulted from modeling analyses performed by the Ozone Transport Assessment Group (OTAG). In May 1999, President Clinton announced new auto emission rules that will require cuts in NO_x emissions from light-duty trucks and more stringent levels of these emissions overall from the fleet, beginning in 2004. SO_x emissions from vehicles will be reduced under the proposed rule to cut the sulfur in gasoline from an average of 300 ppm (parts per million) to 30 ppm by 2004. This measure is recommended primarily to prevent “poisoning” of the catalytic converters in vehicles.

Air Quality-Related Values (AQRVs)—These are the wildland resources that federal land managers are required to protect from air pollution injury. These are generally defined in the CAAA as visibility, flora, fauna, water quality, soils, wildlife, odor and ecosystem integrity. These are being further defined by the FLMs to include lists of sensitive indicators and the levels of pollution that will affect these indicators. The FLAG effort is designed to coordinate the development of lists of sensitive indicators and pollution levels of concern, known as “critical loads,” “critical levels,” “screening level values” or “thresholds.” The most current information is summarized in the FLAG, Phase 1 report (FLAG 1999), with more detailed information included in an array of synthesis documents prepared in the last decade by the USFS (Adams and others 1991; Fox and others 1989; Haddow and others 1998; Peterson and others 1993; Peterson and others 1992; Stanford and others 1991; Turner and others, in preparation), the NPS (Binkley and others 1997; Eilers and others 1994; Peterson and others 1998) and the FWS (Porter 1996).

Natural resources and scenic values most at risk from regional air pollution include: the effects of fine particles on visibility, the effects of ozone on native vegetation, the effects of deposition on surface waters, estuaries and terrestrial systems and the bioaccumulative effects of toxics, such as mercury and chlorinated organics, on aquatic organisms.

Deposition of Sulfur and Nitrogen as a Case Study

History of Deposition Research

“Acid rain,” the deposition of acidic compounds of nitrogen (N) and sulfur (S), was first recognized to have ecological consequences as a result of early studies in Europe. In the U.S., monitoring of precipitation chemistry began in 1978, in response to scientific concern about this stressor. The wet deposition network, known as National Atmospheric Deposition Program (Lynch and others 1995), is the longest running environmental chemistry network in the U.S. The Canadians, concerned that U.S. air pollution was affecting their natural resources, also set up a deposition chemistry

network and defined a “target load” of wet sulfate deposition of 20 kg/ha/year to control damage to lakes in the eastern provinces (Environment Canada 1998). The Canadians have since refined their assessment of the response of lakes to acidic deposition and have set a critical load of 8 kg/ha/year to protect the most sensitive systems.

The measurement and monitoring of deposition inputs in North America has progressed beyond monitoring of wet deposition alone to include national networks to measure dry deposition (Clean Air Status and Trends Network, CASTNet), daily wet deposition inputs (Atmospheric Integrated Research Monitoring Network, AIRMon) and mercury (Mercury Deposition Network, MDN) (Sweet and others 1998). Additional deposition data are available from short-term and regional networks to measure cloudwater (Mountain Cloud Chemistry Network and CASTNet), snow-pack chemistry in the Rockies (Rocky Mountain Synoptic Snow Network) (Ingersoll 1995) and various forms of deposition in California (Blanchard and others 1996; Blanchard and Tonnesen 1993; California Air Resources Board 1993). Class 1 areas in the U.S. are relatively well-characterized with respect to rain, but few parks and wilderness areas monitor clouds, fog, dry deposition or toxic air contaminants on a routine basis as part of a national, quality-assured networks (Federal Land Managers AQRV Working Group 1999).

Research on the effects of acid deposition began in earnest in the U.S. with the passage by Congress of the Acid Precipitation Act of 1980. This legislation authorized a \$500 million research program over a 10-year period. During that time, the National Acid Precipitation Assessment Program (NAPAP) provided funding and direction for 12 federal agencies and hundreds of scientists, both within agencies and outside. This was one of the first experiments with “policy-relevant” research and assessment (Winstanley and others 1998). The final assessment (National Acid Precipitation Assessment Program 1991) and 13 State-of-Science and Technology documents were the products of this scientific effort, which was not without its critics. The results of this research, monitoring and modeling exercise was the Clean Air Act Amendments of 1990, which called for a 10-million ton reduction in SO_x emissions and a two million ton reduction in NO_x emissions in order the reverse the effects on lakes, streams, fish and watersheds soils that were documented in the eastern U.S. and Canada. One provision in the CAAA reauthorized NAPAP to perform periodic assessments of the effectiveness of these emission reductions. The first of these “follow-up” assessments was completed in 1998 (National Acid Precipitation Assessment Program 1998); the next is scheduled for 2000.

At the end of the first 10 years of NAPAP, the scientific community realized that they had not paid sufficient attention to the impacts of nitrogen deposition on freshwaters, terrestrial systems and estuaries (Fenn and others 1998; Vitousek and others 1997). The Forest Response Program (Bernard and Lucifer 1990) and the Episodic Response Program (Wigington and others 1990) were just starting to publish results when NAPAP came to the end of its 10-year funding. Since 1990, there have been additional studies to investigate streamwater episodic acidification and the further development of models to improve prediction of surface water acidification (i.e., MAGIC model: Model of the

Acidification of Groundwater in Catchments) (Cosby and others 1995; Sullivan and Cosby 1995).

Since much of the deposition monitoring and effects research under NAPAP focused on the eastern U.S., there was little information generated about how deposition affects the western U.S., the location of the largest number of class 1 parks and wilderness areas. With the exception of the Western Lake Survey (Eilers and others 1988; Landers and others 1987), little attention was paid to low-acid neutralizing waters in the Sierra Nevada, Cascades, and Rocky Mountains. Even less is known about forest and grassland status relative to deposition in the Southwest and the Colorado Plateau, home to a large number of class 1 parks and wilderness areas. Coastal wilderness areas managed by the FWS were not included in research programs funded under NAPAP. In subsequent assessments and programs, these data gaps are being filled. The EPA’s report on deposition standards (U.S. EPA 1995) began to recognize the role of nitrogen in altering ecosystem processes in Western mountains. There was also a realization that wet deposition inputs in the sensitive, high-elevation areas of the West were dominated by snow and required a different type of deposition monitoring and a different approach to effects research (McGurk and others 1989; Sickman and Melack, 1998; Williams and others 1996b).

Without a national research program dedicated to investigating deposition effects on natural resources, we are left with a patchwork of research and monitoring programs, with NAPAP existing on paper as the “clearinghouse” for research results. Under the CAAA of 1990, NAPAP is still required to carry out periodic assessments of the effects of deposition, with funding and personnel to write the assessments provided by federal agencies. The next assessment, scheduled for the year 2000, will further investigate the progress of ecosystem recovery with reductions in SO_x emissions.

The focus of deposition research and data analysis has shifted to regional assessments, such as the Southern Appalachian Assessment (Southern Appalachian Mountains Initiative 1999). These regional assessments make use of existing models and field data on ecosystem response to deposition. No new data are generated during these exercises. To advance the science of deposition effects on resources, the FLMS rely on the “science” and research arms of their respective agencies; U.S. Geological Survey for the Department of the Interior, and USFS research stations for the USFS. The FLMS have also been able to attract research and monitoring programs funded by other federal agencies, such as Environmental Monitoring and Assessment Program (EMAP), PRIMENet and Global Change programs under the EPA-Office of Research and Development and the Long Term Ecological Research program, funded by the National Science Foundation. Parks and wilderness areas have also served as “ground truth” sites for NASA satellites and remote sensing instruments, such as LANDSAT, SAR (Synthetic Aperture Radar) and AVHRR (Advanced Very High Resolution Radiometer). Some of the NASA-Earth Observing System investigations have focused on class 1 areas, such as Sequoia-Kings Canyon National Parks (CA), Glacier National Park (MT) and Rocky Mountain National Park (CO). With the launch of the TERRA earth-observing platform in 1999, there are more opportunities for collection

Table 1—Milestones in science and policy related to nitrogen and sulfur deposition.

1970	Clean Air Act (CAA)
1977	CAA Amendments (CAAA); Class 1 areas designated
1978	National Atmospheric Deposition Program (NADP) initiated
1980	National Acid Precipitation Assessment Program (NAPAP) starts
1990	CAAA passed; acid rain control program
1991	NAPAP Integrated Assessment published
1991	US/Canada Air Quality Accord signed
1992	Southern Appalachian Mountains Initiative (SAMI) begun
1995	EPA's Deposition Standards Report published
1997	New NAAQS for Ozone and PM-2.5 announced
1997	Class 1 Area managers form Federal Land Managers' Air Quality Related Values Work Group (FLAG) to address consistency in approach to AQRV protection
1998	NAPAP Assessment Report published
1998	NO _x SIP Call announced by EPA
1999	Final Regional Haze Regulations for Protection of Visibility in National Parks and Wilderness Areas.
1999	New vehicle emission rules; fuel standards
1999	Court decision questions basis for new NAAQS

of remote-sensing data on sensitive mountain ecosystems in the West.

For a tabular history of both research and regulatory developments in the area of deposition effects on resources, see Table 1.

Deposition Monitoring and Research Results

What We Know About Deposition

Deposition of N and S and Its Effects—A thorough discussion of regional wet and dry deposition and its effects on watersheds and surface waters is found in Charles (1991), with information on class 1 areas containing sensitive lakes and streams in the eastern U.S. (Mid-Atlantic Highlands) and in the western U.S. (Rockies, Cascades and Sierra Nevada). An update of “what we know” is included in the recent NAPAP assessment (1998). Chemical species in deposition that determine the “dose” to the ecosystem are: hydrogen ion (pH), sulfate, nitrate and ammonium.

In general, acidity in rain and snow can affect soil fertility and nutrient cycling processes in watersheds. Acidity in rain and snow can result in acidification of low-acid neutralizing capacity (ANC) lakes and streams, either of a chronic nature or episodically. In the mountainous areas in the western U.S., the total loading of wet deposition is high, but the concentrations of hydrogen ion at present are low, resulting in a relatively small total load of solutes to these systems. However, in the eastern U.S. at “high” elevations in parks and wilderness areas of the Southern Appalachians, total deposition of acidity and solutes is high due to a combination of inputs from dry, wet and cloudwater deposition (Johnson and Lindberg 1992). The other factor that must be considered in estimating the “load” of hydrogen ion to the ecosystem is the timing of the precipitation. In high-elevation regions, especially in the West, much of the annual

precipitation is snow, which accumulates in a seasonal snowpack and then melts during a relatively short period in the spring. Any acidity in the snowpack that is not buffered in-situ is likely to come out as a concentrated “pulse” of acidic meltwater. This snowpack melting phenomena tends to exacerbate the effect of chemical loading to the pack (Bales and others 1993; Wigington and others 1996).

Another important chemical species in deposition is nitrogen. Deposition of excess nitrogen (nitrate and ammonium) to both terrestrial and aquatic systems can result in: (1) fertilization or eutrophication, and (2) episodic acidification of streams and lakes (Stoddard 1994). The role of ammonium in acidification and nitrogen leakage from ecosystems had been largely ignored in discussions of pollutant impacts in the eastern U.S. (NAPAP 1991). However, in Western locations, such as the Sierra Nevada and the Colorado Rockies, the ratio of nitrate to ammonium in wet deposition is frequently 1. The reaction of nitric acid with ammonia gas emitted from feedlots and fertilized fields results in formation of ammonium nitrate particles, which can be carried long distances before being deposited in remote watersheds. When this buffered compound reaches soils and surface waters, the ammonium is preferentially taken up by biota, thus generating acidity. It is possible for ammonium nitrate transformation and transport to deliver nitrogen species to parks and wilderness areas in some regions of the country, such as the Sierra Nevada and the Front Range of Colorado, depending on the pattern of local ammonia emissions relative to the supply of nitric acid vapor.

Changing Composition of Deposition—We now have sufficient years of data as part of NTN/NADP to plot trends in wet deposition. Lynch and others (1995; 1996) performed an analysis of the trends in wet deposition chemistry for the period 1983-94 and then continued to track wetfall trends for eastern U.S. sites to look for evidence of the 1995 SO_x emission reductions required under the CAAA. The general pattern nationally was a trend toward decreases in sulfate in rain and snow, with little change in nitrate concentrations. Over large areas of the eastern U.S. there were 10-25% decreases in sulfate wet deposition, especially downwind of the Ohio River Valley. Some sites in the network showed increasing concentrations of ammonium. The surprise in the analysis was a general decrease in the concentrations of base cations (calcium, magnesium, sodium and potassium) in rainfall, especially in the Northeast. This general trend was also noted in Europe (Hedin and Likens 1996; Hedin and others 1994). Even wilderness sites in “background” areas, such as Denali National Park (AK) and the Pacific Northwest (Olympic, Mount Rainier and North Cascades National Parks) showed low concentrations of inorganic nitrogen, but with evidence of an increasing trend (Lynch and others 1995). It is not likely that the small emission reductions of NO_x required under the CAAA (a cut of 2 million tons) will result in reductions of N species in rain.

Differences in Types of Deposition—Most of the research and monitoring on deposition focused on regions where rain inputs are the major form of deposition, especially in the Northeast, where most of the acidified waters are found. As NAPAP progressed, there was more information available on major chemical loading coming into sensitive ecosystems in the form of dry deposition, snow loading and cloudwater. On ridges and mountain tops in the eastern

U.S., there can be considerable deposition of S and N from cloudwater (Johnson and Lindberg 1992; Lovett and others 1999; Vong and others 1991). However, these inputs are extremely variable in time and space, depending on the characteristics of the forest canopy. Because of the large heterogeneity and presence of "hotspots" in dry deposition and cloudwater deposition across landscapes, it is likely that models or statistical extrapolation may be preferable to direct monitoring of these inputs (Lovett and others 1999). The role that dry deposition plays in chemical loading to deserts and aridlands, such as the Colorado Plateau parks and wilderness areas, is now being investigated at sites throughout the Southwest, where both NTN/NADP and CASTNet sites have been installed.

The other major form of chemical loading to sensitive ecosystems is snow, which accumulates in seasonal snowpacks and then melts during a short period in the spring (Campbell and others 1995; Elder and others 1991). In extreme cases, such as in the alpine of the Sierra Nevada and the Cascades, as much as 90% of the total precipitation annually can be in the form of snow. However, there is interannual variability in these inputs, with the prospect that Western mountains will receive more rain and less snow because of increasing global temperatures. Large episodes of snowmelt runoff in the spring affects stream and lakewater hydrology and chemistry. In the Western mountains, researchers have observed loss of ANC and depression in pH in surface waters, caused by both elution of ions and dilution by snowmelt (Sickman and Melack 1998; Stoddard 1995; Turk and Campbell 1987).

Reducing Sulfur Emissions Affects Surface Waters—

The recent NAPAP assessment (1998) points out that the reduction in sulfur emissions under the CAAA has been translated into a reduction in sulfate concentrations in deposition and in stream and lake waters in the northeastern U.S. and Canada (Stoddard and others 1998). What was unexpected was the general lack of recovery of pH and ANC in many of the affected water bodies in this region. There are a number of hypotheses to explain this phenomena, including the reduction in base cations in deposition (Hedin and others 1994) and the leaching loss of cations from the soil, resulting in less buffering of incoming acidity (Lawrence and Huntington 1999). The general conclusion is that the reductions in SO_x emissions may be inadequate to improve the acid-base status of freshwaters in the eastern U.S. and Canada. The Canadians arrived at the same conclusion in their acid rain assessment (Environment Canada 1998), and are calling for another round of sulfur emission reductions in both countries, and a revision in the critical loads of S needed to protect the most sensitive lakes in the eastern provinces, in areas such as Kejimikujik National Park (Nova Scotia).

Identity of Sensitive AQRVs—The considerable research on natural ecosystems pursued under NAPAP, the Great Waters Program and state agency programs, such as the California Air Resources Board's, Acid Acidity Protection Program (CARB 1993), has given us a general list of ecosystem components that respond to deposition of sulfur and nitrogen.

1. Freshwater lakes and streams, having ANCs less than 50 ueq/l.

2. Aquatic biota, especially fish (Bulger and others 1998), zooplankton (Engle and Melack 1995) and aquatic invertebrates (Kratz and others 1994).

3. High-elevation watersheds soils, especially in alpine areas and in the spruce-fir zone in the eastern U.S. (Brooks and others 1996; Eager and Adams 1992).

4. Estuaries, which respond to nitrogen inputs by producing algal blooms, oxygen depletion of bottom waters and loss of fish and shellfish (U.S. EPA 1994; 1997).

Indicators of Surface Water Acidification—Lake and stream chemistry responds to increases in deposition of N and S. The chemical changes include loss of ANC, lowered pH, increases in sulfate concentrations and increases in aluminum. Both chronic and episodic changes in water chemistry can affect aquatic organisms, including fish, plankton and aquatic insects. But the response of aquatic biota is variable, depending on other environmental factors, such as drought, floods, organic content of the water and available refugia for fish. The most successful way to identify sensitive biota and to determine their response to acidification is to conduct controlled and replicated in-situ or laboratory experiments (Barmuta and others 1990; Kratz and others 1994). Controlled experiments indicated the lack of response of amphibian species in the Sierra Nevada to episodic acidification (Bradford and others 1994), even though this was a plausible hypothesis at the outset of the study.

Indicators of N Fertilization and N Saturation—The AQRVs that were not well-defined under the first NAPAP (1991) are indicators of estuary health, which respond to nitrogen inputs. Through a combination of monitoring, research and modeling as part of the Great Waters Program (U.S. EPA, 1994; 1997), there is an increased awareness of how deposition of nitrogen in the form of nitrate and ammonium to both water surfaces and watersheds is affecting the biological and chemical status of estuaries and near-coastal waters. A number of FWS wilderness areas along the Atlantic and Gulf Coasts include significant estuary resources that may be affected by deposition of nitrogen (Dixon and Esteves 1998).

Most of the research on estuary response to N inputs has been conducted in the Chesapeake Bay, the largest estuarine system in the contiguous U.S., with a watershed of almost 64,000 square miles, encompassing 1/6 of the Eastern seaboard. Recent results have been obtained from integrated modeling of deposition of nutrients to the bay surface and to the watershed using the Regional Acid Deposition Model, along with water quality and sediment exchange modeling. The models show that a reduction of 20-30% in N and P loadings would result in improvement in dissolved oxygen status. Other models indicate that 30-40% of the N that reaches the bay was deposited from the atmosphere either directly on the water or to the extensive watershed (National Acid Precipitation Assessment Program 1998).

Grassland species diversity and ecosystem function were investigated in a series of N addition experiments in the upper Midwest (Tilman and others 1997; Wedin and Tilman 1996). Simulated N deposition resulted in a change in species diversity, favoring the more opportunistic and "weedy" species, although overall biomass was not significantly

affected. These kinds of ecosystem process experiments are valuable, but it is difficult to devise an easily monitored indicator based on these findings.

Watershed Processes Control Chronic and Episodic Acidification—Most deposition comes in contact with soils before entering surface waters. The severity and type of acidification are determined by hydrologic flow paths through watershed soils. These flowpaths are influenced by climate, precipitation and soil strata. To understand acidification dynamics, researchers have used both naturally occurring isotopes of hydrogen, oxygen, nitrogen, and sulfur and labelled compounds (Kendall and others 1995; Williams and others 1996a).

Depending on the regional hydrology and deposition regimes, sensitive systems may be subject to either chronic and episodic acidification. We find chronically acidic lakes and streams in the eastern U.S., with some streams in Shenandoah National Park having ANC of 0 or less year round, the major acid anion being sulfate. Low-ANC systems found at high elevations in both Eastern and Western wilderness areas and parks are susceptible to episodic acidification associated with intense rains or spring snowmelt. Under this scenario, acidic rain events or the first “pulse” of acidic water from snowpack melting enter low-ANC waters and depress pH and ANC to critical levels. Some of this depression in pH and ANC is a result of dilution of surface waters by snowmelt (Campbell and others 1995; Melack and Sickman 1995). Evidence of acidic episodes have been collected in lakes and streams of Shenandoah, Great Smoky Mountains, Rocky Mountain, Sequoia-Kings Canyon and Yosemite National Parks. Any class 1 area with low-ANC surface waters and a seasonal snowpack can experience episodes.

Unresolved Issues and Research Gaps

Why ANC Is Not Recovering in the East—It appears that the level of sulfur emission control required by the CAAA will not permit the most acidified lakes in regions like the Adirondacks (NY) to recover. There are questions about why this recovery is not occurring. Signs point to loss of base cations (calcium and magnesium) from the soils and the reduction of these same base cations in rainfall. Continued deposition and surface-water monitoring are needed, along with improved response models, before the “right” levels of deposition are identified. The Canadians are calling for another round of sulfur reductions, and possibly nitrogen oxide reductions, to permit the recovery of their sensitive lakes in the Eastern provinces (Environment Canada 1998).

Fate of Nitrogen in Ecosystems—The concept of “nitrogen saturation” of ecosystems was introduced in the late 1980s, at the close of NAPAP (Aber and others 1989). Because N is an essential nutrient for plant growth, it has been more difficult to determine why N is “leaking” out of systems all over the world. A number of stresses and natural processes, including fire, land use, disturbance and insect infestation, can cause the terrestrial systems to “leak” N to streams and lakes (Fenn and others 1998). In alpine regions of the Rockies, the extent of seasonal snow cover will influence the amount of N leaving the terrestrial system in the

spring snowmelt (Williams and others 1996b). Even in the Northeast, where N loading is high, there is evidence that climate is an important control of N cycling (Mitchell and others 1996).

In streams monitored in the northeastern U.S. and in the Mid-Appalachian Highlands, nitrate is now observed at high concentrations during hydrologic episodes and during baseflow periods, indicating that the supply of nitrogen has exceeded the capacity of the soils and vegetation to absorb it (Stoddard 1994). There are a number of explanations for this nitrogen “leakage,” including the maturation of forests, effects of insect infestation and excess nitrogen supply in deposition. Recent investigations in Shenandoah National Park have attempted to separate out the effects of nitrogen flux to upland systems due to deposition from the impact of nitrogen cycle disruption from a gypsy moth infestations. At these affected watersheds, the export of nitrogen via stream-water has resulted in increased frequency of acidic episodes, known to affect native fish species (Bulger and others 1998).

In forests of the arid Southwest, recent data suggest that fire is the most important factor influencing the N cycle (Johnson and others 1998). Most of the forested areas in the West receive low to moderate N deposition, with the exception of southern California (Fenn and others 1996). Studies in Little Valley, Nevada indicate that N fluxes via fire and post-fire N-fixation greatly exceeded atmospheric deposition and leaching of N.

Because of these different controls on nitrogen cycling throughout class 1 areas, there is a need for continued monitoring and research to determine the role of deposition and to define “critical loads” or thresholds of N to protect ecosystem function.

Do We Have the Right Indicators for AQRVs?—Researchers are looking for the best “indicator” of ecosystem response to increasing inputs of S and N. We have moved past the concept of “dead lakes” and “dead fish” to consider what metrics should be used to determine the health of a sensitive system and its response over time to changes in deposition. There is a general acceptance that chemical endpoints, such as pH and ANC of streamwaters or the calcium/aluminum ratio in soil waters, have characteristics that make them good choices as indicators.

The next challenge is to tie changes in these chemical parameters to ecological processes or biological populations that people and land managers “care about,” such as frogs, fish or spruce trees. The current state of science cannot make that connection, with the exception of the effects of acidification on fish populations. And even with the fish and acidification relationship, there are enough confounding factors, such as habitat quality, food supply, predation and competition, to make the dose/response relationship less than straightforward. The selection of sensitive indicators will also require that the species or ecosystem process have a predictable response to deposition, one not confounded by other environmental responses (Hacker and Neufeld 1993).

How to Explore Links Among Climate Change, UV Radiation and Regional and Global Pollutants—We tend to compartmentalize air quality effects research, when in reality, these stressors can interact to give us effects that we did not anticipate. One recent example is the interaction of climate, UV radiation and acid deposition in the boreal

forest areas of Canada. Research on lakes indicates that acidity in deposition reduces the amount of dissolved organic matter in lakes, allowing UV radiation to penetrate deeper, thus increasing exposure to potentially sensitive aquatic biota, such as phytoplankton, fish and frogs larvae (Leavitt and others 1997; Schindler and others 1996; Yan and others 1996). Increases in temperature and incidence of drought can also affect the way that lakes, streams and wetlands respond to acidification, depending on local conditions.

Another unexpected finding was based on long-term data collected at a watershed study site in Olympic National Park, at the western edge of North America. It was assumed that this site would be “unaffected” by air pollution because of the lack of identifiable “upwind” sources. A recent intensive monitoring experiment (Jaffe and others, in press) and analysis of long-term precipitation and streamwater data at the Hoh Rainforest site in Olympic National Park (Edmonds and Murray 1999) suggest that dust and industrial air pollutants are being transported in the spring from the Asian continent to North America.

In both cases, these unexpected air pollution stressor interactions were discovered after analysis of long-term monitoring and effects data not necessarily collected for this purpose. These cases, among others, point to the importance of long-term data collection at intensive sites, especially in parks and wilderness areas that are relatively protected from changes in land use and local pollution (Herrmann and Stottlemeyer 1991; Stottlemeyer and others 1998).

Scaling Up to Landscapes and Bioregions—FLMs cannot do detailed research and monitoring in all class 1 areas. They need to be able to extrapolate both deposition loading and indicator responses based on information gathered at other sites and GIS-based extrapolation techniques. There have been a number of attempts to integrate point data and process information in the form of regional assessments of the effects of air pollutant on FLM resources. These include the Southern Appalachian Assessment (SAMAB 1996), the Inner Columbia River Basin study (Haynes and others 1998), the Southern Appalachian Mountains Initiative (SAMI 1999) and the Sierra Nevada Ecosystem Project (SNEP 1996). Building on watershed-based research and monitoring (Herrmann and Stottlemeyer 1991) and network deposition estimates, and using effects models, FLMs can estimate impacts. Verification monitoring is essential to validate this use of models, such as MAGIC or NuCM. Such a GIS overlay of stressors and forest responses was used in assessing the effects of ozone on forests in the Southeast (Hogsett and others 1997). Another method of “scaling up” includes the use of remote sensing to estimate the regional distribution of resources, such as forest cover type.

Monitoring and Research Methods Appropriate for Wilderness—In keeping with the mandates of the Wilderness Act, most FLMs are reluctant to permit intrusive research and monitoring activity in parks and wilderness areas. There has been development of research and analysis methods that allow for extrapolation of monitored data collected at points outside of wilderness. Vertucci and Eilers (1993) describe a method of lake sampling that is less rigorous than the Western Lake Survey, but which does allow an FLM to “screen” potential AQRVs for sensitivity.

There has also been use of passive air quality monitors that allow for extrapolation of data collected at more sophisticated monitoring stations outside of wilderness boundaries. Most importantly perhaps, the methods of drawing deposition isopleths and using models to estimate deposition along elevation gradients hold promise for estimating both the air pollution levels and indicator responses without extensive monitoring and manipulative research. Experiments to demonstrate dose/response relationships are often conducted on lands adjoining wilderness areas, where this activity is more appropriate.

How to Estimate Total Deposition—To apply critical loads approaches to protect AQRVs, it is necessary to calculate annual deposition loads of S and N to sensitive regions. To make these estimates, FLMs need to consider how to include information on dry deposition, snow, fog and cloudwater.

Snowpack monitoring is a method of estimating the total wet and dry loads to wilderness areas that does not require active samplers (Heuer and others 2000). There is long-term snowpack monitoring at a number of Western watershed sites, but only one regional snow deposition sampling network currently in place, the Rockies Dividewide Snow Survey along the Continental Divide in Montana, Wyoming, Colorado and New Mexico, carried out by the U.S. Geological Survey, in cooperation with the USDA-FS, State of Colorado and National Park Service (Ingersoll 1995). Since the spring of 1993, researchers have collected snowpack samples during the period of maximum snow accumulation to estimate the total loading during the period of approximately October to March. Synoptic snow monitoring projects in the western U.S. have provided estimates of regional solute deposition during the winter period along the Cascade and Sierra crest and throughout the Sierra Nevada (McGurk and others 1989).

Cloudwater and fogwater can contribute significantly to total loading of solutes in some parts of the U.S. in certain types of environments. In high-elevation areas of eastern North America, cloudwater impaction can account for an equivalent amount of loading of sulfate and nitrate as other forms of wet precipitation (for example, at Noland Divide in Great Smoky Mountains National Park, (Johnson and Lindberg 1992)). Research on cloudwater deposition has included limited years of monitoring by the Mountain Cloud Chemistry network (Vong and others 1991), and the CASTNet subnetwork, which collects samples at three high-elevation sites in the east during the summer: Clingman’s Dome, Great Smoky Mountains National Park (TN/NC), Whitetop Mountain (VA) and Whiteface Mountain (NY). Measurements of cloudwater deposition in Western mountains have been confined to short-term research projects in the Sierra Nevada (Sequoia-Kings Canyon National Parks (CA)) and the Rockies (Mt. Werner (CO)). Because of the harsh monitoring environments, especially in winter, high-elevation cloudwater monitoring is not practical.

It is likely that experiments in measuring and modeling of deposition along elevational gradients in the both the East and the West will lead to methods of estimating total deposition, without the need to go to heroic lengths to measure all forms of deposition everywhere (Lovett and others 1999). Once the deposition models are developed,

they can be used to estimate “total” deposition to sensitive environments.

Use of Research and Monitoring Results by Managers

The states and the EPA are the authorities that regulate emissions of deposition precursors, NO_x and SO_x . The FLMs can advise these agencies on the need to control pollution entering wilderness areas and parks. FLMs can intervene with EPA and the states under the National Environmental Policy Act to review environmental impact statements. They can also use existing data on air pollution and its effects in the review of State Implementation Plans and in the review of new source permits under the CAAA provisions for New Source Review (NSR). FLMs can also certify impairment to visibility caused by emissions from existing sources. Of primary importance to their strategy to prevent damage to AQRVs, FLMs need to provide information and education to regulators, the general public and the media as a way of calling attention to adverse impacts in parks and wilderness areas.

FLMs are often at the forefront of alerting the public and regulators of new air pollution threats to class 1 areas. The USFS and NPS were among the first to provide information on the role of N species in degrading visibility and affecting deposition quality in the Rocky Mountains. In parts of the West, N species in deposition can be equally weighted between nitrate (NO_3) and ammonium (NH_4). Ammonia (NH_3) emitted from agricultural operations, fertilizers, industrial operations (power plants and fertilizer manufacturing facilities) and animal feedlots are likely to contribute to the overall loading of N in locations, such as the western slope of the Sierra Nevada (Blanchard and others 1996) and the eastern slope of the Rockies (Heuer and others 2000). This issue came to the attention of air managers in Colorado through the efforts of the NPS and the USFS in discussions on the effects of nitrogen loading to Rocky Mountain National Park and wilderness areas of the Front Range (Williams and Tonnessen, in press; Williams and others 1996b).

The following discussion includes a number of specific cases where FLMs, through unilateral action or as members of affected groups, have used research and monitoring data to influence regulatory actions to clean up SO_x and NO_x emissions.

Progress in Managing Deposition

Adverse Impact Determinations—FLMs routinely use deposition monitoring data and effects information in permit reviews, part of their responsibility to “prevent significant deterioration” due to new sources. In only two cases has the NPS recommended that the states or the EPA declare “adverse impact” of air pollution on resources. In the 1980s, both Shenandoah National Park (VA) and Great Smoky Mountains National Park (TN/NC) were surrounded by proposed sources requesting permits, while existing deposition was already affecting streams and soils in these two class 1 areas (Shaver and others 1994). In both cases, the states disagreed with the NPS finding of “adverse

impact,” and new source permits were granted. However, the data on effects were persuasive enough that the Southeastern states and EPA organized the Southern Appalachian Mountains Initiative (SAMI) as a forum for coming up with regional air pollution control strategies to protect class 1 areas (SAMI 1999).

The USFS also reviews new source permits for proposed emission sources near class 1 wilderness areas. The USFS has developed “screening documents” for different regions, ecosystems and regional pollutants of interest (Fox and others 1989). These reports provide guidance to the USFS on the levels of air pollutants likely to cause effects on terrestrial, aquatic and visibility resources within the national forest system. The USFS has applied the concept of level of acceptable change (LAC) to resources (Peterson and others 1992). This is similar to the concept of “adverse impact,” but sets numerical goals that serve as thresholds of damage to resources. For example, for aquatic resources in the Sierra Nevada, California, the USFS recommends that “significant deterioration” be considered likely with a long-term reduction of ANC of between 5-10 ueq/l (Peterson and others 1992). Threshold LAC values are based on an extensive literature on effects of pollutants on AQRVs.

Regional Air Quality Groups and Assessments—

Discussions of regional air pollution impacts on visibility and other AQRVs have resulted in new experiments in regional air management. This is necessary to deal with air pollution transported to remote parks and wilderness areas, such as ozone, fine particles and acidic deposition. Under the Clean Air Act Amendments of 1990, Congress provided a general mechanism for dealing with interstate pollution problems, via section 176A. This section gives the EPA Administrator authority to create interstate transport commissions, “Whenever, on the Administrator’s own motions or by petition from the Governor of any state, the Administrator has reason to believe that the interstate transport of air pollutants from one or more states contributes significantly to a violation of a national ambient air quality standard in one or more other states, the Administrator may establish, by rule, a transport region for such pollutant that includes such states” (CAAA of 1990, section 176A).

The CAAA called for the creation of the Grand Canyon Visibility Transport Commission (GCVTC), a group of eight states and four tribes concerned with air quality in the southwestern U.S. This group completed their final report to the EPA in 1996 and recommended strategies to improve visibility on the Colorado Plateau. The commission further urged the EPA to create and fund a new Western air management group, known as Western Regional Air Partnership.

Another regional air consortium recently celebrated its seventh anniversary. The Southern Appalachian Mountains Initiative (SAMI) deals with effects on AQRVs from regional air pollutants transported to the 10 class 1 areas located in the eight SAMI states. This group was charged by the EPA with coming up with a comprehensive regional strategy to deal with air pollution issues affecting resources in the 10 class 1 areas. A final integrated assessment is expected in 2001.

It is likely that this approach to regional air management, with emphasis on class 1 areas, will continue and expand as we look for options to protect AQRVs in these

areas, especially in light of the new regional haze rules promulgated by EPA to protect class 1 visibility. EPA is planning to fund two to four additional regional air management partnerships to help plan for restoration of natural background visibility throughout the U.S.

Controls on Existing Power Plants—Since the “visibility goal” was endorsed by Congress in the CAAA of 1977, there have been a number of attempts by FLMs to get Best Available Retrofit Technology (BART) on uncontrolled power plants. Special “attribution” studies were performed to link SO₂ and NO_x emissions to visibility impairment and other adverse impacts on AQRVs at the following parks and wilderness areas: Grand Canyon and Canyonlands National Parks affected by the Navaho Generating Station (National Research Council 1993); Mount Rainier Wilderness and Alpine Lakes Wilderness affected by Centralia Power Plant, Washington; Grand Canyon National Park and Glen Canyon National Recreation Area being affected by the Mohave Power Plant, Nevada (Green 1999); and the Mt. Zirkel Wilderness being affected by the Craig and Hayden Power plants, Colorado (Jackson and others 1996). In each case, with the exception of the Mohave Power Plant, which is still in negotiations, there was agreement to control emissions. However, in all of these cases, the effect of S and N deposition on ecological resources was not the deciding factor in the clean-up decision. However, there is no question that ecological systems in these affected parks and wilderness areas will benefit from the emission reductions.

What Managers Need

Long-Term Monitoring and Index Sites—The best way for FLMs to develop long-term databases on stressors and ecosystem responses is to participate in interagency programs that allow for leveraging of resources. The FLMs have the advantage of managing relatively “unaffected” sites where monitoring programs can operate without local disturbance or likely change in land use. FLMs should think of the parks, refuges and forest lands as “outdoor laboratories,” where research and monitoring can be supported, often with in-kind services. There are existing networks of long-term environmental monitoring sites, many located adjacent to wilderness areas on national forests, parks and wildlife refuges.

The USFS has a network of experimental forests including, Fernow Experimental Forest (WV), Hubbard Brook Experimental Forest (NH), Fraser Experimental Forest (CO), H.J. Andrews Experimental Forest (OR) and Coweeta Experimental Forest (GA). The USFS is also host to a number of Long-Term Ecological Research sites (LTER), funded by the National Science Foundation. LTER sites include experimental forests, the Pawnee Grasslands and Niwot Ridge (CO). At these sites, outside of wilderness, extensive monitoring and research manipulations can be carried out, producing data that can be applied to wilderness area resources (Adams and others 1997).

In the coastal zone, there are a number of research sites maintained by NOAA and EPA, including the recently organized Coastal Index Site Network (CISNET) and the National Estuary Program (NEP). NPS units serve as sites for a number of long-term networks and index site networks,

including the Prototype Parks Monitoring Program (NPS Inventory and Monitoring program funding); the small watersheds program (USGS funding) (Herrmann and Stottlemeyer 1991); the Water, Energy, and Biogeochemical Budgets program (USGS); and the Park Research and Intensive Monitoring of Ecosystems Network (PRIMENet) (EPA and NPS funding) (Summers and Tonnessen 1998).

PRIMENet is the first project to be jointly funded by the EPA and the National Park Service to address the linkages between environmental stressors and ecosystem responses. PRIMENet is designed to monitor major environmental stressors, such as UV, air pollution, contaminants and climate and to relate changes in these stressors to ecological indicators at 14 parks, representing a range of ecosystems (Figure 7).

AQRV Inventories and Monitoring—FLMs are acutely aware that they do not have a good inventory of the natural resources on the lands they manage. The National Park Service realized that it was not carrying out the mandate to identify and then monitor the condition of their resources, as directed by Congress. The scope of the problem was laid out in articles by Stohlgren and others (1995); and Stohlgren and Quinn (1992). This realization led to funding of a long-term ecological monitoring program in the National Park Service, with the centerpiece of the program being a network of 22 “prototype parks” that develop monitoring protocols, in cooperation with the USGS, and then transfer these methods to other parks with similar ecosystems and landscape classification. A complement to the prototype parks program is funding for a comprehensive set of natural resource inventories for the more than 260 park units with resource concerns and issues. Resources to be inventoried are Air Quality-Related Values or sensitive indicators of air pollution.

Development and listing of sensitive AQRVs in class 1 areas is one of the tasks taken on by the Federal Land Managers Air Quality-Related Values Work Group (FLAG 1999). This effort by the USFS, FWS and NPS will continue into Phase 2 of the program.

Dose/Response Information—There are now well-developed methods and models to determine the amount of deposition needed to change surface-water chemistry. There are also a limited number of biological populations that have been tested for response to acidification in lakes and streams. Linking these different types of models was used to assess fish viability in Southeastern streams, using field data from the Shenandoah Watershed Study and the Virginia Trout Stream Sensitivity Study (Bulger and others 1998). The MAGIC model was used in this assessment to forecast the effects of different deposition scenarios on surface water quality. It is currently being modified and tested at watershed sites in parks and wilderness areas in the Rockies, the Sierra Nevada and the Cascades, with the expectation that this model will be used for regional assessment of class 1 areas in the West.

Dose/response data are harder to come by for terrestrial effects of deposition. The Nutrient Cycling Model (NuCM) was developed in the eastern U.S. to forecast the change in soil water chemistry, and indirectly to assess forest health, with different loadings of N and S. It was used in the

Southeast in the spruce-fir zone, the terrestrial system most under stress from deposition (Johnson and others 1996).

Biota that show responses in the lab and the field to acidification include zooplankton, stream invertebrates, fish and amphibians. Experiments to determine biological response to chronic or episodic acidification have been conducted in parks and national forests, including work at Emerald Lake in Sequoia-Kings Canyon National Park (CARB 1993). Zooplankton species, such as *Daphnia rosea*, were found at reduced levels in lakes that have experienced acidification (Barmuta and others 1990). Among the benthic invertebrates found in streams, the mayfly larva, *Baetis* spp., are adversely affected by acidic episodes (Kratz and others 1994). These two species are important as food items for native fish in high-elevation aquatic systems. Native fish species, such as cutthroat trout and rainbow trout, can be sensitive to acidic waters, depending on the life stage exposed to acidic episodes. In general, fish population viability is expected to be reduced below pH 6 (Baker and others 1990). Two amphibian studies conducted in the Rockies point to the direct effects on salamander eggs of acid episodes and possible community level responses of competing amphibian species. Harte and Hoffman (1989) exposed the eggs of tiger salamanders (*Ambystoma tigrinum*) to experimental increases in episodic acidification and determined that they had an LD-50 pH of 5.6, which is within the range of snowmelt pHs encountered in the Rocky Mountains. In a study of coexisting populations of tiger salamanders and chorus frogs (*Pseudacris triseriata*), Kiesecker (1996) reports that changes in development rates in these larvae can be effected by depressed pHs in pond water, leading to changes in predatory success.

It is not always practical for FLMs to conduct these costly experiments in all sensitive systems. The development of empirical models, along with the transfer of these models to similar ecosystems, is the only feasible method to allow for estimation of loss of biota with increasing acidification, although this approach is accompanied by uncertainty.

Critical Loads, Critical Levels, and Thresholds—

Existing standards for air pollution (NAAQS) will not work to protect ecosystems and biota from deposition. Ambient concentrations in air do not translate well into deposition of N and S to watersheds, forests and estuaries. Therefore, we need to use the “critical loads” approach that has been developed by the Europeans and the Canadians. NAAQS may be considered as “critical levels” of pollutants that may affect human health or natural resources (Bull 1991). However, this kind of ambient gaseous or particle standard is aimed toward inhalation risk, rather than the risk of pollutant deposition effects on resources (such as N deposition effects on surface waters). It is likely that a more stringent PM-2.5 standard will reduce the load of particles transported into high-elevation regions and then deposited to sensitive resources via wet or dry deposition. However, this is not one of the major pathways considered in setting these standards.

Critical loads are deposition levels above which natural resources can be negatively affected. The European countries and Canada have been at the forefront of setting critical loads and target loads to protect their forests, soils, lakes and streams from deposition of S and N. The difference between these two levels can be explained in the policy

context. Critical loads are the levels below which no effect to sensitive resources is expected; target loads are the amount of deposition that will result in an “acceptable level” of damage to resources. For example, Dise and Wright (1995) calculate that below a deposition rate of 10 kg/ha/yr of N, no significant nitrogen leaching should occur in European forests; above 25 kg/ha/yr, there is significant leaching. Therefore, 10 kg/ha/yr would be the critical load, and some value between that and 25 kg/ha/yr could be chosen at the target load. It should be noted that existing critical load values are site specific and based on intensive site investigations.

In the U.S., the CAAA of 1990, section 404, called for the EPA to prepare a report on the feasibility and the environmental effectiveness of setting an acid deposition standard to protective sensitive aquatic and terrestrial resources. The completed report includes a number of modeling analyses that project the effect of reductions in both S and N deposition in areas well studied during the National Acid Precipitation Assessment Program (U.S. EPA 1995). The conclusions of the EPA’s analysis are that: (1) the uncertainties associated with effects of nitrogen on ecosystems are such that critical loads cannot be set at this time; (2) there had been no policy decision regarding the level of acceptable damage to systems; and (3) any critical load standards would have to be set on a regional basis and then enforced with regional pollution abatement strategies.

Some states have taken the lead in addressing transport and deposition of secondary pollutants, such as acid deposition and ozone. Minnesota is the only state that currently has an air quality standard to protect sensitive lakes from acid deposition. This state set a limit on total annual sulfate deposition of 11 kg/ha/year in order to keep the pH of precipitation above 4.7 to protect sensitive lakes (Orr and others 1991, 1992). The California legislature passed a statute called the “Atmospheric Acidity Protection Act,” which called for a program of research and monitoring of acid deposition and atmospheric acidity in both urban and rural areas, with an assessment requirement (CARB 1995). The California Air Resources Board was to determine the need for an atmospheric acidity standard to protect both human health and natural systems. That determination has not been made to date.

Ability to Scale Up to Regional Systems—For a number of natural resource management issues, assessments need to be done on a bioregional basis because of the often contiguous management by different state, federal and private organizations. Air management is an issue that requires regional assessments. In the southeastern U.S., the FLMs collaborated, under the auspices of the Southern Appalachian Man in the Biosphere, to determine the condition of federal land resources in the Southeast. This Southern Appalachian Assessment included a report on regional air pollutants and their effects on class 1 resources (SAMAB 1996). It made use of GIS tools and extrapolation techniques for estimating the distribution of air pollutants, such as ozone and deposition, over the landscape. The Southern Appalachian Mountains Initiative (SAMI 1999) is performing a more detailed modeling and assessment exercise for the Southern Appalachian mountain regions, with a particular focus on eight states and the 10 class 1 areas within its boundaries: West Virginia: Dolly Sods and Otter Creek Wilderness Areas; Virginia: Shenandoah National Park and

James River Face Wilderness Areas; North Carolina: Shining Rock and Linville Gorge Wilderness Areas; North Carolina and Tennessee: Great Smoky Mountains National Park and Joyce Kilmer-Slickrock Wilderness Area; Georgia: Cohutta Wilderness Areas; Alabama: Sispey Wilderness Area; and the states of Kentucky and South Carolina, which have no class 1 areas located within the Southern Appalachian bioregion. The stressors of interest in this assessment include fine particles, ozone and acid deposition. An integrated modeling approach is being pursued, with final assessments of emission management options due in 2001.

There have been two bioregional science assessments performed in the West at the request of Congress: the Inner Columbia River Basin assessment (Haynes and others 1998) and the Sierra Nevada Ecosystem Project (SNEP 1996). The objective of these assessments was to determine the current status of natural resources and the trends in their condition, and to propose alternative strategies for land and resource management, using existing data organized with geographic information systems. The goal of the assessments was to balance the social and economic needs of the regions with the need to preserve ecosystem integrity on private, state and federal lands managed by the USFS, the NPS, FWS and the Bureau of Land Management. Both of these ecosystem assessments included an analysis of regional and local air pollution and the effects of these stressors on natural resources. Schoettle and others (1999) present an analysis of air resources for the ICRB region, which includes all or part of the states of Montana, Wyoming, Idaho, Washington, Oregon and a small slice of California and Nevada. The SNEP (1996) evaluated the Sierran bioregion, including parts of California and Nevada. The air issues identified as important in the Sierra Nevada included ozone injury to native tree species and the impairment of visibility due to fine particles from fossil fuel combustion, windblown dust, forest fires and residential wood burning. Although declines were noted in some fauna, such as the mountain yellow-legged frog, the array of stresses leading to loss of biodiversity in the Sierra is the subject of additional research and long-term monitoring.

In the Northern Hemisphere, there are three organizations that have performed assessments of transboundary air pollution. These include the trilateral Commission on Environmental Cooperation (CEC 1998), the International Air Quality Advisory Board of the International Joint Commission (IAQAB 1998) and the U.S./Canada Air Quality Committee. These groups have shown different levels of interest in class 1 area issues related to air pollution in the border region. The U.S./Canada Air Quality Committee, created under the U.S./Canada Air Quality Agreement to control acid rain in the two countries, is also active in planning for control of the other transboundary pollutants, such as fine particles and ozone, and submits biennial progress reports to the governments (U.S./Canada Air Quality Committee 1998).

Integrated Modeling—It is becoming more important to link models to allow regulators to forecast the results of emissions reductions on resources. To do “scenario testing,” it is necessary to follow a proposed change in emissions as it translates into changes in deposition and then determine effects on ecosystems, visibility, human health and socioeconomic factors.

There have been a number of efforts to do this model linkage, including the Canadian’s use of RAISON model (Lam and others 1998), the NAPAP effort to develop and use the TAF (Tracking and Analysis Framework) (NAPAP 1998) and SAMI’s development and testing of their own series of models, including MAGIC and NuCM for assessing effects on water chemistry and soil and forest health (SAMI 1999). Another step in ecological modeling has been used in an assessment of deposition effects on fish in the streams of the Virginia mountains. After “scenario testing” using MAGIC (Cosby and others 1985; Sullivan and Cosby 1995), Bulger and others (1998) linked projected water chemistry changes to changes in fish population status using an empirical model. A survey of other fish response models is available in Baker and others 1990. This is an important step—to link the water chemistry variables that we can measure in the field with a biological response that the public and land managers care about.

Information Management and Data Display Tools—

With the vast array of data and information available for class 1 parks and wilderness areas, there is now a need for computer-based methods to organize, access and synthesize these data sets. The NPS, FWS and USFS all have projects ongoing to organize air monitoring and effects data. The NPS and FWS data management system is called AQUIMS (Air Quality Information Management System) (Nash and others 1996). All class 1 areas managed by the NPS and FWS are listed in the database, along with natural resource and air quality information. AQUIMS also includes annotated bibliographies on deposition and ozone. AQUIMS is now incorporated into a larger NPS data management system, known as SYNTHESIS. The USFS-Air Resource Management Program is developing an Air Module (NRIS-AIR) to link to the Natural Resource Information System. This system will incorporate a broad array of data collected by the USFS and cooperators for assessing air pollution effects on resources in national forests and grasslands.

Geographic information systems are useful tools for managers to access and organize these data. This tool has been used extensively in bioregional assessments, including the Southern Appalachian Assessment (SAMAB 1996), the Sierra Nevada Ecosystem Project (1996) and Inner Columbia River Basin Assessment (Haynes and others 1998).

Decision Support and Expert Systems—It is not enough to organize and display data on air pollution levels and indicator responses. FLMs need interpretation and “expert” judgment to understand how pollutants may be injuring resources. Decision-support systems, or “expert systems,” provide this type of interpretation of data. The NPS and FWS are developing an interactive expert systems module in AQUIMS to interpret deposition and ozone effects information (Nash and others 1997). The deposition module, developed with input from a team of experts on aquatic and terrestrial effects, will allow FLMs to input existing surface water quality data for lakes and streams to determine: (1) current acidification status, (2) likely cause of high concentrations of acid anions (SO_4 or NO_3), (3) sensitivity of waters to increases in N or S deposition, and (4) display the results on a GIS that color-codes acidification states of fresh water in the class 1 area.

FLM Strategies

FLMs alone cannot control air pollution transported to parks or wilderness areas. They are required to work with a myriad of interests outside their borders to control local, regional, hemispheric and global air pollution. Some possible options for FLMs to join the larger community of “stakeholders” to protect class 1 resources and scenic values include:

1. Participating in regional air assessment groups and partnerships (such as WRAP, SAMI).
2. Alerting the public to resource threats through education and interpretive programs and “leading by example” in cleaning up sources of air pollution with the park or wilderness area.
3. Advising regulators on levels of air pollution that can affect sensitive AQRVs, e.g., NAAQS, critical loads and levels and threshold of injury.

Table 2—Air quality websites.

National Park Service, Air Resource Division
http://www.nature.nps.gov/ard
PRIMENet
http://www.forestry.umn.edu/primenet
USFWS/Air Quality Branch
http://www2.nature.nps.gov/ard/fws/fwsaqb.htm
USDA-FS, National Air Resource Management
http://www.fs.fed.us/r6/air/natarm
Environmental Protection Agency, Office of Air
http://www.epa.gov/oar
Regional Haze Rules, EPA-OAQPS
http://www.epa.gov/ttn/oarpg
Western Regional Air Partnership (WRAP)
http://www.wrapair.org
Environmental Protection Agency, Deposition to estuaries
http://www.epa.gov/owow/oceans/airdep
US Geological Survey, Acid Rain Program
http://bqs.usgs.gov/acidrain
NTN/NADP Program
http://nadp.sws.uiuc
Mercury Deposition Network
http://nadp.sws.uiuc/mdn
CASTNet Program
http://www.epa.gov/ardpublic/acidrain/castnet
IMPROVE Program (optical data)
ftp://alta_vista.cira.colostate.edu
NAPAP Assessment
http://www.nncc.noaa.gov/CENR/NAPAP/NAPAP_96.htm
U Georgia National UV Monitoring Center
http://oz.phyast.uga.edu
EPA UV Monitoring Site
http://www.epa.gov/uvnet
Interagency UV Monitoring Site
http://www.arl.noaa.gov/research/programs/uv.html

4. Attracting research and monitoring funds and good research groups to conduct targeted studies in class 1 areas, or similar reserves where data can be extrapolated.

5. Assisting in the collection of air quality and AQRV data to protect resources from transported fine particles, ozone and deposition; this includes providing scientific infrastructure and access for research groups to research sites in parks, wildernesses, or adjacent lands.

6. Making information available to the public via websites (Table 2).

References

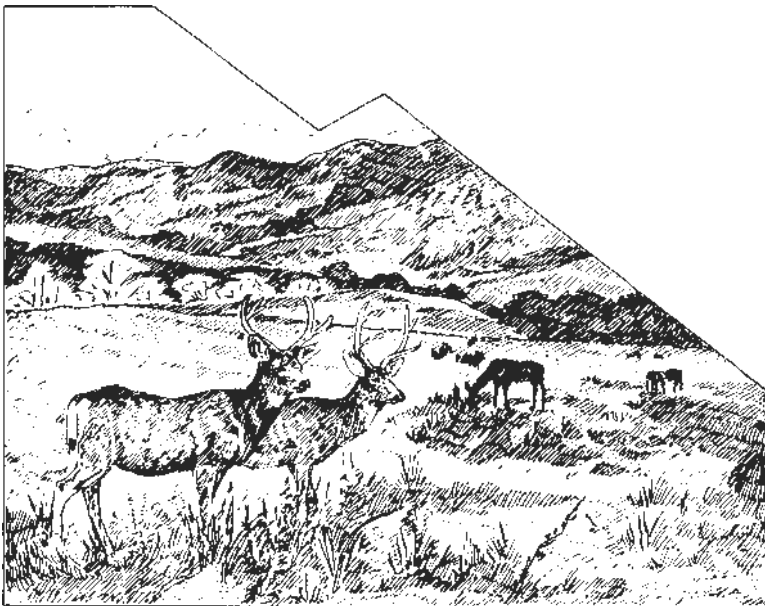
- Aber, J.D.; Nadelhoffer, K.J.; Steudler, P.; Melillo, J.M. 1989. Nitrogen saturation in northern forest ecosystems: excess nitrogen from fossil fuel combustion may stress the biosphere. *BioScience* 39: 378-386.
- Adams, M.B.; Angradi, T.R.; Kochenderfer, J.N. 1997. Stream waters and soil solution responses to 5 years of nitrogen and sulfur addition at the Fernow Experimental Forest, West Virginia. *Forest Ecology and Management* 95: 79-91.
- Adams, M.B.; Nichols, D.S.; Federer, C.A.; Jensen, K.F.; Parrott, H. 1991. Screening Procedure to Evaluate Effects of Air Pollution on Eastern Region Wildernesses Cited as Class I Air Quality Areas. General Technical Report NE-151. U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station, Radnor, Pennsylvania.
- Barmuta, L.; Hamilton, S.; Cooper, S.; Kratz, K.; Melack, J. 1990. Responses of zooplankton and zoobenthos to experimental acidification in a high-elevation lake (Sierra Nevada, California, U.S.A.) *Freshw. Biol.* 23, 571-586.
- Bales, R.; Davis, R.; Williams, M. 1993. Tracer release in melting snow—diurnal and seasonal patterns. *Hydrological Processes* 7: 389-401.
- Baker, J.P.; Bernard, D.P.; Christensen, S.W.; and others. 1990. Biological effects of changes in surface water acid-base chemistry, NAPAP Report 13. In: National Acid Precitation Assessment Program, Acidic Deposition. State of Science and Technology Vol. II. 381 p.
- Barnard, J.; Lucier, J. 1990. Changes in forest health and productivity in the United States and Canada. State of Science and Technology Report 16. National Acid Precitation Assessment Program, Washington, D.C.
- Binkley, D.; Giardina, C.; Dockersmith, I.; Morse, D.; Scruggs, M.; Tonnessen, K. 1997. Status of air quality and related values in Class I National Parks and Monuments of the Colorado Plateau. U.S. Department of the Interior, National Park Service, Air Resources Division, Denver, Colorado.
- Blais, J.M.; Schindler, D.W.; Muir, D.; Kimpe, L.; Donald, D.; Rosenberg, B. 1998. Accumulation of persistent organochlorine compounds in mountains of western Canada. *Nature* 395: 585-588.
- Blanchard, C.; Michaels, H.; Tanenbaum, S. 1996. Regional estimates of acid deposition fluxes in California for 1985-1994. Final Report. Contract 93-332. California Air Resources Board, Sacramento, CA.
- Blanchard, C.; Tonnessen, K.A. 1993. Precipitation-chemistry measurements from the California Acid Deposition Monitoring Program, 1985-1990. *Atmospheric Environment* 27A: 1755-1763.
- Bradford, D.F.; Swanson, C.; Gordon, M. 1994. Effects of low pH and aluminum on amphibians at high elevation in the Sierra Nevada, California. *Can. J. Zool.* 72: 1272-1279.
- Brooks, P.; Williams, M.; Schmidt, S. 1996. Microbial activity under alpine snowpacks, Niwot Ridge, Colorado. *Biogeochemistry* 32: 93-113.
- Brosset, C.; Lord, E. 1991. Mercury in precipitation and ambient air: a new scenario. *Water, Air and Soil Pollution* 56: 493-506.
- Bulger, A.; Cosby, J.; Webb, R. 1998. Acid Rain: Current and projected status of coldwater fish communities in the Southeastern U.S. In: The context of continued acid deposition. Trout Unlimited, Arlington, VA. 28 p.
- Bull, K.R. 1991. The critical loads/levels approach to gaseous pollutant emission control. *Environ. Pollut.* 69: 105-123.

- Bunyak, J. 1993. Permit application guidance for new pollution sources. Natural Resources Report NPS/NRAQD/NRR-93. U.S. Department of the Interior, National Park Service, Air Quality Division, Denver, Colorado.
- California Air Resources Board (CARB). 1993. The Atmospheric Acidity Protection Program: Annual report to the Governor and Legislature, 1992. Sacramento, California.
- Campbell, D.H.; Clow, D.W.; Ingersoll, G.P.; Mast, M.A.; Spahr, N.E.; Turk, J.T. 1995. Processes controlling the chemistry of two snowmelt-dominated streams in the Rocky Mountains. *Water Resources Research* 31(11): 2811-2821.
- Chappelka, A.H.; Samuelson, L.J. 1998. Ambient ozone effects on forest trees of the eastern United States. *New Phytologist* 139: 91-108.
- Charles, D. (ed). 1991. Acid deposition and aquatic ecosystems, Regional case studies. Springer-Verlag, New York.
- Colborn, T.; Clement, C. (eds). 1992. Chemically-induced alteration in sexual and functional development: the human-wildlife connection. Princeton, NJ: Princeton Scientific Publishing Co., Inc.
- Commission for Environmental Cooperation (CEC). 1998. Continental pollutant pathways: an agenda for cooperation to address long-range transport of air pollution in North America. Commission for Environmental Cooperation, Montreal, Canada.
- Committee on Environment and Natural Resources (CENR). 1997. Integrating the Nation's environmental monitoring and research networks and programs: a proposed framework. National Research and Technology Council, Washington, DC.
- Cosby, B.J.; Norton, S.A.; Kahl, J.S. 1996. Using a paired-watershed manipulation experiment to evaluate a catchment-scale biogeochemical model. *Science of the Total Environment* 183: 49-66.
- Cosby, B.J.; Wright, R.F.; Gjessing, E. 1995b. An acidification model (MAGIC) with organic acid evaluated using whole-catchment manipulations in Norway. *Journal of Hydrology* 170: 101-122.
- Dise, N.; Wright, R.F. 1995. Nitrogen leaching from European forests in relation to nitrogen deposition. *Forest Ecology and Management* 71: 153-161.
- Dixon, L.K.; Esteves, E.D. 1998. Chassahowitzka National Wildlife Refuge—Status and Trends. In: NADP Technical Committee Meeting, Abstracts of Papers, St. Petersburg, FL: 55.
- Eager, C.; Adams, M.B.(eds). 1992. Ecology and decline of red spruce in the Eastern United States. Springer-Verlag, New York. 417 p.
- Edmonds, R.L.; Murray, G.L.D. 1999. Suspected asian dust influences stream chemistry in an old-growth ecosystem in the Hoh River Valley. (Abstract) Ninth Annual Review, Center for Streamside Studies, University of Washington, Seattle.
- Eilers, J. M.; Rose, C.L.; Sullivan, T.J. 1994. Status of air quality and effects of atmospheric pollutants on ecosystems in the Pacific Northwest Region of the National Park Service. Technical Report NPS/NRAQD/NRTR-94/160. U.S. Department of the Interior, National Park Service, Air Quality Division. Denver, Colorado.
- Eilers, J.M.; Brakke, D.F.; Landers, D.H.; and Kellar, P.E.; 1988. Characteristics of lakes in mountainous areas of the western United States. *Verh. Int. Verein. Limnol.* 23: 144-151.
- Elder, K.; Dozier, J.; Michaelsen, J. 1991. Snow accumulation and distribution in an alpine watershed. *Water Resources Research* 27: 1541-1552.
- Eldred, R.A.; Cahill, T.A. 1994. Trends in elemental concentrations of fine particles at remote sites in the United States of America. *Atmospheric Environment* 28 (5): 1009-1019.
- Engle, D.; Melack, J. 1995. Zooplankton of high elevation lakes of the Sierra Nevada, California: potential effects of chronic and episodic acidification. *Arch. Hydrobiol.* 133: 1-21.
- Environment Canada, 1998. 1997 Canadian acid rain assessment, Volume 1, Summary of results. Ministry of the Environment, Ottawa, Canada.
- Federal Land Managers' Air Quality Related Values Work Group (FLAG). 1999. Phase 1 Report, (Draft), Lakewood, CO.
- Fenn, M.; Poth, M.; Aber, J.; Baron, J.; Bermann, B.; Johnson, D.; Lemly, A.; McNulty, S.; Ryan, D.; Stottlemeyer, R. 1998. Nitrogen excess in North American ecosystems: a review of geographic extent, predisposing factors, ecosystem responses, and management strategies. *Ecological Applications* 8: 706-733.
- Fenn, M.E.; Poth, M.A.; Johnson, D.W. 1996. Evidence for nitrogen saturation in the San Bernardino Mountains in southern California. *Forest Ecology and Management* 82: 211-230.
- Fox, D.G., A.M. Bartuska, J.G. Byrne, E. Cowling, R. Fisher, G.E. Likens, S.E. Lindberg, R.A. Linthurst, J. Messer and D.S. Nichols. 1989. A screening procedure to evaluate air pollution effects on Class I Wilderness Areas. General Technical Report RM-168. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Ft. Collins, Colorado.
- Grand Canyon Visibility Transport Commission. 1996. Recommendations for improving western vistas. Report of the Grand Canyon Visibility Transport Commission to the U.S. Environmental Protection Agency. 91 p.
- Green, M. 1999. The project MOHAVE tracer study: study design, data quality, and overview of results. *Atmospheric Environment* 33: 1955-1968.
- Groisman, P.Y.; Karl, T.R.; Knight, R.W. 1994. Changes of snow cover, temperature and radiative heat balance over the Northern Hemisphere. *Journal of Climate* 7: 1633-1656.
- Hacker, W.D.; Neufeld, H.S. 1993. The false positive in bioindicators of air pollution. *J. of Arboriculture* 18: 249-250.
- Haddow, D.; Musselman, R.; Blett, T.; Fisher, R. 1998. Guidelines for evaluating air pollution impacts on wilderness within the Rocky Mountain Region: Report of a Workshop, 1990. General Technical Report RMRS-GTR-4. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Ft. Collins, Colorado.
- Harte, J.; Hoffman, E. 1989. Possible effects of acid deposition on a Rocky Mountain population of the tiger salamander, *Ambystoma tigrinum*. *Conservation Biology* 3: 149-158.
- Haynes, R.W.; Graham, R.T.; Quigley, T.M. 1998. A framework for ecosystem management in the Interior Columbia Basin. *Journal of Forestry* 96 (10): 4-9.
- Hedin, L.O.; Likens, G.E. 1996. Atmospheric dust and acid rain. *Scientific American* 275(6): 88-92.
- Hedin, L.O.; Granat, L.; Likens, G.E.; Buishand, T.A.; Galloway, J.N.; Butler, T.J.; Rohde, H. 1994. Steep declines in atmospheric base cations in regions of Europe and North America. *Nature* 367: 351-354.
- Herrmann, R.; Stottlemeyer, R. 1991. Long-term monitoring for environmental change in U.S. National Parks: a watershed approach. *Environmental Monitoring and Assessment* 17: 51-65.
- Heuer, K.; Tonnessen, K.; Ingersoll, G. 2000. Comparison of precipitation chemistry in the Central Rocky Mountains, Colorado, USA. *Atmospheric Environment* 34: 1713-1722.
- Hogsett, W.E.; Weber, J.E.; Tingey, D.; Herstrom, A.; Lee, E.H.; Laurence, J.A.. 1997. An approach for characterizing tropospheric ozone risk to forests. *Environmental Management* 21 (1): 105-120.
- Ingersoll, G. 1995. Maximum-accumulation snowpack chemistry at selected sites in northwestern Colorado during spring 1994. U.S. Geological Survey, Open File Report 95-139, Denver, CO.
- International Air Quality Advisory Board (IAQAB). 1998. Special Report on Transboundary Air Quality Issues. International Joint Commission, Windsor, Ontario.
- Jaffe, D.; Anderson, T.; Covert, D.; Kotchenruther, R.; Trost, B.; Danielson, J.; Simpson, W.; Berntsen, T.; Karlsdottir, S.; Blake, D.; Harris, J.; Carmichael, G.; Uno, I. Transport of asian air pollution to North America. *Geophysical Research Letters* (in press).
- Jackson, L.; Geiser, L.; Blett, T.; Gries, C.; Haddow, D. 1996. Biogeochemistry of lichens and mosses in and near Mt. Zirkel Wilderness, Routt National Forest, Colorado: influences of coal-fired power plant emissions. Open File Report 96-295. U.S. Department of the Interior, U.S. Geological Survey.
- Johnson, D.W.; Susfalk, R.B.; Dahlgren, R.A.; Klopatek, J.M. 1998. Fire is more important than water for nitrogen fluxes in semi-arid forests. *Environmental Science and Policy* 1: 79-86.
- Johnson, D.W.; Susfalk, R.B.; Brewer, P.F. 1996. Simulated responses of red spruce forest soils to reduced sulfur and nitrogen deposition. *Journal of Environmental Quality* 25: 1300-1309.
- Johnson, D.W.; Lindberg, S.E.(eds). 1992. Atmospheric deposition and forest nutrient cycling: a synthesis of the integrated forest study. Springer-Verlag, New York. 707 p.

- Johnson, D.W.; van Miegroet, H.; Lindberg, S.E.; Todd, D.E.; Harrison, R.B. 1991. Nutrient cycling in red spruce forests of the Great Smoky Mountains. *Can. J. For. Res.* 21: 769-787.
- Kendall, C.; Campbell, D.H.; Burns, D.A.; Shanley, J.B.; Silva, S.R.; Chang, C.C.Y. 1995. Tracing sources of nitrate in snowmelt runoff using the oxygen and nitrogen isotopic compositions of nitrate. In: Tonnessen, K.A.; Williams, M.W.; Tranter, M. (eds), *Biogeochemistry of seasonally-covered catchments*. IAHS Publ. No. 228: 339-347.
- Kiesecker, J. 1996. pH-mediated predator-prey interactions between *Ambystoma tigrinum* and *Pseudacris triseriata*. *Ecological Applications* 6 (4): 1325-1331.
- Kratz, K.; Cooper, S.; Melack, J. 1994. Effects of single and repeated experimental acid pulses on invertebrates in a high altitude Sierra Nevada stream. *Freshwater Biology* 32: 161-183.
- Lam, D.C.L.; Puckett, K.J.; Wong, I.; Moran, M.D.; Fenech, G.; Jeffries, D.S.; Olson, M.P.; Whelpdale, D.M.; McNicol, D.; Marian, Y.K.G.; Minns, C.K. 1998. An integrated acid rain assessment model for Canada: from source mission to ecological impact. *Water Qual. Res. J. Canada* 33: 1-17.
- Landers, D.; Gubala, C.; Verta, M.; Lucotte, M.; Johansson, K.; Vlasova; and Lockhart, W.L. 1998. Using lakes sediment mercury flux ratios to evaluate the regional and continental dimensions of mercury deposition in arctic and boreal ecosystems. *Atmospheric Environment* 32 (5): 919-928.
- Landers, D.; Eilers, J.; Brakke, D.; Overton, W.; Kellar, P.; Silverstein, P.; Schonbrod, R.; Crowe, R.; Linthurst, R.; Omernik, J.; Teague, S.; Meier, E. 1987. Characteristics of lakes in the western United States, volume I, population descriptions and physio-chemical relationships. EPA/600/3-86/54a. U.S. Environmental Protection Agency, Washington, D.C. 176 p.
- Lawrence, G.; Huntington, T. 1999. Soil-calcium depletion linked to acid rain and forest growth in the eastern United States. USGS Report WRIR 98-4267. February 1999.
- Lear, G.; Frank, N. 1998. The long, dry road to total deposition: discovering EPA's Clean Air Status and Trends Network. In: NADP Technical Committee Meeting, Abstracts of Papers, St. Petersburg, FL: 19.
- Lefohn, A.; Shadwich, D.S.; Somerville, M.C. [and others]. 1992. The characterization and comparison of ozone exposure indices used in assessing the response of loblolly pine to ozone. *Atmospheric Environment* 26A: 287-298.
- Leavitt, P.R.; Vinebrooke, R.D.; Donald, D.B.; Smol, J.P.; Schindler, D.W. 1997. Past ultraviolet radiation in lakes derived from fossil pigments. *Nature* 388: 457-459.
- Lovett, G.M.; Thompson, A.W.; Anderson, J.B.; Bowser, J.J. 1999. Elevational patterns of sulfur deposition at a site in the Catskill Mountains, New York. *Atmospheric Environment* 33: 617-624.
- Lynch, J.; Bowersox, V.; Grimm, J. 1996. Trends in Precipitation Chemistry in the United States, 1983-94: an Analysis of the effects in 1995 of Phase I of the Clean Air Act Amendments of 1990, Title IV. Open-File Report 96-0346. U.S. Department of the Interior, U.S. Geological Survey.
- Lynch, J.; Bowersox, V.; Grimm, J. 1995. Trends in precipitation chemistry in the United States: a national perspective, 1980-1992. *Atmospheric Environment* 29: 1231-1246.
- Madronich, S. 1993. The atmosphere and UV-B radiation at ground level. In: *Environmental UV Photobiology*. Plenum Press, NY. 1993.
- Malm, W. 1992. Characteristics and origins of haze in the continental United States. *Earth Science Reviews* 33: 1-36.
- McGurk, B.; Berg, N.; Marks, D.; Melack, J.; Setaro, F. 1989. Monitoring atmospheric deposition in California's Sierra Nevada: a comparison of methods. *International Assoc. Hydrological Sciences, Publ #179*: 71-79.
- Melack, J.; Sickman, J. 1995. Snowmelt-induced chemical changes in seven streams in the Sierra Nevada. In: Tonnessen, K.A.; Williams, M.W.; Tranter, M. (eds), *Biogeochemistry of seasonally snow-covered basins*. IAHS Publication 228. International Association of Hydrological Sciences, Wallingford, UK, 1995: 221-234.
- Miller, P.R.; Stolte, K.W.; Duriscoe, D.M.; Pronos, J. 1996. Evaluating ozone air pollution effects on pines in the western United States. General Technical Report PSW-GTR-155. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. Albany, California.
- Mitchell, M.; Driscoll, C.T.; Kahl, J.S.; Likens, G.E.; Murdoch, P.S.; Pardo, L.H. 1996. Climatic control of nitrate loss from forested watersheds in the northeast United States. *Environmental Science and Technology* 30: 2609-2612.
- Myneni, R.B.; Keeling, C.D.; Tucker, C.J.; Ashrar, G.; Nemani, R.R. 1997. Increased plant growth in the northern high latitudes from 1981-1991. *Nature* 386: 698-702.
- Nash, B.L.; Tonnessen, K.A.; Heuer, K.H.; Davis, D.D.; Saunders, M.C.; Miller, B.J. 1997. Using the AQUIMS air quality decision support system to identify natural resources at risk from air pollution. *Proc. of the Amer. Society for Photogrammetry and Remote Sensing*, Seattle, WA: 362-369.
- Nash, B.L.; Saunders, M.C.; Miller, B.J.; Tonnessen, K.A.; Davis, D.D.; and others. 1996. AQUIMS: a computerized system for interpreting and integrating information on air pollution effects on natural resources. *Proc. of Eco-Informa '96*; Lake Buena Vista, FL: 575-580.
- National Atmospheric Deposition Program (NADP). 1999. NADP: 1997 wet deposition maps. Illinois State Water Survey, Champaign, IL.
- National Acid Precipitation Assessment Program (NAPAP). 1991. Acid deposition: state of science and technology. Summary Report of the U.S. NAPAP. Washington, D.C.
- National Acid Precipitation Assessment Program (NAPAP). 1998. NAPAP biennial report to Congress: an integrated assessment. National Science and Technology Council, CENR, Silver Spring, MD, May 1998.
- National Research Council. 1993. Protecting visibility in National Parks and wilderness areas. Washington, D.C.: National Academy Press. 446 p.
- Neufeld, H.S.; Lee, E.H.; Renfro, J.R.; Hacker, W.D.; Yu, B-H. 1995. Sensitivity of seedlings of black cherry (*Prunus serotina* Ehrh.) to ozone in Great Smoky Mountains National Park: I Exposure-response curves for biomass. *New Phytologist* 130: 447-459.
- Orr, E.; Swain, E.; Strassman, R.; Bock, D. 1991. Sulfur emissions and deposition in Minnesota: 1990 biennial report to the Legislature. Air Quality Division, Minnesota Pollution Control Agency, St. Paul, MN. 23 pp.
- Orr, E.J.; Strassman, R.L.; Swain, E.B. 1992. Minnesota wet sulfate deposition standard: 1991 compliance report. Air Quality Division, Minnesota Pollution Control Agency. St. Paul, Minnesota.
- Peterson, D.L.; Schmoldt, D.L.; Eilers, J.; Fisher, J.; Doty, R. 1993. Guidelines for evaluating air pollution impacts on Class I Wilderness Areas in California. General Technical Report PSW-GTR-136. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. Albany, California.
- Peterson, D.L.; Sullivan, T.J.; Eilers, J.M.; Brace, S.; Horner, D.; Savig, K. 1998. Assessment of air quality and air pollutant impacts in National Parks of the Rocky Mountains and Northern Great Plains. U.S. Department of the Interior, National Park Service, Air Resources Division, Denver, CO.
- Peterson, J.; Schmoldt, D.; Peterson, D.; Eilers, J.; Fisher, R.; Bachman, R. 1992. Guidelines for evaluating air pollution impacts on Class I Wilderness Areas in the Pacific Northwest. General Technical Report PNW-GTR-299. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. Portland, Oregon.
- Porter, E. 1996. Air quality and air quality related values in Chassahowitzka National Wildlife Refuge and Wilderness Area. U.S. Fish and Wildlife Service, Air Quality Branch. Denver, Colorado.
- Schindler, D.W.; Curtis, J.P.; Parker, B.R.; Stainton, M.P. 1996. Consequences of climate warming and lake acidification for UV-B penetration in North American boreal lakes. *Nature* 379: 705-708.
- Schoettle, A.; Tonnessen, K.; Turk, J.; Vimont, J.; Amundson, R.G. 1999. An assessment of effects of human-caused air pollution on resources within the Interior Columbia River Basin. General Technical Report PNW-GTR-447. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR.
- Seinfeld, J.H., 1989. Urban air pollution: state of science. *Science* 243: 745-762.

- Shaver, C.; Malm, W. 1996. Air quality in Grand Canyon. In: Halvorsen, W.L.; Davis, G.E. (eds). Science and ecosystem management in the National Parks. University of Arizona Press, Tucson, AZ: 227-250.
- Shaver, C.; Tonnessen, K.; Maniero, T.M. 1994. Clearing the air at Great Smoky Mountains National Park. *Ecological Applications* 4: 690-701.
- Sickman, J.; Melack, J. 1998. Nitrogen and sulfate export from high elevation catchments of the Sierra Nevada, California. *Water, Air, and Soil Pollution* 105: 217-226.
- Sierra Nevada Ecosystem Project (SNEP). 1996. Summary of the Sierra Nevada Ecosystem Project Report. Centers for Water and Wildland Resources, Davis, CA.
- Sisler, J.F.; Malm, W.C.; Gebhart, K.A.; Pitchford, M.L. 1996. Spatial and seasonal patterns and long term variability of the composition of the haze in the United States: an analysis of data from the IMPROVE Network. ISSN No. 0737-5352-32. Cooperative Institute for Research in the Atmosphere, Fort Collins, Colorado.
- Southern Appalachian Man and the Biosphere Cooperative (SAMAB). 1996. The Southern Appalachian assessment: Atmospheric Technical Report. USDA-Forest Service. July 1996.
- Southern Appalachian Mountains Initiative (SAMI). 1999. SAMI, 1999 Interim Report. SAMI, Asheville, NC. April 1999.
- Stanford, J.; Acheson, A.; Brakke, D.; Eversman, D.; Savig, K.; Eilers, J. 1991. A screening procedure to evaluate air pollution effects in Region 1 wilderness areas, 1991. General Technical Report RM-GTR-294. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. Ft. Collins, Colorado.
- Stoddard, J. 1995. Episodic acidification during snowmelt of high elevation lakes in the Sierra Nevada Mountains of California. *Water, Air and Soil Pollution* 85: 353-358.
- Stoddard, J. 1994. Long-term changes in watershed retention of nitrogen: its causes and aquatic consequences. In: L.A. Baker. *Environmental chemistry of lakes and reservoirs. Advances in Chemistry Series No. 237.* American Chemical Society, Washington, D.C: 223-284.
- Stoddard, J.L.; Driscoll, C.T.; Kahl, J.S.; Kellogg, J.H. 1998. Can site-specific trends be extrapolated to a region? An acidification example for the northeast. *Ecological Applications* 8(2): 288-299.
- Stohlgren, T.J.; Quinn, J.F.; Ruggiero, M.; Waggoner, G.S. 1995. Status of biotic inventories in US National Parks. *Biological Conservation* 71: 97-106.
- Stohlgren, T.J.; Quinn, J.F. 1992. An assessment of biotic inventories in western U.S. National Parks. *Natural Areas Journal* 12 (3): 145-153.
- Stottlemeyer, R.; Toczydlowski, D.; and Herrmann, R. 1998. Biogeochemistry of a mature boreal ecosystem: Isle Royale National Park, Michigan. Scientific Monograph NPS/NRUSGS/NRSM-98/01. U.S. Department of the Interior, Denver, CO.
- Summers, K.; Tonnessen, K. 1998. Linking monitoring and effects research: EMAP's intensive site network program. *Environmental Monitoring and Assessment* 51: 369-380.
- Sullivan, T.J.; Cosby, B.J. 1995. Testing, improvement and confirmation of a watershed model of acid-base chemistry. *Water, Air and Soil Pollution* 85: 2607-2612.
- Sweet, C.W.; Prestbo, E.; Burnette, R. 1998. Wet deposition of total mercury at Mercury Deposition Network (MDN) sites in 1996 and 1997. In: NADP Technical Committee meeting, abstracts of papers, St. Petersburg, FL, October 26-29, 1998.
- Tevani, M. (ed). 1992. UV-B radiation and ozone depletion: effects on humans, plants, microorganisms, and materials. Lewis Publishers, Boca Raton, FL.
- Tilman, D., J. Knops, D. Wedin, P. Reich, M. Ritchie and E. Siemann. 1997. The influence of functional diversity and composition on ecosystem processes. *Science* 277: 1300-1302.
- Turk, J.; Campbell, D. 1987. Estimates of acidification of lakes in the Mount Zirkel Wilderness Area, Colorado. *Water Resources Research* 23: 1757-1766.
- Turner, R. S.; Fox, D.G.; Bartuska, A.; Jackson, W.; Bayle, B. Draft. Screening procedures to evaluate effects of air pollution on Forest Service wilderness in the Southern Region.
- U.S./Canada Air Quality Committee. 1998. U.S./Canada Air Quality Agreement, 1998 Progress Report. U.S. Government Printing Office, Washington, D.C.
- U.S. EPA. 1998. National air quality and emissions trends report, 1997. EPA 454-R-98-016. Office of Air Quality Planning and Standards, Research Triangle Park, N.C.
- U.S. EPA. 1997. Deposition of air pollutants to the Great Waters. Second Report to Congress. Washington, D.C.
- U.S. EPA. 1996a. Review of national ambient air quality standards for ozone: Assessment of scientific and technical information. OAQPS Staff Paper. EPA-452/R-96-007. U.S. EPA, Office of Air Quality Planning and Standards. Research Triangle Park, NC.
- U.S. EPA. 1996b. Air quality criteria for ozone and related photochemical oxidants. Vol. 2. EPA/600/P-93/004bF. U.S. EPA, Office of Research and Development, Research Triangle Park, NC.
- U.S. EPA. 1995. Acid deposition standard feasibility study: final report to Congress. EPA/430-R-95-001a. Washington, D.C. 120 p.
- U.S. EPA. 1994. The EPA Great Waters program: an introduction to the issues and the ecosystems. EPA-453/B-94/030. U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards, Durham, NC.
- van Miegroet, H.; Johnson, D.W.; Cole, D.W. 1992. Analysis of N cycles in polluted versus unpolluted environments. In: Johnson, D.W.; Lindberg, S.E. (eds). *Atmospheric deposition and forest nutrient cycling: a synthesis of the integrated forest study.* Springer-Verlag, New York: 199-202.
- Vertucci, F.A.; Eilers, J.M. 1993. Issues in monitoring wilderness lake chemistry: a case study in the Sawtooth Mountain, Idaho. *Environmental Monitoring and Assessment* 28: 277-294.
- Vitousek, P.M.; Aber, J.D.; Howarth, R.W.; Likens, G.E.; Matson, P.; Schindler, D.W.; Schlesinger, W.H.; Tilman, D.G. 1997. Human alteration of the global nitrogen cycle: sources and consequences. *Ecological Applications* 7(3): 737-750.
- Vong, R.J.; Sigmon, J.T.; Mueller, S.F. 1991. ES&T Critical Review: Cloudwater deposition to Appalachian forests. *Environmental Science and Technology* 25: 1014-1021.
- Wedin, D.A.; Tilman, D.G. 1996. Influence of nitrogen loading and species composition on the carbon balance of grasslands. *Science* 274: 1720-1723.
- Wigington, P.J.; DeWalle, D.R.; Murdoch, P.S.; Kretser, W.A.; Simonin, H.A.; Van Sickle, J.; Baker, J.P. 1996. Episodic acidification of small streams in the northeastern United States: ionic controls of episodes. *Ecological Applications* 6(2): 389-407.
- Wigington, P.J.; Davies, T.D.; Tranter, M.; Eshleman, K. 1990. Episodic acidification of surface waters due to acidic deposition. State of Science and Technology Report 12. National Acid Precipitation Assessment Program, Washington, D.C.
- Williams, M.W.; Brooks, P.; Mosier, A.; Tonnessen, K.A. 1996a. Mineral nitrogen transformations in and under seasonal snow in a high-elevation catchment in the Rocky Mountain, United States. *Water Resources Research* 32: 3161-3171.
- Williams, M.W.; Baron, J.; Caine, N.; Sommerfeld, R.; Sanford, R. 1996b. Nitrogen saturation in the Colorado Front Range. *Environmental Science and Technology* 30: 640-646.
- Williams, M.W.; Bardsley, T.; Rikkers, M. 1998. Overestimation of snow depth and inorganic nitrogen wetfall using NADP data, Niwot Ridge, Colorado. *Atmospheric Environment* 32: 3827-3833.
- Williams, M.W.; Tonnessen, K.A.. Setting critical loads for inorganic nitrogen deposition in the Colorado Front Range, USA. *Ecological Applications*. (in press).
- Williamson, C.E.; Zagarese, H.E. 1994. Impacts of UV-B on pelagic freshwater ecosystems. Wiley and Sons, NY.
- Wilson, L.; Vallee, M.; Tarasick, D.; Kerr, J.; Wardle, S. 1992. Operational forecasting of daily total ozone and ultraviolet-B radiation levels for Canada. Atmospheric Research Service, Environment Canada. Research Report No. 92-004.
- Winstanley, D.; Lackey, R.T.; Warnick, W.L.; Malanchuk, J. 1998. Acid rain: science and policy making. *Environmental Science and Policy* 1: 51-57.
- Yan, N.D.; Keller, W.; Scully, N.M.; Lean, D.R.; Dillion, P. 1996. Increased UV-B penetration in a lake owing to drought-induced acidification. *Nature* 381: 141-143.

2. Recreation Impacts and Management



Effects of Soil Compaction on Root and Root Hair Morphology: Implications for Campsite Rehabilitation

L. Alessa
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Abstract—Recreational use of wild lands can create areas, such as campsites, which may experience soil compaction and a decrease in vegetation cover and diversity. Plants are highly reliant on their roots' ability to uptake nutrients and water from soil. Any factors that affect the highly specialized root hairs ("feeder cells") compromise the overall health and survival of the plant. We report here initial data in our investigation of how soil compaction affects plant roots, using the common bean as a dicot model. Soil compaction decreases overall plant growth and causes changes in root hair morphology and the F-actin cytoskeleton, critical to the function of root hairs. In addition, rates of cytoplasmic streaming, which facilitate nutrient and water uptake, are reduced in root hairs from compacted treatments. When plants were removed from compacted soils, higher amounts of total C, N and Ca were found compared to those of controls. We discuss these data in the context of rehabilitation methods in impacted wilderness areas.

Plant growth and resilience strongly depend on the ability of the roots to anchor in substratum and uptake nutrients. Nutrient uptake is facilitated by modification of the maturing root epidermal cells into specialized extensions called root hairs, which extend into the soil by tip growth and facilitate the transport of soil nutrients and microelements, as well as water, via passive and active transport across their plasma membranes. Root hair development and function are highly dynamic and restricted to a specialized zone of the maturing root. Thus, they are highly responsive to changes in the physical and chemical status of the soil.

Since root hairs grow by tip growth, conditions that affect soil pore size or produce point localized pressure may affect not only their morphology, but also their function. Tip growth in root hairs appears to be controlled by a cellular component known as actin. The polymerization of globular (G) actin monomers into filamentous (F) actin forms a highly organized network throughout the cortex of root hairs. This network is very sensitive and responsive to changes in the external environment of the cell and facilitates cytoplasmic streaming, which is critical to the uptake and distribution of nutrients to the body of the plant.

Soil compaction, bulk density and strength are important factors affecting both shoot and root growth of plants, and roots growing in soil are able to respond to changes in these soil properties to some extent (Dexter 1987). Nonetheless, plants subjected to soil compaction are more susceptible to water stress and soil-borne diseases (Smucker and Erickson 1987). Furthermore, the possible reduction in plant-associated fungi and bacteria present in the soil combined with a retardation of root hair structure and function, may result in a rapid decline of flora. Roots growing in compacted soils may also be damaged by lack of oxygen (Schumacher and Smucker 1984) and by the accumulation of toxins (Crawford 1982). In the vicinity of recreational campsites, an increase in soil compaction and a decrease in vegetation cover have been documented (Marion and Cole 1996). However, little attention has been paid, both in terms of biological mechanism and remediation, to the changes that occur in the morphology and physiology of dicot roots. These data are crucial not only to monitoring and assessment of impacts, but in prescribing methodologies for rehabilitation of impacted areas. This is especially true since user impacts may vary depending on the type of soil, the diversity (types) of vegetation cover and the general features of terrain. Thus, the prescription of generalized assessment and rehabilitation protocols for heavily impacted campsites may not be effective in all areas.

Materials and Methods

Soil and Planting Conditions

Experiments were conducted using a sandy loam soil (approximately 67% sand, 23% silt and 10% clay) in a rooftop greenhouse with an automatic drip water system modified to an area sprinkler system. Soil was placed in plastic cylinders approximately 14 cm in diameter, and four treatments were applied: 1) no compaction 2) light compaction (~0-2.5 MPa) 3) medium compaction (~2.6-4.5 MPa) and 4) heavy compaction (~4.6-6.0 MPa).

Compaction and Penetrometer Resistance

Compaction was achieved using a 15 kg weight and assessed using a Haughn penetrometer. Soil penetrometer resistance was measured twice for pots, before and after the plant growth experiment. Additional measurements of penetrometer resistance were made on each pot at the time of plant harvest. A hand-held penetrometer equipped with a conical steel probe of 2.5 mm in diameter (with relieved

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shaft) was used in all measurements. During soil penetration of the probe, mass equivalence of force (M, kg) for each cylinder was determined using an electronic balance with resolution 5 g. Values of M were recorded to a soil depth of 75 mm, and maximum values of penetrometer resistance (Mpa) were estimated from the maximum values of M according to the equation $P = 4 \text{ Mg/pd}^2$.

Planting and Growth

Seeds of common bean (*Phaseolus vulgaris* L. var. Vista) were sterilized using a weak solution of sodium hypochlorite (10%) in distilled water and dried under sterile conditions. One seed was planted per cylinder to eliminate artifacts that may have occurred due to competition from neighbouring roots. Seeds were covered with approximately 4 mm of sandy loam soil and gently packed. In each of the six replicates, 20 cylinders were used. Five were used as controls, and five pots were used for each treatment (low, medium and high compaction). After planting, pots were placed on a perforated metal sheet on a 3 x 1 m planting table equipped with drains. Temperature was controlled by automatic windows and vents, as well as automatic heat lamps. Measurements of temperature indicated that temperatures remained relatively constant at 24 degrees C during the day and 14 degrees C during the night. Soils were not supplemented with fertilizers.

Plant Root Analysis

Seedlings were harvested 21 days after planting. The entire soil column was removed from the cylinder, and plants were gently separated out and laid flat on a white plastic tray. Soil adhering to roots was removed as much as possible by gentle washing with lukewarm distilled water. The root system of each seedling was examined under a Leitz AM443 dissecting microscope to obtain quantitative measurements of numbers of lateral roots, lateral root length and primary root length. Qualitative assessment of root hairs was made under a Zeiss Axiophot microscope equipped with DIC optics, and digital images were captured using a top-mounted Zeiss CCD camera.

Staining for F-actin (Cytoskeleton)

Coverslips were affixed to plant roots so that some root hairs lay flat against the glass. Root hairs were then permeabilized by incubating them with 0.01% w/v (0.1 mg/ml) saponin in distilled water for one hour. They were then rinsed and labeled with rhodamine phalloidin (RP), a fluorescent probe that specifically binds F-actin, by placing 100 μL of 6.6×10^{-7} M RP. A stock solution of 6.6×10^{-6} M RP was prepared in methanol and stored at -20°C . Prior to staining, the RP was desiccated and reconstituted in ASW to produce a working solution of 6.6×10^{-7} M. Root hairs were incubated in RP for at least 12 hours. Immediately prior to microscopic observations, cells were rinsed for one hour in distilled water. Controls for the specificity of RP binding were conducted by incubating root hair cells with an excess of unlabeled phalloidin (2×10^{-6} M) in ASW for six hours prior to staining with RP. Phalloidin has higher specificity than RP

for F-actin and therefore should greatly reduce or eliminate RP staining. All procedures involving staining for F-actin were done at 23°C .

Microscopy

Rhodamine phalloidin-labeled root hairs were observed on a Zeiss compound microscope equipped with epifluorescence using a narrow bandpass emission filter (605 ± 27 nm band width filter; Chroma Technology, Brattleboro, VT).

Analysis of Root Exudation

For analysis of root exudation, plants were carefully removed from pots and the remaining soil carefully examined for plant organic debris using a dissecting microscope. Five g of soil was mixed with 100 ml of distilled water and mechanically shaken for 20 minutes. For N and Ca, the supernatant (liquid fraction) was used; for C, dry soil was used. Total nitrogen was assayed by Kjeldahl digestion and ammonium analysis using standard Autoanalyzer techniques (Technicon Industrial Systems). Cation analyses were performed on a Perkin-Elmer Atomic Absorption Spectrophotometer. Calcium and carbon were determined with furnace and flame analysis, respectively, under optimized conditions. Changes in pH were determined by using a hand-held pH indicator under constant conditions of moisture and temperature.

Results

Effects of Compaction on Primary and Lateral Roots

A representative of seeds grown under control (no compaction) conditions is shown in fig. 1A. Overall morphological differences between control plants and those grown under moderately compacted conditions (approximately 4 MPa) consisted of the following, generally conserved, features for compacted treatments: plants were 1) shorter (as measured from soil surface to apical tip), 2) possessed reduced leaf surface area and 3) rarely possessed straight stems (see fig. 1B). With respect to root morphology, increasing compaction (low>medium>high) resulted in an increase in the length of the primary root and a corresponding decrease in the number of lateral roots. This inverse relationship varied from replicate to replicate but followed a fairly consistent pattern (fig. 2). Sub-apical swelling was observed in most primary and lateral roots ($n=112$), regardless of their average distal diameter. Diameters of lateral root tips were of at least half of those of primary root tips. A representative root tip showing the morphology of typical sub-apical swelling is shown in fig. 3A. Root tip squashes revealed that an increase in the thickness of the cortex, but not the stele, contributed to the increase in sub-apical root tip diameter (not shown). It was also observed that the root cap appeared to be consistently thicker; however, this potential morphological effect of compaction was not quantified. Since none of the plant roots had reached the bottom of the cylinder, there were no plants that were considered to be experiencing artifacts from impedance other than soil strength. Increasing soil compaction resulted in a

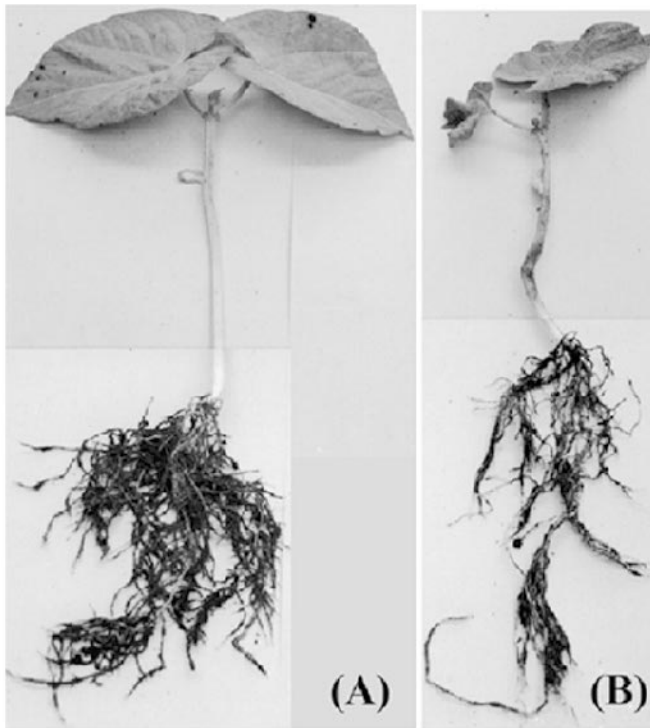


Figure 1—Representative control plant (A) vs. plant grown in compacted soil (approximately 4 MPa), (B). Compaction resulted in plants with decreased leaf surface areas and lower dry weights, as well as longer primary roots and fewer lateral roots.

proportional increase in sub-apical swelling of the primary root tip (fig. 3B), in addition to the observed increase in primary root length.

Effects of Compaction on Root Hairs

Root hairs were examined under a compound microscope equipped with differential interference contrast (DIC) optics

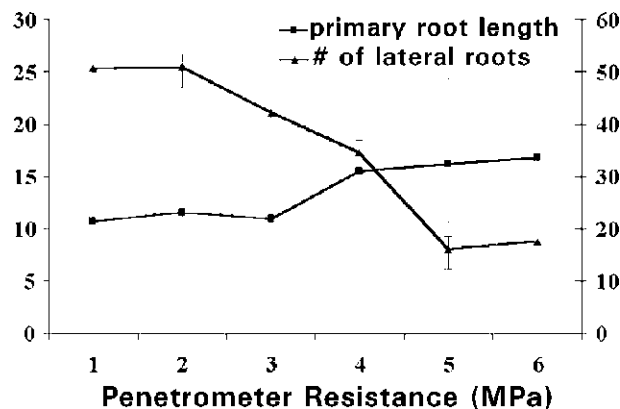


Figure 2—Graph showing inverse relationship between primary root length and number of lateral roots. As primary root length increases, the number of lateral roots decreases.

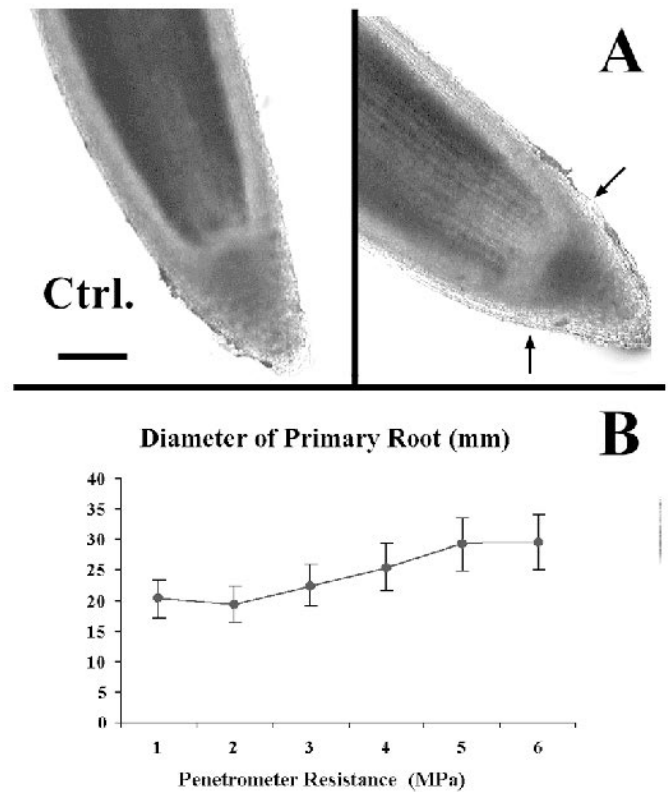


Figure 3—A. Micrograph of representative control (left) and compacted root tip (right). Sub-apical swelling is apparent in the latter (arrows), and the overall root diameter is increased. Bar=2 mm. B. Graph demonstrating an increase in primary root diameter with increasing soil compaction.

in order to obtain a clearer image of these extremely fine cells. Root hairs observed in roots from control plants appeared as shown in fig. 4A, left image. Overall length to width ratios were on the order of approximately 22:1 in controls and approximately 3:1 for those observed from moderately compacted roots (fig. 4B, left image). Cytoplasmic streaming rates were measured from videotaped analysis (table 1) and decreased proportionately with increasing soil compaction. To investigate the status of the filamentous actin network, root hairs were permeabilized with saponin and labelled with rhodamine phalloidin (fig. 4A and B, right images). The F-actin cytoskeletal network in root hairs from control plants consisted of bundled filaments that ran parallel to the long axis of the cell (fig. 4A, right image). Cytoplasmic streaming occurred along these bundles at high rates (see table 1) throughout the cell. Comparatively, root hairs from plants grown in moderately compacted soil possessed an F-actin network that appeared disorganized. Bundles often were observed to terminate along various points (fig. 4B, right image), and cytoplasmic streaming occurred sporadically, both spatially and temporally, in association with the fragmented cytoskeletal array. As soil compaction increased, the length of root hairs decreased, and the overall width (diameter) increased (fig. 5). Few root hairs with a decreased length to width ratio possessed

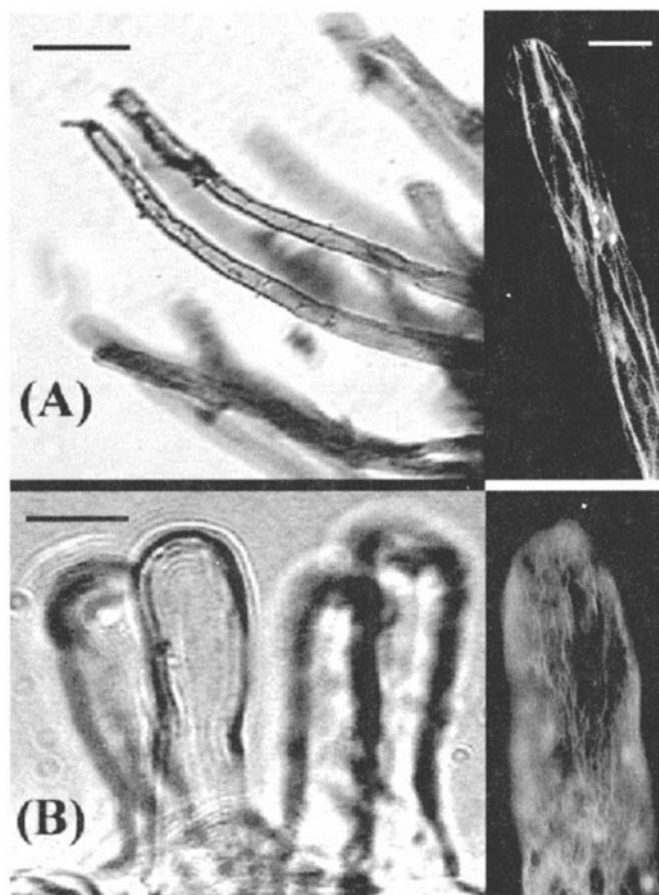


Figure 4—Left image. Root hair morphology in control plant roots (A). Root hairs possess a high surface area to volume ratio (longer lengths and smaller diameters), whereas root hairs from treated plants possess a smaller surface area to volume ratio (B, left image). Right images: fluorescence micrographs showing the F-actin cytoskeleton of root hairs. F-actin in control root hairs has well-defined bundles, oriented to the long axis of the cell (A, right image). F-actin in treated root hairs exhibits signs of disorganization (B, right image). A, left image, Bar=1 mm, right image, Bar=200 μm. B, left image, Bar=1 mm, right image, Bar=750 μm.

F-actin arrays that resembled controls and supported continuous, vigorous cytoplasmic streaming.

Root Exudation

In order to investigate whether compaction resulted in increased root exudation, carbon (C), nitrogen (N) and calcium (Ca) present in soils were measured after seedlings had been removed. Compaction resulted in an increase in C, N and Ca present in the soil of pots after seedlings were removed. These values were expressed as a percent increase from controls. A decrease in pH was also noted in soils subjected to moderate and heavy, but not mild compaction. These data are summarized in table 2.

Discussion

Soil compaction, whether caused by trampling or impact from machines, has adverse effects on plant health.

Table 1—Effects of compaction on rates of cytoplasmic streaming in root hairs.

Compaction ^a	Rate of cytoplasmic streaming (μm/min)
CONTROL	22.5 ± 5.7
MILD	20.8 ± 9.4
MODERATE	14.2 ± 2.7
HEAVY	9.9 ± 6.1

^aControl: no compaction; mild: 1-2 MPa; moderate: 2.1-3.45 MPa; heavy: 3.5-5.0 MPa.

Compaction may result in 1) an increase in bulk density, 2) the elimination or decrease of biologically available pore space (into which fine root processes may extend) (Kooistra and others 1992), 3) a change in soil gas balance and changes in soil moisture status and regulation (Kuss 1986). These effects can be linked to morphological and physiological changes at the level of the plant root. Morphological changes appear to include restriction of root extension and shoot growth, as well as modifications of the root pattern and root diameter (Ikeda and others 1997).

The impacts of compaction on the status of vegetation cover and soil structure and function may be severe (Hammit and Cole 1987) and almost always involve soil compaction (Marion and Cole 1996). Often, the relationship between compaction and impact is positive but nonlinear due to differences in soil structure, vegetation diversity and abiotic factors such as hydraulic balance and topography. Thus, rehabilitation methods that focus on mechanical but not biological reparation may not overcome impact-induced biological deficits such as changes in root function and loss of microbial biomass and the sustenance of initial natural recovery processes such as vegetation succession (Wardle 1992).

Primary and Lateral Root Systems

Our data are consistent with those reported by Pietola and Smucker (1998), in that compaction actually increased

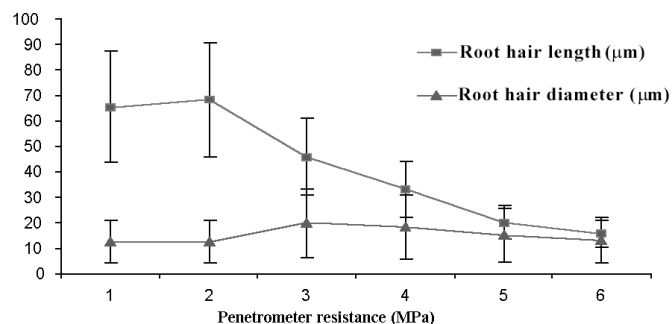


Figure 5—Graph showing changes in root hair length and diameter of *P. vulgaris* grown in compacted soils. Root hair length decreases, and diameter increases. Cytoplasmic streaming in these cells is greatly reduced (see table 1), and the F-actin cytoskeleton becomes disorganized (see fig. 4). $p < 0.05$.

Table 2—Effect of compaction on total carbon (C), nitrogen (N) and Calcium (Ca) present in soils.

Compaction ^a	Total C ^b	Total N ^c	Total Ca ^d	pH
CONTROL	0.0	0.0	0.0	7.06 ± .25
MILD	9.7 ± 0.3	8.2 ± 0.9	N.D.	7.23 ± 0.11
MODERATE	12.5 ± 1.4	11.3 ± 0.7	2.3 ± 0.8	6.8 ± 0.17
HEAVY	32.8 ± 6.9	29.2 ± 1.3	10.1 ± 1.4	6.6 ± 0.15

^aControl: no compaction; mild: 1-2 MPa; moderate: 2.1-3.45 MPa; heavy: 3.5-5.0 MPa.

^{b,c,d}Reported as percent increase from control.

primary root length, but they are contrary to reports that describe a decrease of primary root lengths of other plants grown in compacted soils (Atwell 1988). The data also suggest that soil compaction that results in a change in lateral root number, as well as root hair function may have greater effect on overall plant health than when primary root growth alone is inhibited in deeper soil layers (Goodman and Ennos 1999). Thus, plants may respond to changes in soil structure (compaction) that are fairly mild to moderate (compaction of the topmost layer of soil). They also imply that rehabilitation of sites that have been stripped of topsoil layers, thus exposing mineral soil, may be far more challenging and complex than that of sites which are only moderately impacted.

Roots Hairs

Our data show that, in the system studied, structure and function of root hairs are affected by soil compaction (fig. 4). In young roots, the epidermal root hairs, which greatly increase the absorbing surface area of the root, absorb water and minerals. Root hairs are relatively short-lived and may reach maturity within hours. The production of new root hairs occurs just beyond the region of elongation and at approximately the same rate as that at which the older root hairs are dying off at the upper end of the root hair zone. Any factors that compromise the structure and function of these cells will affect the status of overall plant health. Since these cells are relatively short-lived and generated quite rapidly, seedlings germinated on soil that has not been adequately de-compacted may be unable to adequately anchor in soil and uptake nutrients and water. Furthermore, epidermal cells of the root produce a mucigel that enables the root hairs to establish close physical contact with soil particles (Ulehlova and others 1988). If soils are de-compacted so that large air spaces are present, this contact, critical to uptake, may be greatly reduced. This mucigel is also hypothesized to have other functions such as facilitating carbon sequestering near the root, facilitating the passage of root processes through soil (Ulehlova and others 1988) and attracting soil microorganisms to the vicinity of the root (Ikeda and others 1997).

Exudation

An increase in root exudates from maize and cereal roots grown in compacted soils has been reported by

Boeuf-Tremblay and others (1995) and Barber and Gunn (1974), respectively. Root exudation may be described as a generalized stress response to conditions, for example, where the root physical structure has been compromised or where a toxicity response has been initiated such as in plants grown in low pH soils containing aluminum (Taylor 1995).

Our data are consistent with those of Boeuf-Tremblay (1995) and Barber and Gunn (1974), in that compaction increased the production of root mucilage, which contributed to an increase in total C, N and Ca levels of soils surrounding the root. The function of increased root exudation may include, but not be limited to, 1) chelating toxic compounds by changing the localized pH (as appeared to be the case for aluminum toxicity) and 2) providing an environment which favours the aggregation of culturable bacteria (Ikeda and others 1997). This latter function has interesting implications for plants growing under compacted soil conditions and for seedlings introduced to sites for rehabilitation. Soil compaction may not only affect plant growth, it may also affect the diversity and numbers of soil micro-organisms (Zabinski and Gannon 1997). The functions of these diverse organisms are poorly understood; however, they may function in roles such as nitrogen fixation, the decomposition of debris, the stimulation of root growth and the accumulation of nutrients in the vicinity of the maturing root (Perry and Amaranthus 1990; Turkington and others 1988). Zabinski and Gannon (1997) report that bacterial and fungal components of the soil community were severely disrupted in soil from campsites. Thus, plants growing on increasingly compacted soils, which may be increasing root exudates in order to attract micro-organisms, may be unable to do so due to the absence or decline of the latter. A caveat, however, is that soil micro-organisms may be highly dependent on the presence of vegetation, which may provide a carbon source for substrate utilization (Rovira 1995). Unfortunately, our understanding of the dynamics of the plant root-soil microorganism relationship is extremely limited, so our ability to speculate on the nature of this, possibly nonlinear, biology is severely restricted.

Implications for Rehabilitation of Recreation Wilderness

Although preliminary, our data provide evidence that moderate soil compaction affects the structure and functioning of a dicot plant root system. These data provide evidence for a possible mechanism by which compaction of soils in the area of campsites may cause a decrease in vegetation cover. Thus, in areas where vegetation is partially removed, remediation efforts that involve the introduction of nutrients to the impacted area (such as the raking of humic soils and detritus) after mechanical de-compacting of soils may not be effective. Not only are root hairs unable to uptake nutrients, as is suggested by a decrease in cytoplasmic streaming, the plant root exhibits increased exudation of organic N, C and Ca-containing compounds, possibly as a generalized stress response (Dexter 1987) or as a mechanism to attract root-associated, culturable bacteria (Barber and Gun 1974; Boeuf-Tremblay and others 1995). We are particularly interested in the changes that occur in the root hairs due to soil compaction. If these cells do not function

optimally, plant growth will be compromised even if seeds germinate in disturbed areas. Thus, monitoring the status of root hairs has implications for predicting the effectiveness of rehabilitation in impacted areas and may be accomplished easily, with minimal gear and time. Mechanical loosening of soils may be beneficial in attempting to reestablish a functional soil matrix; however, outcomes other than the ideal "ratio" of soil aggregate to pore size may occur. These include, but are not restricted to, 1) loosening of only the top few centimetres of soil with sub-layers remaining compacted, 2) over-loosening of soil such that root-soil contact is decreased, 3) loosening of soil so that aggregates consist of large, compacted pieces of soil with large air spaces between them.

Ongoing Research

Currently, we are extending and bifurcating the study to 1) an investigation of dicot roots of woody and nonwoody plants from soils in sites that have been compacted by foot and livestock traffic, as well as the behaviour of root processes in de-compacted field plots that have been re-seeded and planted in an attempt to restore vegetation cover and, 2) a study involving rehabilitation methods that transfer soil normal flora from the perimeter of heavily used sites to the impacted area in order to facilitate seedling growth. The effects of soil compaction on root structure and function are complex. Here, we report data for a specific type of soil and dicot species. Compaction effects may vary depending on soil type, soil moisture status, plant type, etc. Thus, a great deal of research is needed to identify the potentially common mechanisms that may underlie varying responses to soil compaction.

References

- Atwell, B.J. 1988. Physiological responses of lupin roots to soil compaction. *Plant and Soil*. 111:277-281.
- Barber, D.A. and Gunn, K.B. 1974. The effect of mechanical forces on the exudation of organic substances by the roots of cereal plants grown under sterile conditions. *New Phytologist*. 73:39-45.
- Boeuf-Tremblay, V.; Plantureux, S. and Guckert, A. 1995. Influence of mechanical impedance on root exudation of maize seedlings at two development stages. *Plant Soil*. 172:279-287.
- Crawford, R.M.M. 1982. Physiological responses to flooding. In O.L. Lange and M. Aragno (eds.). *Encyclopedia of plant physiology*. Vol.12B. *Physiological plant ecology II*. Springer Verlag, Berlin, Germany.
- Dexter, A.R. 1987. Mechanics of root growth. *Plant and Soil*. 98:303-312.
- Goodman, A.M. and Ennos, A.R. 1999. The effects of soil bulk density on the morphology and anchorage mechanics of the root systems of sunflower and maize. *Annals of Botany*. 83:293-303.
- Hammit, W.E. and Cole, D.N. 1987. *Wildland recreation ecology and management*. John Wiley and Sons. New York, New York, USA.
- Ikeda, K.; Toyota, K.; Kimura, M. 1997. Effects of soil compaction on the microbial populations of melon and maize rhizosphere. *Plant and Soil*. 189:91-96.
- Kooistra, M.J.; Schoonderbeek, D.; Boone, F.R.; Veen, B.W. and van Noordwijk, M. 1992. Root-soil contact of maize, as measured by a thin-section technique. II. Effects of soil compaction. *Plant Soil*. 139:119-129.
- Kuss, F.R. 1986. A review of major factors influencing plant responses to recreation impacts. *Environmental Management*. 10:637-650.
- Marion, J.L. and Cole, D.N. 1996. Spatial and temporal variation in soil and vegetation impacts on campsites. *Ecological Applications*. 6(2):520-530.
- Perry, D. and Amaranthus, M. 1990. The plant-soil bootstrap: Microorganisms and reclamation of degraded ecosystems. In J. Berger (ed.). *Environmental restoration: Science and strategies for restoring the earth*. Island Press. Washington D.C.
- Pietola, L. and Smucker, A.J.M. 1998. Fibrous carrot root responses to irrigation and compaction of sandy and organic soils. *Plant and Soil*. 200:95-105.
- Rovira, A.D. 1995. Interactions between plant roots and soil microorganisms. *Annual Review of Microbiology*. 19:241-266.
- Schumacher, T.E. and Smucker, A.J.M. 1984. Effect of soil compaction on root growth and uptake of phosphorus. *Plant and Soil*. 77:15-28.
- Smucker, A.J.M. and Erickson, A.E. 1987. Anaerobic stimulation of root exudates and diseases of peas. *Plant Soil*. 99:423-433.
- Taylor, G.J. 1995. Overcoming barriers to understanding the cellular basis of aluminum resistance. *Plant and Soil*. 171:89-103.
- Turkington, R.; Holl, F.B.; Chanway, C.P.; and Thompson, J.D. 1988. The influence of microorganisms, particularly *Rhizobium*, on plant competition in grass-legume communities. In A.J. Davy; Hutchings, M.J. and Watkinson, A.R. (eds.). *Plant population ecology*. Blackwell Scientific Publications, Oxford, U.K.
- Ulehlova, B; Kunc, F. and Vancura, V. 1988. Nutrition and energy sources of microbial populations in ecosystems. In V. Vancura and F. Kunc (eds.). Elsevier Press, Amsterdam.
- Wardle, D. A. 1992. A comparative assessment of factors which influence microbial biomass, carbon and nitrogen levels in soil. *Biological Reviews of the Cambridge Philosophica*. 67(3):321.
- Zabinski, C.A. and Gannon, J.E. 1997. Effects of recreational impacts on soil microbial communities. *Environmental Management* 21(2):233-238.

Twenty-Eight Years of Wilderness Campsite Monitoring in Yosemite National Park

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Mark Fincher
Jan van Wagtendonk

Abstract—The research, resource management and wilderness staffs in Yosemite National Park recently completed the third 10-year cycle of a wilderness campsite impact monitoring program. Initial results indicate an overall improvement in conditions due to a strong restoration program, decreased use and increased visitor education. Lessons learned point to the necessity for ample and appropriate data collection and consistent techniques over time. This paper discusses the methodology and findings of this 28-year project.

The lure of Yosemite has resulted in profound effects on both those who visit the park and the natural environments it encompasses. As John Muir said, “the galling harness of civilization drops off...” those who visit, but their very presence has also modified the landscape. Public land managers have long recognized that recreational use may pose pervasive and intractable threats to resources, but they have grappled with just how to measure, monitor and manage those impacts.

In the early 1970s, the research, wilderness and resource staffs realized the need to improve their understanding of how recreation affected ecosystems and the effectiveness of management. Over the course of the next 28 years, the staff undertook three wilderness-wide inventory and monitoring studies focusing on campsite impacts. Our objectives in this undertaking were three-fold:

- 1) Establish a baseline for natural conditions and variation;
- 2) Determine when, where and why significant change occurs, and track that change over time;
- 3) Understand the relationship of natural conditions, visitor experience, and wilderness resource management.

Background

Yosemite is one of the most heavily used wilderness areas in the National Wilderness Preservation System. Wilder-

ness recreational use peaked in 1975, with a record high of 219,000 visitor use nights (van Wagtendonk 1981). After dropping to a low of less than half that number in 1983, current trends show a leveling of use with a slight downward trend, fluctuating around 117,000 use nights in the past 10 years (National Park Service 1999) (Fig. 1). In the early 1970s, park managers were not fully aware of the magnitude and impact of the increasing hordes. A formal backcountry management district was established in 1973, with a small but dedicated staff to patrol trails, perform light maintenance and issue wilderness permits.

In 1973, Yosemite started restricting use by travel zones, determined from the area of the zone, the number of miles it contained, its ecological fragility and social density standards (van Wagtendonk 1986). Today, wilderness visitation is managed by a trailhead quota system, established in 1977 after extensive research on capacity and use (van Wagtendonk and Coho 1986). The quota system allows for spatial and temporal distribution of use (dispersed camping is allowed in most areas of the Yosemite Wilderness) and provides a means to limit access to areas exceeding appropriate levels of use. It also serves as an important educational tool, giving staff the opportunity to convey minimum impact regulations to visitors.

Increasing use and public complaints regarding impacts and crowding quickly made managers aware of the need for more information. The first survey of campsite impacts was made in 1972. Daniel Holmes and a team of 31 others formed the Wilderness Research Group through the University of California at Berkeley. In cooperation with the National Park Service, Holmes and his crew covered the more than 700,000 acres of the Yosemite backcountry, surveying almost every area receiving human use. Detailed descriptions and maps were made of more than 7000 campsites, over 800 miles of trail and all waterways that receive use (Holmes 1972).

The primary objectives of the study were 1) To describe, both quantitatively and qualitatively, the range of visible environmental damage from users in the Yosemite Wilderness, and 2) describe the physical distributions of impact from those users.

A need for further study, coupled with an interest in assessing change over time, prompted the Yosemite Resource Management staff to resurvey the entire wilderness 10 years later. Between 1981 and 1986, Charisse Sydorik and a varied team of mostly volunteers went back over those 700,000 acres, this time making more detailed studies and maps of 5,547 campsites and 1,048 miles of trail (Sydorik 1986). The methodology used in this study was a slightly modified model of the system developed in Sequoia and

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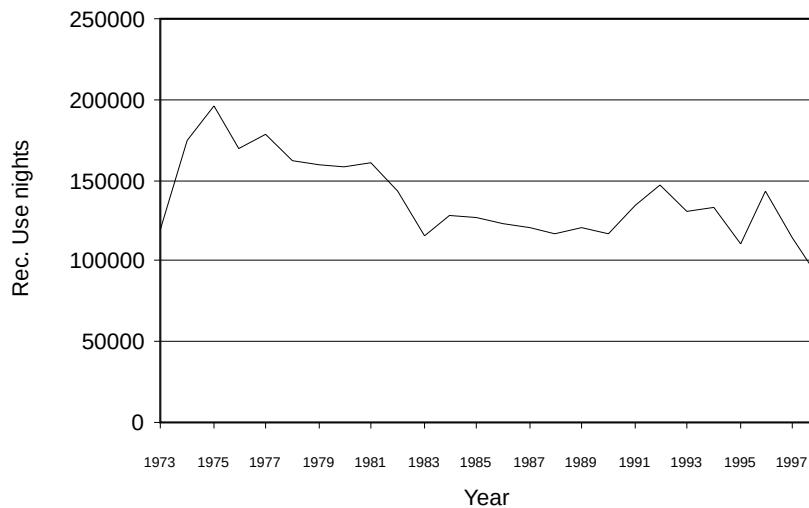


Figure 1—Recorded visitor use nights for Yosemite Wilderness, 1973 to 1998.

Kings Canyon National Parks by Parsons and McLeod (1980) and became known as the Wilderness Inventory and Monitoring System, or WIMS. WIMS measured eleven impact criteria: firewood availability, tree root exposure, visual obtrusiveness, vegetation density, vegetation composition, total campsite area, barren core, litter and duff, campsite developments, mutilations, and social trails. Descriptive information about the local environment was also recorded, including vegetation type and foundation, distance from water, crowding and management recommendation. Maps and photo documentation were done for each site. This information became an important tool for a newly formed restoration program in the Park.

Ten years passed, and understanding the value of the historic data on hand, the research, resources and wilderness staffs at Yosemite undertook WIMS 2, (or Son of WIMS as it was somewhat affectionately called) in 1992. The purpose of this study was to combine and replicate as much as possible of both previous studies from the 1970s and 80s to further evaluate the change in recreational impacts over time.

The scope of the project was reduced from its wilderness-wide approach, and 34 target campsites were selected. These sites needed to be dispersed throughout the Park and have been surveyed in both the Holmes 1972 study and the WIMS 1980s study. A variety of sites, including heavily used or stock camps, moderately used and lightly used or cross country camps were pulled from both sets of data. Both studies were evaluated for comparables, and a modified monitoring system was developed, rating the campsites with the same criteria used in both projects to the greatest extent possible. Two new criteria were added to address human waste and stock impacts, and two different techniques were used to quantify vegetation density. Detailed maps and photo sets were also completed, trying to match documentation from the 1980s, both spatially and seasonally.

In order to assess a larger picture of the extent of campsite proliferation and impacts, each area surrounding the target site was mapped and measured using a

two-prong condition-class rating. Using the lake basin, trail junction or defined area around each target site, surveyors rated and mapped every site in the area for: 1) developments (primarily fire rings) and 2) vegetation loss. These parameters were chosen because they were ecologically important and had been rated in both previous studies. This method gave managers an idea of the recreational “health” of an entire area and could be used to assess management actions such as restoration, closures and quotas, as well as make reasonable comparisons to the more extensive data from the previous 2 surveys. The monitoring of the WIMS 2 sites was completed in 1998, with the 34 target campsites evaluated and over 700 campsites recorded in the areas surrounding them.

Data Analysis

Between 1972 and 1999, data were collected three times: Holmes, WIMS and WIMS 2. However, the methodology for data collection was changed part way through WIMS 2, resulting in four kinds of data. The post-change data for WIMS 2 is referred to as WIMS 2.1. Certain adjustments were needed to compare the data sets.

Target Site Analysis

Comparing data from the different data sets was easy for the target site analysis because, with one exception, WIMS 2 used the exact same criteria as Holmes and WIMS. In that case, the mutilations score for WIMS 2 was divided into mutilations to rocks and soil and mutilations to vegetation.

Area/Campsite Class Analysis Using Adjusted Data

WIMS 2 measured the number of campsites and their condition class in each area. In order to assess change in

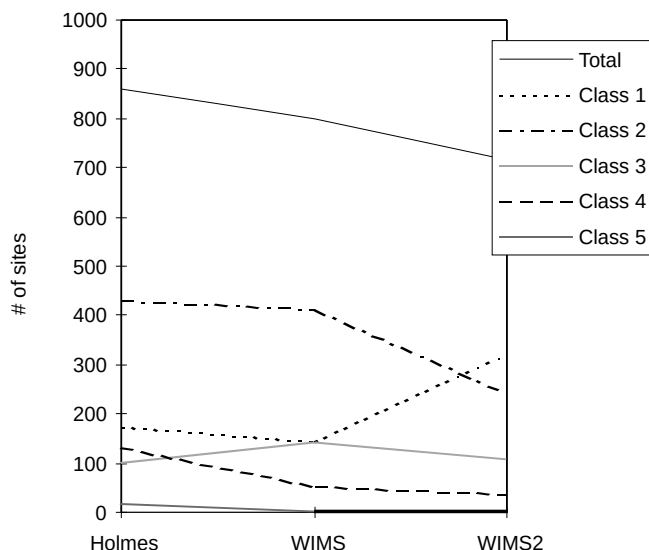


Figure 2—Change in number of sites by class.

these sites over time, an attempt was made to directly compare the data from all data sets. To accomplish this, the data from Holmes, WIMS and WIMS 2.1 were adjusted to match the criteria for the five condition classes for WIMS 2. Condition classes ranged from barely discernible (class 1) to heavily developed and impacted (class 5).

Damage Total Calculation

In an effort to produce a single number that roughly describes the cumulative amount of impact to an area due to campsites, WIMS 2 field personnel considered the relative impact of a site of each condition class. The result of this consideration was the following: A class 2 site causes twice as much impact as a class 1 site, a class 3 site causes three times as much impact as a class 1 site, a class 4 site causes six times as much impact as a class 1 site, and a class 5 site causes 18 times as much impact as a class 1 site.

Area Campsite Class Analysis Using Original Data

The condition-class data were also considered in their original form so they could be compared to the verbal descriptions for each condition class and to allow comparisons of how that verbal description of the mean condition class changed over time. Unfortunately, no composite condition class numbers or corresponding verbal descriptions have been found for the original WIMS data, so this comparison is of limited value.

Results and Discussion

Campsite Class

From Holmes to WIMS 2, the number of sites decreased 17%. Between the Holmes and WIMS surveys, class 3 sites

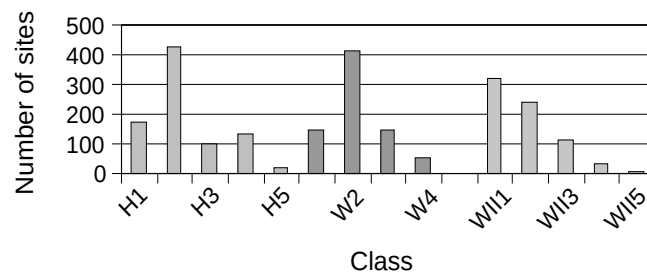


Figure 3—Number of sites by class.

increased, while all other classes decreased. Between the WIMS and WIMS 2 surveys, class 2, 3, and 4 sites all decreased substantially, while class 1 sites increased 124% (Fig. 2).

The increase in class 1 sites occurred during the period when restoration of campsites was a priority for management. Much of this increase is probably due to sites that have been restored but are still discernible. Some of them may also be single-use sites that were established when sites near water were removed by management (Fig. 3).

The number of sites considered undesirable (classes 3, 4 and 5) decreased 41% (Fig. 4). The damage total decreased 43% (Fig. 5).

Restoration crews worked in seven of the 34 areas surveyed and significantly reduced impacts. Between WIMS and WIMS 2, in those areas not visited by the restoration crews, the number of sites increased 16% , mostly due to an increase in class 1 sites. (Figs. 6 and 7).



Figure 4—Number of sites considered undesirable.

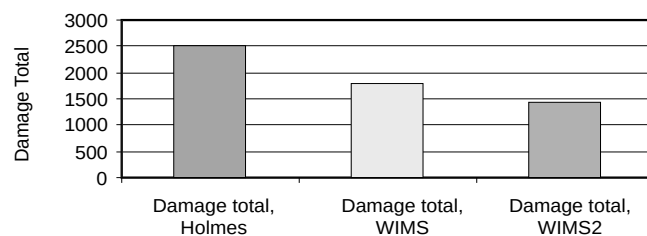


Figure 5—Damage total.

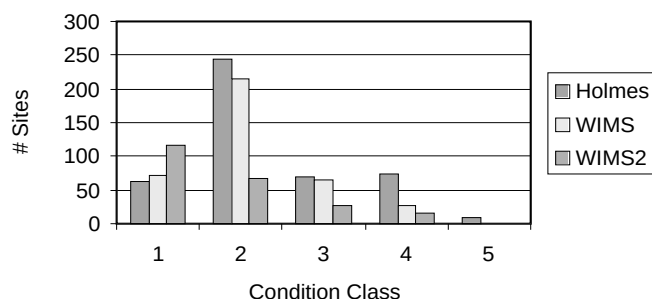


Figure 6—Number of sites by class, restored areas.

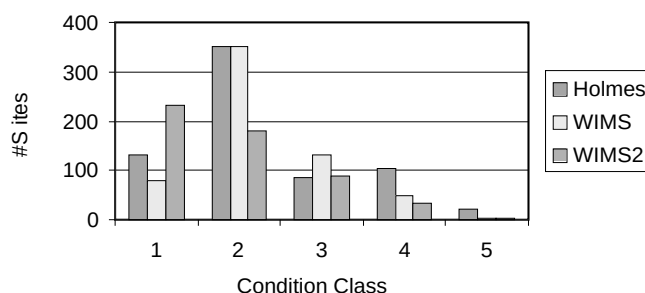


Figure 9—Number of sites by class, trail area.

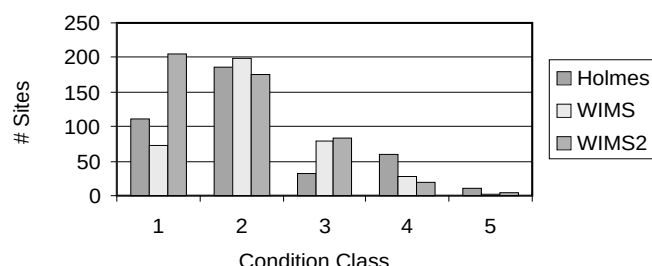


Figure 7—Number of sites by class, unrestored areas.

The cross-country camping areas were an exception to the general trend of reduced impacts. Between WIMS and WIMS 2, the average condition class remained the same, while the total number of sites increased 22%. This increase is probably due to two factors: 1) Full-time restoration crews have worked only in trailside areas, and 2) wilderness patrol rangers spend very little time in the cross-country areas (Figs. 8 and 9).

Campsite Impacts

Three criteria from Holmes and nine criteria from WIMS were repeated during WIMS 2 at the 34 target sites. Between WIMS and WIMS 2, root exposure, firewood scarcity and access trail impacts all increased significantly. In the WIMS survey, these three criteria had the lowest scores. Vegetation density impacts also increased significantly (Fig. 10).

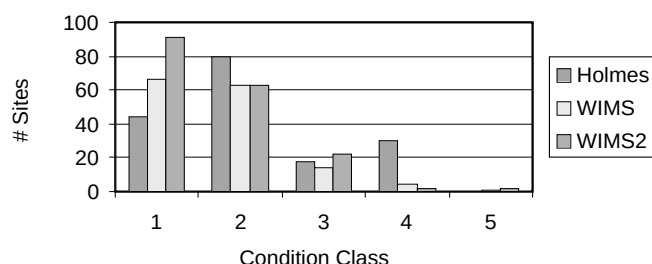


Figure 8—Number of sites by class, cross country areas.

Management Implications

It is clear that management efforts are reducing the impacts of campsites in the Yosemite Wilderness. Now that formal restoration crews have completed work in the most impacted areas, it will be vital to monitor and maintain those areas. Although the total number of class 1 sites increased, some previously recorded class 1 sites disappeared over the decade between surveys. It is extremely important that these restored sites are kept from further use and have time to heal.

Continued educational and restoration efforts will be needed to sustain the reduction in class 3, 4, and 5 sites. Requiring stock groups to use designated sites is being considered as well.

Our monitoring system would explain more about the effects of our management actions if we had differentiated between “healing” class 1 sites and new, single-use class 1 sites. Further monitoring is needed to determine if the increase in class 1 sites was caused by the systematic removal of sites near water and trails, or by other management actions.

In addition, a more focused approach to campsite restoration, concentrating on the complete removal of single-use sites and the continued reduction of larger sites, may reduce the increase of class 1 sites. This work can be accomplished by individuals such as the patrol rangers or volunteers rather than large groups.

More patrols and restoration efforts are needed in the cross-country areas. The increase in the number of sites in these areas is of particular concern, due to the sensitivity of lightly used areas to small amounts of change and the importance of keeping the trail-less areas in a more pristine condition. Banning campfires in the cross-country areas is being considered.

The increase in root exposure, firewood scarcity and access trail impacts are potentially worrisome and warrant close monitoring. These impacts, on average, are currently below the threshold that would trigger a change in management action for the Park as a whole. In one area, however, an immediate management action was implemented to close an area to campfires, based on survey information. Additional areas will be watched closely for continued change. These impacts are cumulative and only get worse with continued use. The decrease in vegetation densities is also of concern because it indicates increased trampling and potential soil

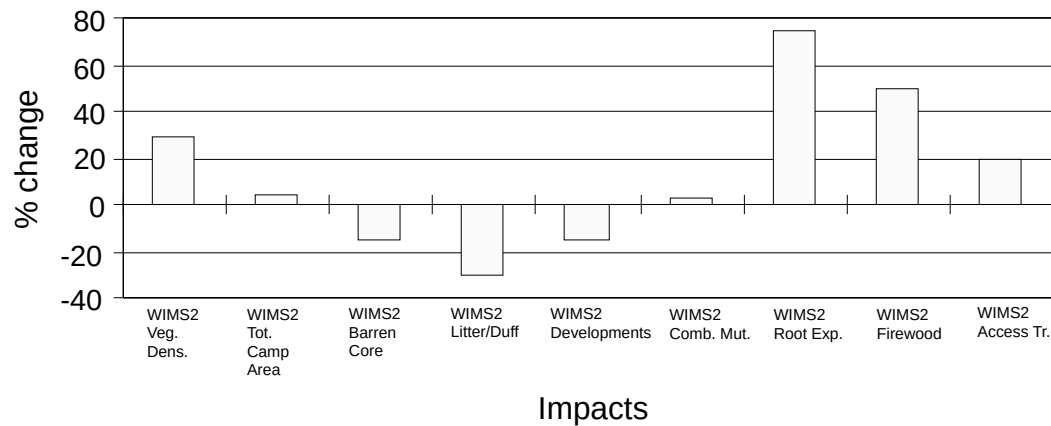


Figure 10—Percent change in impacts.

compaction, which may lead to more serious ecological impacts.

Lessons Learned

It is vital that we monitor the changing conditions of our resource. The baseline data recorded almost 30 years ago have been invaluable, creating a picture of the status of our wilderness before we actively began managing it. If you haven't recorded baseline conditions yet, start tomorrow. Do it in a way that is scientifically sound; covers all significant bases, not just those you are worried about now; and make sure the results can be replicated.

To be efficient, it is imperative that you determine the pertinent questions. Data glut is a danger, but so is going to all the time and effort to get to a site and not recording information that would be valuable. For example, we could not determine the effectiveness of our educational/regulatory message about camping 100' from water because we neglected to measure distance from water in the WIMS 2 study.

After deciding what to monitor, define or quantify the parameters very thoroughly. While it is important to measure some indicators precisely, many can be measured quickly but appropriately if the parameters and rating criteria are well quantified. This was a particular problem with the Holmes data, as it was unclear just what the descriptors meant: How far from a campsite did you look at firewood availability, or what was a "large" fire ring?

Once you settle on a system, try to stick to it. The WIMS crew spent an entire summer "truthing" their system and then started recording the data that were kept. The WIMS 2 process changed the method of determining vegetation densities mid-study, complicating comparisons. It was difficult to analyze the Holmes data because WIMS used such a different system. Try to fine-tune your system before you start.

Monitoring, especially using the adjective classes, primarily shows trends that indicate when more research is needed. The mid-point value is really the only thing you can measure, which serves as a gross filter to identify which fine filter actions will be appropriate. Monitoring does not answer all management questions, but it does indicate trends or warning flashes that need to be looked at more closely.

Mapping and locating sites become increasingly important over time. Photo documentation should be done as a series narrowing in on the site, and GPS is the tool we wish could have been used in the 1970s and 80s. All is lost if you can't find the site.

And finally, perhaps the most important message: Use it now, but keep doing it. Yosemite needs to start Grandson of WIMS, or WIMS the Third, in 2000 to 2010 to continue our assessment over time. This is particularly important to track the trends we are seeing now, to appraise the effectiveness and longevity of efforts such as expensive restoration projects, and to continue assessing the appropriateness of management of this wild and important resource.

References

- Holmes, D. O. 1972. Yosemite backcountry inventory, summer 1972. USDI, National Park Service, Yosemite National Park, CA. Final Report. 2295 p.
- Parsons, D. J., and S. A. McLeod. 1980. Measuring impacts of wilderness use. *Parks*. 5(3):8-12.
- Sydoriak, C. A. 1986. Yosemite wilderness trail and campsite impact monitoring system. USDI, National Park Service, Yosemite National Park, CA. 25 p.
- van Wagtenonk, J. W. 1981. The effects of use limits on backcountry visitation trends in Yosemite National Park. *Leisure Sci.* 4(3):311-323.
- van Wagtenonk, J. W. 1986. The determination of carrying capacities for the Yosemite Wilderness. P. 456-461 in *Proc. National Wilderness Res. Conf.: Current Res.* USDA Forest Service Gen. Tech. Rep. INT-212. 553 p.
- van Wagtenonk, J. W., and P. R. Coho. 1986. Trailhead quotas: rationing use to keep wilderness wild. *J. Forestry*. 84(11):22-24.

Camping Impact Management at Isle Royale National Park: An Evaluation of Visitor Activity Containment Policies From the Perspective of Social Conditions

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Abstract—A survey of backcountry and wilderness campsites at Isle Royale National Park reveals that the park's policies for managing visitor impacts have been remarkably effective in limiting the areal extent of camping-related disturbance. However, the dense spatial arrangement of designated campsites within backcountry campgrounds has also contributed to problems with visitor crowding and conflict. Only 9% of the sites had no other sites visible, while 22% had three or more other sites visible. Mean intersite distance was only 76 feet, and 34% of the sites are within 50 feet of another site. Visitor education programs and selected relocation of sites could reduce these social problems.

National Park Service legal mandates and administrative policies prescribe a management paradox for administering recreational use in backcountry and wilderness areas. Park staff are charged with managing naturally functioning ecosystems and processes substantially free from human influence, yet these protected areas must also be managed for recreational visitation. Even low levels of hiking or camping activity have been shown by research to cause substantial degradation to vegetation and soils (Cole 1995). Camping-related impacts are an even greater concern in federally designated wilderness areas, which direct managers to maintain resource conditions that are "untrammeled by man...protected and managed so as to preserve its natural conditions" (16 USC 1131-1136).

However, managers must recognize that some camping impacts are inevitable with wilderness visitation. The challenge is to minimize the number of campsites and the extent and severity of impact at each site. As described in this paper, Isle Royale National Park (ISRO) represents one of the best examples of camping activity containment for minimizing camping impacts in wilderness areas. Activity containment policies seek to reduce recreation impacts by spatially concentrating visitor activities to limit the area of resource disturbance. ISRO park managers have accomplished this by carefully locating and constructing designated campsites

to sustain heavy camping visitation while limiting associated resource impacts.

Although ISRO's visitor activity containment policies have successfully limited the areal extent of camping disturbance, high campsite densities have contributed to social problems of visitor crowding and conflict. The Wilderness Act specifies that wilderness areas should offer "outstanding opportunities for solitude or a primitive and unconfined type of recreation" (16 USC 1131-1136). This paper examines this social "visitor experience" mandate relative to wilderness camping, as illustrated with data from the ISRO campsite survey.

Solitude at the Wilderness Campsite

Camping activities represent a significant component of the overall wilderness experience. The majority of a wilderness area visit may occur on the campsite, where parties interact with each other and the environment, cook, eat, sleep and engage in other spiritual or contemplative activities. The campsite itself represents a temporary home within the wilderness, where visitors perceive the existence of territorial boundaries isolating them from other people. Therefore, visitors are often less tolerant of contact with other visitors on or around their campsites than they are on common use areas like trails (Cole and others 1987) or destination areas (Cole and others 1997). The number of parties, group size and type of user group also affect visitor perceptions of acceptable numbers of encounters with other visitors on campsites (Roggenbuck and others 1993). For example, more people or certain types of groups may make more noise. In addition, different activity groups, such as non motorized and motorized users, may exhibit incompatible camping behaviors. In response to unwanted encounters in camping areas, visitors may engage in avoidance behavior, either by selecting campsites farther away from other occupied campsites or by choosing a more heavily screened campsite (Lee 1977).

Wilderness managers can directly or indirectly influence social settings and opportunities for camping solitude through their camping management policies, site selection criteria, site management practices and visitor education messages. Dispersed camping policies, for example, permit visitors to select camping areas or sites that potentially increase opportunities for solitude. However, management experience and research studies have shown that dispersal policies are

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generally ineffective, often because visitors fail to disperse very far from trails, other campsites or popular attraction features (Leung and Marion, in press). For example, a survey of backcountry and wilderness campsites at Shenandoah National Park found a large number of campsites ($n = 768$), two-thirds of which were illegal according to the park's dispersed camping regulations (Williams and Marion 1995). Conversely, containment camping policies, such as designated site camping, can restrict visitor freedom and may create or exacerbate problems with crowding and conflict.

Study Area

Isle Royale National Park, established in 1940, is located in the northwest corner of Lake Superior, 73 miles from Houghton, Michigan, and 22 miles from Grand Portage, Minnesota. The Park's terrain was formed by glaciers and includes exposed rocky ridges interspersed by numerous ponds and streams. One of the primary attractions and features of interest in the Park are its moose and wolf populations, but the island also supports many other wildlife and fish species (USDI 1994). Approximately 99% of the Park's land area is designated as wilderness. Because ISRO is managed as a wilderness area, pets and wheeled vehicles are prohibited in the Park, and no motorized vessels can travel on the inland lakes, with motorized boating permitted only on Lake Superior. The area was also designated as an International Biosphere Reserve in 1980.

The Park is open from mid-April until the end of October, with transportation from the mainland available by boat or floatplane. In 1996, the Park received approximately 13,000 visitors, with 54% primarily engaged in hiking, 31% in power boating, 9% in canoeing, 3% in sailing and 3% in kayaking (ISRO 1996). Backcountry visitation has been steadily increasing and, at over 50,000 overnights/year, ranks 10th among National Park Service (NPS) units (USDI 1996a). More importantly, ISRO has the highest number of backcountry overnights of all NPS units when figured on a per acre basis and considering that the Park is closed for half of each year.

Camping Policies and Regulations

Park camping policies and regulations require that visitors camp only at one of 36 designated campgrounds, which are accessed by hiking trails and/or boats. Campgrounds contain a combination of three-sided shelters, individual campsites or group campsites. Larger groups (7-10 individuals) must specify and adhere to an itinerary and camp only at group campsites; groups of six or fewer may use either shelters or individual sites on a first-come first-served basis. If a campground is full, visitors are advised to find alternate campgrounds or double up with other parties, as long as they do not exceed the site capacities. To reduce problems with crowding and conflict, visitors are also advised to use equipment with natural colors and to avoid unnecessary noise and other disruptive activities.

Methods

Conditions on all designated wilderness and non wilderness campsites were assessed during the summer of 1996. Elements of photographic, condition class and multi-indicator measurement-based approaches were combined for campsite inventory and impact assessments (Farrell and Marion 1998). This approach emphasizes field procedures that are efficiently applied yet yield reliable campsite condition measurements for a variety of campsite attributes. Inventory attributes included distance to nearest other campsite, distance to campground trail, number of other sites visible, site visibility from campground trail, site visibility from formal park trail, vegetation type, percent canopy cover and type of site use. Impact attributes included percent vegetative cover onsite and offsite, percent exposed soil, number of damaged trees onsite, number of tree stumps onsite, total campsite area, number of fire sites and number of human waste sites. A comprehensive procedural manual was developed to guide present and future field staff in taking consistent measurements.

Results and Discussion

Within the Park's 36 campgrounds, survey staff located and assessed 244 sites, including 113 individual campsites, 43 group campsites, and 88 shelters (hereafter referred to as sites). Site distribution between wilderness and non wilderness is approximately equal: 116 (48%) campsites and shelters are in wilderness and 128 (52%) are in non wilderness.

Campgrounds are located primarily around the island's perimeter. A principal advantage of this spatial arrangement is that it concentrates visitor activities, reducing human presence in large areas of the island's interior. Resource protection is enhanced by reducing wildlife habitat fragmentation and minimizing potential interference with wolves, moose and other wildlife. Site clustering also increases the efficiency of maintenance and visitor contact/enforcement activities and the provision of facilities like boat docks. However, site clustering also has negative aspects. While visitors have ample opportunities for experiencing solitude while hiking, the large number and close proximity of sites in many backcountry campgrounds reduces opportunities for solitude while camping. Site clustering gives visitors fewer options for designing alternative itineraries and less flexibility in altering travel plans while in the backcountry.

Natural Resource Protection

Data from the 1996 assessment of camping impacts reveal the success of camping containment at ISRO from the perspective of natural resource protection. Conditions on 211 sites (86%) were quite acceptable, with condition class ratings of 1, 2, or 3. The majority of sites were rated class 3, characterized by extensive organic litter and/or vegetation disturbance but with soil exposed only in primary use areas. Soil was exposed more extensively on only 33 sites (14%) and no sites were rated class 5, characterized by obvious soil

erosion. Median campsite size was only 554 ft² (23 x 23 ft), with an average disturbed area of 3.8 ft² per annual overnight stay. Similar data from other wilderness and backcountry areas indicates that these numbers are exceptionally low (Farrell and Marion 1998). For example, median size for designated campsites at Great Smoky Mountains National Park is 1,039 ft², with an average of 5.7 ft² disturbed area per annual overnight stay (Marion and Leung 1997).

Median percent vegetation loss on sites was 61% (mean = 62%). Nearly 80% of the sites lost more than 80% of their estimated original cover; vegetation loss of this magnitude is common on designated campsites. Conversely, the areal extent of vegetation loss was relatively small; 170 campsites (70%) lost less than 500 ft², with another 88 sites (36%) losing less than 250 ft². Area of exposed soil was also relatively small, ranging from 6 to 1,906 ft², with a median of 159 ft². Nearly two-thirds of the sites (65%) had less than 250 ft² of exposed soil, with the majority (82%) under 500 ft².

The principal factors for ISRO's success in limiting the areal extent of camping-related resource disturbance are campsite location and design. ISRO campgrounds are generally located in gently sloping terrain, where visitor activities are naturally constrained to the limited areas of flat ground on campsites. Most campsites consist of one to three tent pads created through cut-and-fill work to provide gently outsloped terraces. These campsite construction practices provide strong visual cues to identify the intended use areas. Campsites in flatter terrain are commonly outlined with embedded logs along at least two sides. In addition, many of the sites have been colonized by trampling-resistant grasses, at least in peripheral use areas. The obvious change in vegetation composition, from grasses to herbs, provides another visual cue demarcating site boundaries. Statistical analyses reveal that site facilities, such as shelters and picnic tables, also help to concentrate use and impacts (Farrell and Marion 1998).

Maintaining Desired Social Conditions

Although successful from the perspective of natural resource protection, camping activity containment has contributed to social problems at ISRO campsites. A survey of ISRO backcountry visitors, conducted by the University of Minnesota Cooperative Park Studies Unit, revealed that visitors consider both crowding and conflict at campgrounds to be salient issues. Crowding-related problems included "Seeing too many other hikers in the campgrounds" (ranked 2nd out of 64 items), "Being able to find a vacant shelter" (ranked 4/64), "Seeing too many other watercraft on Lake Superior" (ranked 5/64), "Finding an available campsite" (ranked 6/64) and "Campsites or shelters too close together in campgrounds" (ranked 13/64). Conflict-related problems included "Too much motorboat noise" (ranked 1/64), "Motorboat noise in narrow harbors and bays" (ranked 3/64), and "Noisy people at campgrounds with docks" (ranked 9/64). While most visitors did not consider these issues a problem, they remain highly ranked among the extensive list of potential issues provided for visitor comment (Pierskalla and others 1996).

ISRO recently completed the final version of its General Management Plan, during the process of which raised the following camping management concerns:

Visitors with different recreational objectives often find themselves in conflict, primarily at campgrounds. Increasing visitation is resulting in resource impacts and in crowding of some campgrounds, docks and trails...some visitors complain that there are too few backcountry campsites on the island, and they are concerned about having to share campsites (USDI 1996b).

Our survey data confirmed and explained these issues and concerns, discussed here in terms of crowding and conflict, and carrying capacity.

Crowding and Conflict—Crowding and conflict are expressed in our data by number of other sites visible, intersite distance, distance to campground trail, site visibility from campground and formal trails and type of campsite user (hiker, non motorized and motorized boaters). Generally, the overall potential for camping solitude is higher for wilderness campsites (N = 116) than nonwilderness campsites (N = 128). However, a review of data for these selected indicators reveals that users are still likely to experience crowding and conflict at either wilderness or non wilderness campsites.

The number of other sites visible from each campsite or shelter ranged from zero to six, with a mean of 1.8. Only 22 (9%) of the sites have no other sites visible, while 19 sites (8%) have four or more other visible sites (table 1). Three or more sites are visible from 46 (36%) of the nonwilderness sites, compared to only eight (7%) of the wilderness sites. For more than half of the wilderness sites, one or no sites are visible, compared to only one-third of the nonwilderness sites.

Intersite distance ranges from 0 to 334 feet, with a mean of 76 feet. In agreement with intersite visibility findings, intersite distances in wilderness areas range from 0 to 334 feet with a mean of 82 feet; in non wilderness areas, mean distance to the nearest other site is 71 feet. However, in wilderness areas, nearly one-third (27%) of campsites are within 50 feet of each other, while nearly three-quarters (73%) are within 100 feet of each other (table 1).

Distance to campground trail ranges from 0 to 352 feet with a mean of 64. The majority of sites (83%) were within 100 feet of a campground trail (table 1). In nonwilderness, campground trail distance was shorter (0 to 42 feet with a mean of 55 feet) than in wilderness (0 to 352 feet with a mean of 73 feet). However, within wilderness, 77% of the campsites are still within 100 feet of the campground trail.

Most sites (218, 89%) are visible from the campground trail (table 1). Of the 116 wilderness sites, 98 (85%) are visible from campground trails. Of the 128 nonwilderness sites, 120 (94%) are visible.

Conversely, a majority of sites are not visible from formal park trails (123 sites, 56%) (table 1). In wilderness, 38 sites (33%) are visible from formal park trails compared to 57 sites (45%) in nonwilderness.

Compared to other backcountry and wilderness areas, ISRO campsites are more densely packed together, with closer proximity and greater site intervisibility. For example, within the Jefferson National Forest, 59% of wilderness

Table 1—Number and percent of nonwilderness and wilderness campsites for indicators of social conditions.

Social indicators	Nonwilderness campsites (N = 128)		Wilderness campsites (N = 116)	
	Number	Percent	Number	Percent
Other sites visible (#)				
0	8	6	14	12
1	34	27	52	45
2	40	31	42	36
3	28	22	7	6
• 4	18	14	1	1
Distance to nearest other site (ft)				
0-50	51	40	31	27
51-100	55	43	53	46
101-150	13	10	21	18
151-200	4	3	7	6
>200	5	4	4	3
Distance to campground trail (ft)				
0-50	76	59	58	50
51-100	38	30	31	27
101-150	6	5	12	10
151-200	3	2	6	5
>200	5	4	9	8
Site visibility from campground trail				
Yes	120	94	98	85
No	8	6	18	15
Site visibility from formal trail				
Yes	57	45	38	33
No	70	54	53	46
Missing data	1	1	25	21

campsites have no other campsites visible, compared to only 12% at ISRO (Leung and Marion 1995). Similarly, 64% of backcountry campsites at Big Bend National Park and 21% of backcountry campsites at Great Smoky Mountains National Park have no other campsites visible (Williams and Marion 1997; Marion and Leung 1997).

Visitors have different expectations and behaviors that may lead to conflict between user groups, such as kayakers and motorboat users. A common method for addressing the problem of conflicting uses is to spatially separate different user groups. However, most of Isle Royale campgrounds may be easily accessed by water using canoes, kayaks and motorized boats and by land via hiking trails. Multiple access by boats and by trail is the most common access category (136 sites, 56%). In addition, wilderness boundaries stop at the shoreline, so visitors traveling by motorboat can easily access wilderness campsites. One-quarter of the wilderness sites (N = 30) are accessible by motorboats. At ISRO, a variety of different user groups must share common campgrounds, which lack clear distinctions between groups that may have incompatible behaviors, such as motorized and nonmotorized users.

Carrying Capacity—Visitor crowding and conflict problems at ISRO are further confounded by increasing use. Backcountry visitation has risen 37% over the past decade. Campground occupancy data indicates that most campground capacities (number of groups vs. number of campground sites) are exceeded on one or more nights each year, forcing groups to double up on campsites or create illegal

sites. Ten campgrounds exceeded their capacities (according to permit data) on more than 20 nights in 1995 (ISRO 1996).

High campsite occupancy rates indicate a number of potential problems. First, visitors who arrive at a full campground are more likely to be tempted to camp illegally, particularly if they are unable or unwilling to travel farther to another campground. Second, those who share campsites, as recommended by Park staff, degrade their experience and may contribute to site expansion. Third, visitors camping in full campgrounds may feel crowded or experience greater conflict. Interactions with others and noise levels are generally higher with higher densities of people, and the sense of being on a remote wilderness island is lost.

Management Recommendations and Conclusions

ISRO's visitor activity containment policies have been successful in limiting the areal extent of camping disturbance. However, high campsite densities have contributed to social problems of visitor crowding and conflict, which are further compounded by carrying capacity issues. Park managers and planners may wish to reexamine the current distribution of campsites and campgrounds as they affect current or desired visitor distribution patterns.

Relevant management recommendations to address social problems include the following: (1) visitor education programs encouraging visitors to select designated campsites

that are farthest away from currently occupied sites, (2) selected site relocations applying site selection techniques to increase intersite distances and (3) creating additional campsites within preexisting campgrounds located out of sight and at least 100 feet from existing sites. Carrying capacity concerns present a more difficult challenge. Options include (1) setting travel zone quotas to shift visitation in time or space to force a better match between the distribution of visitors and existing campsites, (2) constructing additional campsites in areas with perpetual shortages, or (3) limiting total visitation.

Visitor education programs like *Leave No Trace* have been developed to help managers prevent or reduce resource and social impacts. A park brochure could be developed to address specific camping management concerns like promoting solitude. Park staff could also remind visitors to select campsites that are farthest away from other parties.

In addition, standards for intersite visibility and distances should be considered to reduce the potential for crowding and conflict within wilderness. Examples include campsites not visible or at least 150 feet from formal park trails, intersite campsite distances of at least 50 feet and no more than one other site visible. Site selection criteria could then be applied by managers to select campsites that promote visitor solitude and close or discourage use of other campsites.

Creating additional campsites would reduce the potential for both crowding and conflict. Conflict problems at some existing campgrounds could be resolved by designating them for specific user types, such as campgrounds restricted to hikers or campgrounds restricted to powerboaters. This may necessitate the creation of additional campgrounds for the alternate use type.

Altering visitor distribution through time or space can address carrying capacity concerns. For example, in the Boundary Waters Canoe Area Wilderness (BWCAW), entry point quotas based on visitor travel models are used to maintain site occupancy rates of 60-85% in each travel zone. ISRO has relatively few backcountry entry points, and access to some is more difficult due to constraints on the frequency and timing of ferry boats. However, the BWCAW approach may still be feasible if boating schedules and access points could be altered to improve visitor distribution patterns relative to available campsites. This option allows visitors the freedom to travel where and when they want, a benefit which is largely offset by the "cost" of a greater area of disturbance associated with campground sites that go unused each night.

Additional campsites could also be constructed at campgrounds with overcapacity problems. Alternately, new campgrounds might be established in the vicinity of overcrowded campgrounds. The construction of new campsites or campgrounds would alleviate current and future overcrowding, but would also increase the area of disturbance associated with camping activities, and does not address concerns of future overcrowding.

Constructing additional sites to accommodate ever-increasing demand has been the traditional response of ISRO managers. However, it is appropriate to question this policy as it permits a potentially never-ending process of recreation expansion into previously undisturbed areas. Given the limited land area on the island and the sensitive issue of

fragmentation of wolf habitat, such a policy is ultimately non sustainable. Thus, limitation of backcountry visitation will ultimately need to be considered.

National Park Service backcountry and wilderness areas are administered under dual legal mandates that require managers to achieve an acceptable balance between resource protection and recreation provision. Some degree of environmental degradation is inevitable where recreational visitation is permitted. Managers are challenged to develop recreation resource management policies that can sustain both high quality recreational experiences and environmental conditions. Although ISRO has effectively minimized natural resource impacts via camping concentration, social problems like crowding, conflict and carrying capacity concerns require additional management actions to improve the quality of the visitor experience.

References

- Cole, David N. 1995. Disturbance of natural vegetation by camping: Experimental applications of low-level stress. *Environmental Management* 19(3):405-416.
- Cole, David N.; Watson, Alan E.; Hall, Troy E.; Spildie, D.R. 1997. High-use destinations in wilderness: Social and biophysical impacts, visitor responses, and management options. Ogden, UT: USDA Forest Service, Intermountain Research Station. 30p.
- Cole, David N.; Petersen, M.E.; Lucas, R.C. 1987. Managing wilderness recreation use: Common problems and potential solutions. Ogden, UT: USDA Forest Service, Intermountain Research Station. 60p.
- Farrell, Tracy A.; Marion, Jeffrey L. 1998. An evaluation of camping impacts and their management at Isle Royale National Park. Blacksburg, VA: USGS Patuxent Wildlife Research Center, Virginia Tech Cooperative Park Studies Unit. 97p.
- Isle Royale National Park, National Park Service, U.S. Department of the Interior (*unpublished data*). 1996. Isle Royale National Park visitor data. Houghton, MI: USDI National Park Service.
- Lee, Robert G. 1977. Alone with others: The paradox of privacy in wilderness. *Leisure Sciences*. 1(1): 3-19.
- Leung, Yu-Fai; Marion, Jeffrey L. 1995. A survey of campsite conditions in eleven wilderness areas of the Jefferson National Forest. Blacksburg, VA: USDI National Biological Service, Virginia Tech Cooperative Park Studies Unit. 79p.
- Marion, Jeffrey L.; Leung, Yu-Fai. 1997. An assessment of campsite conditions in Great Smoky Mountains National Park. Blacksburg, VA: USGS, Patuxent Wildlife Research Center, Cooperative Park Studies Unit. 127p.
- Leung, Yu-Fai; Marion, Jeffrey L. In Press. Spatial strategies for managing visitor impacts in National Parks. *Journal of Park and Recreation Administration*
- Pierskalla, C.; Anderson, D.H.; Lime, D.W. 1996. Isle Royale National Park 1996 visitor survey. Draft Report to the USDI National Park Service, Denver Service Center and Isle Royale National Park. St. Paul, MN: University of Minnesota, Department of Forest Resources, Cooperative Park Studies Unit. 133p.
- Roggenbuck, J.W.; Williams, D.R.; Watson, A.E. 1993. Defining acceptable conditions in wilderness. *Environmental Management*. 17(2): 187-197.
- Williams, Peter B.; Marion, J.L. 1995. Assessing campsite conditions for Limits of Acceptable Change management in Shenandoah National Park. Blacksburg, VA: USDI National Park Service, Virginia Tech Cooperative Park Studies Unit. 138p.
- U.S. Department of the Interior, National Park Service. 1994. Isle Royale National Park - Resources Management Plan. Houghton, MI.
- U.S. Department of the Interior, National Park Service. 1996a. 1996 Statistical abstract. Denver, CO : National Park Service Public Use Statistics Program Center.
- U.S. Department of the Interior, National Park Service. 1996b. Isle Royale National Park - General Management Plan Newsletter 3.

Managing Coastal Recreation Impacts and Visitor Experience Using GIS

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Abstract—A campsite monitoring program was initiated in Gwaii Haanas National Park Reserve/Haida Heritage Site to determine baseline levels of visitor impacts. These data were necessary to evaluate visitor management strategies and to act as reference points to measure changes in impacts over time. Using GIS, survey data were integrated with an ecological land classification, archaeological databases and a visitor use database. Analyses showed that although the campsites impacted only 0.0007% of the land base, 53 of the 75 campsites were ranked as either extremely or highly sensitive to human disturbance. The implications of this information to visitor management are discussed.

Gwaii Haanas National Park Reserve/Haida Heritage Site is a 1,475 square kilometer (570 square mile) wilderness area located on the southern end of Haida Gwaii/Queen Charlotte Islands, in British Columbia, Canada (fig. 1). Access to the area is by water or air only, as no roads exist. Gwaii Haanas is cooperatively managed by Parks Canada and the Council of the Haida Nation; two members from each organization form the Archipelago Management Board (AMB), which is responsible for all aspects of planning, management and operation of Gwaii Haanas.

In 1995, the AMB became concerned that it did not have a good understanding of the level of impacts related to camping activities in Gwaii Haanas. Based on this uncertainty, the AMB decided that a proactive strategy had to be developed to ensure that visitor use of the area was not significantly impacting ecological and cultural heritage. The strategy had two facets:

1. Freeze visitor use at current levels until a baseline of visitor impacts could be determined.
2. Initiate a campsite monitoring program to determine the extent of impacts and monitor changes to those impacts over time.

Since that time, the AMB has also initiated the development of a backcountry management plan, and decisions

related to camping activities rely heavily on the results of this monitoring program. This paper summarizes the baseline data collected at 75 sites between 1996 and 1998, which were integrated into Gwaii Haanas' geographic information system (GIS); it also itemizes the recommendations which formed part of the backcountry management plan.

Camping Behavior in Gwaii Haanas

Although Gwaii Haanas currently has a random camping policy, certain sites have become heavily used, creating visible impacts. Based on research conducted by Vaske and

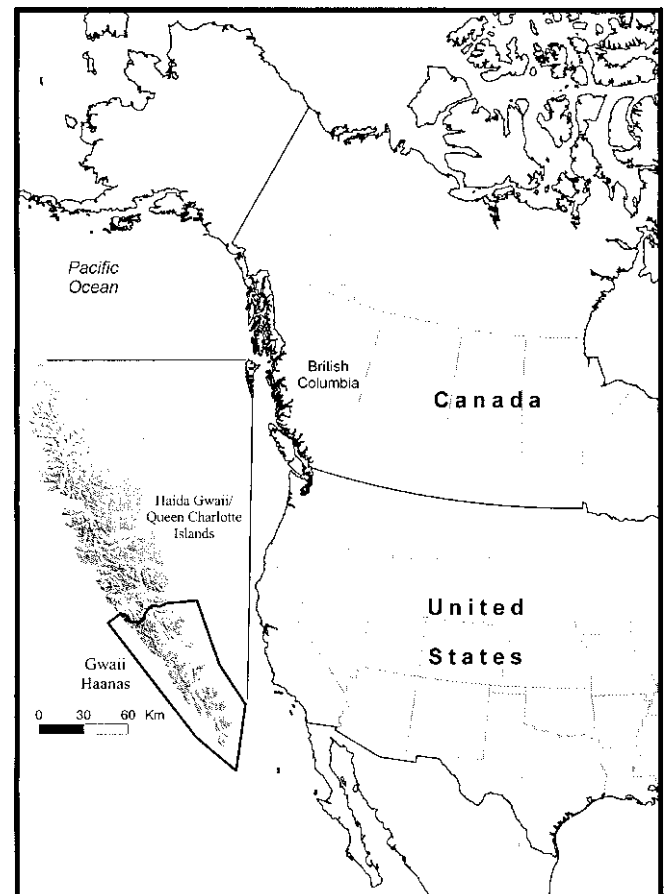


Figure 1—Location of Gwaii Haanas National Park Reserve/Haida Heritage Site.

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others (1996), three main types of “visitor experience” areas have been defined in Gwaii Haanas:

1. Access areas: areas that are particularly safe entry and departure points.
2. Attraction sites: areas that offer unique opportunities for education, spiritual introspection and the solitary or shared experience of a special place and living Haida culture.
3. Wild places: the majority of Gwaii Haanas, which can be defined as wilderness.

Most independent visitors and commercial tour guides plan their trips around pick-up and drop-off schedules at access areas, as well as visits to specific attraction sites. Thus, many groups may stay at or near these areas to facilitate access, resulting in multiple use of campsites throughout the summer. In addition, many guides establish standard routes for their tours and camp repeatedly on the same sites. This is particularly the case when large groups are involved, since sites with adequate space for multiple tents are uncommon.

Independent visitors (those who travel without the assistance of a licensed guide) who do pre-trip research or inquire with others familiar with Gwaii Haanas, such as their transport company, may also find out where the most favorable campsites are located. They then plan their travel routes and overnights based on that information. Therefore, some sites are receiving a lot of use based on ‘local knowledge’. In addition, topography dictates that not all areas are viable campsites. Therefore, the random camping policy is more accurately described as an ‘undesigned’ camping policy.

Methodology

Development of Ecological Campsite Monitoring Methodology

Dr. Jeffrey Marion and Tracy Farrell of Virginia Technical University developed the ecological monitoring methodology, based on field testing in Gwaii Haanas, peer review and staff input (Marion and Farrell 1996). Modification of existing monitoring techniques by Gwaii Haanas staff tailored the methodology to the Gwaii Haanas environment.

Campsites were identified by field staff based on local knowledge of previous camping activities.

Development of a Cultural Heritage Monitoring Methodology

Parks Canada staff developed a framework for monitoring cultural heritage at campsites. The methodology was based on the Cultural Resource Management Policy (Parks Canada 1994), the Gwaii Haanas Draft Terrestrial Area Strategic Management Plan (Archipelago Management Board 1996) and standardized archaeological methods. Presentation of the framework at a Parks Canada wilderness conference and a cultural resource management workshop provided review prior to commencement of monitoring.

Data Collection and Analysis

The methodology for this project requires that each campsite be evaluated a minimum of two times. The initial evaluations of the campsites were conducted in August of 1996, 1997 and 1998. This established the baseline data for the project, which included the campsite’s location, present condition and sensitivity to impact. Subsequent evaluations will be compared to the baseline data to monitor change over time. A multidisciplinary team consisting of Parks Canada wardens, patrol officers and archaeologists performed the initial fieldwork for each individual site.

Inventory/Impact Parameters

Measurements of physical attributes were taken at each site (fig. 2), a permanent pin was placed at the centre of the site’s primary use area, and photographs and videotape were taken as visual records of current conditions. At many campsites, there were several distinct use-areas (fig. 3), which required that each use-area be surveyed separately. The variable radial transect method was chosen to measure the area of impact. A sighting compass and a Sonin measuring device or metric measuring tape (when rain prevented use of the electronic device) were used to record bearings and distances from a central point to points on the perimeter of each use-area (fig. 4). All sites were georeferenced using the 1:20,000 Gwaii Haanas base map, which employed a spatial referencing system based on a UTM extended zone 9 projection and an NAD 1983 Datum.

Analyses of the data were done to determine the total and the average level of physical impacts at campsites. Median values were used because they provide better estimates of central tendency when the effect of outliers are present, as is the case with most campsite monitoring data (Marion 1994).

Condition Classes

Campsite condition class ratings were assigned to provide an overall picture of each campsite’s condition (table 1). Again, each use-area was surveyed separately. Because the condition class ratings are category variables, it was not possible to determine a mean or median condition class for each campsite. The use-area with the highest condition class was chosen to represent the campsite as a whole, in order to err on the side of caution when identifying existing impacts, as well as monitoring impact changes.

Ecological Features

Impacts to ecological features were estimated by overlaying data collected at each campsite with the Gwaii Haanas Ecological Land Classification (Westland Resources Group 1994) database on GIS. Indicators were sensitive ecosystems, erosion-prone terrain, seabird colonies, Peale’s peregrine falcon (*Falco peregrinus pealei*) aeries, bald eagle (*Haliaeetus leucocephalus*) nests, harbor seal (*Phoca vitulina*) and Steller sea-lion (*Eumetopias jubatus*) haul-outs and rookeries and salmon (*Oncorhynchus* sp.) streams.

Campsite Description Form

Campsite Number: _____ Date: _____ Inventoried By: _____

Located on Map? ☐ Map Code: _____

Location Description: _____

Primary Pin Reference Information

Primary Pin Location Coordinates from (GIS, GPS, Map)	Description	Landmarks	Bearing	Distance (m)
1. Easting: _____ North: _____	1. _____	_____	_____	_____
2. _____	2. _____	_____	_____	_____
3. _____	3. _____	_____	_____	_____

Inventory/Impact Parameters

Substrate of Landing Area (B = bedrock, C = cobble, S = sand, D = dirt/soil) ☐

Shoreline Developments (#) Tables ☐ Benches ☐ Chairs ☐ Floats ☐

Other Campsites Visible (#) ☐

Tree Canopy Cover (1=0% 2=1-5% 3=6-25% 4=26-50% 5=51-75% 6=76-95% 7=96-99% 8=100%) ☐

Overstory Species: Dominant ☐ Codominant ☐ Codominant ☐

Vegetation Groundcover Onsite (M=Moss H=Herbs G=Grasses/sedges O=Organic litter S=Shrubs) ☐

Shoreline Condition Class (m) Class 1: ☐ Class 2: ☐ Class 3: ☐

Trails By Condition Class (m) Class 1: ☐ Class 2: ☐ Class 3: ☐

Fire Sites (#) Campsite: ☐ Beach: ☐ Litter/trash: ☐

Exotic Plants

☐ <i>Cirsium brevistylus</i> Short-Styled Thistle	☐ <i>Cirsium vulgare</i> Bull Thistle	☐ <i>Cirsium arvense</i> Canada Thistle	☐ <i>Aira praecox</i> Early Hairgrass
☐ <i>Digitalis purpurea</i> Foxglove	☐ <i>Urtica dioica</i> Stinging Nettle	☐ <i>Matricaria matricarioides</i> Pineapple Weed	

Comments: _____

Figure 2—General parameters measured for Gwaii Haanas campsites.

Campsite Description Form

Campsite Number: _____ Date: _____ Inventoried By: _____

UA #: _____ From: _____

Bearing: _____ Distance (m): _____

Reference Points: 1. _____ 2. _____ 3. _____

Boundary Points: 1. _____ 2. _____ 3. _____ 4. _____ 5. _____ 6. _____ 7. _____ 8. _____

Condition Class (0-5) ☐

On/Off ☐

Vegetative Ground Cover ☐

Unconsolidated Organic Duff ☐

Consolidated Organic Duff ☐

Exposed Primary Substrate ☐

UA #: _____ From: _____

Bearing: _____ Distance (m): _____

Reference Points: 1. _____ 2. _____ 3. _____

Boundary Points: 1. _____ 2. _____ 3. _____ 4. _____ 5. _____ 6. _____ 7. _____ 8. _____

Condition Class (0-5) ☐

On/Off ☐

Vegetative Ground Cover ☐

Unconsolidated Organic Duff ☐

Consolidated Organic Duff ☐

Exposed Primary Substrate ☐

UA #: _____ From: _____

Bearing: _____ Distance (m): _____

Reference Points: 1. _____ 2. _____ 3. _____

Boundary Points: 1. _____ 2. _____ 3. _____ 4. _____ 5. _____ 6. _____ 7. _____ 8. _____

Condition Class (0-5) ☐

On/Off ☐

Vegetative Ground Cover ☐

Unconsolidated Organic Duff ☐

Consolidated Organic Duff ☐

Exposed Primary Substrate ☐

UA #: _____ From: _____

Bearing: _____ Distance (m): _____

Reference Points: 1. _____ 2. _____ 3. _____

Boundary Points: 1. _____ 2. _____ 3. _____ 4. _____ 5. _____ 6. _____ 7. _____ 8. _____

Condition Class (0-5) ☐

On/Off ☐

Vegetative Ground Cover ☐

Unconsolidated Organic Duff ☐

Consolidated Organic Duff ☐

Exposed Primary Substrate ☐

UA #: _____ From: _____

Bearing: _____ Distance (m): _____

Reference Points: 1. _____ 2. _____ 3. _____

Boundary Points: 1. _____ 2. _____ 3. _____ 4. _____ 5. _____ 6. _____ 7. _____ 8. _____

Condition Class (0-5) ☐

On/Off ☐

Vegetative Ground Cover ☐

Unconsolidated Organic Duff ☐

Consolidated Organic Duff ☐

Exposed Primary Substrate ☐

UA #: _____ From: _____

Bearing: _____ Distance (m): _____

Reference Points: 1. _____ 2. _____ 3. _____

Boundary Points: 1. _____ 2. _____ 3. _____ 4. _____ 5. _____ 6. _____ 7. _____ 8. _____

Condition Class (0-5) ☐

On/Off ☐

Vegetative Ground Cover ☐

Unconsolidated Organic Duff ☐

Consolidated Organic Duff ☐

Exposed Primary Substrate ☐

Coverage Categories (0%, 1-5%, 6-25%, 26-50%, 51-75%, 76-95%, 96-99%, 100%)

Midpoints: 0 3 16 38 63 86 98 100

Figure 4—Use-area parameters measured for Gwaii Haanas campsites.

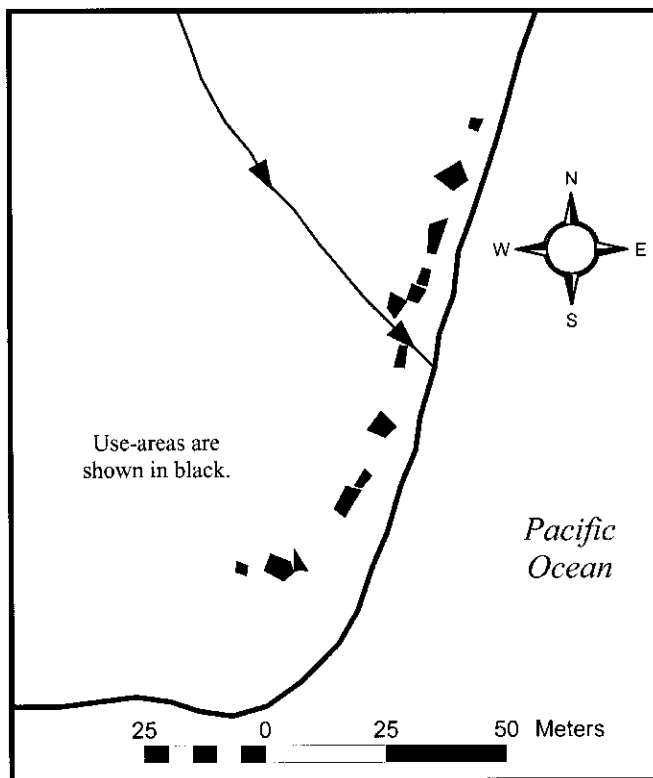


Figure 3—Example of use-areas that compose a single campsite.

Sensitivity to erosion rankings was based on GIS analysis, using the biophysical inventory and ground truthing. Presence of peregrine falcon aeries within 1 km increased a site's sensitivity rating, as falcons aggressively defend nesting areas and may abandon their nests if disturbed. Eagles are more tolerant of disturbance than peregrine falcons, but the extent of their tolerance is unclear. Thus, campsites were identified as being within either 500 m (highly sensitive) or 250 m (extremely sensitive) of an eagle nest. Seal and sea

Table 1—Condition class rating definitions used for the Gwaii Haanas campsite monitoring program.

Class	Definition
0	Campsite barely distinguishable; no or minimal disturbance of vegetation and/or organic litter. Often an old site that has not seen recent use.
1	Campsite barely distinguishable; slight loss of vegetation cover and/or minimal disturbance of organic litter.
2	Campsite obvious; vegetation cover lost and/or organic litter pulverized in primary use areas.
3	Vegetation cover lost and/or organic litter pulverized on much of the site; some bare soil exposed in primary use areas.
4	Nearly complete or total loss of vegetation cover and organic litter; bare soil widespread.
5	Soil erosion obvious, as indicated by exposed tree roots and rocks and/or gullyng.

lion haul-outs were noted if they were in the general vicinity of the campsites. A complete inventory of sensitive seal and sea lion habitat has not yet been completed, but integration of future inventories with campsite information should yield a better understanding of potential impacts to these species. The presence of salmon streams was recorded if they were within or 100 m (extremely sensitive) or 250 m (highly sensitive) of a campsite. The sensitivity was not related to the salmon themselves, but to the presence of black bears (*Ursus americanus carlottae*) feeding on the fish during spawning season.

An overall ranking of ecological sensitivity was also determined to establish priorities for management action. The qualitative categories are provided in table 2.

Cultural Features

Parks Canada's extensive archaeological databases were available on GIS and facilitated the analysis of cultural heritage site sensitivity. An archaeologist inventoried each campsite, conducted a literature review and provided recommendations to minimize camping impacts. The sensitivity and significance of the archaeology and spirituality of the sites were recorded after consultation with Haida elders.

Cultural heritage features at or near the site were identified, and these included:

- Historic Haida village sites, burial caves, human remains.
- House pits, house beams, habitation sites, campsites.
- Canoe runs, fish weirs, culturally modified trees.
- Fire broken rock, cultural rock mounds.
- Terrestrial and intertidal lithic shell middens.

As with ecological sensitivity, an overall ranking of cultural sensitivity was determined for each site. The same qualitative categories that were used for ecological sensitivity (table 2) were used to rank the overall cultural sensitivity of each campsite.

Visitor Use Levels

In addition to the ecological and cultural data, visitor use data were used to aid in the overall ranking of the individual campsites. Beginning in 1998, both commercial

operators and independent visitors submitted trip logs, which included maps to identify campsites used on trips. The total number of user-nights was calculated for each campsite, and these data were correlated with levels of impact at specific campsites surveyed in that same year. In the future, it may also be possible to relate impacts to cumulative use over time, but current data are insufficient to do such an analysis.

Cumulative Impacts

This assessment was done using results from the previous five analyses. The objective was to estimate the overall extent of impacts to ecological features, cultural features, and visitor experience and to identify sites that were extremely or highly sensitive to impacts. Sites that were ranked as extremely or highly sensitive became the priority sites for management action.

An extremely sensitive site was one that triggered one of the following criteria:

- Received an extreme rating for any ecological indicator, for spiritual sensitivity or significance or for archaeological sensitivity or significance; or
- Had a weighted median condition class of 3 or greater, received more than 150 user-nights/year and was located on a sensitive ecosite or had an impact area of greater than 110 m² and received more than 150 user-nights/year.

In comparison, the criteria for a highly sensitive site were:

- Received a high rating for spiritual sensitivity or significance or for archaeological sensitivity or significance.
- Had a weighted median condition class of 3 or greater or had a weighted median condition class of 2 and had received more than 100 user-nights.

Results and Discussion

Seventy-five campsites were monitored through the 1996, 1997 and 1998 field seasons (fig. 5). The majority of the campsites were located along the relatively protected east coast, with the remainder in the Houston-Stewart Channel area. The east coast receives the majority of use because its numerous bays and inlets provide more protection from Gwaii Haanas' unpredictable weather. There are not as many suitable camping locations along the west coast of Gwaii Haanas because it is a steep and rocky coastline, with pounding surf and lengthy stretches of water where landing is not possible. Very few user-nights are spent on the west coast, and thus campsite monitoring is currently not being done in that portion of Gwaii Haanas.

Inventory/Impact Parameter Analysis

Table 3 provides a summary of the cumulative inventory/impact measurements at the 75 campsites measured between 1996 and 1998. The total area in itself (1 ha, or 0.0007%) is not significant relative to the entire land mass of Gwaii Haanas. The median number of use-areas per campsite was five, with a median cumulative impact area of 13.21m² (table 4). The use-areas had a median loss of

Table 2—Categories of overall ecological sensitivity for Gwaii Haanas campsites.

Sensitivity category	Definition
Extreme	At least one indicator is rated as extremely sensitive, and site visit verified that there are significant impacts or potential for significant impacts.
High	The majority of indicators are rated as highly sensitive, and site visit verified that there are significant impacts or potential for significant impacts.
Medium	The majority of indicators are rated at medium sensitivity, and site visit verified that there are moderate impacts or potential for moderate impacts.
Low	The majority of indicators are rated at low sensitivity, and site visit verified that there are minimal impacts or minimal potential for impacts.

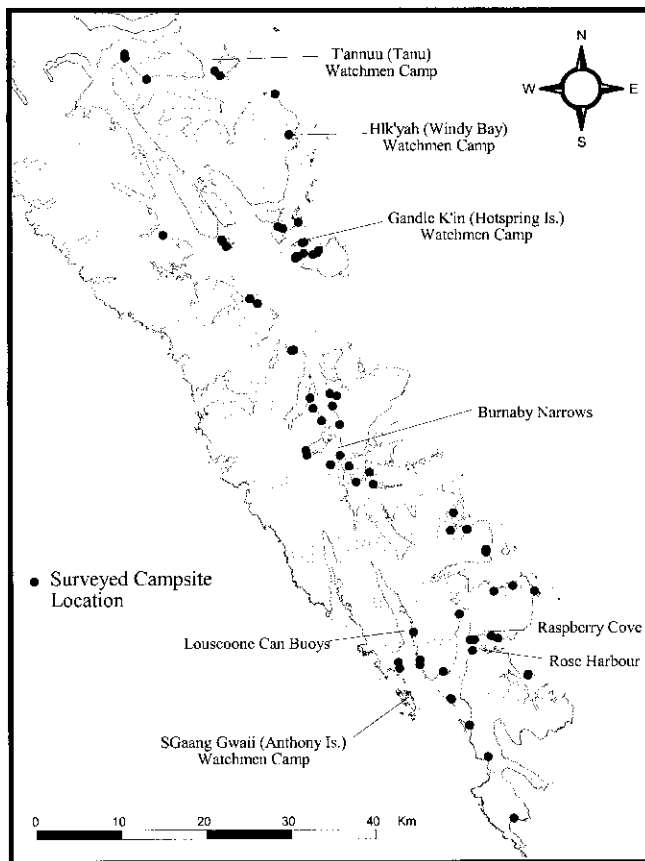


Figure 5—Map of Gwaii Haanas showing 75 surveyed campsites in relation to attraction sites and access areas.

0.72 m² of vegetation and resulted in a median of no exposed substrate. The median differences in the percentages of loose (unconsolidated) organic duff and consolidated organic duff were 35% and 0%, respectively. This indicates that there was 35% less loose organic duff (such as twigs and leaves) on the use-areas compared to the immediate vicinity of the use area. This is generally a result of the organic duff material being pushed off to the boundaries, making the use-area clear of any debris. The value of zero for consolidated organic duff indicates that the majority of the sites were still covered with loose material or vegetation, thus preventing the damage to the consolidated duff layer.

Table 3—Cumulative inventory/impact parameter measurements of Gwaii Haanas campsites.

Factor	Cumulative total impact
Number of camping areas	75
Number of use-areas	453
Campsite area	10352.75 m ²
Vegetation loss	2023.65 m ²
Exposed soil	498.91 m ²
Shoreline disturbance	270.62 m ²
Fire rings	83

Table 4—Median inventory/impact values calculated for Gwaii Haanas campsite use-areas.

Criteria	Median value	Range
Number of use-areas per campsite	5	1 to 23
Area of use-areas	13.21 m ²	2.51 to 255.68 m ²
Area of vegetation loss	0.72 m ²	0 to 214.77 m ²
Area of exposed 1° substrate	0 m ²	0 to 98 m ²
Percent difference in loose organic duff	–35	0 to –98
Percent difference in consolidated organic duff	0	0 to –95

In reviewing the cumulative impact of all use-areas per site, an average campsite covered a median area of 106 m², but had no human-caused shoreline disturbance (table 5). The median number of human developments (beach furniture and fire rings) were zero and one, respectively.

Condition Class Assessment

The use-areas had a median condition class rating of 2, but a substantial proportion had a ranking of 3 (table 6). The median condition class rating of 2 indicates that the campsites were generally obvious—that is, there was some lost vegetation cover and/or pulverized organic litter in the primary use areas.

The condition class descriptors (table 1) were compared to the vision for the Gwaii Haanas draft strategic management plan, which states “... visitors from all over the world begin to arrive. Each one of them shares the sensation of being the first person to set foot here.” Based on this comparison, it was decided that a condition class rating of 0 or 1 meets the plan’s vision for environmental protection and visitor experience. Sites with a condition class rating

Table 5—Median criteria values obtained for Gwaii Haanas campsites.

Criteria	Median value	Range
Total campsite area	106.02 m ²	6.53 to 367.06 m ²
Shoreline disturbance	0 m	0 to 112 m
Pieces of beach furniture	0	0 to 41
No. fire rings	1	0 to 4

Table 6—Condition class ratings for all use-areas.

Condition class	Number of use-areas	Percentage of total use-areas
0	63	13.9
1	148	32.7
2	133	29.4
3	105	23.1
4	4	0.9
5	0	0

of 2 are acceptable, but sites with a rating of 3, 4 or 5 require management action.

Approximately 24% of the individual use-areas exceed the acceptable condition class of 2. There were, however, only four use-areas rated at a condition class of 4, where complete loss of vegetation has occurred and bare soil has been exposed on the majority of the site. These use-areas are located in four separate campsites, which are associated with an attraction site. In addition, two sites are access areas where visitors often begin and end their trips. This confirms that campsites near attraction sites and access areas experience more of an impact than average campsites in Gwaii Haanas.

The use-area with the highest condition class was chosen to represent the campsite as a whole. To use a simple evaluation of central tendency would misrepresent the extent of impact as it relates to the management goal, because many, slightly impacted use-areas could mask the presence of one extensively impacted use-area. The measure also would not be very sensitive to changes, as the condition of most use-areas would have to increase in order to register an increase in campsite condition. Choosing the highest condition class provides management with a more sensitive indicator of changes in campsite condition.

The condition classes for the 75 campsites are provided in table 7. Based on this analysis, 52% of the campsites have use-areas that exceed the acceptable impact standards of Condition Class 2 and thus require some level of management action.

Ecological Assessments

Although the campsites only covered 0.0007% of Gwaii Haanas, this may represent a significant portion of sensitive ecosites. Queries of the Gwaii Haanas biophysical inventory, using GIS, provided information regarding the relationship of campsites to sensitive ecological heritage. Table 8 summarizes the number of campsites that were rated in each category.

Analysis to determine overall ecological sensitivity showed that four (5%) campsites were rated as being extremely sensitive to ecological impact, and 26 (35%) were ranked as being highly sensitive. In addition, four sites that received a overall ranking of "medium" were located in sensitive ecosites with a total area of less 100 ha (considered rare), and thus were given additional consideration.

Table 7—Distribution of Gwaii Haanas campsites according to condition class.

Condition class	Number of campsites	Percentage of campsites
0	3	4
1	18	24
2	16	21
3	36	48
4	3	4
5	0	0

Table 8—Overall ecological sensitivity ratings for Gwaii Haanas campsites.

Ranking	Number of campsites	Percentage of campsites
Extreme	4	5
High	26	35
Medium	34	45
Low	11	15

Cultural Assessment

Archaeological assessments reveal that 77% of the campsites monitored are associated with known cultural heritage. Twenty-six campsites are associated with extremely or highly sensitive cultural heritage (table 9).

Visitor Use Assessment

For the purposes of this report, a campsite was considered 'high use' when it received 100 or more user-nights in a season. At this stage of the monitoring program, this number is relatively arbitrary, but it was decided to choose a figure and refine it as more information becomes available to correlate use levels to levels of impact.

In 1997, the median number of user-nights for campsites was 27, with a range of 0 - 472. In 1998, the median increased slightly to 32 user-nights, while the range decreased to 0 - 273. It is important to note that the number of user-nights is a conservative estimate, since only about 40% of independents return trip logs indicating overnight locations. In addition, the trip log maps were small-scale, and thus there are unknown errors related to the accuracy of where people indicated their campsites. If a campsite was mapped within 200 m of a surveyed campsite's primary pin, the user-nights associated with that mapped site were counted under the surveyed site. Clearly, then, the missing independent trip logs could result in an underestimate of user-nights, while the inclusion of user-nights within 200 m of the primary pin could result in an overestimate. These shortcomings are recognized, and work is in progress to refine the collection of campsite locations from users.

There are several campsites that receive substantially higher than average use throughout the season. The user-night distribution pattern for the 1998 season (fig. 6) demonstrates that each of the high-use campsites were closely associated with an attraction site (SGaang Gwaii, Burnaby

Table 9—Overall cultural sensitivity ratings for Gwaii Haanas campsites.

Ranking	Number of campsites	Percentage of campsites
Extreme	6	8
High	20	27
Medium	23	31
Low	12	16
To be determined	14	19

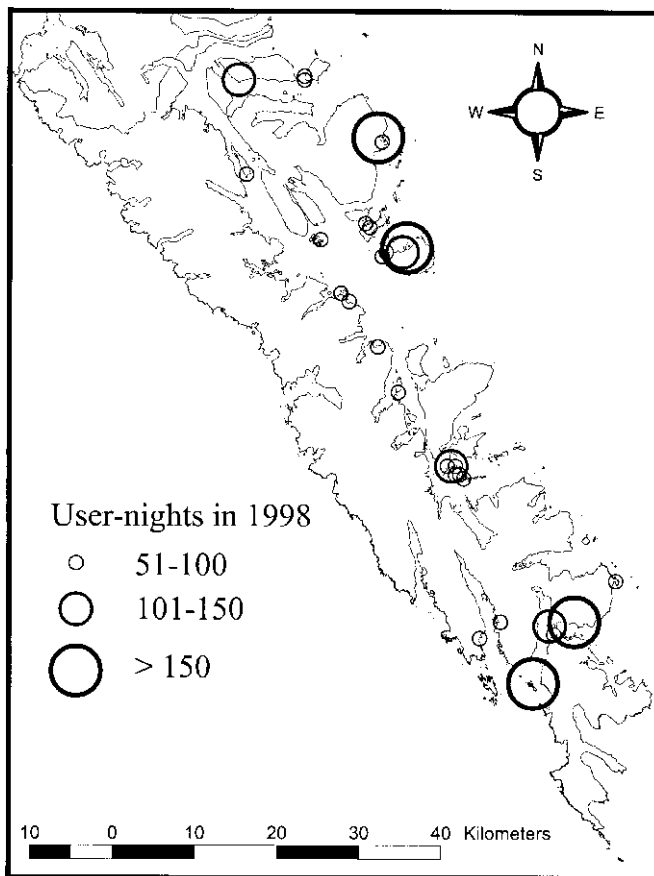


Figure 6—Campsites with high use in 1998 typifying concentration of camping activities at certain sites.

Narrows, Hotspring Island, Windy Bay and T'aanuu) or with an access site (Louscoone Inlet, Rose Harbour, Raspberry Cove).

Cumulative Impact Assessment

Based on the methodology described earlier, the cumulative impacts from camping in Gwaii Haanas are extreme or high at 52 of the 75 surveyed sites (fig. 7). Although only a small fraction of Gwaii Haanas is being impacted by camping, that activity is concentrated in areas that are either sensitive to use, or at levels where impacts exceed acceptable standards set by management.

Management Recommendations

Based on the cumulative analysis, management action must be taken at a majority of the surveyed campsites. Because each site has unique characteristics, it is not possible to develop a general management strategy that can be applied equally to all sites. Use levels have already been limited in Gwaii Haanas, and 37 of the campsites of concern are identified as such due to ecological or cultural sensitivity—that is, any use at all is a concern. Therefore, each site must be evaluated separately to determine what management action is appropriate.

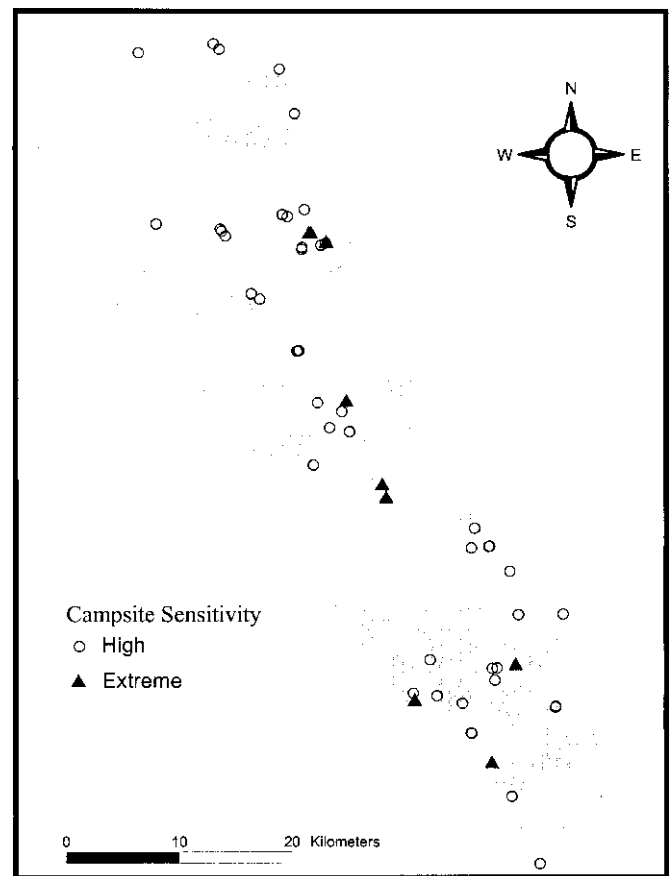


Figure 7—Surveyed campsites that were ranked as being either extremely or highly sensitive to human disturbance.

There are six general management prescriptions that can be applied to each site:

1. Accept current level of impact.
2. Actively restore the site.
3. Close temporarily (for example, when surface-nesting birds are breeding).
4. Close temporarily until site recovers to acceptable level.
5. Close permanently.
6. Harden site.

Site hardening will be considered a last resort, as it conflicts with the Gwaii Haanas strategic management plan's goal of minimizing infrastructure in this wilderness environment.

In developing prescriptions at each site, consideration must be given to public safety. Management is encouraging people to visit sites such as Sgaang Gwaii and Gandle K'in (Hotspring Island). Travel to these sites involve a committed crossing for kayakers, who do a majority of the camping in Gwaii Haanas. Since winds generally increase during the afternoon, many kayak groups try to travel to attraction sites in the morning or early evening. To minimize travel time, therefore, camping will occur close to the attraction sites. If management forced a reduction of use close to attraction sites (and similarly to access areas), it would potentially be increasing visitors' risk by requiring longer travel times in suboptimal conditions.

The results of the campsite inventory provide management with a better understanding of visitor preferences for choosing campsites. This is also valuable management information, since it provides the AMB with a better understanding of the limitations related to finding appropriate camping areas. The following is a list of general campsite characteristics for Gwaii Haanas.

1. Campsites have a medium gradient gravel beach that allows for relatively easy access to the adjacent flat terrain suitable for camping at all tide levels.

2. Campsites are made up of a number of dispersed/decentralized use-areas linked by trails. This configuration permits camping with forest protection during poor weather and use of open areas for tenting, cooking and group gatherings when weather conditions are favorable. Use-areas are typically round or simple polygon shapes that are just slightly larger than the footprint of a lightweight three-person tent. This is consistent with an area of disturbance of a tent and gear storage.

3. The greatest amount of ground disturbance occurs at kitchen areas and at tent entrances. The kitchen area tends to be a place where groups gather, with a lot of movement occurring during setup, food preparation and cooking. The movement can quickly scuff away delicate surface vegetation, like mosses. This may be of particular concern for larger groups or when longer stays occur in one campsite. As people enter and exit through tent entrances vegetation, can be scuffed away.

4. Visitors tend to camp in locations where distances from landing to camp are as short as possible, since the activities of packing, unpacking and the hauling of gear are repeated many times during a multiday coastal camping trip. Therefore, campsites tend to be developed close to high tide marks.

5. Availability of fresh water is a consideration but not a necessity, as most visitors carry their own water supply. When a water supply is available, campsites are generally as close as possible to that supply.

6. Protection from the elements is a preferred quality in a campsite. It may provide a break from the wind and rain, or a sheltered harbor free of driving surf. Exposure to the weather and sea conditions, in addition to the type of shoreline leading to a campsite, all affect the degree of risk for accessing or departing from a campsite.

7. A beach consisting of sand, gravel or cobble with a gradient that allows for convenient access to the campsite at all tide levels is preferred. If the gradient is too shallow, a visitor would have a long to haul to get their gear and boats to the campsite. Too steep a gradient makes landing difficult. Visitors normally attempt to time arrival and departure from campsites at high tide, thus reducing the distance gear must be hauled from boats to camp. However, depending on travel routes, it may be necessary to do the exact opposite in order to take advantage of favorable tidal currents.

8. Sites that can accommodate larger groups are more limited, and thus impacts are more extensive (a) because of an increase in use-areas and (b) because the limited nature of this type of campsite results in higher reuse.

Considering the public safety issues, visitor behavior patterns and preferred campsite characteristics, it is clear that random camping is not occurring in Gwaii Haanas. It is

also clear that moving to a strict designated camping policy has significant public safety implications and could cause people to push themselves to reach a particular site, rather than stopping whenever they are tired or the weather worsens. The potential for visitor conflicts also increases, since limiting the number of sites would force increased contact among groups.

As the AMB evaluates each campsite that has been given an extreme or high ranking for overall sensitivity, consideration will be given to the realities of topography, weather and visitor behavior. The latter can be modified to some extent through the visitor orientation program (random camping messages have been enforced since this program was initiated in 1996). However, research on visitor behavior in Gwaii Haanas (Vaske and others 1996) indicates that a majority of visitors prefer designated camping to minimize impacts, as opposed to dispersing camping to achieve the same objective. One option may be to accept higher levels of impacts in "zones" surrounding attraction sites and access areas, while keeping the original standard (condition class 2) for the remaining portions of Gwaii Haanas. If this were done, the AMB may consider designating some campsites in these zones and encourage visitors to restrict their camping activities to these areas in order to minimize overall impacts. Outside these zones, visitors would continue to be encouraged to camp in areas where there is no evidence of previous camping activities to keep use levels, and therefore impacts, minimal.

Summary

The Gwaii Haanas campsite monitoring program has provided valuable information in assessing the impacts of visitor activities on the ecological and cultural heritage of the area, as well as management's ability to provide a high-quality wilderness experience to its visitors. Although a random camping policy encourages visitors to camp where there is no previous evidence of use, baseline data show that visitors frequently reuse the same sites due to proximity to attraction sites or access areas, or to favorable characteristics of the campsite itself.

The monitoring protocol was developed using both quantitative and qualitative variables in order to provide a comprehensive picture of current conditions. Analysis of these baseline data has been critical to the development of management strategies for visitor use in Gwaii Haanas, and resurveying of these sites in the future will provide information to determine if management objectives are being met. In the analyses presented in this paper, the qualitative factors have played a predominant role. This caused difficulties in analysis, since qualitative factors generally cannot be analyzed statistically. Therefore, there remains significant subjectivity in interpreting the results. Ultimately, however, all management decisions are subjective - the line must be drawn somewhere. The advantages of this monitoring program and its application to the backcountry management program are that:

1. The establishment of indicators and standards set baselines of acceptability; they may be imperfect, but they do play a critical role in "forcing" managers to think about specific methods of evaluating management strategies.

2. As the campsite monitoring program continues, additional information will allow analyses and standards to be refined in accuracy, thus improving the effectiveness of the management actions on which this information is based.

GIS has been a powerful tool in allowing the AMB to understand the intricacies of managing visitor impacts in Gwaii Haanas. The analyses presented here are doubtless a very simplistic description of a complicated interaction of factors, but the process is nevertheless extremely valuable in assisting managers to make the best possible decisions with the information at hand. The powerful analyses also provide opportunities for managers to begin to answer questions that were previously considered unanswerable.

Acknowledgments

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References

- Archipelago Management Board. 1996. Gwaii Haanas National Park Reserve & Haida Heritage Site draft strategic management plan for the terrestrial area. Public planning program newsletter No. 3. 42 p.
- Marion, Jeffrey L. 1994. Changes in Campsite Condition: Results from Campsite Monitoring at Delaware Water Gap National Recreation Area. Technical Report NPS/MARDEWA/NRTR - 94/063.
- Marion, Jeffrey L.; Farrell, Tracy. 1996. A Backcountry Campsite Inventory and Monitoring Program for Gwaii Haanas National Park Reserve/Haida Heritage Site. Department of Canadian Heritage, Parks Canada Research/Resources Management Report. 33 p.
- Parks Canada. 1994. Parks Canada guiding principles and operational policies. Minister of Supply and Services Canada. Cat. No. R62-275/1994E. 125 p.
- Vaske, J.J. Donnelly, M.P., Freimund, W.A., Miller, T. 1996. The 1995 Gwaii Haanas visitor survey. Project report for Gwaii Haanas National Park Reserve/Haida Heritage Site. Human dimensions in natural resources unit (HDNRU Rep. No. 26) Colorado State University, Fort Collins. 68 p.
- Westland Resources Group. 1994. Gwaii Haanas ecological land classification. Prepared for Environment Canada, Canadian Parks Service, Western Region, Calgary, Alberta. 227 p.

Thirty-Year Monitoring of Subalpine Meadow Vegetation Following a 1967 Trampling Experiment at Logan Pass, Glacier National Park, Montana

Ernest Hartley

Abstract—This long-term study, monitoring visitor impact on subalpine vegetation beginning in 1967, revealed that after 30 years all treatment plots had returned to pre-treatment ratios of vegetation (all species combined), organic litter and bare ground. Higher trampling intensities produced longer term impacts. Vegetation cover recovered in 19 to 25 years when trampled 15 times per week for six weeks in 1967 compared to 25 to 30 years where trampled 50 times per week. The long-term consequences of human trampling on dry meadow vegetation cannot be assessed from short-term observations.

Glacier National Park, like all sites in the National Park System, was set aside as a preserved area to be maintained “unimpaired for the enjoyment of future generations.” In the words of the 1964 Wilderness Act, these “federally owned areas...shall be administered for the use and enjoyment of the American people in such manner as will leave them unimpaired for future use and enjoyment as wilderness...” The Wilderness Act goes on to define a wilderness as an area “where the earth and its community of life are untrammelled by man, where man himself is a visitor who does not remain.”

The purpose of this 30-year research project was to ascertain any significant departure from unimpaired conditions triggered by human activities and to estimate the rate and direction of that change in the subalpine dry meadow vegetation at Logan Pass.

Study Area

Glacier National Park straddles the Continental Divide in northwestern Montana and includes over one million acres (approximately 424,000 hectares) of rugged, Rocky Mountain terrain. A visit to the Park usually includes a stop at Logan Pass, elevation 6,680 feet (2,036 meters), the highest point on the Going-to-the-Sun Road. Of the areas near treeline, Logan Pass receives by far the most visitors.

In the mid 1960s, while still a University of Montana undergraduate, I became concerned about the unrestrained activities of visitors at Logan Pass. Later, as a graduate student, I began

studying human activities as an ecological factor. Field and laboratory work for the early part of this study are reported in Hartley (1976). I have returned to Logan Pass about every five years since the completion of that work to re-sample the permanent treatment plots established in 1967. Thus, sampling data are now available for the thirty year sequence from 1967 to 1997. Additional sampling is anticipated for the year 2002 (35th year).

Sixty-seven million people visited Glacier National Park in its first 89 years, 1910 to 1998 (Williams and others, 1999). Attendance has averaged 1.94 million visitors per year over the last 10 years (table 1). Heyward and others (1984)

Table 1—Glacier National Park, Montana, total annual visits: 1910-1998.

1910	4,000		1958	706,841	
1911	4,000		1959	722,338	6,440,016
1912	6,257		1960	724,538	
1913	12,138		1961	739,982	
1914	12,168		1962	966,100	
1915	13,465		1963	811,214	
1916	12,839		1964	624,100	
1917	15,050		1965	847,104	
1918	9,086		1966	907,839	
1919	18,956	107,959	1967	884,049	
1920	22,449		1968	964,493	
1921	19,736		1969	1,051,200	8,520,619
1922	23,935		1970	1,241,603	
1923	33,988		1971	1,303,073	
1924	33,372		1972	1,392,145	
1925	40,063		1973	1,398,958	
1926	36,901		1974	1,406,643	
1927	41,745		1975	1,571,393	
1928	53,454		1976	1,662,678	
1929	70,742	376,385	1977	1,656,212	
1930	73,783		1978	1,601,131	
1931	59,846		1979	1,446,236	14,680,072
1932	53,202		1980	1,475,538	
1933	76,615		1981	1,786,843	
1934	116,965		1982	1,666,431	
1935	143,240		1983	1,555,717	
1936	210,072		1984	1,580,935	
1937	194,522		1985	1,524,585	
1938	153,528		1986	1,579,191	
1939	170,073	1,251,846	1987	1,660,737	
1940	177,307		1988	1,817,733	
1941	179,082		1989	1,821,523	16,469,233
1942	63,080		1990	1,987,000	
1943	23,469		1991	2,096,966	
1944	36,192		1992	2,199,767	
1945	67,179		1993	2,141,704	
1946	201,145		1994	2,152,989	
1947	324,396		1995	1,839,518	
1948	281,562		1996	1,720,805	
1949	478,839	1,832,251	1997	1,708,877	
1950	485,950		1998	1,830,944	17,678,570
1951	500,125		1999		
1952	630,949		2000		
1953	633,480				
1954	608,230				
1955	674,004				
1956	718,938				
1957	759,161				
				67,356,951	67,356,951

In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management; 1999 May 23–27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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estimated that 59% of total Park visitation crossed Logan Pass on the Going-to-the-Sun Road, and 36% of total Park visitation stopped at the Logan Pass Visitor Center. Recent estimates by Park officials indicate that nearly 80% of total Park visitors, or approximately 1,600,000 visitors, utilize the Logan Pass area each year: most of them during the brief, 100 day, vascular plant growing season between mid-June and mid-September. The inherent conflict between the photosynthetic and reproductive cycles of subalpine flora and the influx of several thousand visitors per day is most severe in July and August, on the weekends and at midday (fig. 1).

While the problems of recreational damage to high mountain vegetation have been recognized and described for over 85 years, detailed scientific research into these problems has been undertaken only during the past three to four decades (Price 1985; Hammitt and Cole 1998). Statistically quantified and designed visitor impact studies were rare when this study was initiated in 1967. More recently, a multitude of recreational impact studies have been conducted around the world, providing valuable baseline information geared to help resource managers evaluate carrying capacity and to

establish the limits of acceptable change for many different ecosystems.

In reviewing visitor impact studies, particularly those involving trampling treatments, it was discovered that most described the initial rates of deterioration, but very few continued plant community analysis or monitored any kind of recovery beyond the first two or three years. Three European exceptions were studies continued for four to eight years (Bayfield 1979; Grabherr 1982; Lance and others 1989). Bayfield's (1979) trampling study recorded data for eight years, and the author concluded, "observation over a substantial period seems necessary to assess the responses of slow growing mountain vegetation to disturbance by trampling." Cole (1985) pointed out the inadequacies of applying only one year of trampling treatments and has initiated a long-term study, in which trampling and monitoring are being applied year after year until year-to-year change in vegetation and soil conditions becomes minimal.

Methods

The research design included experimental treatment plots and trail-side vegetation sampling using a point quadrat sampling method.

Experimental treatment plots were placed in plant communities similar to those found around the Logan Pass Visitor Center and along nearby trails to measure rates of vegetational change from *known quantities* of trampling treatments. During the 1967 growing season, the nine subplots (each one meter square) within the 3 x 3 meter plots were given nine combinations of treatments: three levels of trampling treatments per week [0-15-50] for six weeks, and three levels of clipping [0-1-2] (fig. 2). Plots were designed to separate the impact of trampling on plants and soil from the impact of removing or picking flowers and leaves without soil compaction. Four plots were placed on near level terrain on Caribou Ridge, south of the Logan Pass Visitor Center. Each plot was oriented in a different compass direction. In 1997, the plot locations were recorded by Global Positioning Systems. The plots were sampled using point-quadrats in a stratified random pattern. On each sampling date, 100 sampling points per subplot were recorded: a total of 900 sampling points per plot. Data from the replicated plots were pooled for analysis. The plot data reported herein represent 36,000 random point samples from the 3 x 3 meter plots from the years 1967 (5400), 1969 (8100), 1973 (2700), 1982 (4500), 1986 (5400) and 1997 (7200).

Vegetation of trail-side plant communities was sampled to determine changes in vegetation brought about by long-term, *unquantified*, off-trail trampling. Sampling was accomplished by a point-quadrat method at decimeter intervals along line transects running perpendicular to the trail axes. During the summers of 1967, 1968 and 1969, 40,000 points were sampled along the heavily used trails in the Logan Pass area (Hartley 1976). In 1997, one of those sites was revisited near the head of Highline Trail, just 20 meters north of the Going-to-the-Sun Road. This study consisted of 30 transects: 15 transects east of the trail and 15 transects west of the trail. Each transect was 3 meters long, and placed 0.5 meter apart, with sampling points each decimeter for a total of 900 sampling points in this study.

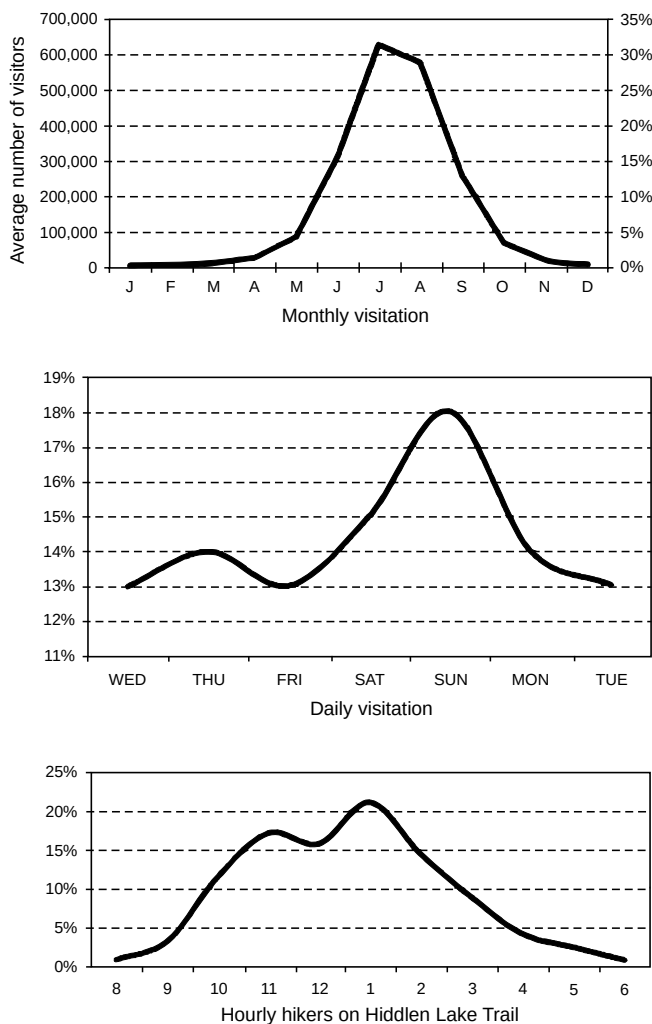


Figure 1—Glacier National Park visitation patterns. Top = monthly; middle = daily; bottom = hourly.

	T-0	T-15	T-50
C-0	Subplot 1 T-0 C-0	Subplot 2 T-15 C-0	Subplot 3 T-50 C-0
C-1	Subplot 4 T-0 C-1	Subplot 5 T-15 C-1	Subplot 6 T-50 C-1
C-2	Subplot 7 T-0 C-2	Subplot 8 T-15 C-2	Subplot 9 T-50 C-2

T = Number of trampling treatments per week (for 6 weeks), summer of 1967.

C = Number of clipping treatments, summer of 1967.

Plot sampling years and number of years since treatments.			
1968	1 year	1982	15 years
1969	2 years	1986	19 years
1973	6 years	1987	20 years
1976	9 years	1992	25 years
1977	10 years	1997	30 years

Figure 2—1967 Experimental plot design for trampling and clipping treatments.

Results: Plot Data

Analysis of pooled sampling data from individual subplots provides recovery data as a function of treatment intensity. Tables 2 to 4 and figure 3 illustrate ratios of living vegetation (VG), organic litter (LT) and bare ground (BG) in treatment subplots from 1967 through 1997. These ratios obtained from the point-quadrat sampling procedure present

an overview of plant community status following trampling and clipping treatments.

1967—The First Year

Table 2 presents pooled data from three treatment plots sampled in 1967 before treatments commenced and again late in the season, after trampling and clipping treatments were completed. Total vegetation groundcover (VG) from all vascular plant species combined in all subplots and all plots averaged 78.2% at the first, pre-treatment sample. By the end of August, average groundcover had dropped to 23.6%: a conspicuous decrease in living vegetation with a corresponding increase in organic litter (LT-dead plant material) and bare ground (BG-soil surface). In the control Subplot 1 (T-0, C-0, upper left), the percentages of vegetation, litter and bare ground remained relatively constant through the season, but in the heaviest impact (Subplot 9 (T-50, C-2, lower right), the vegetational cover decreased through the first growing season from 77% to 11%, while bare ground increased from 3% to 45%. Litter increased from 19% to 44%.

In subplots receiving only trampling treatments (Subplots 2 and 3), 15 tramplings per week reduced vegetation cover 43% below controls by the end of the 1967 growing season, whereas 50 tramplings per week reduced cover only an additional 4-6%. Thus, the major initial impact had already occurred at 15 treatments. In subplots receiving only clipping treatments (Subplots 4 and 7), one clipping reduced cover 40% below controls by the end of the year, whereas two clippings reduced cover 44%.

The data further show that no clipping and heavy trampling (Subplot 3, T-50, C-0) resulted in very similar percentages of cover, litter and bare ground, as did light clipping and light trampling (Subplot 5, T-15, C-1). Conversely, heavy clipping and light trampling (Subplot 8, T-15, C-2) gave season-end results similar to light clipping and heavy trampling (Subplot 6, T-50, C-1).

At season's end, Subplot 9 (T-50, C-2), receiving the heaviest treatments, averaged only 11% cover, 44% litter

Table 2—Early and late 1967 percentages of vegetation, litter, and bare ground in experimental treatment subplots.

	Early pre-treatment 14 July 1967				Late post-treatment 21 August 1967			
	T-0	T-15	T-50	Mean	T-0	T-15	T-50	Mean
----- Percent -----								
Vegetation (VG)								
C-0	75.7	75.7	83.0	78.1	66.7	23.3	17.3	35.8
C-1	84.0	77.7	78.0	79.9	26.3	17.7	13.0	19.0
C-2	77.0	75.7	77.3	76.7	22.3	14.7	10.7	15.9
Mean	78.9	76.3	79.4	78.2	38.4	18.6	13.7	23.6
Litter (LT)								
C-0	18.7	21.3	14.7	18.2	30.7	66.0	62.7	53.1
C-1	12.7	17.3	18.7	16.2	68.0	62.7	53.3	61.3
C-2	18.0	18.3	19.3	18.6	67.0	63.3	44.0	58.1
Mean	16.4	19.0	17.6	17.7	55.2	64.0	53.3	57.5
Bare ground (BG)								
C-0	5.7	3.0	2.3	3.7	2.7	10.7	20.0	11.1
C-1	3.3	5.0	3.3	3.9	5.7	19.7	33.7	19.7
C-2	5.0	6.0	3.3	4.8	10.7	22.0	45.3	26.0
Mean	4.7	4.7	3.0	4.1	6.3	17.4	33.0	18.9

and 45% bare ground. Vegetation was only 16% of that found in controls.

Recovery After Two Years (1969)

VG replacement made a strong comeback in the two year period between 1967 and 1969 in most subplots. The pooled T-50 (Subplots 3, 6, 9), with mean cover that had dropped from 79% to 14% in 1967 returned to 60% in 1969 (table 3). The pooled T-15 treatments (Subplots 2, 5, 8) whose mean cover dropped from 76% to 19% in 1967 returned to 69% two years later. T-0 treatments (Subplots 1, 4, 7) decreased from 79% to 38% in 1967 increased to 83% in 1969.

Litter among clipping treatments (rows) and trampling treatments (columns) showed almost no differences two years after treatments. The highest litter means occurred in Subplots 4 and 7, which were clipped but not trampled.

Bare ground percentages varied widely among trampling treatments with more moderate differences among clipping treatments.

Recovery After Six Years (1973)

Vegetation had increased approximately 5% among T-15 subplots and 9% among T-50 subplots since the 1969 sample four years earlier, while BG had decreased 10-15% in trampled plots (table 3).

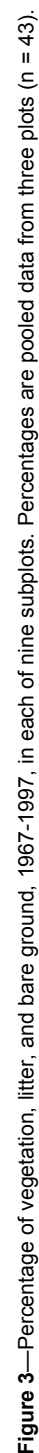
Subplot 9 (T-50, C-2) had regained cover from 55% to 63% since the 1969 sample. The untrampled subplots had lost cover from the sample four years earlier, probably from seasonal fluctuations in soil moisture. Plant cover had increased in all other treatment subplots (average 7% $\pm$ 8) since the 1969 sample.

Table 3—1969 and 1973 percentages of vegetation, litter, and bare ground in experimental treatment subplots.

	2nd year post-treatment July, September 1969				6th year post-treatment July 1973			
	T-0	T-15	T-50	Mean	T-0	T-15	T-50	Mean
	----- Percent -----							
Vegetation (VG)								
C-0	87.9	68.0	61.2	72.4	87.3	78.3	75.7	80.4
C-1	80.9	70.4	63.7	71.7	76.7	74.3	67.3	72.8
C-2	80.2	69.8	55.4	68.5	73.3	71.7	63.0	69.3
Mean	83.0	69.4	60.1	70.8	79.1	74.8	68.7	74.2
Litter (LT)								
C-0	8.4	9.6	8.4	8.8	10.7	11.3	12.0	11.3
C-1	12.9	6.9	7.0	8.9	16.0	10.0	15.3	13.8
C-2	12.7	6.4	6.3	8.5	13.7	16.7	21.0	17.1
Mean	11.3	7.6	7.3	8.7	13.4	12.7	16.1	14.1
Bare ground (BG)								
C-0	3.7	22.4	30.3	18.8	2.0	10.3	12.3	8.2
C-1	6.2	22.7	29.3	19.4	7.3	15.7	17.3	13.4
C-2	7.1	23.8	38.2	23.0	13.0	11.7	16.0	13.6
Mean	5.7	23.0	32.6	20.4	7.4	12.6	15.2	11.7

Table 4—1982 and 1997 percentages of vegetation, litter, and bare ground in experimental treatment subplots.

	15th year post-treatment July, August 1982				30th year post-treatment July, August 1997			
	T-0	T-15	T-50	Mean	T-0	T-15	T-50	Mean
	----- Percent -----							
Vegetation (VG)								
C-0	86.6	81.2	82.0	83.3	83.9	85.9	88.8	86.2
C-1	87.6	86.6	78.4	84.2	85.4	87.0	87.0	86.5
C-2	83.2	82.0	77.8	81.0	80.0	87.1	85.1	84.1
Mean	85.8	83.3	79.4	82.8	83.1	86.7	87.0	85.6
Litter (LT)								
C-0	12.6	17.6	15.8	15.3	9.6	12.6	10.0	10.8
C-1	11.6	12.6	18.6	14.3	10.6	11.8	12.5	11.6
C-2	15.4	16.4	19.8	17.2	10.1	11.6	12.6	11.5
Mean	13.2	15.5	18.1	15.6	10.1	12.0	11.7	11.3
Bare Ground (BG)								
C-0	0.8	1.2	2.2	1.4	6.5	1.5	1.3	3.1
C-1	0.8	0.8	3.0	1.5	4.0	1.3	0.5	1.9
C-2	1.4	1.6	2.4	1.8	9.9	1.3	2.3	4.5
Mean	1.0	1.2	2.5	1.6	6.8	1.3	1.3	3.2



Recovery After 15 Years (1982)

Vegetation cover in all subplots (n = 36) averaged 82.8% in 1982 after 15 years recovery (range 78-88), a 9% increase since 1973 (table 4). BG had reached an all-time low (mean 1.6%)

Subplot 9 (T-50, C-2) had gained 15% VG and lost 14% BG since 1973. All other treatment plots had gained 3% to 11% cover.

Recovery After 30 Years (1997)

Vegetation cover in all subplots averaged 85.6% [range 80-89%] (table 4). Recovery, *as measured only by total vegetation cover*, appeared to be complete 25 to 30 years post-treatment. The lesson to be learned here is that short-term disturbances can lead to long-term recovery. These findings are consistent with recreation ecology research studies from campsites and trails (Marion 1996).

Litter and bare ground had returned to normal levels, except in some subplots where increases in BG and decreases in VG were observed. These perturbations were brought about primarily by the excavations of Columbian ground squirrels (*Citellus columbianus*) in recent years. Grizzly bear (*Ursus arctos horribilis*) excavations were common near the plots, but fortunately the bears did not dig up any of the 3 x 3 meter plots.

Figure 3 summarizes the field sampling results reported in the tables above, but in this presentation, vegetation, litter and bare ground percentages are plotted within each of the nine treatment subplots through the 30 year period. The graphs are arranged in figure 3 in the same order as the treatment subplots were arranged in the field plots (fig. 2). The rate and degree of recovery can therefore be compared between any of the treatment combinations at various points in time. These graphs illustrate the positive relationship between the intensity of treatment and the number of growing seasons required for a return to natural conditions: In untrampled Subplots 4 and 7, VG, LT and BG return to pre-treatment levels within two years, in 15 years where clipped once, and in 19 years where clipped twice. Light trampling required 19 to 25 years for natural return of groundcover, whereas heavy trampling required 25 to 30 years. The graphs also show pairs of subplots yielding similar results from different combinations of treatments, as in Subplots 3 and 5, and Subplots 6 and 8. These pairs share comparative plotted curve patterns, sometimes differing in degree.

A comparison of the vegetation curves in these graphs also reveals which treatment combinations yield the greatest return of cover in the shortest period of time. Here, the Subplots are arranged from the shortest recovery time to the longest: 4-7-2-5-3-6-8-9. Thus, clipping-only renders the least long-term effects, followed by light trampling and no clipping, intermediate levels of both and finally the heavily trampled and clipped subplots.

Results: Trail-Side Vegetation Sampling

A very clear reduction in species diversity was observed in the vegetation along the Highline Trail's edge. An increase

in the number of species was positively correlated with an increase in distance from the trail. Only seven species were sampled in the first 1/2 meter—dominated by Sedges (*Carex nigricans*) and (*Carex phaeocephala*). At 1.5 to 2.0 meters from the trail, 20 species were sampled—dominated by Glacier Lily (*Erythronium grandiflorum*) and Woodrush (*Luzula wahlenbergii*). At 2.5 to 3.0 meters from the trail, Arnica (*Arnica alpina*) and Glacier Lily (*Erythronium grandiflorum*) dominated among the 24 species recorded (Hartley 1999). Trail side distribution of Glacier Lily (*Erythronium grandiflorum*) and Fleabane (*Erigeron peregrinus*) are plotted in figure 4. Sedges and grasses dominated the plant community at trail-side (fig. 5 top), while herbaceous dicots (fig. 5 bottom) were predominate at greater distances from the trail.

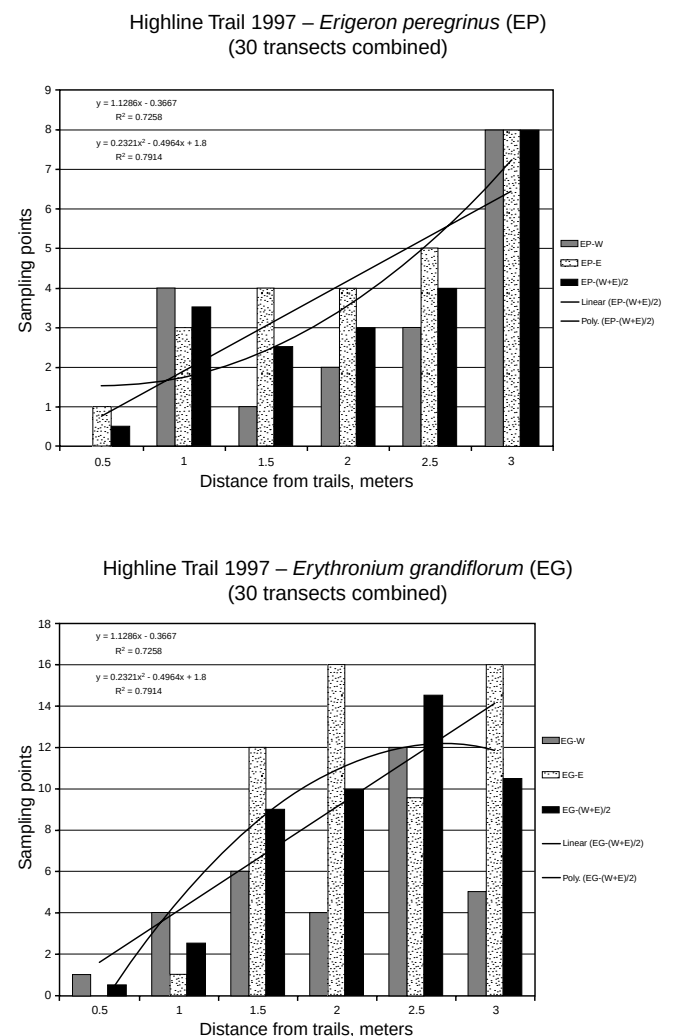


Figure 4—Trail-side distribution of *Erigeron peregrinus* and *Erythronium grandiflorum* on Highline Trail at Logan Pass 7 August 1997: EP-W = *Erigeron peregrinus*, west side of trail; EP-E = *Erigeron peregrinus*, east side of trail; EG-W = *Erythronium grandiflorum* sampled from west side of trail; EG-E = *Erythronium grandiflorum*, east side of trail; Linear = linear regression line and equation; Poly = second order polynomial regression line and equation.

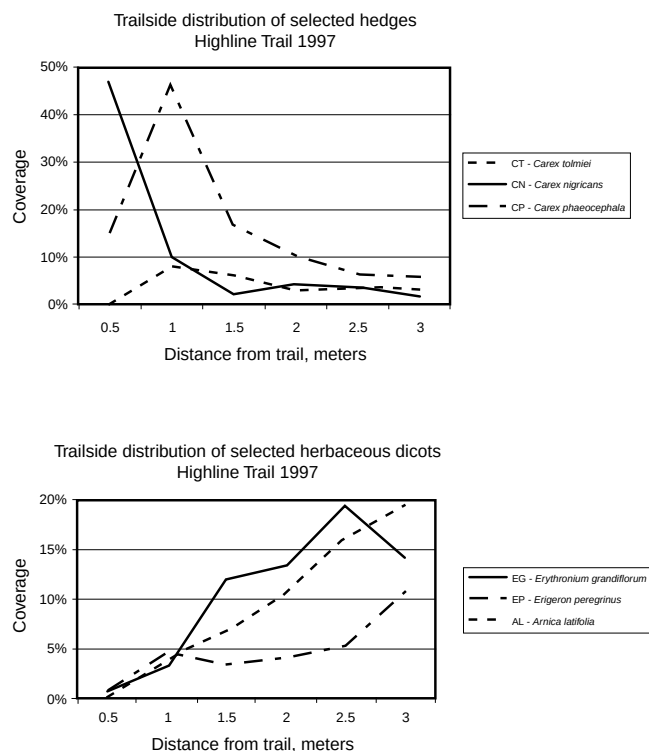


Figure 5—Trail-side distribution of selected graminoids (top) and selected herbaceous dicot species (bottom) on Highline Trail at Logan Pass 7 August 1997: AL = *Arnica latifolia*; CT = *Carex tolmiei*; CN = *Carex nigricans*; EG = *Erythronium grandiflorum*; CP = *Carex phaeocephala*; EP = *Erigeron peregrinus*.

Discussion

Vascular plants growing at Logan Pass and in other alpine and subalpine areas with high visitor use are subject to injury and destruction by visitor activities. Alpine ecosystems are particularly vulnerable to the presence of people in any numbers. There is a high degree of incompatibility between use by people, directly or indirectly, and the maintenance of ecosystem integrity in high mountains (Billings 1973,1979). The dry meadow plant communities appear to withstand single individuals or even a few people crossing them at one time. Very light use may leave imperceptible traces, but even a small group walking in single file across the Logan Pass meadows will leave a path of trampled plants. Data from experimental trampling plots revealed that a single individual walking in a Glacier Lily field once each week created a 9% reduction in vegetation cover by the end of the growing season (Hartley 1979). Such damage is temporary if the action is not repeated, but research strongly indicates that other visitors will see the path and further contribute to trampling impacts. For example, an informal path was discovered in 1969 near the Logan Pass Visitor Center. Photographic monitoring occurred through the years as the path continued to receive foot traffic. It persisted for about 20 years unabated. When last observed in 1997, after a decade of abandonment, the path was recovering by natural revegetation thanks to Park Service protection.

Trampling destroys photosynthetic tissue and triggers an energy flow rate decrease through the plant. Consequently, stored carbohydrate in the underground portions of subalpine plants is stored at a lower than normal level. Overwinter survival and vigorous, early growth the following year is therefore jeopardized. This depleted condition reduces the plant's ability to produce photosynthetic tissue the following year, so plants are smaller and produce fewer flowers in the season following trampling treatments. Continued trampling triggers further breakdown of the system leading to death of the plant (Hartley 1976). It is unknown how many growing seasons are required for return to normal metabolic rates, but current observations strongly suggest that full recovery may require many decades. Liddle and Kay (1987) rightly differentiated between survival after damage versus recovery after damage. From the data reported here, it is evident that most species of the dry meadow plant community survived the 1967 trampling treatments. Recovery after damage from visitor impact, however, took two to three decades.

Sensitivity and subsequent recovery of individual plants is a function of more than trampling intensity, and more than time. Cole (1995) has described the importance of plant physiognomy, the position of perennating buds, and the stem-leaf architecture in determining the resistance, resiliency and tolerance of plants to trampling pressure. Visitor impact on vegetation and soils is also affected by the size and frequency of hiking groups and their care in staying on existing trails. Hikers exhibit a common tendency of stepping off the trail to socialize with other members of their party or to allow the passing of oncoming hikers. In heavily used narrow trails, both parties are known to step off the trail neglecting to use the trail provided. Such activity is generally due to a lack of awareness of trampling impacts. Nevertheless, the larger the hiking groups, the greater the destruction to plant life beside the trail.

During 1967, 50 trampling treatments per week for six weeks removed little more plant cover (4-6%) than 15 trampling treatments per week. If managers were only concerned about the initial removal of vegetation, they might reasonably conclude that there was little need to be concerned about the different trampling impact of these two levels of treatments. Such a conclusion might be drawn from a one- or two-year study. It is important to emphasize, however, that the greater the impact of trampling, the greater number of years required for return to normalcy. For example, recovery required 5 to 10 years longer in the subplots trampled 50 times than in those receiving only 15 trampling treatments. In 1986 and 1992, vegetation cover had peaked in subplots trampled 15 times, but not until 1992 and 1997 did cover reach its highest level in the subplots trampled 50 times.

The rate of natural vegetation cover replacement was more rapid during the first two years following trampling than in later years. Early post-treatment recovery was evidenced by the relatively steep curves in years 0 to 6 followed by a leveling of the curves between 6 and 30 years. The impact of the clipping treatments alone was visible for only the first two years. The combined effects of trampling and clipping, however, were evident for at least 19 years. After 30 years, all subplots had returned to pre-treatment ratios of vegetation, litter and bare ground. Therefore,

managers must consider the time elapsed for short-term and long-term recovery.

Long-term data are now recognized as crucial to our understanding of environmental change and management (Gosz 1998). Such studies can only occur, however, if planned and coordinated. Essential to this process is the permanent marking of treatment plots so they can be relocated at a future time for observation and re-sampling. This study could not have occurred had we not marked the corners of each plot and subplot with long spikes in 1967. Most corner markers have stayed in place through 30 years of freeze-thaw cycles and the activities of ground squirrels, grizzly bears and human beings. The area was also mapped detailing the directional and distance relationships between plots. The National Park Service recorded the position of the treatment plots in 1997 with global positioning system (GPS) equipment, so the plots can be more easily located in the future.

Throughout this paper recovery has been described in terms of total vegetation cover—all species combined. A more detailed and beneficial description of recovery includes the results of plant community analysis and the response patterns of individual species to trampling treatments. The author is preparing a paper to describe the plant community characteristics and species interactions observed in the treatment plots through the 30-year study.

The 1997 Highline Trail data describe a three-meter zone on either side of this busy trail used by hundreds of visitors each day—some were on short nature walks, while others were backcountry hikers heading to Granite Park Chalet or Canada. The trail is also easily accessed by large numbers of the motoring public who park at the nearby Logan Pass Visitor Center and walk across the Going-to-the-Sun Road to the Highline Trail.

The vegetation adjacent to this trail's border is dominated by Sedges (*Carex nigricans*) and (*Carex phaeocephala*). The side-stepping off the trail had almost eliminated the herbaceous dicots from this plant community. It had not been so severe as to reduce the area to bare ground. The high concentration of *Carex* species and other graminoids next to the trail demonstrate that these species exhibit a high resilience—they have the capacity to return the next season after trampling (Cole, 1995). Glacier Lily, Fleabane, *Arnica*, and other herbaceous dicots have a low tolerance to trampling. Hence, the plant community beside a busy trail differs in physiognomy and species mix when contrasted with the plant community a few meters from the trail. The trail has been widened by off-trail trampling through the years: when this site was first sampled in 1967, the trail was slightly more than 1 meter wide. In 1997 it was about 1.5 meters wide. This may be a section of trail where high hiker traffic warrants a hardening of the site by construction of a boardwalk or paving the trail surface.

Year to year fluctuations of available soil moisture were, no doubt, contributing influences in the 30 year sampling data. Alternating periods of drought or abundant moisture probably triggered seasonal shifts in recovery trends throughout the study. For instance, in 1967 the pre-treatment vegetation sample produced a total vegetation cover of 78.2%. A growing season with less soil moisture than average may have been responsible for the fact that this first measure of cover was approximately 10% lower than normal.

In 1997, a year with record high snowpack recordings on nearby Flattop Mountain, SNOTEL data recorded 67.9 inches of snow water equivalent—well above the 43.3 inch average during the previous sampling years (Klasner 1999). Corresponding record high recordings of overall groundcover, species frequencies and flower counts may have indicated recovery from trampling or vigorous growth stimulated by the higher than average moisture. Such elevated levels of leaf, stem and flower production, as observed in 1997, could have masked residual trampling effects or average recovery rates. On the other hand, abundant moisture triggering vigorous plant growth might be expected to accelerate the overall recovery process. The 35th year sampling in the year 2002 may clear up this ambiguity. Correlating growth patterns with available moisture was difficult in the absence of adequate weather data.

Wilderness resource managers of high-elevation natural areas can more effectively decide what constitutes acceptable carrying capacities and acceptable biotic alterations caused by visitor activities if they know the quantitative relationships between various levels of use and their resultant levels of impact on the biota. The most important intended contribution of this research project was to quantify the prolonged recovery period required to repair trampling damage in a slow-growing plant community exhibiting relatively low resilience to visitor activity. It is hoped that the study contributes to that understanding.

Conclusions

The following list summarizes the major findings and implications of this study.

- 1) In high mountain ecosystems subjected to heavy visitor use, rates of disturbance, can occur rapidly in a day or a season. In contrast, rates of natural recovery may occur slowly over decades or centuries.

- 2) After 30 years, all subplots subjected to experimental trampling treatments, had returned to pre-treatment ratios of vegetation, litter and bare ground.

- 3) The study demonstrated that higher trampling intensities produced longer term impacts.

- 4) Fifty trampling treatments per week for six weeks in 1967 removed little additional ground cover [4-6%] than 15 trampling treatments, but natural replacement of cover in subplots trampled 50 times required 5 to 10 years longer than those trampled 15 times.

- 5) Vegetation cover peaked in subplots trampled 15 times after 19 to 25 years, but in the subplots trampled 50 times cover peaked after 25 to 30 years.

- 6) Clipping produced short-term impacts on groundcover lasting two or three years; trampling produced long-term impacts lasting two or three decades.

- 7) Recovery rates of subalpine dry meadow vegetational cover were more rapid in the first two years following trampling than during the 28 years that followed.

- 8) These data describe recovery in terms of total vegetation cover including all species present. A more satisfactory measure of recovery is obtained from detailed plant community analysis. A report of individual species responses and species interactions to trampling and clipping treatments will be presented in a future paper.

9) The long-term implications of human impacts on wilderness plant communities cannot be learned using short-term observations. Researchers should consider the real time invested in the actual field study in contrast to the real time required for full recovery. Most two- and three-year trampling impact studies can tell resource managers only the early stages of disturbance and recovery. Treated plots should be permanently marked and monitored through a substantial portion of the recovery stages.

10) There is no question that long-term studies have much higher financial and human resource costs, but the results obtained in long-term studies will equip resource managers with more reliable data upon which to base their management decisions.

11) Long-term studies tend to provide abundant and more useful data for determining recreational carrying capacity. They also more clearly establish the Limits of Acceptable Change in visitor-dense wilderness areas and high mountain ecosystems such as those at Logan Pass.

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References

Bayfield, Neil G. 1979. Recovery of Four Montane Heath Communities on Cairngorm, Scotland, from Disturbance by Trampling. *Biological Conservation*. 15:165-179.

Billings, W. D. 1973. Arctic and Alpine Vegetations: Similarities, Differences and Susceptibility to Disturbance. *Bioscience*. 23(12):697-704.

Billings, W. D. 1979. Alpine Ecosystems of Western North America. *Special Management Needs of Alpine Ecosystems*: 6-21.

Cole, David N. 1985. Recreational Trampling Effects on Six Habitat Types in Western Montana. Research Paper INT-350. Inter-mountain Research Station, Ogden, Utah.

Cole, David N. 1995. Experimental trampling of vegetation. II. Predictors of Resistance and Resilience. *Journal of Applied Ecology*. 32:215-224.

Gosz, James R. 1998. Long-term Research on a Global Scale. In: *The International Long Term Ecological Research Network - 1998*. US LTER Network, Department of Biology, University of New Mexico, Albuquerque.

Grabherr, G. 1982. The Impact of trampling by tourists on a high altitudinal grassland in the Tyrolean Alps, Austria. *Vegetatio*. 48:209-217.

Hammit, William E. & David N. Cole. 1998. *Wildland Recreation: Ecology and Management*, Second Edition. John Wiley & Sons, New York.

Hartley, Ernest. 1976. Man's Effects on the Stability of Alpine and Subalpine Vegetation in Glacier National Park, Montana. Ph.D. Dissertation, Duke University, Durham, NC 258 pp.

Hartley, Ernest. 1979. Visitor Impact on Subalpine Meadow Vegetation in Glacier National Park, Montana. *Proceedings of the First Conference on Scientific Research in the National Parks*: 1279-1286.

Hartley, Ernest A. 1999. Visitor Impact at Logan Pass, Glacier National Park: A Thirty-year Vegetation Study. In: *On the Frontiers of Conservation: Discovery, Reappraisal & Innovation. The 10th Conference on Research and Resource Management in Parks and on Public Lands. Proceedings of the George Wright Society*, Asheville, North Carolina.

Heyward, Patricia, R., Gerald Wright and Edward E. Krumpe, eds. 1984. *Glacier National Park Biosphere Reserve: A History of Scientific Study. Volume I: A Narrative description of Scientific Studies*. U.S. Man and the Biosphere Program U.S. MAB Report No. 9. 137 p.

Klasner, Fritz. (1999). Personal communication, Ecologist, Glacier National Park.

Lance, Art N., Ian D. Baugh and John A. Love. 1989. Continued Footpath Widening in the Cairngorm Mountains, Scotland.

Liddle, M. J. & A. M. Kay. 1987. Resistance, Survival and Recovery of Trampled Corals on the Great Barrier Reef. *Biological Conservation*. 42:1-18.

Marion, Jeffrey L. 1996. Access and Management: Public Land Perplexities—Balancing Resource Protection and Recreation; Recreation Ecology Research Findings: Implications for Wilderness and Park Managers. In *Proceedings of the National Outdoor Ethics Conference*, April 18-21, 1996, St. Louis, Missouri: 188-196.

Price, Martin F. 1985. Impacts of Recreational Activities on Alpine Vegetation in Western North America *Mountain Research and Development*. 5(3):263-277.

Williams, Tara. (1999). Personal communication, Ecologist, Glacier National Park.

Assessing Soil Erosion on Trails: A Comparison of Techniques

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Abstract—Reports of trail degradation have been increasing in different wildernesses. This impact has become a common concern among managers. Deteriorating tread conditions of trails are increasing, as is concern at protected areas worldwide. In order to make objective and timely trail resource decisions, managers need to have effective and efficient methods of assessing trail erosion. Various approaches to assessing trail erosion, the limitations and utility of each and implications for management are discussed.

Trail deterioration, in the form of trail erosion, is a common problem in wilderness and other backcountry areas and is an impact indicator that warrants the attention of managers (Cole 1983). Trail erosion significantly affects ecological, social and managerial environments.

Ecological Significance

Erosion can result in aquatic system disturbance, excessively muddy trails, widening of trails, tread incision and braided or multiple trails and can lead to the creation of undesired trails (Hammitt and Cole 1998, Marion and others 1993). Unlike disturbed vegetation and compacted soil, soil erosion is the only trail degradation indicator, relatively speaking, that does *not* recover naturally over time. A study of 106 National Park Service units found that almost 50% of all park managers indicated that soil erosion on trails was a problem in many or most areas of the backcountry. Trail widening was cited by 31% of park managers, and 29% rated the formation of braided or multiple trails and the creation of undesired trails as serious problems (Marion and others 1993).

Social Significance

The impacts of soil erosion include undesirable trail conditions, which can adversely affect the recreational experience. Deeply eroded, muddy, multiple or undesired trails may lead to a variety of social problems. Trails that are severely eroded may have significant amounts of exposed

roots, which can decrease the functional utility of the trail; the scars left by eroded trails may be considered a visual impact and adversely affect the visitors' experience. Braided trails commonly found in open meadows create a visual impact sometimes noticeable from miles away. These impacts and the decrease in the functional utility of trails due to factors of trail erosion have been found to affect the quality of recreational experiences (Vaske and others 1982).

Managerial Significance

Trail erosion caused by recreational use threatens the resource protection mandates of federal land managers. Managers of wilderness areas are legally mandated to assess recreational impacts. Management guidelines provide National Park Service managers with the most specific guidance in implementing legislation. The *Natural Resources Management Guideline* (National Park Service 1991) states that "park managers must know the nature and condition of the resources in their stewardship, have the means to detect and document changes in those resources, and understand the forces driving the changes" (chapter 5:20). A second *Natural Resources Inventory and Monitoring Guideline* (National Park Service 1992) states that it is the policy of the NPS to "assemble baseline inventory data describing the natural resources under its stewardship, and to monitor those resources forever [and] detect or predict changes that may require intervention" (chapter 1:1).

Several studies on trail conditions, specifically trail erosion, have been conducted (Bayfield and Lloyd 1973, Bratton and others 1979, Coleman 1977, Garland 1990, Helgath 1975, Rinehart and others 1978), from which valid assessment methods have been developed. This paper presents different approaches to assessing trail erosion and discusses the utility and management implications of each.

Literature Review

Assessment of trail erosion is fairly well-represented in the trail impact literature, which is to say that there have been numerous reported methods used to assess trail erosion. The literature presents nearly a dozen different terms related to methods of assessing trail erosion. They range from proactive estimations of potential soil loss to reactive methods that result in precise measurements of actual loss. The nine most widely applied methods are reviewed here.

Cole (1989) discusses the use of the *condition class* method. This rapid assessment method involves a series of condition descriptions determined by management objectives (fig. 1). The trail system is then systematically sampled, and trails/segments are classified according to the predetermined

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Trail Condition Classes

- Class 0: Trail barely distinguishable; no or minimal disturbance of vegetation and/or organic litter.
- Class 1: Trail distinguishable; slight loss of vegetation cover and/or minimal disturbance of organic litter.
- Class 2: Trail obvious; vegetation cover lost and/or organic litter pulverized in primary use area.
- Class 3: Vegetation cover lost and/or organic litter pulverized within the center of the tread, some bare soil exposed.
- Class 4: Nearly complete or total loss of vegetation cover and organic litter within the tread, bare soil widespread.
- Class 5: Soil erosion obvious, as indicated by exposed roots and rocks and/or gulying

Figure 1—Condition class descriptions used to assess trail conditions (source: Jeffrey L. Marion).

condition classes. Sampling has been done at differing intervals by various researchers. Bayfield and Lloyd (1973) every 50m, Bratton and others (1979) sampled every 500m, Marion and others (1997) every 300m, Root and Knapik (1972) every 152m.

Leonard and Whitney (1977) and Cole (1983) describe in detail the *cross-sectional area* method, in which once a sampling location is identified, a taught line, rope, cord, wire or rigid bar is placed across the trail and attached to two fixed points. These points should be permanent and far enough off the trail to allow for future erosion and or the development of multiple treads. At fixed intervals along the horizontal transect line, vertical measurements are taken to the tread surface. Care must be taken to keep the horizontal rope, wire or bar taut, level and elevated above vegetation. The cross-sectional area below the taut line or bar can then be calculated (fig. 2). Future measures will indicate the amount and rate of change that has occurred.

Published research indicates that the use of the *quadrat* assessment method on trails is limited. A quadrat is typically a square device made of varying materials, which is then made to look like a checkerboard by subdividing the frame with string. This device is placed on the tread surface, at sampling points determined by a sampling scheme, and conditions are then estimated on a percentage basis.

Census of active erosion, as described by Leung and others (1997) and Farrell and Marion (1999), is a subjective rapid assessment method, requiring experience and expertise in trail design and construction. Actively eroding trail segments is one type of erosional event, which will appear to develop constantly over present time, and a substantial loss may occur over a years time or a couple of months. The erosion is continuing its downward movement to the bedrock. An assessment of the trail system is done by walking the trail and tallying the actively eroding segments.

Census of erosional events, a rapid assessment method, is considered a subset of *active erosion*, described by Leung and others (1997), and Marion (1997a). The first step is to define in precise terms exactly what will be considered an erosional event (that is, at least 10 feet long and 1 foot deep). An erosional event is considered an inactive event that has stabilized as the downward erosional process hits the more resistant subsoil, regolith layer or bedrock. A census of the trail system is then conducted by tallying the number and length of erosional events while walking the trail.

Rinehart and others (1978) used *stereo photography* to assess trail conditions. This method involves taking stereoscopic pairs of photos at a sampling location determined by a sampling scheme. Trail transects are established following procedures similar to the cross-sectional method. However, instead of taking vertical measurements to the trail tread, stereo photos are taken and the cross-sectional area is computed with a digitized stereo plotter. Rinehart used a 2- x 2-inch camera mounted to a stereo board that accommodated various film sizes. Before each photo is taken, a target card is placed on the trail for scale, and the board is leveled and kept exactly 15 feet away from the trail transect.

Maximum tread incision, is a method in which a surveyor conducts incision measurements at a series of points along a trail, which is determined by a sampling scheme. One method of measuring incision is to identify the post-construction tread surface and take a vertical measurement to the deepest section of the current tread surface. A modification of the procedure is to identify the level of the current tread and take a vertical measurement to the deepest point of the tread surface.

Coleman (1977) used *aerial photographs* to evaluate trail conditions over a 19-year period. A 1953 photo of a popular trail was analyzed using a Hilger and Watts 5x Print Magnifier to measure path width, at a scale of approximately 1:10,000. This instrument is capable of measuring to 1/10mm. Trail sections were sampled from this photo and compared with a 1973 photo of the same trail segment.

Kuss and Morgan (1980; 1984) applied the *Universal Soil Loss Equation* (USLE) to assess soil loss. This method is a

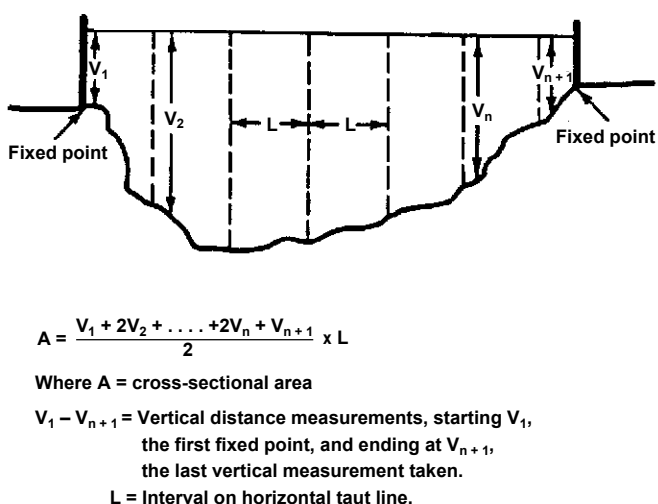


Figure 2—Layout of trail transect and formula for calculating cross-sectional area (source: David N. Cole).

model and an estimation of potential soil loss, and it is useful in planning and designing trail systems. Its utility as a measure of real soil loss is limited and should be used with caution outside of Eastern agricultural lands in which the empirical relationships were developed. For these reasons, this method is not discussed further.

Wallin and Hardin (1996) estimated trail-related soil erosion in Ecuador and Costa Rica using an experimental design. Using a modified *McQueen rainfall simulator*, the study compared on- and off-trail infiltration rates and particle dispersion due to the simulated rainfall. Like Kuss and Morgan's method, this method is an estimation of potential erosion and therefore is not discussed further.

More comprehensive reviews of the trail impact literature are provided in Hammitt and Cole (1998), Cole (1981) and Hendee and others (1990). Table 1 summarizes the assessment methods previously discussed.

Methods

An analysis of published research resulted in the development of a trail erosion matrix, comparing methods of assessing trail erosion with evaluation criteria. Three scientists

with expertise and experience in assessing recreation-based trail erosion were consulted. Each scientist independently rated each assessment method (1 = very low, 5 = very high) against five evaluative criteria. The five criteria were developed with the assistance of experts in the field (table 2). The average total assessment scores were computed using the formula $Total = E + P + A + MU - LTR$ where (E=efficiency, P=precision, A=accuracy, MU=management utility and LTR=level of training required). The LTR criterion is reverse coded due to the negative aspects (time and cost) of training. Assessment scores were calculated and rank ordered.

Results

The range of possible scores are -1 to 19. The *condition class* method of assessing trail erosion was found to have the highest score of 11.68, while the *aerial photo appraisal* method had the lowest score of 6.0 (table 3). The condition class method, in addition to having the best overall ranking, also has the best score on level of training required (2.33, meaning a low level of training is required), but this method ranked the lowest on management utility (2.67).

Table 1—Summary of assessment methods by type, with corresponding selected references.

Assessment method	Description	Selected references
<i>Condition Class Assessment</i> Condition Class	Descriptive classes are defined and assigned to trails/segments.	Cole and others (1997)
<i>Morphometric Assessments</i> Cross-sectional Area	Sampling points are determined by a sampling scheme, then measurements are taken vertically from a horizontal datum, which is attached to fixed points on both sides of the trail.(1983)	Leonard and Whitney (1977) Cole
Maximum Tread Incision Post-construction (MIP)	Incision measurements are performed at a series of points along a trail that is determined by a sampling scheme, from Post-construction height to tread surface.	Marion (1997)
Maximum Tread Incision Current Tread (MIC)	Incision measurements are performed at a series of points along a trail segment.	Marion (1997)
<i>Census/Tally Assessments</i> Census of Erosional Events	Erosional events are defined, followed by a census of those problems.	Marion (1994)
Census of Active Erosion	"Active erosion" is defined, followed by a complete census of those problems.	Farrell and Marion (1999)
<i>Quadrat Assessment</i> Quadrat Measurement	Measurements are performed within quadrats at a series of points that is determined by a point-sampling scheme.	None (for assessing trail erosion)
<i>Photographic Assessments</i> Stereo Photography	On-the-ground photos are taken and evaluated against future photos using a digitized stereo plotter.	Rinehart and others (1978) Warner and Kvaerner (1998)
Aerial Photo Appraisal	Trails are identified and stereoscopically evaluated from aerial photos.	Coleman (1977) Price (1983)

Table 2—Evaluation criteria used to rate the utility of various trail erosion assessment methods.

Evaluation criteria	Description
Level of Training Required	Amount of time required to train a novice in the use of the method.
Efficiency	Amount of time and financial resources required to apply the method.
Precision	The ability to consistently replicate results. Will ten individuals using the same method report identical results?
Accuracy	How close to the “true” value can you get?
Management Utility	Will the results gathered from a particular method be relevant to resource management and planning decisions?

These results differ from past research and hypotheses put forth in the literature. For example, it has been stated in the literature that the *cross-sectional area* method

is probably the most useful measure for managers in that the technique is replicable, requires relatively little training, and provides results that are easy to use and interpret (Cole 1983).

Results reported here indicate that not only did the *cross-sectional area* method tie for third place with an overall score of 10.33, the individual scores on management utility and level of training required were 3.00 (neutral) and 3.67 (neutral-high), respectively. These data indicate 1) that individuals with experience and expertise with trail assessment methods are either not in agreement or 2) that there

has been an evolution in thought regarding this particular method over the past 15 years.

Further investigation revealed that, when controlling for research-oriented criteria, precision and accuracy (table 4), the cross-sectional area method dropped in ranking from third to eighth, while condition class and census of erosional events continued to rank one and two, respectively.

Discussion

Monitoring of trail conditions can be useful for many reasons. Trail conditions, rate of change and trends can be identified. This information can be used to evaluate the acceptability of current conditions and whether or not trail

Table 3—Comparison of trail erosion assessment methods based on evaluation criteria, and summary of ratings showing individual rater scores and their average, (1 = very low to 5 = very high).

	Level of training required	Efficiency	Precision	Accuracy	Management utility	Average total* score	Kruskal-Wallis** mean rank
Condition Class	1 3 3 2.3	5 5 4 4.7	3 2 4 3.0	3 5 3 3.7	2 2 4 2.7	11.3	23.7
Census of Erosional Events	3 4 4 3.7	4 4 4 4.0	3 3 2 2.7	3 4 3 3.3	5 5 4 4.7	11.0	22.0
Cross-sectional Area	4 4 2 3.7	1 2 1 1.3	5 4 5 4.7	5 5 5 5.0	3 4 2 3.0	10.3	18.0
Maximum Incision Post-construction (MIP)	3 3 3 3.00	4 4 2 3.33	2 3 4 3.00	2 3 4 3.00	4 4 4 4.00	10.3	17.2
Census of Active Erosion	5 4 4 4.3	4 3 4 3.7	2 3 2 2.3	3 3 3 3.0	5 5 4 4.7	9.3	11.8
Quadrat Measurement	3 4 3 3.3	2 2 2 2.0	4 3 4 3.7	4 4 4 4.0	3 4 2 3.0	9.3	11.8
Maximum Incision Current Tread (MIC)	3 4 3 3.3	4 2 2 2.7	3 2 4 3.0	3 2 4 3.0	4 3 2 3.0	8.3	11.2
Stereo Photography	5 5 3 4.3	1 1 3 1.7	3 4 3 3.3	3 5 2 3.3	2 4 3 3.0	7.0	5.5
Aerial Photo Appraisal	4 4 5 4.3	3 3 3 3.0	1 3 3 2.3	1 3 2 2.0	2 4 3 3.0	6.0	4.8

*Average total score = (Efficiency + Precision + Accuracy + Management Utility) - (Level of Training Required)

** Kruskal-Wallis $c^2 = 17.643$; $p = .024$.

Table 4—Summary of survey results comparing trail erosion assessment methods based on evaluation criteria, while controlling for accuracy and precision (1 = very low to 5 = very high).

	Level of training required	Efficiency	Management utility	Average total* score
Condition Class	2.3	4.7	2.7	5.0
Census of Erosional Events	3.7	4.0	4.7	5.0
Maximum Tread Incision Post-construction (MIP)	3.0	3.3	4.0	4.3
Census of Active Erosion	4.3	3.7	4.7	4.0
Maximum Tread Incision Current Tread (MIC)	3.3	2.7	3.0	2.3
Quadrat Measurement	3.3	2.0	3.0	1.7
Aerial Photo Appraisal	4.3	3.0	3.0	1.7
Cross-sectional Area	3.7	1.3	3.0	0.7
Stereo Photography	4.3	1.7	3.0	0.3

* Average total score = (efficiency + management utility)—(level of training required).

management programs, including maintenance and reconstruction, have been effective.

Condition Class Method

In many areas, field assessments of impact are desirable, but it is not feasible to spend more than a couple of minutes at each sampling location. This is usually the case in large, dispersed recreation areas. The first step is to define, in precise terms, exactly what the condition class ratings will be. Defining condition class ratings is a subjective, time-consuming process, however, results are useful but limited. A major limiting factor in the utility of the condition class method is that it relies on a single qualitative measure. However, the *condition class* method of assessing trail erosion requires little training and is a rapid, accurate, efficient method that results in somewhat limited data as to the character of the trail system. The utility of these data for managers is questionable and should be considered a second or third alternative to methods with greater management usefulness.

Census of Erosional Events

The census of erosional events method can accurately assess trail systems. Terminology must be identified and defined in terms of exactly what will be considered an erosional event. This method is applied using a systematic sampling scheme, and it is accurate, and efficient, and the results are relevant to managers who must make appropriate and timely trail resource decisions. Limitations include the need for a high level of training due to the qualitative nature of an erosional event and the potential lack of inter-rater reliability. However, this method allows relatively rapid assessment of a trail system and produces information on the frequency, extent and distribution of erosional event problems; this would explain the high score received on

management utility. The data in this paper suggest that trail system monitoring be most effectively done using a combination of methods.

Maximum Incision Post-Construction (MIP)

This method is increasing in use as indicated by its mention in recent theses, dissertations and journal publications. Measuring incision from the post-construction height is an effective method of monitoring system-wide trail erosion. This point measurement technique allows prompt assessment of trail conditions and their spatial variations. The data collected provide information that managers can use to make trail resource decisions.

This method is limited due to the subjectivity of identifying the post-construction tread height, measurement error and inter-rater variability. The time required to train technicians remains a concern of managers. This method was rated relatively neutral (3.0) across all five criteria and yet resulted in the third highest overall rating. It should be noted that of all nine methods, the MIP method is the only one that received perfect inter-rater reliability of 4.0 on management utility. This seems to suggest that there is more agreement about the usefulness of data collected using the MIP method than any other method.

Cross-Sectional Area Method

Soil erosion is the single most important, managerially significant trail degradation indicator. The *cross-sectional method* is probably the most frequently used, replicable method for monitoring purposefully located trail segments. This method may also be applied to systematically sampled locations for monitoring entire trail systems. The erosion or deposition of soil can be measured with very high precision and accuracy with this method. The data collected using this method are adequate for managers making trail

management decisions. However, there are a number of limitations to the *cross-sectional method*. First, the training required is high, and the method is extremely time-consuming and therefore an inefficient method for monitoring trail systems. When monitoring a trail system with a systematically sampled scheme, this method becomes inefficient in terms of time (equipment is often heavy and difficult to transport) and financial resources. In addition, it involves a number of assumptions, including ability to relocate the fix points precisely, reference line elevated above surrounding vegetation, the line is kept taut, a level is used for the vertical measurements, the taut line is repositioned the same height above the fixed points, vertical measurements are taken at the same interval, and the vertical measurements are taken starting from the same side. For these reasons, training is the single most important factor in the proper application of this method. Adequate training is costly and thus a major limiting factor for managers.

Certain wilderness areas, however, may have only a few problem locations within their trail system. Monitoring of these locations using the *cross-sectional method* would be quite appropriate with the proper training and experience. This method is accurate, precise, gives managers relevant information about amount of soil loss/deposition and rate of loss, and identifies any trends that may be developing. Furthermore, a well-trained surveyor should be able to make management suggestions about how to mitigate the continued soil loss.

Census of Active Erosion

This problem assessment method is efficient and results in data useful to managers. Before this method can be implemented, managers must determine what constitutes “active erosion.” This step is crucial to the effectiveness of this method. Defining in precise terms what exactly is to be considered active erosion is a considerable task and a limitation of this method. The subjective distinction between active and inactive erosion can be mitigated with precise definitions developed before the method is implemented. The qualitative definition of “active erosion” leads to inter-rater variability. This is a major concern for managers who have to deal with high employee turnover. Furthermore, extensive employee training is required to ensure accuracy. The *census of active erosion* method has its benefits however. The method is efficient, in terms of time and financial resources and accurate, and it results in information on the frequency, extent and distribution of active erosion problems. Trail data relevant to managers can be obtained using this method and should be considered as a trail monitoring method.

Quadrat Measurement

Published research indicates that the use of quadrats to assess trail conditions is limited. The use of the quadrat method may become more widespread as indicated by its overall score of 9.34 (table 3). Relocation of sampling points, measurement error and field/training time limit the efficiency of this method. However, our results indicate that the quadrat method is accurate and precise, and the results are

managerially significant. This method has significant management utility, and the results are adaptable to an indicator/standards-based management framework.

Maximum Incision Current Tread (MIC)

The current tread incision measurement is a variation of the MIP method. This rapid assessment method is more subjective, in that identification of the current tread height is, often times, more difficult than identifying the post-construction height. This would explain the lower efficiency rating of MIC as compared to MIP. This lower efficiency rating caused a decrease in the management utility rating, which adversely affected the overall rating. Although this method is similar to MIP and has comparable limitations and usefulness, MIC ranked eighth overall compared to a third place ranking of MIP. This method, along with MIP, can be effectively used in an indicator/standards-based management framework, and it is an effective method of monitoring trail erosion and should be considered for monitoring trail systems.

Stereo Photography

“Stereo photographs taken with an ordinary camera mounted on a shop-made tripod attachment proved valuable in studying trail entrenchment...” (Rinehart and others 1978). The use of stereo photography to monitor trail systems is questionable, although it does have advantages. Backcountry areas with short seasons may be well-suited for this method. Spending the short season in the field taking photos and leaving the more time-consuming and tedious plotting of trails until later would be an efficient use of time. Also, stereo photographs illustrate current conditions and trends, a feature that is especially useful in orienting and training new personnel (Rinehart and others 1978). Stereo photographs identify actual change in tread conditions rather than forcing one to interpret numerical measurements that can conceal compensating changes. For example, “if a trail becomes wider and also fills in with material eroded elsewhere and deposited in the transect... the transect area might remain unchanged (Rinehart and others 1978). Other methods would interpret this as an unchanged condition, and stereo photos would accurately identify the dynamic process of trail erosion.

However, the utility of this method to managers is questionable. Disadvantages of stereo photographs include vegetation occasionally obscuring the view of the transect, field limitations due to inclement weather and relocating transects. Rinehart and others (1978) suggest measuring from the trailhead using a calibrated bicycle wheel to relocate transects. However, the inter-rater reliability of using measuring wheels should be of concern. In unpublished field tests Marion (1997b) demonstrated the significant lack of inter-rater reliability using various diameter measuring wheels.

Although the stereo photography method is relatively accurate and precise, it lacks efficiency and requires a high level of training. Managers should be versed in numerous assessment methods before implementing stereo photography as a method of monitoring trail systems.

Aerial Photo Appraisal

If suitable coverage is available over a sufficient period of time, aerial photography can be an efficient method of measuring trail erosion. As implied, this method has potentially significant limitations. Coleman (1977) suggests that identifying real trends from mere fluctuations can be done effectively with this method. Using aerial photography on a popular path in England, she documented a level of accuracy much greater than that required for defining paths. However, due to limitations such as varying weather conditions and canopy cover, aerial photography is typically an ineffective method in most of the United States. When interpreting aerial photographs, the distinction between trampled, dying, dead or damaged vegetation and eroded segments is far from obvious (Coleman 1977). This limits the interpretation to the visible extent of change and therefore, may vary seasonally in some types of vegetation. Visible extent of trail alterations may be extremely relevant to managers. In contrast, lack of accuracy, precision and efficiency significantly detracts from the utility of this method. Furthermore, the financial commitment and high level of training necessary to interpret photos raises serious concern about its utility. This method should not be implemented as a single monitoring method. However, in combination with other methods, aerial photography may enhance the data that managers use to make trail resource decisions.

Conclusions

This study looked at nine different methods of assessing trail erosion. When determining which method to implement, resource managers must first identify their resource standards. Human judgments, in the form of standards and indicators, are needed before the appropriate method can be determined. Thoughtful and timely development of those standards and indicators are of fundamental importance to proper management of trail systems in the backcountry.

Managers often lack adequate information on the nature, severity and causes of erosion-related problems and on the management approaches (assessment methods) that have successfully reduced such problems (Manning and others 1996). Moreover, little or no formal effort or few if any programs exist that are specifically designed to foster communication among natural resource managers. Consequently, information about trail erosion and alternative solutions are not effectively gathered, analyzed and shared (Manning and others 1996). This lack of information sharing results in considerable confusion and inefficiency.

We believe that natural resource managers can use these findings for improving impact assessment and monitoring programs. First, the extensive, systematic list of erosion assessment methods developed in this paper can be a useful guide. Understanding and awareness of the methods available can help managers make better trail resource decisions and result in more effective management. Moreover, the table of assessment methods should help stimulate managers' thinking about alternative solutions to managing trail erosion-related problems. Typically, a number of potential management practices can be applied to assess trail erosion, and these management practices vary in their strategic

purpose and directness. Managers should be aware of and give serious consideration to all potential trail erosion assessment methods before implementing a trail-monitoring program. It is our hope that this paper assists managers in recognizing the assessment methods available for measuring soil erosion, and that we have provided some order to the confusion.

References

- Bayfield, N. G.; Lloyd, R. J. 1973. An approach to assessing the impact of use on a long distance footpath- the Pennie Way. *Rec. News Supp.* 8: 11-17.
- Bratton, Susan P.; Hickler, Mathew G.; Graves, James H. 1979. Trail erosion patterns in Great Smokey Mountains National Park. *Environmental Management*. 3: 431-445.
- Cole, D. N. 1981. Managing Ecological Impacts at Wilderness Campsites: An Evaluation of Techniques. *Journal of Forestry* 79(2): 86-89.
- Cole, D. N. 1983. Assessing and monitoring backcountry trail conditions. Research Paper INT-303. U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT.
- Cole, D. N. 1989. Wilderness Campsite Monitoring Methods: A Sourcebook. General Technical Report INT-259. USDA Forest Service, Intermountain Research Station, Ogden, UT.
- Cole, David N.; Watson, Alan E.; Hall, Troy E., and Spildie, D. R. High-Use Destinations in Wilderness: Social and Biophysical Impacts, Visitor Responses, and Management Options. Ogden, UT: USDA Forest Service, Intermountain Research Station; 1997; Research Paper INT-RP-496. 30p.
- Coleman, Rosalind A. 1977. Simple techniques for monitoring footpath erosion in mountain areas of north-west England. *Environmental Conservation*. 4: 145-148.
- Farrell, T. A. and Marion J.L. 1999. Article currently 'in preparation'.
- Garland, G. G. 1990. Technique For Assessing Erosion Risk From Mountain Footpaths. *Environmental Management* 14: 6, pp. 793-798.
- Hammit, W. E. and Cole, D.N. 1998. *Wildland Recreation: Ecology and Management* (2nd Ed.). New York: John Wiley & Sons.
- Helgath, Sheila F. 1975. Trail deterioration in the Selway-Bitterroot Wilderness. Res. Note INT - 193. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.
- Hendee, John C., George H. Stankey and Robert C. Lucas. 1990. *Wilderness Management*. North American Press: Golden, CO.
- Kuss, F. R. and Morgan J.M. 1984. Using the USLE to Estimate The Physical Carrying Capacity of Natural Areas For Outdoor Recreation Planning. *Journal of Soil and Water Conservation* 39: (6) pp. 383-387.
- Kuss, F. R. and Morgan J.M. 1980. Estimating The Physical Carrying Capacity of Recreation Areas: A Rationale For the Application of the Universal Soil Loss Equation. *Journal of Soil and Water Conservation* 25: pp. 87-89.
- Leonard, R. E.; Whitney, A. M. 1977. Trail transect: a method for documenting trail changes. Res. Pap. NE-389. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station.
- Leung, Y. and Marion, J.L. and Ferguson, J.Y. 1997. Methods for assessing and monitoring backcountry trail conditions: an empirical comparison. Making Protection Work: Proceedings of the 9th George Wright Society Conference on Research and Resource Management in Parks and on Public Lands, ed. D. Harmon, pp. 406-414. The George Wright Society, Hancock, MI.
- Manning, R. and N.L. Ballinger, J.L. Marion, J.W. Roggenbuck 1996. *Recreation Management in Natural Areas: Problems & Practices, Status & Trends*. *Natural Areas Journal* 16(2): 142-146.
- Marion, J. L., J. W. Roggenbuck and Manning, R. E. 1993. Problems and practices in backcountry recreation management: a survey of National Park Service managers. *Natural Resources Report NPS/NRVT/NRR-93/12*. U.S. Dept. of the Interior, National Park Service, Natural Resources Publication Office, Denver, CO.

- Marion, J.L. 1994. An assessment of trail conditions in Great Smoky Mountains National Park. Research/Resources Management Report. U.S. Department of the Interior, National Park Service, Southeast Region, Atlanta, GA.
- Marion J. L. 1997a. Trail and Campsite Assessment in Patagonia, Chile. (Unpublished).
- Marion J. L. 1997b. Accuracy of various measuring wheels. (Unpublished).
- Price, Martin F. Management planning in the Sunshine Area of Canada's Banff National Park. *Parks*. 1983; 7(4): 6-10.
- Rinehart, Robert P.; Hardy, Colin C.; Rosenau, Henry G. 1978. Measuring trail conditions with stereo photography. *Journal of Forestry*. 76: 501-503.
- Root, J. D., and L. J. Knapik. 1972. Trail conditions along a portion of the Great Divide trail route. Alberta and British Columbia Rocky Mountains. Res. Counc. Alberta Rep. 72-75, Edmonton, Alta.
- U.S. Department of the Interior, National Park Service 1991. Natural Resources Management Guidelines. NPS-77. Government Printing Office, Washington, D.C.
- U.S. Department of the Interior, National Park Service 1992. Natural Resources Inventory and Monitoring Guideline. NPS-75.
- Vaske, J. J., Gaefe, A. R. and Dempster, A. 1982. Social and environmental influences on perceived crowding. In: *Proceedings of the Third Annual Conference of the Wilderness Psychology Group*, ed. F. E. Boteler, pp. 211-227. West Virginia University, Division of Forestry, Morgantown, WV.
- Wallin, T. R., and Harden, C. P. 1996. Estimating Trail-Related Soil Erosion in the Humid Tropics: Jatun Sacha, Ecuador, and La Selva, Costa Rica. *Ambio* 25:517-522.
- Warner, W. S. and Kvaerner, J. Measuring trail erosion with a 35 mm camera. *Mountain Research and Development*. 1998; 18 (3):273-280.

Sanitation in Wilderness: Balancing Minimum Tool Policies and Wilderness Values

Paul R. Lachapelle

Abstract—Officials with the four wilderness managing agencies are faced with balancing wilderness preservation values and the minimum tool policies of their respective agencies. One example is the management of sanitation, particularly human waste and the often intrusive infrastructure that accompanies its treatment and disposal. Because the treatment and disposal of human waste is a potentially serious public health hazard if mismanaged, it sometimes requires an elaborate infrastructure, including buildings and use of helicopters or pack stock. A paradox exists between public health concerns and the use of a minimum tool allowed by the agency to deal with human waste treatment and disposal. What is needed is a framework for balancing these interests to make explicit various options available to scientists and managers. This paper provides a matrix and related flow chart for considering various sanitation techniques while incorporating minimum tool options and concerns about related impacts.

The issue of sanitation in wilderness presents a troublesome paradox. On one hand, managers and scientists with the four wilderness-managing agencies must provide for the preservation of wilderness character while protecting the resource from impacts, including surface and ground water pollution caused by improper human waste disposal. The implementation of permanent structures to treat or store waste and the consistent use of helicopters or pack stock to transport waste or materials presents an interesting, albeit unusual perspective from which to examine the legal and ethical framework of wilderness.

The Wilderness Act of 1964 (Public Law 88-577) includes the characterizing phrases “untrammeled by man,” “retaining its primeval character” and “man’s work substantially unnoticeable,” yet it also explicitly states that the areas are to be managed with “no use of motor vehicles, motorized equipment...no other form of mechanical transport, and no structure or installation...except as necessary to meet minimum requirements for the administration of the area.” The notion of “minimum requirements” in wilderness areas mandated to be managed for “the preservation of their wilderness character” presents some ambiguity. The choice

of a minimum tool is largely at the discretion of the land manager. Hendee (1990) refers to the “minimum tool rule” as “the minimum regimentation necessary to achieve established wilderness management objectives” and depends “on a manager’s judgment about the degree of regulation necessary to achieve objectives and the likely effectiveness of various regulatory and nonregulatory actions in certain situations.” Thus, management decisions can be based on subjective judgements, personal values or even administrative convenience.

Managers may neglect sanitation issues at specific sites or may implement a sanitation strategy with an emphasis on mechanized transport or an elaborate infrastructure that is incompatible with social values or biophysical constraints. Several studies of wilderness managers have indicated that steps to improve resource conditions are taken only after “substantial damage...had occurred” (Shindler 1992). Cole (1996) asserts that managers have been reluctant to attack problems directly, stating, “Two oft-cited wilderness management principles, that indirect management techniques are best and that use limits should be a last resort, have become so entrenched in the wilderness community that they have paralyzed many management programs.” However, a new wave of purist sentiment has occupied recent discussion regarding management objectives in wilderness. Nash (1996) describes the wilderness experience as “delicate” and one that is “vulnerable to seemingly insignificant disturbance.” Even the amount of noise heard that comes from outside of wilderness can elicit high levels of concern among wilderness recreationists (Shafer and Hammitt 1995). Noss (1991) posits that our desire to manage wilderness is “exceedingly arrogant” and thus what is needed is recognition of a humility value that represents “self-imposed restraint in a society that generally seeks to dominate and control all of nature.” Recognizing restraint will prove increasingly difficult as use and intensity of wilderness continue to grow.

Problem Statement

Since 1965, recreation use in wilderness has grown by nearly 400 percent (Hampton and Cole 1995), increasing substantially during the 1990s in most wilderness areas and likely to intensify (Cole 1996). The protection of water resources is a vital component of wilderness integrity, and thus researchers commonly look to water to quickly determine the state of health of an entire watershed or ecosystem (Herrmann and Williams 1987). Several surveys reveal that the public believe preserving water quality is the most important wilderness value and reason for wilderness

In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O’Loughlin, Jennifer, comps. 2000. Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management; 1999 May 23–27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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protection (Cordell and others 1998; Kloefer 1992). Both standing and free-flowing water in wilderness is often the focal point of backcountry recreation; it tends to be limited and subject to ever-increasing consumptive, polluting and competing uses (Aukerman 1986). Research shows that certain backcountry locations with pristine-looking water can be contaminated with pathogenic organisms (Tippets 1999; Aukerman and Monzingo 1989; Suk 1986; Varness 1978). New and potentially dangerous organisms such as *Giardia lamblia* and *Cryptosporidium* are particularly worrisome because of their disabling effects and prevalence in some backcountry locations (Perry and Swackhammer 1990). While it has been difficult in the past to discern whether recreation is the cause of fecal contamination of water, new techniques have become more sophisticated. Human fecal contamination in recreation settings has been documented using a method that extracts the DNA from coliform bacteria to determine the source (human, beaver, horse, etc.) of the pollution (Tippets 1999).

The primary concerns of human waste disposal are, 1) the transmission of disease-causing organisms and, 2) the aesthetic concerns of improper human waste disposal or the accompanying sanitation infrastructure. The public is shown to be increasingly intolerant of sanitation problems. In their study of social and ecological normative standards, Whittaker and Shelby (1988) found that the standard for human waste represented a no-tolerance norm, in which 80 percent of the respondents reported that it was never acceptable to see signs of human waste. Increased use has led to increased social and biophysical impacts, particularly in sites not conducive to the decomposition of human waste. A recent study reports that 25 percent of National Park Service managers find human waste to be a common problem in many or most areas, and 43 percent consider it a serious problem in a few areas (Marion and others 1993). Increasing wilderness use, the severity of public health issues and lack of tolerance by the public combined with biophysical constraints, changing social values toward wilderness and limited human waste treatment and disposal techniques creates a complex situation for managers and scientists who must determine the application of a minimum tool.

Discussion

The matrix (table 1) and related flow chart (figure 1) were created to help managers and scientists design and maintain sanitation programs and infrastructures while incorporating minimum tool options and concerns about related impacts such as aesthetics, noise, trail erosion and the social acceptability of the option. Information contained in the matrix and flow chart were gathered from the limited quantity of research on water quality and human waste management in backcountry settings and makes explicit the technology or technique to treat and dispose of human waste, minimum tool options and related impacts. The flow chart presents various scenarios and actions relating to sanitation management options. The matrix establishes descriptions and related impacts of various sanitation techniques. Determinations of opportunity classes are based on Stankey and others (1990) and designed to define resource, social and managerial conditions considered

desirable and appropriate in wilderness. Opportunity classes associated with techniques are approximated to gauge the severity of obtrusiveness. Within the matrix, Class I implies little or no evidence of site management, while Class IV implies extensive use of onsite management and site modification.

Numerous organizations including the USDA Forest Service, Leave No Trace and the National Outdoor Leadership School detail the positive and negative attributes of various sanitation techniques in the backcountry. Clearly, no means of human waste disposal in the backcountry is without ramifications, and no one method can be unconditionally recommended for every situation. Even urine, which is ordinarily sterile, can attract wildlife that defoliate plants and disturb soils. (Hampton and Cole 1995; Cole 1989). Good judgement is the key to proper human waste disposal. Hampton and Cole (1995) maintain that disposal techniques are best when they: 1) diminish human, animal and insect contact, 2) encourage decomposition, and 3) avoid polluting water sources. The fate of pathogenic organisms in human waste deposited on or in soils is highly variable and depends on numerous factors including soil type, moisture and temperature.

The "cat hole" method allows for aerobic decomposition by microbial activity within individual shallow holes in the ground. Hampton and Cole (1995) report that this is the preferred method in nearly every outdoor environment. However, research has documented the ineffective breakdown of coliform bacteria using this technique (Temple and others 1982). Use of the cat hole procedure should not be attempted in areas with less than optimal conditions for decomposition, including moderate temperatures, presence of organic matter in the soil and low chance of being found by potential users. The group trench latrine is a technique in which the waste is buried in a shallow trench used by a small group. This technique can also apply to parties camping in snow conditions. However, waste deposited in permanent snow conditions will most likely take hundreds if not thousands of years to decompose (Ells 1997). The smear method, also known as surface disposal, is a technique in which the waste is spread thinly on the surface to allow aerobic decomposition by microbial activity and breakdown by ultraviolet radiation. The method works well in low-use locations where others are not likely to find the waste (Cole 1989). The individual pack-out method is gaining popularity in high-use areas. The waste is double-bagged, or single-bagged and placed in a tube. However, because of social acceptability issues, compliance is often low (Drake 1997). Numerous commercial options are available for the pack-out of group waste (Meyer 1994). The waste is sealed in an ammunition can or other secure receptacle and then carried out. This method is most common on river trips where the receptacle can be placed in a boat.

Treatment and disposal techniques that generally require a structure (outhouse) include pit toilets. Pit toilets offer a simple and relatively low maintenance method of waste treatment. However, these toilets are often anaerobic, characterized by slow decomposition and producing ammonia which is odorous. In addition, their use can affect water quality, depending on water table and flow path characteristics (Leonard and Plumley 1979). Composting

Table 1—Matrix of sanitation and minimum tool options in wilderness.

Technology/ technique	Description	Appropriate use	Minimum tool requirement	Potential social impact	Potential biophysical impact	Opportunity class
Cat hole	Waste is buried in individual shallow holes to allow decomposition by microbial activity	Low-use locations (esp. Desert, Temperate)	Educational displays; Periodic monitoring	Aesthetic concerns	Surface or ground water pollution; Issues related to urination	Class I
Smear (Surface disposal)	Waste is spread thinly on surface to allow decomposition by microbial activity and UV radiation	Low-use locations	Educational displays; Periodic monitoring	Aesthetic concerns	Surface or ground water pollution; Issues related to urination	Class I
Group Trench Latrine	Waste is buried in shallow trench by group to allow decomposition by microbial activity	Low-use locations (esp. Temperate)	Educational displays; Periodic monitoring	Aesthetic concerns	Surface or ground water pollution; Issues related to urination	Class I
Individual Pack-out	Waste is carried-out in tube or bag	High-use locations (esp. Alpine)	Educational displays; Periodic monitoring; Provision of bags or tubes	Social acceptability of method	Surface or ground water pollution; Issues related to urination	Class I
Group Pack-out	Waste is carried-out in ammo can or bag	High-use locations (esp. Riparian)	Educational displays; Periodic monitoring; Provision of ammo can or bags	Social acceptability of method	Noncompliance can lead to surface or ground water pollution; Issues related to urination	Class I
Pit	Waste is deposited in unlined hole in ground by multiple parties and decomposes anaerobically	High-use locations	Latrine structure; Field Staff required for on-site maintenance	Odor; Aesthetic concerns of latrine structure	Surface or ground water pollution	Class II to IV
Composting	Waste decomposes through mesophilic or thermophilic methods using wood chips	High-use locations (esp. Temperate)	Latrine structure; Field Staff required for on-site maintenance; Pack stock or helicopter required transport of wood chips to site and removal of solids	Aesthetic concerns of latrine structure or trail erosion issues	Impacts related to use of pack stock (seed dispersal, trail erosion) and helicopter (wildlife issues)	Class II to IV
Incineration	Waste is cooked in propane-fired chamber	High-use locations	Latrine structure; Pack stock or helicopter required for transport of propane and ash	Odor; Aesthetic concerns of latrine structure; Noise or trail erosion issues related to transport	Impacts related to use of pack stock (seed dispersal, trail erosion) and helicopter (wildlife issues)	Class II to IV

(con.)

Table 1 (Con.)

Technology/ technique	Description	Appropriate use	Minimum tool requirement	Potential social impact	Potential biophysical impact	Opportunity class
Dehydration	Waste is dried in straining system; can use passive solar assistance	High-use locations	Latrine structure; Liquid treatment system; Pack stock or helicopter required for removal of solids	Odor; Aesthetic concerns of latrine structure; Noise or trail erosion issues related to transport	Impacts related to use of pack stock (seed dispersal, trail erosion) and helicopter (wildlife issues); Surface or ground water pollution	Class II to IV
Vault with no liquid separation	Liquid and solid waste is collected in sealed vault	High-use locations	Latrine structure (Wallowa- style); Pack stock or helicopter required for removal of vault	Odor; Aesthetic concerns of latrine structure; Noise or trail erosion issues related to transport	Impacts related to use of pack stock (seed dispersal, trail erosion) and helicopter (wildlife issues)	Class II to IV
Vault with liquid separation	Liquid and solid waste is separated using strainer and liquid treatment system	High-use locations (esp. Temperate)	Latrine structure; Liquid treatment system; Pack stock or helicopter required for removal of solids	Odor; Aesthetic concerns of latrine structure; Noise or trail erosion issues related to transport	Impacts related to use of pack stock (seed dispersal, trail erosion) and helicopter (wildlife issues); Surface or ground water pollution	Class II to IV

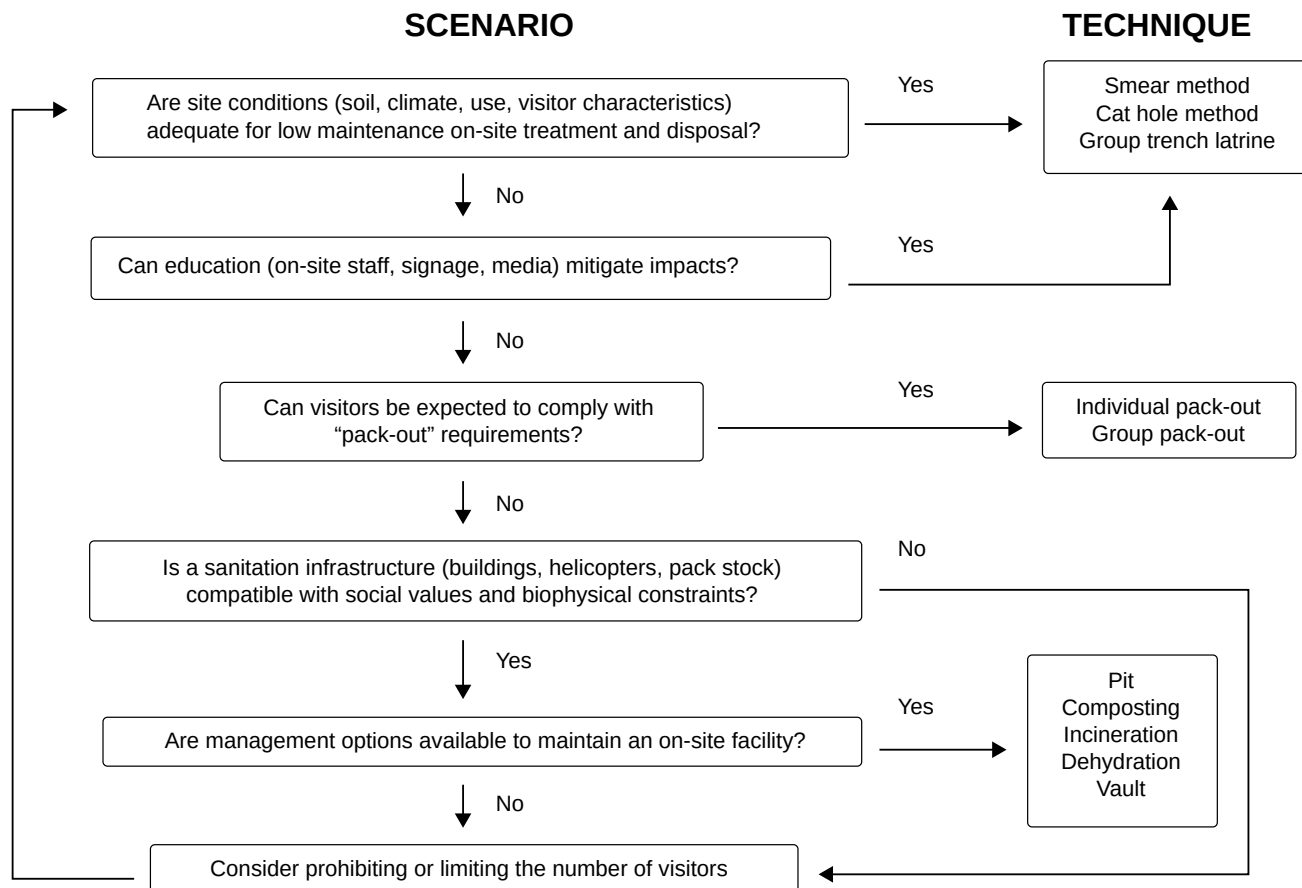


Figure 1—Flow chart of sanitation options for wilderness managers.

toilet options involve the aerobic breakdown of waste in a sealed bin or tank. These methods operate favorably in locations where the climate is temperate and there is regular maintenance. Numerous composting methods have been tested and used in various applications (Lachapelle 1997; Land 1995a; Davis and Neubauer 1995; Yosemite National Park 1994; Weisberg 1988; Jensen 1984; Cook 1981). Although not a popular option, incineration offers an alternative that has been applied in backcountry settings. Mechanical difficulties have often been cited as a limiting factor. The use of dehydrating toilets is especially popular in extreme conditions such as alpine or desert locations (Drake 1997; Mt. Rainier National Park 1993; McDonald and others 1987). Surface and ground water pollution can result from liquid discharge and the dehydrated solids must still be removed from the site. Vault toilets can either incorporate a liquid treatment system or be large enough to accommodate the liquid. (Land 1995b; Leonard and others 1981). Helicopters or pack stock such as mules are generally used in these situations because of the great weight and volume factors of transporting the waste. However, pack stock may contribute to fecal contamination of surface and ground water sources while the use of helicopters may intensify social impacts.

Conclusions and Recommendations

Several trends suggest that managers and scientists must design and maintain sanitation programs and infrastructures with an emphasis on incorporating minimum tool options and concerns about related impacts. First, use and impact have intensified and are expected to grow. Second, there is little research on sanitation and related public health concerns that result from wilderness use. Third, monitoring programs appear to be lacking. Marion and others (1993) found that only 52 percent of national parks had implemented some type of water quality monitoring program. Herrmann and Williams (1987) cite four reasons for a lack of water quality research in wilderness as the difficulty of access to sites, difficulty in discriminating the effects from background water quality levels, the magnitude of the action to the consequence and the limited opportunity for control in the wilderness environment.

Options for managers and scientists are often limited depending on social values and biophysical constraints. Cole and others (1987) describe five strategies for managers when dealing with human waste issues as 1) reducing

use (prohibiting or limiting the number of visitors), 2) modifying the location of use (locate facilities on durable sites), 3) modify type of use and visitor behavior (education), 4) increase resistance of the resource (provide sanitation infrastructure), and 5) maintain or rehabilitate the resource (remove waste from toilets). The matrix and flow chart incorporate these strategies in order to make explicit various sanitation techniques, minimum tool options and related impacts. Since these options present the manager with numerous potential management actions, they must all be considered in relation to social values and biophysical constraints. While a reduction in use can conceivably lessen the sanitation impact, Cole and others (1997) report that reduction levels can sometimes result in more negative than positive consequences. This has been described as the "toothpaste effect," in which limits on one area may expand to other areas when "pressed" by management actions (Cole 1993). Priorities should be well-developed in order to identify, monitor and publicly report the internal and external threats to wilderness values (McCool and Lucas 1990).

Increasingly, issues associated with visitor use and intensity, the severity of public health impacts and lack of tolerance by the public regarding sanitation has created a complex situation of determining methods of balancing minimum tool requirements and wilderness values. The difficult issue of sanitation options in wilderness would benefit from increased discussion and research. The situation remains a challenge for managers and scientists who strive to ameliorate the issues associated with sanitation, increasing use and changing values toward wilderness.

References

- Aukerman, R. 1986. Overview of the session on water. In: Proceedings of the National Wilderness Research Conference: Current Research. Gen. Tech. Rep. INT-220. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 215.
- Aukerman, R.; Monzingo, D. L. 1989. Water treatment to inactivate Giardia. *Journal of Forestry*. 87:18-21.
- Cole, D. N. 1989. Low-impact recreational practices for wilderness and backcountry. Gen. Tech. Rep. INT-265. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 131 p.
- Cole, D. N. 1993. Wilderness recreation management: we need more than bandages and toothpaste. *Journal of Forestry*. 91(2):22-24.
- Cole, D. N. 1996. Wilderness recreation in the United States: trends in use, users and impacts. *International Journal of Wilderness*. 2(3):14-18.
- Cole, D. N.; Peterson, M. E.; Lucas, R. C. 1987. Managing wilderness recreation use: common problems and potential solutions. Gen. Tech. Rep. INT-230. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 60 p.
- Cole, D. N.; Watson, A. E.; Hall, T. E.; Spildie, D. R. 1997. High-use destinations in wilderness: social and biophysical impacts, visitor responses, and management options. Gen. Tech. Rep. INT-RP-469. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 30 p.
- Cook, B. 1981. Field evaluation of composting toilets. In: McClelland, N. I.; Evans, J. L., eds. Individual on-site wastewater systems: proceedings of the seventh national conference. Ann Arbor: National Sanitation Foundation: 83-98.
- Cordell, H. K.; Tarrant, M. A.; McDonald, B. L.; Bergstrom, J. C. 1998. How the public views wilderness: more results from the USA survey on recreation and the environment. *International Journal of Wilderness*. 4(3): 28-31.
- Davis, B.; Neubauer, P. 1995. Manual for bin composting and waste management in remote recreation areas. 4th ed. Waterbury Center, VT: Green Mountain Club. 68 p.
- Drake, R. 1997. Backcountry human waste disposal at Mount Rainier national Park. In: Josephson J., ed. Environmental ethics and practices in backcountry recreation. Canmore, Alberta: Alpine Club of Canada: 21-24.
- Ells, M. D. 1997. Impact of human waste disposal on surface water runoff: the Muir snowfield, Mount Rainier. *Journal of Environmental Health*. 59(8):6-13.
- Hampton, B.; Cole, D. N. 1995. Softpaths: How to enjoy the wilderness without harming it. Stackpole Books, Mechanicsburg, PA. 222 p.
- Hendee, J. 1990. Principles of wilderness management. In: Hendee, J.; Stankey, G.; Lucas, R., eds. Wilderness Management. 2nd ed. Golden, CO: Fulcrum: 181-193.
- Herrmann, R.; Williams, O. R. 1987. Water resources research for wilderness: A state-of-knowledge review. In: Lucas, R. C. comp. Proceedings of the National Wilderness Research Conference: Issues, State-of-knowledge, Future Directions. Gen. Tech. Rep. INT-220. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 191-202.
- Jensen, M. E. 1984. National Park Service remote area toilet facilities: experiences and observations 1983 and 1984. NPS Report No. 85-01-NPS/ESSD. Washington, DC: Environmental Sanitation Program, National Park Service.
- Kloepfer, D. 1992. Keeping it wild: a citizen guide to wilderness management. Washington DC: Wilderness Society. 70 p.
- Lachapelle, P. R. 1997. Composting toilet options: a northeastern United States perspective. In: Josephson J., ed. Environmental ethics and practices in backcountry recreation. Canmore, Alberta: Alpine Club of Canada: 50-59.
- Land, B. 1995a. Composting toilet systems: planning, design, and maintenance. Tech. Rep. SDTDC 9523-1803. San Dimas, CA: U.S. Department of Agriculture, Forest Service, Technology and Development Program. 38 p.
- Land, B. 1995b. Remote waste management. Tech. Rep. SDTDC 9523-1202. San Dimas, CA: U.S. Department of Agriculture, Forest Service, Technology and Development Program. 31 p.
- Leonard, R. E.; Plumley, H. J. 1979. Human waste disposal in eastern backcountry. *Journal of Forestry*. 77(5):349-352.
- Leonard, R. E.; Spencer, E. L.; Plumley, H. J. 1981. Backcountry facilities: design and maintenance. Boston, MA: Appalachian Mountain Club. 214 p.
- Marion, J. L.; Roggenbuck, J. W.; Manning, R. E. 1993. Problems and practices in backcountry recreation management: a survey of National Park Service managers. Natural Resources Report NPS/NRVT/NPR.93/12. Denver, CO: U.S. Department of the Interior, National Park Service, Natural Resources Publication Office. 65 p.
- McCool, S. F.; Lucas, R. C. 1990. Managing resources and people in wilderness: accomplishments and challenges. In: Lime, D. W., ed. Managing America's enduring wilderness resource. St. Paul, MN: Minnesota Agricultural Experiment Station: 64-87.
- McDonald, J.; Stanley, R.; McCauley, D. 1987. Mount Whitney solar toilet. U.S. Department of Agriculture, Forest Service, Engineering Field Notes. 19:13-19.
- Meyer, K. 1994. How to shit in the woods. 2nd ed. Berkeley: Ten Speed Press. 107 p.
- Mount Rainier National Park. 1993. Backcountry toilet technology workshop. Unpublished report. Ashford, WA: U.S. Department of Interior. National Park Service, Mount Rainier National Park.
- Nash, R. 1996. Soul of the wilderness: a wilderness ethic for the age of cyberspace. *International Journal of Wilderness*. 2(3):4-5.
- Noss, R. F. 1991. Sustainability and wilderness. *Conservation Biology*. 5(1): 120-122.
- Perry, J.; Swackhammer, D. 1990. Nice place to visit but don't drink the water. In: Lime, D. W., ed. Managing America's enduring wilderness resource. St. Paul, MN: Minnesota Agricultural Experiment Station: 524-533.
- Shafer C. S.; Hammitt, W. E. 1995. Purism revisited: specifying recreational conditions of concern according to resource intent. *Leisure Sciences*. 17:15-30.
- Shindler, B. 1992. Countering the law of diminishing standards. In: Shelby, B.; Stankey, G.; Shindler, B., eds. Defining wilderness quality: the role of standards in wilderness management-a work-

- shop proceedings. Gen. Tech. Rep. PNW-GTR-305. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 114 p.
- Stankey, G.; McCool, S. F.; Stokes, G. L. 1990. Managing for appropriate wilderness conditions: the carrying capacity issue. In: Hendee, J.; Stankey, G.; Lucas, R., eds. Wilderness management. 2nd ed. Golden, CO: Fulcrum: 215-239.
- Suk, T. J.; Riggs, J. L.; Nelson, B. C. 1986. Water contamination with Giardia in back-country areas. In: Proceedings of the National Wilderness Research Conference: Current Research. USDA Forest Service General Technical Report INT-212. Ogden, UT: Intermountain Research Station: 237-244.
- Temple, K.; Camper, A.; Lucas, R. 1982. Potential health hazards from human wastes in wilderness. *Journal of Soil and Water Conservation*. 37(6):357-359.
- Tippets, N. 1999. Backcountry water quality testing in Grand Teton National Park. Unpublished report. Moose, WY: Grand Teton National Park.
- Varness, K. J.; Pacha, R. E.; Lapen, R. F. 1978. Effects of dispersed recreational activities on the microbiological quality of forest surface water. *Applied and Environmental Microbiology*. 36(1):95-104.
- Weisburg, S. 1988. Composting options for wilderness management of human waste. Unpublished report. Rep. No. K70172. Sedro Woolley, WA: USDI National Park Service, North Cascades National Park, Skagit District. 57 p.
- Whittaker, D.; Shelby, B. 1988. Types of norms for recreation impacts: extending the social norms concept. *Journal of Leisure Research*. 20(4):261-273.
- Yosemite National Park. 1994. Backcountry human waste management: a guidance manual for public composting toilet facilities. El Portal, CA: USDI National Park Service, Yosemite National Park. 83 p.

Wilderness Campsite Conditions Under an Unregulated Camping Policy: An Eastern Example

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Abstract—This study identified and assessed 110 campsites in seven designated wilderness areas in the Jefferson National Forest of Virginia. The campsites were unevenly distributed within each wilderness, concentrating along trail corridors and near popular destination areas. With a few exceptions, most campsites surveyed were in good condition. The findings indicate that management actions should be directed at reducing both the number of campsites and the problems associated with campsite expansion. The Forest's unregulated camping policy could be focused through educational programs to encourage dispersed camping or camping containment to further reduce social and resource impacts.

Managing campsite impacts has always been a challenging task for wilderness managers, who are required by the 1964 Wilderness Act to preserve and enhance the wilderness resource while providing opportunities for solitude and unconfined recreation (Conrad 1997; Washburne and Cole 1983). The success of this task depends in part on the availability and judicious use of objective, timely information on the numbers, distribution and resource conditions of campsites. Impact assessment and monitoring (IA&M) programs for campsites, which can yield such information, are growing in recognition and use. However, these programs are less common in Eastern wilderness areas (McEwen and others 1996; Williams and Marion 1995).

Earlier settlement has left little wilderness in the Eastern United States. Only about four percent of the entire designated wilderness acreage is located in Eastern states (Landres and Meyer 1998). In general, Eastern wildernesses are much smaller (25% of the Western average), and they are closer to population centers (Landres and Meyer 1998). Despite their unique environmental and use attributes, Eastern wildernesses have received less research attention compared with their Western counterparts (Kulhavy and Legg 1998). This lack of information has limited our knowledge of region-specific impact patterns and trends, as well as the ability of wilderness managers to respond with effective campsite-management strategies and actions.

Camping and its associated resource and social impacts have been managed under a number of different policies and strategies (Leung and Marion 1999). Areas containing rare or sensitive natural and cultural resources may be closed to camping. A dispersed camping strategy seeks to reduce the frequency of camping use to avoid or minimize permanent resource impacts or visitor crowding. In more heavily visited areas, such impacts are often limited effectively by restricting camping to established or designated campsites. However, camping is unregulated in most wilderness areas, allowing visitors the freedom to select existing campsites or to create new campsites.

This paper presents results from the development and implementation of a campsite IA&M program for 11 wilderness areas of the Jefferson National Forest, Virginia. A comprehensive survey of wilderness campsites was performed to provide a baseline data set for comparison with future conditions (Leung and Marion 1995). This paper presents selected findings and discusses some implications of the study. In particular, we examine the potential resource and social effects of the Forest's unregulated camping policies for these areas.

Study Area

The Jefferson National Forest was established in 1936. In 1995, the USDA Forest Service combined the Jefferson and adjacent George Washington National Forests to form a single administrative unit. The results and discussion that follow refer to the Jefferson National Forest portion of the unit.

Situated in the Appalachian Mountains of southwestern Virginia, the Jefferson National Forest encompasses more than 1.6 million acres, 41% of which are federally owned. Forest overstory is classified as Appalachian hardwoods, comprising predominantly of upland oak and including poplar, hickory, pine and other hardwoods. The Forest is managed under a multiple-use and sustained-yield mandate designed to maximize the production of goods and services in an environmentally sound manner. Forest uses include timber, recreation, fisheries, wildlife, mineral and energy resources.

The Forest contains 11 wilderness areas with a total size of 57,760 acres (fig. 1). The Appalachian National Scenic Trail and Virginia Creeper National Recreational Trail traverse some parts of the wildernesses. More than 76,000 recreation visitor days (RVDs) were recorded for these wilderness areas in 1993 (Jefferson National Forest, unpublished statistics). About 70% of the total visitation was

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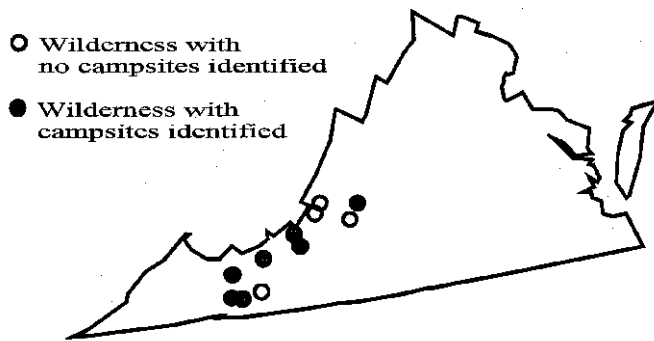


Figure 1—Location of the wilderness areas included in this study.

accommodated by three areas: Lewis Fork, Mountain Lake and Little Wilson Creek Wildernesses (table 1).

An unregulated or “at-large” camping policy has been adopted for these wilderness areas: camping is permitted throughout each area, unless otherwise posted as closed to visitor use. Overnight stay permits are not required. There is no limit on party size, though there is a 21-day limit on the total duration of overnight stays. Wood campfires are allowed. Information on minimum-impact recreation practices is available at ranger offices and visitor centers.

The Forest is in the process of implementing the Limits of Acceptable Change (LAC) planning framework in its wilderness areas. This study was initiated as part of the LAC process, which emphasizes the formulation of indicators and standards (Stankey and others 1985). An earlier survey was conducted by the Forest staff and reported by Marion (Marion 1991b). The current study was considered a refinement of the earlier survey, with substantial changes in survey procedures. It is not the intent of this paper to compare results from the two surveys.

Methods

This study included all 11 wilderness areas of the Jefferson National Forest, two of which fall within the boundaries of the Mount Rogers National Recreation Area (NRA). The high country non-wilderness zone of the Mount Rogers NRA

was also included. Only results from the wilderness areas are presented. The survey procedures were adapted from Marion (1991a), which combined condition class and multi-parameter IA&M approaches in order to document the numbers, distribution and resource conditions of campsites. Extensive searches along trail corridors and at potential use areas were conducted with the Forest staff to identify and locate campsites. A census was considered necessary to establish a baseline database and provide information for wilderness planning activities. At each campsite, boundaries were defined according to vegetation change, plant litter and local topography. Inventory indicators were recorded, including locational information (GPS coordinates and description), site position on slope, distance to water sources, distance to trails and visibility from trails or other campsites. Impact indicators were also assessed, which included site size (area of disturbance), number of fire sites, groundcover vegetation loss, soil exposure, trees with exposed roots, damage to tree trunks, tree stumps, human waste and human trash. Comprehensive descriptions of the field procedures are provided in the final management report (Leung and Marion 1995).

Results

The survey identified a total of 110 campsites distributed in seven wilderness areas. No campsites were identified in four wilderness areas (fig. 1); possible reasons for the lack of campsites include the relative inaccessibility of and low visitation to these four areas.

Nearly three-quarters of the campsites were located within sight of established trails (table 2). Over one-third of the campsites (38%) were located less than 25 feet from trails, another 27% were located between 25 and 100 feet from trails. Site intervisibility was mixed: While 59% of the sites had no other sites visible, 14% had one other site visible, 18% had two other sites visible, and 9% had three other sites visible (table 2). A substantial number of sites (70%) were somewhat distant (> 200 ft) from water sources, although one-quarter were located less than 100 feet. An example of uneven distribution of campsites is shown as figure 2. In the Lewis Fork Wilderness, the vast majority of campsites were located right along trail corridors or at trail junctions (fig. 2).

Table 1—Wilderness areas of the Jefferson National Forest included in this study.

Wilderness	Size	Visitation	Accessibility/level of facility
	<i>acres</i>	<i>RVD^a</i>	
Barbours Creek	5,382	2,650	Accessible; 1 maintained trail
Beartown	5,609	1,540	Very remote; no maintained trails
James River Face	8,886	4,466	Very accessible; 6 maintained trails
Kimberling Creek	5,542	1,320	Accessible; no maintained trails
Lewis Fork	5,618	25,350	Very accessible; several maintained trails
Little Dry Run	2,858	1,950	Very accessible; 1 maintained trail
Little Wilson Creek	3,613	11,700	Remote; 4 maintained trails
Mountain Lake	11,113	15,600	Very accessible; several maintained trails
Peters Mountain	3,328	9,200	Very accessible; several maintained trails
Shawvers Run	3,467	1,350	Accessible; no maintained trails
Thunder Ridge	2,344	1,334	Very accessible; one maintained trail

^aRecreation visitor days in 1993 (Jefferson National Forest, unpublished statistics).

Table 2—Number and percent of wilderness campsites for selected inventory indicators.

Inventory indicator	Wilderness campsites (N = 110)	
	Number	Percent
Site visibility from trail		
Yes	79	72
No	31	28
Distance to formal trail (ft)		
<25	41	38
25-100	30	27
101-200	19	17
>200	20	18
Other sites visible (#)		
0	65	59
1	15	14
2	20	18
3	10	9
Distance to water (ft)		
<25	9	8
25-100	18	16
101-200	6	6
>200	77	70

In addition, campsites tended to proliferate in close proximity to shelters, where water sources and flat grounds are usually available. This distribution pattern was common to other wilderness areas of the Forest.

Survey data revealed that 59% of the campsites were in condition classes 1 and 2, indicative of no discernible soil exposure onsite. However, groundcover vegetation loss was substantial in the James River Face and Lewis Fork Wildernesses, with mean losses of 57% and 60%, respectively. Most campsites were generally small in size; 67% of the sites were less than 500 ft². Damaged trees, root exposure and tree stumps were not serious problems on the

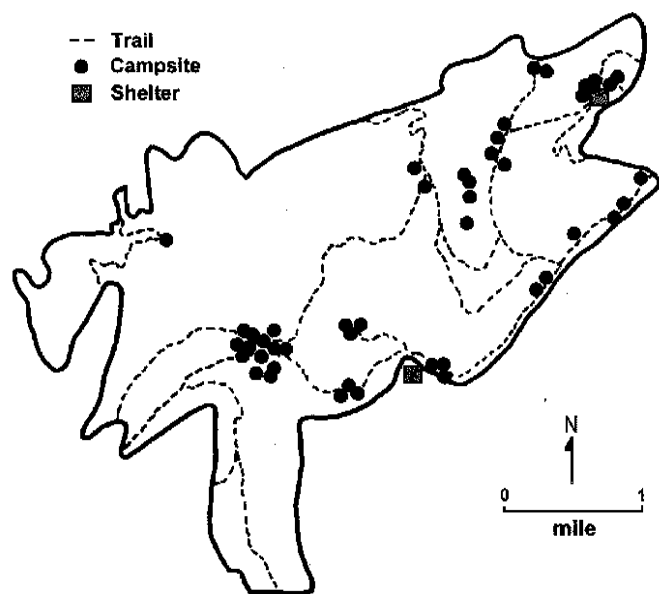


Figure 2—Spatial distribution and clustering of campsites in the Lewis Fork Wilderness.

majority of campsites (table 3). Except in the Beartown Wilderness, campsites tended to have large numbers of radiating social trails, which are indicative of potential problems with campsite expansion and proliferation, as reported in other more heavily visited wilderness areas (Cole 1993; Cole and others 1997).

The uneven distribution of visitation among wilderness areas was reflected by different levels of impact. Campsites in the Lewis Fork and Mountain Lake Wildernesses, the two most visited areas, received greater resource impact than other wilderness areas (tables 1 and 3). In particular, average campsite sizes in these two wilderness areas were larger, indicating a larger area of site disturbance, including groundcover loss and soil exposure.

With respect to aggregate impacts, the Lewis Fork Wilderness had the largest extent of impact on all three aggregate measures of site size, vegetation loss and soil exposure (table 4). Both the moderate level of impact intensity and the large number of campsites contributed to the larger aggregate impact measures.

Management Implications and Conclusions

The findings of this study show that some wildernesses in the Eastern U.S. may receive very low overnight visitation and associated resource impacts, despite the fact that they are relatively close to population centers. The inaccessibility and low use of these areas may facilitate restoration of vegetation and soil in the more resilient Eastern environment (Cole and Marion 1988).

The survey found several higher-use destination areas with larger numbers of campsites, some in tight clusters. Campsite locations reflect the site choices of visitors, as camping is unregulated in these wilderness areas. Although field staff conducted extensive searches of distant and hidden potential camping locations, our results reveal that a majority of campsites were located within sight of established trails. Only 20 (18%) of the campsites were found more than 200 feet from a trail. The Forest staff concurred with these findings, noting that relatively few visitors currently practice dispersed camping. However, due in part to more dense Eastern forest vegetation, campsite intervisibility was relatively low, though site clustering did occur in a few popular areas.

Survey findings suggest that visitors to these lower-use Eastern wilderness areas are not selecting campsites based on a desire for solitude or privacy. In particular, visitors who camp close to trails reduce the potential for solitude of both hikers and campers. Topography presents significant limitations in many areas: Mountainous terrain largely restricts camping to flat ground along stream drainages and on ridgetops. Trails are often routed along these topographic features as well, further limiting the ability of visitors to locate more distant camping locations. Novice visitors may fear getting lost if they venture too far from trails. Other visitors may simply take the first available campsite they see when they reach their destination. Trailside campsites may be more convenient to use than those requiring searches through difficult off-trail vegetation and terrain. Finally, proximity to an attractive destination location, water or the

Table 3—Mean conditions for selected inventory and impact indicators for campsites assessed in the wilderness areas.

Wilderness ^a	Number sites	Visible from trail	Site size	Indicator			
				Vegetation loss	Exp. soil	Damaged trees	Social trails
		%	ft ²	%	%	no.	no.
Beartown	5	100	338	0	0	0.4	0
James River Face	21	76	365	57	37	0.9	5
Lewis Fork	49	63	771	60	29	0.5	4
Little Wilson Creek	18	61	349	32	14	0	3
Mountain Lake	12	92	861	24	35	1.2	5
Peters Mountain	4	100	496	0	27	0.3	6

^aResults from the Kimberling Creek Wilderness were excluded due to insufficient sample size.

Table 4—Aggregate measures for selected impact indicators for campsites assessed in the wilderness areas.

Wilderness ^a	Sum of site size	Indicator sum of vegetation loss	Sum of soil exposure
		ft ²	
Beartown	1,690	0	0
James River Face	7,671	3,809	2,551
Lewis Fork	37,764	17,169	8,780
Little Wilson Creek	6,282	497	222
Mountain Lake	10,322	2,410	3,546
Peters Mountain	1,982	1,404	613
Total	65,711	25,289	15,712

^aResults from the Kimberling Creek Wilderness were excluded due to insufficient sample size.

trail may simply be more important than finding a camping location that enhances solitude.

An unregulated camping policy does provide the freedom and opportunity for visitors to locate a campsite that ensures their solitude while camping. Visitors could be encouraged to hike off-trail to discover a more distant and private campsite or location. However, a second group, one less concerned about solitude, might still show up after the tents are set up and camp close by. Educational efforts for wilderness visitors should address the issue of solitude, directing visitors to camp out of sight or at some minimum distance from other groups.

Campsite locations assessed in this study were generally neither resistant nor resilient to visitor impacts (Marion and Proudman 1999). Most campsites were located under forest canopies on fragile forest herbs; some were located close to streams. Soil from riparian zone campsites can be eroded directly into streams, contributing sediments to aquatic communities. However, with a few exceptions, campsites were generally small in size and in good condition. These findings are probably attributable to the relatively low use levels and small group sizes common to most of the wildernesses surveyed. Campsite expansion and proliferation were evident at several popular locations, as evidenced by large clusters of sites (fig. 2). Enlargement of some core sites was causing them to merge together to form excessively large

camping areas. Management responses are urgently needed for these areas. In particular, controlling the spatial growth of established campsites and minimizing the creation of new campsites at these high-use locations are needed to curb the expansion of resource impacts in these areas. A similar situation can be found in other parts of the country (Cole and others 1997; McEwen and others 1996).

As with the management of social problems, resource impact management under an unregulated camping policy is largely an issue of effective visitor education. In lower use travel zones, resource impacts can be minimized with a dispersed camping strategy that encourages visitors to select resistant pristine sites and employ *Leave No Trace* camping practices (National Outdoor Leadership School 1994). Managers have had relatively low success with dispersed camping, however, due to many of the previously discussed campsite selection factors. In addition, few areas have enough resistant flat locations to sustain such a strategy. Management experience and research suggest that a camping containment strategy minimizes resource impacts more effectively, particularly in moderate to heavy use areas. Educational materials can encourage visitors to use only well-established existing campsites. *Leave No Trace* camping practices, such as concentrating use and impact on the most resistant or disturbed surfaces, can also help reduce impacts. More discussion on these alternative impact management strategies can be found in Cole and others (1987) and Leung and Marion (1999).

This study demonstrates that data generated from campsite IA&M programs can inform and aid in management decision-making, particularly when evaluating the effectiveness of policies, strategies and actions in minimizing visitor impacts. The continuation of such programs is critical for providing timely feedback to wilderness managers who try to balance nature preservation and recreation.

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References

- Cole, David N. 1993. Campsites in Three Western Wildernesses: Proliferation and Changes in Condition Over 12 to 16 Years. Research Paper INT-463. Ogden, UT: USDA Forest Service, Intermountain Research Station. 15p.
- Cole, David N.; Marion, Jeffrey L. 1988. Recreation impacts in some riparian forests of the eastern United States. *Environmental Management*. 12(1): 99-107.
- Cole, David N.; Petersen, Margaret E.; Lucas, Robert C. 1987. Managing Wilderness Recreation Use: Common Problems and Potential Solutions. General Technical Report INT-230. Ogden, UT: USDA Forest Service, Intermountain Research Station. 60p.
- Cole, David N.; Watson, Alan E.; Hall, Troy E.; Spildie, David R. 1997. High-Use Destinations in Wilderness: Social and Biophysical Impacts, Visitor Responses, and Management Options. Research Paper INT-RP-496. Ogden, UT: USDA Forest Service, Intermountain Research Station. 30p.
- Conrad, Richard 1997. National survey highlights agency training needs in the United States. *International Journal of Wilderness*. 3(4): 9-12.
- Kulhavy, David L.; Legg, Michael H. 1998. Wilderness and Natural Areas in Eastern North America: Research, Management and Planning. Nacogdoches, TX: Stephen F. Austin State University, Center for Applied Studies in Forestry. 321p.
- Landres, Peter; Meyer, Shannon 1998. National Wilderness Preservation System database. General Technical Report RMRS-GTR-18. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station. 97p.
- Leung, Yu-Fai; Marion, Jeffrey L. 1995. A Survey of Campsite Conditions in Eleven Wilderness Areas of the Jefferson National Forest. Blacksburg, VA: USDI National Biological Service, Virginia Tech Cooperative Park Studies Unit. 79p.
- Leung, Yu-Fai; Marion, Jeffrey L. 1999. Spatial strategies for managing visitor impacts in national parks. *Journal of Park and Recreation Administration* 17(4): 20-38.
- Marion, Jeffrey L. 1991a. Developing a Natural Resource Inventory and Monitoring Program for Visitor Impacts on Recreation Sites: A Procedural Manual. Natural Resources Report NPS/NRVT/NRR-91/06. Denver, CO: USDI National Park Service, Natural Resources Publication Office. 59p.
- Marion, Jeffrey L. 1991b. Results from the Application of a Campsite Inventory and Impact Monitoring System in Eleven Wilderness Areas of the Jefferson National Forest. Blacksburg, VA: USDI National Park Service, Virginia Tech Cooperative Park Studies Unit. 73p.
- Marion, Jeffrey L.; Proudman, Robert D. 1999. May the forethought (and studies) be with your campsite protection planning! *The Register* 23(2):12-15.
- McEwen, Douglas; Cole, David N.; Simon, Mark 1996. Campsite Impacts in Four Wildernesses in the South-Central United States. Research Paper INT-RP-490. Ogden, UT: USDA Forest Service, Intermountain Research Station. 12p.
- National Outdoor Leadership School 1994. Leave No Trace Outdoor Skills and Ethics: Southeastern States. Lander, WY: National Outdoor Leadership School. 16p. (A series of LNT booklets are also available on the web at: <http://www.lnt.org/>).
- Stankey, George H.; Cole, David N.; Lucas, Robert C. and others 1985. The Limit of Acceptable Change (LAC) System for Wilderness Planning. General Technical Report INT-176. Ogden, UT: USDA Forest Service, Intermountain Research Station. 37p.
- Washburne, Randel F.; Cole, David N. 1983. Problems and Practices in Wilderness Management: A Survey of Managers. Research Paper INT-304. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 56p.
- Williams, Peter B.; Marion, Jeffrey L. 1995. Assessing Campsite Conditions for Limits of Acceptable Change Management in Shenandoah National Park. Technical Report NPS/MARSHEN/NRTR-95/071. Blacksburg, VA: USDI National Biological Service, Virginia Tech Cooperative Park Studies Unit. 138p.

The Consequences of Trampling Disturbance in Two Vegetation Types at the Wyoming Nature Conservancy's Sweetwater River Project Area

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Sharon Kehoe
Dayna Ayers-Baumeister

Abstract—The consequences of human trampling disturbance on two codominant vegetation types at the Wyoming Nature Conservancy's Sweetwater Preserve were examined. Small trampling lanes (1.5m x 0.5m) were established in both vegetation types and trampling treatments ranging from 0 to 800 passes were applied. *Artemisia* (Sagebrush) vegetation type was more sensitive to initial trampling disturbance than the *Equisetum* (Smooth scouring rush) community. After one year, however, both communities closely resembled predisturbance conditions, in terms of relative cover, relative height and percent bare ground. These results suggest that these vegetation types could withstand a moderate amount of visitor use without extensive degradation, although it would be prudent to continue monitoring conditions and regulating use levels to ensure that impacts do not proliferate.

The demand for recreational opportunities in the Rocky Mountain Region has resulted in increases in visitation in wilderness and nonwilderness lands. Many accessible "wildlands," while not designated wilderness, represent important areas ecologically and if managed correctly, could also provide areas for primitive recreation experiences. For example, many areas managed by The Nature Conservancy (TNC), although primarily managed for biodiversity and habitat protection, are near federally protected lands and can offer opportunities for primitive recreation. Extending wilderness management techniques to these areas, where appropriate, would benefit these lands directly and also provide a significant extension of wilderness preservation concepts. Moreover, these areas are often closer to population centers than designated wilderness and have a high

potential for primitive recreation, provided that the human activities can coexist with resource preservation.

Information on the relative tolerance of ecosystems to human use is essential to land management. Frequently there are apparent conflicts between allowing access for recreation and the preservation of natural conditions (Cole 1995a). Experimental trampling of groundcover vegetation has often been utilized as an index of tolerance to human use. It has the advantage of eliminating confounding variables and utilizes small plots of previously undisturbed vegetation. This approach was initiated by Wagar (1964) and has been used on many vegetation types worldwide, including arctic tundra (Monz and others 1996), mountain regions in the United States (Cole 1995 a,b), and heath communities in Scotland (Bayfield 1979). The standard methodology, as suggested by Cole and Bayfield (1993), has been utilized in many of these studies and therefore comparisons across different ecosystem types are possible.

The objective of this project was to investigate the consequences of human trampling on two distinct vegetation types at TNC's Sweetwater Preserve. We conducted experiments in which controlled levels of trampling were applied to plant communities in areas of potential increased recreation use. This technique is particularly applicable to this preserve for several reasons. First, few developed trails exist, and the development of trails is deemed undesirable by management objectives. Second, the vegetation and soils in certain areas of the preserve could be subject to significant disturbance given the current visitor use patterns. Last, regulating use levels below thresholds of disturbance to maintain pristine conditions is a feasible management option for the preserve.

Methods

Study Site

The Sweetwater River Preserve is located roughly at 42° N 108° W at an elevation of 2000 m and totals approximately 1200 ha. The land and conservation easements on an adjacent 600 ha were purchased 1991 by the Wyoming chapter of The Nature Conservancy. The Sweetwater River is a major tributary of the North Platte River and the area represents

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one of the few relatively undisturbed riparian habitats in Wyoming.

Plant Communities

We selected the two codominant plant communities on the preserve for trampling experiments; one with a dominant overstory of *Artemisia tridentata* (big sagebrush) and the other dominated by *Equisetum laevigatum* (smooth scouring rush) and other graminoids such as *Poa* spp. and *Koeleria macrantha* (june-grass). At the initiation of the experimental work the pre-trampling species abundance for experimental plots in both sites (table 1) were assessed with standard techniques. For the purposes of this project, *Artemisia* was not directly affected by the trampling treatments (since it is not a groundcover) and is therefore not included in the results. Both of these communities are in an area selected for potential increased use as they are adjacent to visitor cabins and are in an area of fishing access to the Sweetwater River. Experimental plots were located approximately 100 m apart and roughly 50 m from the river's bank. Soils are a relatively uniform sandy loam.

Experimental Treatments

Trampling—Experimental design for the trampling treatments follows the standard protocols described by Cole and Bayfield (1993). Four replicates of experimental trampling lanes (1.5m x 0.5m) were established in each of the two vegetation types. Lanes were selected within experimental blocks on the basis of suitability of application of trampling and homogeneity of the vegetation. Each replicate consists of five lanes; control (untreated), 25, 75, 200 and 500 trampling passes. A pass is a one-way walk at a natural gait along the lane; the people weigh 60-75 kg and wear lug sole boots. Treatments were applied once during early summer at the time of maximal seasonal biomass. For examinations of the overall ability of vegetation to tolerate recreational use, application of trampling at one time has been shown to be equally as effective as multiple treatments throughout the season. (Bayfield 1979; Cole 1985).

All areas on the preserve are sometimes subjected to cattle grazing at various times during the growing season. Plots were isolated from this potential confounding disturbance by using grazing exclosures, and a complete set of trampling

Table 1—Initial frequency and mean percent cover of the more abundant species in each of the two vegetation types.^a

Species	Vegetation type			
	<i>Equisetum laevigatum</i>		<i>Artemisia tridentata</i>	
	Freq.	Cover	Freq.	Cover
<i>Equisetum laevigatum</i>	98	3		
<i>Koeleria macrantha</i>	88	21		
<i>Poa juncifolia</i> var. <i>juncifolia</i>	88	35		
<i>Poa palustris</i>	83	48		
<i>Elymus trachycaulus</i>	80	11		
<i>Erigeron glabellus</i>	80	16		
<i>Taraxacum officinale</i>	75	22		
<i>Trifolium longipes</i>	73	7		
<i>Astragalus agrestis</i>	70	9		
<i>Agrostis variabilis</i>			100	72
<i>Sporobolus cryptandrus</i>			100	31
Moss			73	3
<i>Erigeron caespitosus</i>			58	2
<i>Carex</i> spp.	65	3	50	14
<i>Iva axillaris</i>			43	2
<i>Elymus lanceolatus</i>	68	17	23	5
<i>Muhlenbergia richardsonis</i>	63	7		
<i>Juncus balticus</i>	53	4		
<i>Erigeron</i> sp.	48	9		
<i>Iris missouriensis</i>	45	15		
<i>Elymus trachycaulus</i> var. <i>andinus</i>	20	5		
<i>Polygonum viviparum</i>	20	9		
<i>Sporobolus</i> sp.	13	6		
<i>Deschampsia caespitosa</i>	8	11		
<i>Agoseris glauca</i>	5	8		
<i>Phleum pratense</i>	5	3		
<i>Stellaria longipes</i>	5	3		
<i>Castilleja flava</i>			15	5
<i>Elymus cinereus</i>			10	5
<i>Chrysothamnus viscidiflorus</i>			8	10
Cryptogam			3	5

^aOnly species with mean cover of at least 2% are included. Frequency is the percent of the forty 30 x 50-cm plots in which the species was found.

lanes were also exposed to potential grazing in each vegetation type.

Trampling Response Variables

Standard indices of trampling effects (Cole and Bayfield 1993) were recorded in each lane in one 30 x 50 cm subplot. Measurements consisted of 1) visual estimates of canopy coverage of each vascular plant species (only green material) and of mosses and lichens; 2) visual estimates of the cover of bare ground, which included mineral soil, organic material and plant litter; and 3) determinations of vegetation height, using a point quadrat frame with five pins five cm apart within the width of the subplot, for a total of 50 pin drops. Every effort was made to standardize and calibrate ocular cover estimates by using 100 random pin drops per subplot as a baseline in initial trial runs, and then basing final ocular estimates on these results. Soil compaction was estimated using a pocket soil penetrometer (Forestry Suppliers, Inc. Jackson, MS 39284-8397 USA) and two random measurements per subplot. Measurements were performed approximately two weeks after trampling to determine the initial resistance and repeated one year later to determine the subsequent resilience.

Data Analysis

For the trampling results, analysis follows the suggested protocols of Cole and Bayfield (1993) where the primary response variable for each vegetation type is relative cover. This is a measure of the proportion of the original vegetation that survives trampling and is adjusted for changes occurring on control plots. It is calculated by summing all the percent covers of individual species to obtain total cover and then calculating relative cover as:

$$\frac{\text{Surviving cover on trampled subplots}}{\text{Initial cover on trampled subplots}} \times \text{cf} \times 100\%$$

Where:

$$\text{cf} = \frac{\text{Initial cover on control subplots}}{\text{Surviving cover on control subplots}}$$

For some widespread individual species, we also calculated relative cover in response to trampling impact. Relative height of the vegetation was calculated by summing the heights and dividing by the number of values greater than zero and then substituting the mean height values in the formula given above for relative cover. Calculations of resistance and resilience indices follow the procedures outlined by Cole (1995a). Statistical analysis was performed with SPSS software (SPSS, Inc. Chicago, Ill, USA).

Results

The *Artemisia* vegetation type (fig. 1a & b) showed little initial resistance to trampling disturbance, with significant decreases in overall cover with as little as 75 trampling passes. The highest level of trampling (500 passes) resulted in approximately 20% relative cover remaining. In the *Equisetum* vegetation type overall responses were similar, but much higher trampling intensities (800 passes) were

required to induce a moderate cover loss of approximately 50% (fig. 1c & d). Both vegetation types demonstrated significant ability to recover (resilience), with almost all of the relative cover measurements close to 100% one year after disturbance. After one year of regrowth, T-test results revealed no evidence of a grazing effect on relative cover for either vegetation type ($t = 1.31$, $p = 0.26$ for *Artemisia* and $t = 3.09$, $p = 0.091$ for *Equisetum*).

Relative height (fig. 2) followed a similar trend as relative cover, but significant decreases occurred with just 25 passes. The *Equisetum* vegetation type was particularly sensitive to trampling in this regard, with relative heights approaching zero with moderate to high levels of disturbance. After one year of recovery, plant heights in the *Equisetum* plots exposed to potential grazing had 44% greater relative height ($t = 5.31$, $p = 0.030$) and *Artemisia* plots had 23% greater relative height ($t = 20.51$, $p < 0.00$) than comparable nongrazed plots.

Responses immediately after trampling are reported for individual species (table 2). In the *Equisetum* vegetation type, *Koeleria macrantha*, *Poa juncifolia*, and *Poa palustris* demonstrated a high resistance to disturbance, with significant cover remaining after even 800 passes. *Erigeron glabellus* and *Equisetum laevigatum* were highly susceptible with almost zero cover remaining after the same level of disturbance. In the *Artemisia* vegetation type, *Agrostis variabilis* was susceptible to disturbance, while *Sporobolus cryptandrus* was moderately resistant.

Although significant increases in bare ground were observed in both vegetation types immediately after trampling (table 3), there were no significant differences remaining one year later. No clear trends were evident in soil penetration resistance, with high levels of trampling disturbance showing no significant effect.

The resistance, resilience and tolerance indices (table 4) demonstrate that both vegetation types are of moderate resistance (in the 50–60% range), of high resiliency (above 70%) and of high tolerance (above 90%). Interestingly, the grazed plots were consistently more resilient than the respective nongrazed plots.

Discussion

Although information is available on the resistance and resilience of plant communities (for example, Cole 1995 a & b), site-specific information on the response of plant communities to human disturbance is desirable when making important management decisions. Applied trampling studies do not exactly mimic the disturbance from actual use, but these approaches are an effective means of examining the responses to short term trampling and they provide an accurate index by which to base visitor use management decisions (Cole and Bayfield 1993).

The overall durability of a vegetation type is a function of its ability to resist the initial disturbance of trampling and its ability for regrowth. The ability of a vegetation type to withstand initial disturbance is termed resistance (Cole and Bayfield 1993; Sun and Liddle 1991). Others such as Grime (1979) refer to this property as inertia. In this experiment, we assessed resistance by measuring plant properties two weeks after initial disturbance.

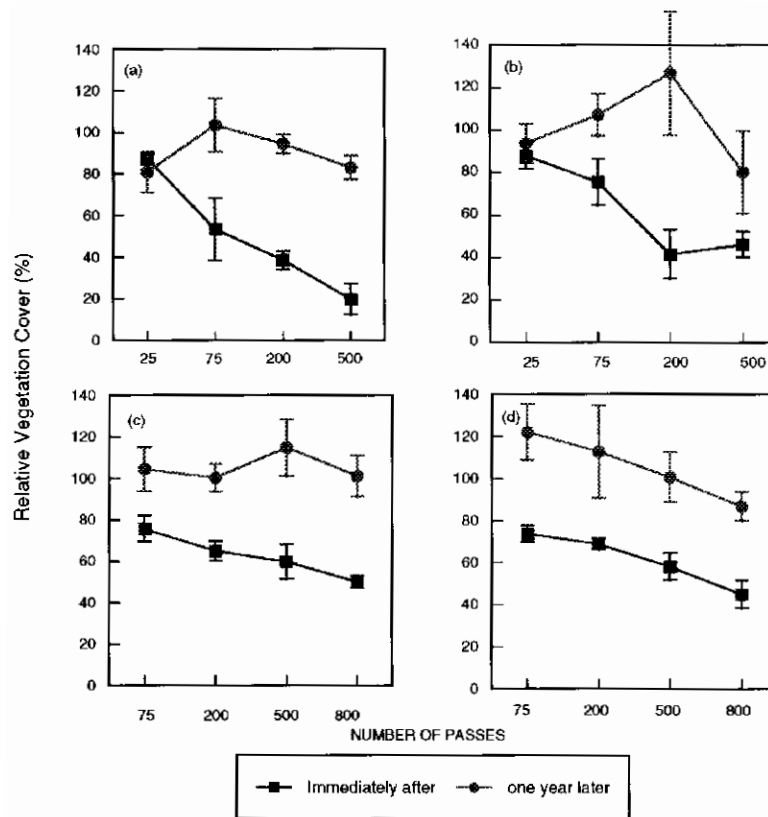


Figure 1—The relationship between vegetation cover and amount of trampling in the a) *Artemisia*, no grazing, b) *Artemisia*, grazing, c) *Equisetum*, no grazing, d) *Equisetum*, grazing vegetation types. Bars are one standard error.

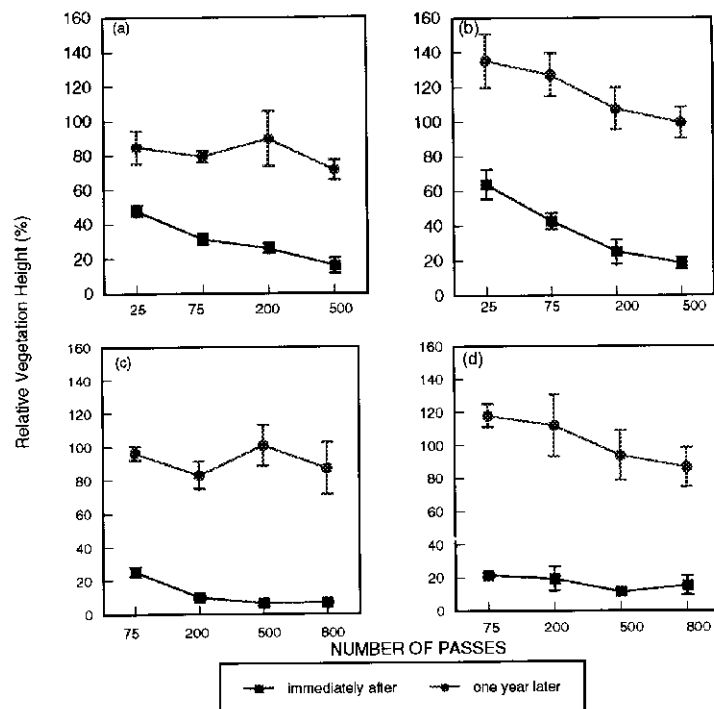


Figure 2—The relationship between vegetation height and amount of trampling in the a) *Artemisia*, no grazing, b) *Artemisia*, grazing, c) *Equisetum*, no grazing, d) *Equisetum*, grazing vegetation types. Bars are one standard error.

Table 2—Relative cover^a of abundant species after trampling and after one year recovery.

Species	After trampling				After one year recovery			
	Number of passes				Number of passes			
<i>Artemisia</i>	25	75	200	500	25	75	200	500
Grazing								
<i>Agrostis variabilis</i>	82	45	25	20	86	96	200	107
<i>Sporobolus cryptandrus</i>	91	118	69	75	83	96	68	54
Without grazing								
<i>Agrostis variabilis</i>	90	55	29	14	57	80	104	82
<i>Sporobolus cryptandrus</i>	118	65	74	44	80	91	102	43
<i>Equisetum</i>	75	200	500	800	75	200	500	800
Grazing								
<i>Elymus trachycaulus</i>	55	100	75	49	91	20	17	19
<i>Erigeron glabellus</i>	80	28	7	0	*	*	*	*
<i>Equisetum laevigatum</i>	44	64	10	5	*	*	*	*
<i>Koeleria macrantha</i>	94	93	94	65	*	*	*	*
<i>Poa juncifolia</i>	86	77	78	83	*	*	*	*
<i>Poa palustris</i>	100	100	87	49	*	*	*	*
<i>Poa</i> spp.	90	87	82	66	162	174	116	127
<i>Taraxacum officinale</i>	75	109	39	11	50	93	51	94
Without grazing								
<i>Elymus trachycaulus</i>	100	66	80	100	15	35	2	13
<i>Erigeron glabellus</i>	53	109	33	29	*	*	*	*
<i>Equisetum laevigatum</i>	67	19	14	6	*	*	*	*
<i>Koeleria macrantha</i>	80	82	78	86	*	*	*	*
<i>Poa juncifolia</i>	64	87	60	67	*	*	*	*
<i>Poa palustris</i>	96	92	96	67	*	*	*	*
<i>Poa</i> spp.	91	90	89	67	111	105	121	118
<i>Taraxacum officinale</i>	88	63	27	75	62	77	33	75

^aRelative cover is the proportion of original cover that survives trampling, adjusted for changes on controls. For the *Artemisia* plots, relative covers were calculated following Cole and Bayfield (1993). For the *Equisetum* plots, relative covers were calculated without using a correction factor due to excessive variability in the control plots.

*Indicates missing species data due to lack of flowering over the course of the season (see Discussion).

The term resilience has been used commonly in the literature (Grime 1979; Cole and Bayfield 1993) as the ability of an ecosystem to recover from disturbance. Here, we assessed resilience by comparing the relative cover immediately after disturbance with the relative cover after one year of recovery. Tolerance is another useful measurement suggested by Cole (1988), Cole and Trull (1993) and Cole and Bayfield (1993), and is a measure of the vegetation to both resist and recover. We measured tolerance by comparing the vegetation cover after one year of recovery with the initial cover prior to disturbance.

The groundcover in the *Artemisia* vegetation is fairly sensitive to trampling, with a 50% overall cover loss occurring at less than 200 trampling passes. This is in contrast with the more initially resistant *Equisetum* vegetation type that does not reach 50% loss, even at 800 passes (fig. 1). Both vegetation types are highly resilient (fig. 1 and table 4), with overall cover approximating predisturbance levels in just one year of regrowth.

In *Equisetum*, vegetation height was significantly reduced with just 75 passes (fig. 2). Due to the morphology of this vegetation type (collectively tall graminoids and horse-tails, in the 30 cm range), plants can be easily flattened by

human use. This may or may not be of important management consequence, given the degree resiliency we observed. It could be problematic for management since areas of disturbance become obvious with just a few passes. These areas will tend to attract more use, and therefore concentrate impact, which could lead to trail formation.

Soils in both sites are essentially unaffected by trampling. This is an indicator that there will be little long-term surface soil compaction. Direct comparisons of the measurements immediately after trampling to those one year later were difficult since the second season was unusually wet in the *Equisetum* plots, and penetration resistances consequently are very low in year two (table 3). This wet season also resulted in a lack of flowering in these plots, and we were therefore unable to correctly identify many individual species (data omitted from table 2).

Work by Cole (1995b) demonstrates that tolerance is largely a function of resilience rather than the initial resistance to disturbance. Similar trends are observed here, where both vegetation types are of moderate resistance (~49 to 66%), but of high resilience and therefore high tolerance (table 4). Growth form has also been identified as a predictor of durability, with chamaephytes (plants with penetrating

Table 3—Exposure of bare ground and changes in soil compaction due to trampling after one year recovery.^a

Treatment	After trampling				After one year recovery			
	Bare ground		Soil penetration resistance		Bare ground		Soil penetration resistance	
	Percent		kg/cm ²		Percent		kg/cm ²	
Grazing								
<i>Artemisia</i> control	10 ± 3.5	a	4 ± .4	a	18.8 ± 10.5	a	4.1 ± .5	a
25 passes	15 ± 5.4	a	4.3 ± .3	a	5.5 ± 2.6	a	3.4 ± .5	a
75 passes	33.8 ± 7.5	a	4.1 ± .5	a	5.1 ± 2	a	4.1 ± .3	a
200 passes	55 ± 14.4	b	3.6 ± .45	a	8.8 ± 3.8	a	3.1 ± .5	a
500 passes	47.5 ± 6.3	ab	3.3 ± .3	a	27.5 ± 8.3	a	4.2 ± .2	a
<i>Equisetum</i> control	.1 ± .06	a	3.3 ± .2	a	.05 ± .05	a	1.3 ± .03	a
75 passes	1.6 ± 1.2	a	3.1 ± .3	a	.05 ± .05	a	1.5 ± .3	a
200 passes	6.3 ± 2.4	a	3.4 ± .3	a	1.25 ± 1.25	a	1.3 ± .2	a
500 passes	11.5 ± 4.1	a	3.5 ± .11	a	1.3 ± 1.2	a	1.5 ± .2	a
800 passes	45 ± 9.6	b	3.6 ± .2	a	3 ± 2.4	a	1.4 ± .2	a
Without grazing								
<i>Artemisia</i> control	8.8 ± 3.8	a	3.3 ± .4	a	2.8 ± 2.4	a	2.6 ± .3	a
25 passes	10 ± 2	a	3.5 ± .13	a	2.8 ± 1.3	a	3.7 ± .7	a
75 passes	41.3 ± 19.6	ab	3.8 ± .46	a	0.8 ± .2	a	3.1 ± .6	a
200 passes	67.5 ± 4.8	b	2.9 ± .65	a	4.0 ± 1	a	3.3 ± .2	a
500 passes	75 ± 6.5	b	3.1 ± .5	a	15.3 ± 8.5	a	3.9 ± .8	a
<i>Equisetum</i> control	0.0 ± 0	a	3.6 ± .07	a	0.0 ± 0	a	.67 ± .04	a
75 passes	.25 ± .25	a	3 ± .2	a	.05 ± .05	a	.96 ± .14	a
200 passes	1.8 ± 1.1	a	3.1 ± .13	a	.13 ± .07	a	.88 ± .14	a
500 passes	9 ± 2.9	a	2.6 ± .3	a	0.0 ± 0	a	1.4 ± .3	a
800 passes	32.5 ± 4.8	b	3.1 ± .2	a	.05 ± .05	a	1.0 ± .3	a

^aMeans not followed by the same letter are significantly different using the modified LSD at $\alpha = 0.05$.

Table 4—Indices of resistance, resilience, and tolerance for the two vegetation types.^a

	<i>Artemisia</i>		<i>Equisetum</i>	
	Grazing	Without grazing	Grazing	Without grazing
Resistance				
Mean relative cover after 0-500 passes	62.71	49.66	66.97	66.82
Resilience				
Mean increase in cover one year after 0-500 passes, as a percent of the damage caused by trampling	105.27	80.82	135.85	74.91
Tolerance				
Mean relative cover one year after 0-500 passes	101.97	90.34	111.84	91.68

^aCalculations follow Cole and Bayfield (1993).

bud above the ground surface) being the least tolerant (Cole, 1995b). In general, these observations were supported here; the vast majority of the overall cover in these plots were composed of cryptophytes, or plants with penetrating buds below the ground surface. Therefore, despite the erect nature of the grasses, particularly in the *Equisetum* plots, regenerative structures remained undisturbed and resilience high.

Our data do not address the overall effects of grazing on these vegetation communities. Trends seem to indicate a possible stimulation of the regrowth response, but this could

have been due to micro site differences in water stress, since regrowth was assessed after a particularly wet season. An additional complication is that actual application of the grazing was not controlled; in other words, these plots are best referred to as having “potential grazing.” Although cattle were on the property, it is not clear to what extent they affected the experimental plots. Nonetheless, the results of the nongrazed plots (within grazing exclosures) are clear, and a more carefully controlled grazing study should be employed to examine the effects of grazing more thoroughly.

Management Implications and Future Research

The results of this work indicate clearly that both studied vegetation types can tolerate a significant amount of human use, without sacrificing the ability to recover in the short term. Off-trail use is currently permitted in the area for fishing access to the Sweetwater River, and current findings show no immediate rationale for changing this practice, provided the overall use does not exceed the ability of the vegetation to recover. Proper visitor education and regulation, in combination with continued monitoring, will help guide future management decisions.

Several important questions remain that should be addressed by future research and monitoring:

- These results indicate that the plots where grazing was possible had greater vegetation height. A more carefully designed study with applied grazing and trampling treatments should be conducted to determine the combined effects of these two treatments.
- Individual species responses, and consequently, plant community changes were not possible to determine, particularly in the *Equisetum* plots. This was due to a lack of flowering of many species due to an especially wet season in year two of the project. It is possible that long-term trampling may have an effect on species composition, and this should be determined with future investigations.

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References

- Bayfield, N.G. 1979. Recovery of four montane heath communities on Cairngorm, Scotland, from disturbance by trampling. *Biological Conservation*. (15): 165-179.
- Cole, D. N. 1985. Recreational trampling effects on six habitat types in western Montana. USDA Forest Service Research Paper, INT-350, Ogden UT. 43 p.
- Cole, D. N.; Trull, S. J. 1992. Quantifying vegetation response to recreational disturbance in the North Cascades, Washington. *Northwest Science*. 66 (4): 229-236.
- Cole, D. N.; Bayfield, N. G. 1993. Recreational trampling of vegetation: standard experimental procedures. *Biological Conservation*. 63: 209-215.
- Cole, D.N. 1995a. Experimental trampling of vegetation. I. Relationship between trampling intensity and vegetation response. *J. Appl. Ecol.* 32: 203-214.
- Cole, D.N. 1995b. Experimental trampling of vegetation. II. Predictors of resistance and resilience. *J. Appl. Ecol.* 32: 215-224.
- Grime, J. P. 1979. Plant strategies and vegetation processes. John Wiley and Sons, New York. 222 pp.
- Monz, C.A.; Meier, G.A.; Buckley, R. C.; Cole, D. N.; Welker, J. M.; Loya, W. M. 1996. Responses of moist and dry arctic tundra to trampling and warmer temperatures. *Bull. Ecol. Soc. Am.* 77(3): 311.
- Sun, D. and Liddle, M. J. 1991 Field occurrence, recovery and simulated trampling resistance and recovery of two grasses. *Biological Conservation*. 57: 187-203.
- Wagar, J.A. 1964. The carrying capacity of wild lands for recreation. Forest Science Monograph 7. Society of American Foresters, Washington, DC.

Meadow Response to Pack Stock Grazing in the Yosemite Wilderness: Integrating Research and Management

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Abstract—Management decisions on meadow preservation and allowable use are, ideally, based on scientific information that describes the relationship between levels of impact and levels of use. This information allows managers to provide the best protection of meadow systems while responding to demands for recreational use of mountain meadows. Monitoring and research activities can be coordinated to support management by gathering information on measurable levels of meadow use, meadow response to different levels of use and cause and effect relationships reflected in meadow response. Based on this information, wilderness managers can decide on the maximum acceptable impacts to meadows that still provide protection.

Meadows occupy less than 10 percent of the montane and subalpine regions of the Sierra Nevada, yet they support disproportionate amounts of biological diversity, ecosystem function and aesthetic interest. They contain high plant diversity, provide wildlife forage and habitat, filter organic inputs to streams and, from a human perspective, provide high aesthetic value. They have long been valued as well for livestock and pack stock forage that lasts late into the summer.

Meadows comprise over three percent of the area of Yosemite National Park. Ratliff (1982) classified central and southern Sierra Nevada meadows into 14 types. Two of the more extensive of these, shorthair sedge (*Carex filifolia* var. *erostrata*) and shorthair reedgrass (*Calamagrostis breweri*), also were recognized by Sumner (1941) and Klickoff (1965). Benedict (1981) described 19 associations from southern

Sierra Nevada meadows, including Ratliff's (1995) tufted hairgrass (*Deschampsia cespitosa*) series, a mid-elevation meadow type. Grazing of mountain meadows in the Yosemite Wilderness is limited to pack animals used to support National Park Service administrative functions, transport small private parties, and conduct clients of permitted commercial pack stations. Currently, commercial outfits dominate use, followed by the Park's concessioner and private parties.

Several qualitative evaluations of meadow condition in the Sierra Nevada have been conducted (Briggs 1966; Ernst 1949; Guse 1969; Sharsmith 1961). Ernst provided a comprehensive picture of the Park's grazing situation in 1948, suggesting remedies for perceived overuse. Sharsmith evaluated selected heavy use areas and commented on "deteriorated conditions," including exotic annual grass invasion, erosion and lodgepole pine (*Pinus contorta* var. *murrayana*) encroachment associated with trampling and concentrated grazing. Guse commented on the lack of timely information on impacts and lack of a consistent mechanism for mitigating impacts in a responsive manner. He recommended meadow use monitoring, as well as research to determine meadow species composition and grazing capacities. Despite this periodic attention to meadow condition, few quantitative studies exist to support management decisions.

Mueggler (1967) documented a decline in herbage production after three successive years of defoliation in mountain grasslands of Montana. DeBenedetti (1980) found that clipping the herbage to a one-inch (2.5 cm) stubble height reduced total nonstructural carbohydrates in the roots of shorthair sedge, shorthair reedgrass and Rocky Mountain sedge (*Carex scopulorum*) by 20 to 40 percent. Stohlgren and others (1989) found a decrease in mesic meadow productivity following herbage removal but an increase in dry meadow productivity in each of four years of herbage removal. We hypothesized that applying different intensities of herbage removal could result in varying reductions in meadow productivity in subsequent years.

Olson-Rutz and others (1996) quantified the impact to meadow plant communities from four different durations of horse grazing over three summers. However, measurements were aimed at detecting the immediate effect of grazing on vegetation (changes in plant height) and not at describing meadow condition in subsequent years. Proulx and Mazumder (1998) found lower species richness in nutrient-poor ecosystems under heavy grazing by ungulates than

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under light grazing intensity. Olson-Rutz and others (1996) found that grass heights were reduced more than forb heights and that grasses were grazed more often than forbs for the four- and eight-hour durations. In this study, we hypothesized that such differential grazing pressure by plant type could result in species composition shifts in subsequent years.

McClaran and Cole (1993) noted that research is needed on the relationships between pack stock use and impacts, specifically how impacts vary among use levels. McClaran (1989) suggested that meadow management could benefit from additional information on how impacts vary among meadow types within use levels, as well as monitoring of current utilization.

Pack Stock Grazing Management

The goals of pack stock management in mountain meadows are to avoid adverse impacts to meadow structure, function, diversity and productivity and to allow access by pack stock users. Management decisions on meadow maintenance and allowable use are, ideally, based on scientific information that describes the relationship between levels of impact and levels of use. This information is needed to provide the best protection of meadow systems while responding to demands for recreational use of mountain meadows and the associated consumption of forage. These goals present a dilemma to managers because unlimited access may result in overuse, with adverse impacts on plant productivity, species composition and vegetative cover. One approach to resolving this conflict is to compromise each goal to some extent. Managers can set minimally acceptable standards for meadow condition. One such standard might be to maintain meadow productivity at no less than 90 percent of its ungrazed level. When minimal conditions are not met, managers must modify stock use levels or other use parameters to protect meadow integrity.

Supporting sound management requires four types of information: the measurable level of use of the system, the system response to different levels of use, the cause and effect relationships reflected in the response and the maximum acceptable impacts to the system that will still provide adequate protection. The first three are obtained through a combination of monitoring and research. The fourth is a judgment based on the available information. With this approach, monitoring assesses and tracks use intensity over time (along with some variables indicating meadow response), research identifies the effects of various use intensities on meadow condition, management sets maximum acceptable impact levels, and managers act on monitoring results to limit impacts on the system.

Various measures of levels of pack stock grazing include the number of animal nights per unit of time and the associated utilization or amount of plant material removed for each level of stock use. The inverse—the amount of meadow vegetation remaining after use by pack stock, termed residual biomass—has been used as a predictor of meadow response (McClaran and Cole 1993). Describing and tracking pack stock use levels is a primary task of meadow monitoring. Monitoring can also gather information on the

system response to determine how meadows are responding to current grazing intensities (such as changes in productivity and ground cover).

Research can provide a systematically derived understanding of cause and effect, identifying what grazing intensities cause various degrees of impact. If the research integrates monitoring parameters into the research design, it can describe the relationship between monitoring measures, such as ground cover and residual biomass, and less efficiently measured but significant aspects of meadow structure and function, such as plant species composition and foliar cover.

Decisions by wilderness and resource managers and the public need to be made about minimally acceptable conditions and maximum acceptable impacts to meadows and to visitor experience. This task, out of the scope of research and monitoring, requires careful balancing of biological and political realities to make difficult decisions about wilderness management.

As an example of the interaction of research, monitoring and management, a maximum of 10% loss of productivity might be selected, based on wildlife needs, associated changes in species composition and soil retention capabilities over the long term. Research may determine that 20 percent utilization causes a 10 percent reduction in productivity for a certain meadow type. If monitoring estimates that current utilization is greater than 20 percent, management would respond by reducing pack stock use levels to allow the meadow to recover.

Management, then, is an iterative process of 1) recognizing research results that define meadow responses to different levels of use and 2) responding to unacceptable impacts exposed by ongoing monitoring aimed at detecting change. This model is applicable to most research and management relationships. In the following sections, we elaborate on the role of monitoring, research and management in this process.

Monitoring

The goals of monitoring include describing the condition of sites, intensity of use and range of variability (as influenced by weather variation, variations in use patterns, and other stressors) and detecting change and the direction of change in the variables measured. Establishing cause and effect is not a goal of monitoring but remains a responsibility of research.

The monitoring program at Yosemite is designed for maximum efficiency and simplicity because wilderness staff implement the monitoring as collateral duties in addition to all other responsibilities while on patrol. The program involves 14 different meadows in 12 areas, ranging in elevation from 1,300 to 3,050 m (4,400 to 10,000 ft). The rangers use permanently marked transects and collect data in two categories: late-season residual biomass to represent grazing intensity—the causal influence—and bare ground cover, representing meadow response to grazing over time.

Ground cover is measured at 150 points along the transects, and standing biomass is clipped from 15 quadrats arranged parallel to the transects. Quadrats are 25 x 25 cm, and plant material is clipped to a height of 1.0 cm. Only plant material produced in the current growing season is collected; litter from previous years is separated out and left on site. Quadrats

are located without repetition for eight years to avoid previously clipped sites.

Transects are relocated through the use of topographic maps, photographs, diagrams and permanent markers located at each end of the transects. Meadow transects are read annually, near the end of the season, to most closely represent the biomass remaining on sites after all of the season's use has occurred. Residual biomass is more practical to collect than the amount of forage removed, and it can be an indicator of use intensity if associated mean annual productivity is estimated.

It is important to monitor the amount of meadow use in terms of animal nights and forage removed before any standards are set. It provides information about when and to what degree any hypothetical standards have been exceeded and the magnitude of management adjustment that might be required. Detecting when the maximum allowable impacts have been exceeded is even more important after standards are established. This helps evaluate current use practices and allows managers to keep impacts below the acceptable standards. Despite the critical role of monitoring for grazing use levels and impacts, monitoring cannot establish cause and effect relationships, and monitoring alone is an insufficient basis for management.

Research

Research is critical to wilderness management because it defines the functional relationships between various levels of stress to natural systems and system responses to those stresses. Our research goal was to define the relationship between grazing intensity and associated changes in meadows. That is, we sought to describe changes in ground cover, meadow productivity, foliar cover and species composition from three different grazing intensities.

Study Design

We focused on three subalpine meadow types that are common and extensive in the Park. The first was a high-elevation, xeric shorthair sedge (*Carex filifolia* var. *erostrata*) type that occurs on well-drained, sandy, granitic soils between 2,600 and 3,300 m (8,500-11,000 ft). These meadows have early-season snowmelt, and they flower, set seed and senesce midway through the growing season. They produce little in the way of biomass (mean = 71 g/m²) and have the lowest species diversity (we found 35 species in the 1,024 square meter study area) of the three types. The second meadow type was a more mesic shorthair reedgrass (*Calamagrostis breweri*) type. It occurs on sites where soils are saturated longer than shorthair sedge meadows, such as floodplains and near ponds. The type ranges in elevation from 2,400 to 3,050 m (8,000 to 10,000 ft) and exhibits moderate productivity (mean = 214 g/m²) and species diversity (55 species on our study site) for a Sierra Nevada meadow. Tufted hairgrass (*Deschampsia cespitosa*) is the dominant species in the third type. These wet meadows are associated with upper montane mixed conifer forests, such as red fir (*Abies magnifica*), and have the greatest number of species (72 species on our study site) and the highest productivity levels (mean = 345 g/m²) of all three types.

The study was designed to describe the changes in annual productivity, ground cover, plant foliar cover (both absolute and species specific) and species composition associated with recreational pack stock grazing at three different intensities. The experiment involved the three meadow types, three grazing intensities and four years of grazing. Within the shorthair sedge and reedgrass types, there were four sets of four plots each; there were three sets of four plots each in the tufted hairgrass type. Prior to grazing each year, ground cover, foliar cover, species composition and productivity measurements were made on 10 subplots (0.0125m² each) per plot. Then three plots within each block of four were grazed, each at a different intensity, with a fourth, ungrazed plot for comparison. This provided four replicates per grazing intensity and four control replicates. Immediately following grazing, we measured groundcover and then clipped for residual biomass, similar to the methods used for monitoring.

Ground cover data were analyzed using one-sample t-tests to compare post-grazing measurements with original condition. We analyzed productivity data by comparing values with original condition as well. We evaluated species composition changes using an index of floristic dissimilarity to compare conditions after grazing with original conditions, standardized for changes on the controls.

We measured groundcover and standing biomass immediately following grazing in order to estimate the grazing intensity achieved and to closely link the causal factor (grazing intensity), the resulting changes in meadows and monitoring variables. We provided this link to the research results within a fairly controlled setting, establishing the expected degree of change, by grazing intensity, for each meadow type.

Results

Our experimental grazing resulted in a number of statistically significant changes in meadow condition. One-sample t-tests showed that bare ground increased after two years of grazing in all three meadow types (table 1). Plant foliar cover decreased in a significant way only in the tufted hairgrass type (p <0.05). While there was a measurable shift in species

Table 1—Number of years of grazing required to cause statistically significant^a changes in various meadow attributes in three meadow types in the Sierra Nevada.

	Shorthair sedge	Shorthair reedgrass	Tufted hairgrass
	----- years -----		
Basal bare cover	1	2	2
Basal vegetative cover	2	>4 ^b	>4 ^b
Foliar cover	>4 ^b	>4 ^b	3
Species composition	2	2	2
Species richness	>4 ^b	>4 ^b	>4 ^b
Productivity	2	2	2

^ap <0.05.

^bNo adverse impact was found, suggesting either that more than four years of grazing is needed or grazing at the intensities applied would not have an adverse impact.

composition after two years of grazing, as reflected in a similarity index, the most pronounced and consistent change across all meadow types was in productivity. Productivity declined by the second year of grazing in all three types.

The meadow productivity results are directly applicable to pack stock management if used to anticipate productivity reductions associated with observed levels of use. We graphed reduction in productivity after the third year of grazing against grazing intensity (proportion of biomass removed by grazing). These data allow us to estimate the proportion of biomass that could be removed each year by grazing and cause reductions in productivity of 10, 25, and 50 percent (fig. 1). If the maximum acceptable reduction in productivity is set at 10 percent for the shorthair sedge type, animals could remove 36 percent of standing biomass. If a 25 or 50 percent reduction in productivity is acceptable, animals could remove 45 or 57 percent of forage, respectively. Forage removal of 20, 49, and 99 percent in the shorthair reedgrass type would result in similar reductions in productivity (table 2). Limiting the decline in productivity to 10 percent for the tufted hairgrass type provides for up to 17 percent forage removal. Productivity declines of 25 and 50 percent are associated with 42 and 84 percent forage removal in this type.

A commonly used rule of thumb for grassland vegetation is to leave 50 percent of biomass at the end of the grazing period to maintain nutrient levels through decomposition (Frandsen 1961). This level of pack stock use would cause about a 30 percent loss of productivity in shorthair sedge meadows, a 25 percent decline in productivity in shorthair reedgrass meadows and a 28 percent decline in wet meadows dominated by tufted hairgrass. Ratliff (1985) concluded that utilization should not exceed 35 percent of average herbage production in drier types and 45 percent in more mesic types to maintain meadow productivity. He stated that the 50 percent rule cannot be considered a safe utilization guide for all meadows of the Sierra Nevada.

Although utilization—the amount of plant material removed by grazing—was the best predictor of meadow condition, it is difficult to measure. However, Park staff can

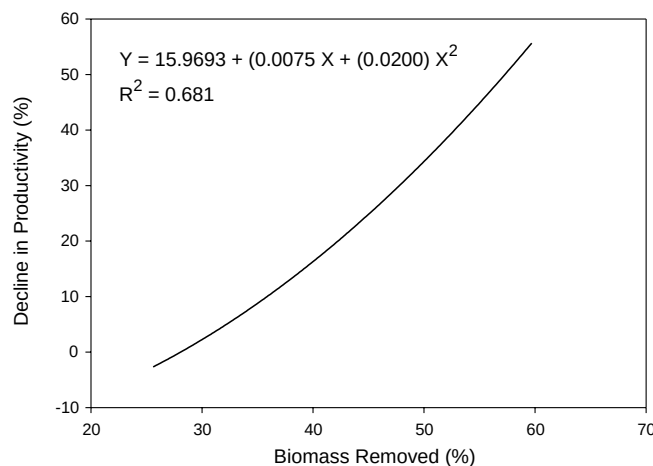


Figure 1—Proportional reductions in productivity associated with proportional amounts of forage removal by grazing in shorthair sedge meadows at Yosemite National Park, California.

Table 2—Maximum grazing intensity (proportion of biomass removed) that would cause specific levels of reduction in productivity.

Decline in productivity	Shorthair sedge	Shorthair reedgrass	Tufted hairgrass
----- percent -----			
10	36	20	17
25	45	49	42
50	57	99	84

monitor residual biomass and use it as an indicator of use intensity if they use mean annual productivity levels by meadow type to estimate utilization and associate these estimates with monitoring measures. Utilization can be estimated from

$$U = [(p-r)/p] \times 100$$

where U = utilization, p = mean annual productivity by meadow type, and r = residual biomass.

All three meadows used in this study had not been grazed for several decades. As such, they may be more representative of pristine conditions and support species that are more susceptible to grazing than meadows that were recently, periodically or continuously grazed during that time. Therefore, it is possible that grazed areas may have a different, possibly more conservative response to grazing intensity than ungrazed meadows. However, these results speak to the severity of impacts when grazing is introduced to previously ungrazed, pristine areas.

Management

Wilderness managers have responsibility for setting maximum acceptable impact levels and must decide what percent decline in productivity is acceptable. Research can only indicate the ramifications of such a decision. Managers must also set policy, communicate guidelines and implement restrictive actions when standards are not met. The monitoring component of the Yosemite program provides a consistent examination of Park meadows over time under actual use patterns. However, it is only through consistent monitoring that trends can be documented and change can be detected quickly enough for management to respond.

Research and monitoring, then, work in conjunction to provide managers with timely information on meadow condition and trends. They also provide a set of expected responses associated with different levels of use, which are in turn associated with meadow condition. Using this combined information, managers will be better prepared to set sound meadow use policies and protect wilderness meadows for the future.

Acknowledgments

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References

- Benedict, N. B. 1981. The vegetation and ecology of subalpine meadows of the southern Sierra Nevada, California. Ph.D. Dissertation. University of California, Davis. 128 p.
- Briggs, G. S. 1966. A report on backcountry conditions and resources, with management recommendations, Yosemite National Park. National Park Service. Unpub. Manuscript. 217 p.
- DeBenedetti, S. H. 1980. Response of subalpine meadow communities of the southern Sierra Nevada, California, to four clipping regimes. In: Thirty-third annual meeting abstracts and position statements, February 11-14, 1980. San Diego, CA: Society for Range Management. 29 p.
- Ernst, E. F. 1949. The 1948 saddle and pack stock grazing situation of Yosemite National Park. Unpub. Manuscript. 77 p.
- Frandsen, W. R. 1961. How to "feed" range forage plants, enable them to make more efficient use of soil moisture and nutrients and produce more forage. In: Stockmen's handbook, Washington State University stockmen's short course. 11th ed. Pullman, WA: Department of Animal Science, Institute of Agricultural Sciences, Washington State University: 74-76.
- Guse, N. G. 1969. A personal criticism of stock grazing practices in Yosemite National Park. National Park Service. Unpub. Manuscript. 15 p.
- Klikoff, L. G. 1965. Microenvironmental influence on vegetational pattern near timberline in the central Sierra Nevada. Ecological Monographs 35: 187-211.
- McClaran, M. P. 1989. Recreational pack stock management in Sequoia and Kings Canyon National Parks. Rangelands 11(1): 3-8.
- McClaran, M. P.; Cole, D. N. 1993. Packstock in Wilderness: impacts, monitoring, management and research. Ft. Collins, CO: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. Gen. Tech. Rep. INT-301.
- Mueggler, W. F. 1967. Response of mountain grassland vegetation to clipping in southwestern Montana. Ecology 48(6): 942-949.
- Olson-Rutz, K. M.; Marlow, C. B.; Hansen, K.; Gagnon, L. C.; and Rossi, R. J. 1996. Packhorse grazing behavior and immediate impact on a timberline meadow. Journal of Range Management 49: 546-550.
- Proulx, M.; Mazumder, A. 1998. Reversal of grazing impact on plant species richness in nutrient-poor vs. nutrient-rich ecosystems. Ecology 79(8): 2581-2592.
- Ratliff, R. D. 1982. A meadow site classification for the Sierra Nevada, California. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. Gen. Tech. Rep. PSW-60. 16 p.
- Ratliff, R. D. 1985. Meadows in the Sierra Nevada of California: state of knowledge. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. Gen. Tech. Rep. PSW-84. 52 p.
- Sharsmith, C. W. 1961. A report on the status, changes and comparative ecology of selected back country meadow areas in Yosemite National Park that receive heavy visitor use. Unpub. Manuscript. 58 p.
- Stohlgren, T. J.; DeBenedetti, S. H.; Parsons, D. J. 1989. Effects of herbage removal on productivity of selected high-Sierra meadow community types. Environmental Management 13(4): 485-491.
- Sumner, L. E. 1941. Special report on range management and wildlife protection in Kings Canyon National Park. Unpub. Manuscript at Sequoia and Kings Canyon National Parks. Three Rivers, CA.

Human Impact Surveys in Mount Rainier National Park: Past, Present, and Future

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Darin D. Swinney

Abstract—Three survey methods were utilized to describe human impacts in one wilderness management zone of Mount Rainier National Park: wilderness impact cards, social trail and campsite surveys, and condition class surveys. Results were compared with respect to assessment of wilderness condition and ecological integrity. Qualitative wilderness impact cards provided location of point impact such as litter, human waste, and campsites. They did not provide data related to ecological integrity and were limited by their inconsistent implementation. Systematic social trail and campsite data provided quantitative estimates of bare ground impacts. Condition class surveys provided spatial documentation of wide range of impacts. Selection of a method is dependent on good articulation of monitoring goals and funding limitations.

Each year approximately two million people visit Mount Rainier National Park (Johnson and others 1991). For many visitors, the primary destination is within the subalpine parkland or alpine zone. Collectively, these two ecological zones only comprise about 35% of the entire Park, yet they absorb over half of all visitors, and use is generally concentrated within two to three summer months. Some specific areas of the Park, such as Paradise and Muir Corridor, have been popular attractions since the early 1890s—before the Park's establishment in 1899. In the 1890s, recreationists from the Seattle and Tacoma areas supported the establishment of the Park as a means of protecting the area from recreational impacts (Catton 1996). This intense use for over 100 years has resulted in many human impacts such as campsites, informal trails and large denuded or eroded areas.

Surveys of human impacts have been conducted since the late 1960s (John Wilcox, personal communication). Survey methods have ranged from qualitative descriptions and photographs to detailed measurements. In 1988, with the passage of the Washington Wilderness Act, 97% of Mount Rainier National Park was designated as wilderness. The park's Backcountry Management Plan was replaced with a Wilderness Plan that included Limits of Acceptable Change indicators and standards (Samora 1989). Since 1994, concurrent with the Park's development of the General

Management Plan, LAC and VERP indicators and standards have been reexamined (National Park Service 1997; Stankey and others 1984).

Currently, three impact survey methods are used to document recreational impacts in wilderness areas of Mount Rainier National Park: wilderness impact cards, social trail and campsite surveys and condition class surveys. The objectives of our study were: to survey one study area using all three methods, to compare results from the three methods, to evaluate the results with respect to ecological integrity and wilderness conditions, and develop recommendations for future monitoring of wilderness areas.

Methods

Study Area

Mount Rainier National Park is located on the western slope of the Cascade Range, 60 miles southeast of the Seattle-Tacoma metropolitan area. It encompasses 235,622 acres and extends from old-growth forest (1,730 feet) through subalpine and alpine communities to the mountain's summit at 14,410 feet. The study area is referred to as Muir Corridor and is located within the Muir Snowfield Wilderness management zone, on the south central flank of Mount Rainier (figure 1). The area encompasses 425 acres and extends from treeline at 6,800 feet to Anvil Rock at 9,000 feet elevation. Five broad community types can describe vegetation of the area: heather, fellfields, sedge fellfields, talus, and snowbed (Edwards, 1980). Heather communities are dominated by one or two ericaceous species: *Phyllodoce empetrifloris* (red heather), *P. glanduliflora* (yellow heather), or *Cassiope mertensiana* (white heather). Radiocarbon dating of buried plant stems has documented the persistence of some communities for 7,000 to 10,000 years (Edwards, 1980). Fellfields are flat, rocky areas with small, discrete clumps of plants growing among the rocks. Sedge fellfields are located in wet areas and often have continuous cover of several short sedge species. Talus slopes are steep rocky areas with larger rocks than fellfields. Plants grow in protected cracks and crevices. Snowbed communities grow on the edge of late melting snowbeds. Some areas may remain under snow for several years during cool periods.

Approximately 8,000 climbers and an estimated 4,000-6,000 dayhikers travel through the area each summer. For many, the destination is Camp Muir at 10,000 feet elevation, the most popular climbing camp. Although park regulations require hikers to stay on maintained trails or snowfields, many visitors walk off-trail across fragile vegetation. Since 1989, camping has only been allowed at Camp Muir and on snow-covered areas in Muir Corridor. However, rangers

In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management; 1999 May 23-27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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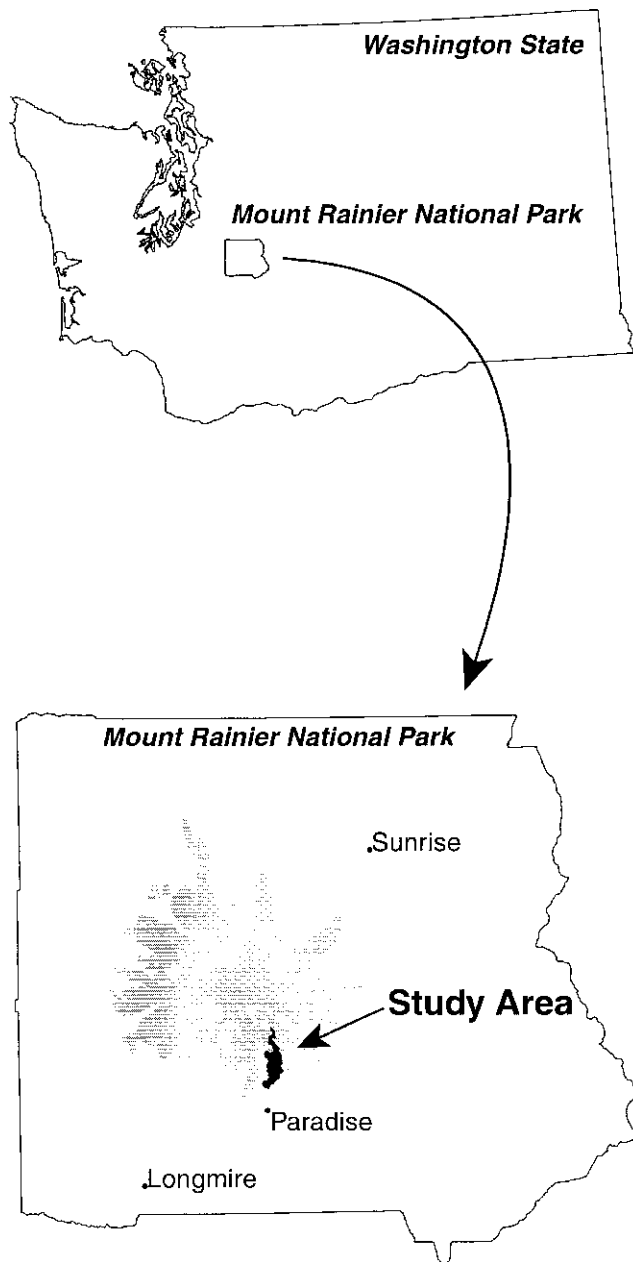


Figure 1—Location of the study area in Mount Rainier National Park.

frequently find people camping on nonsnow sites in the corridor. Both hikers and climbers camp in the zone, but the majority of overnight use is by climbers (fig. 2). Off-trail hiking, illegal camping and camping prior to 1989 has resulted in extensive resource damage. Impacts range from trampled plants to removal of entire plant communities and severe erosion.

History of Human Impact Surveys

John Dalle-Molle initiated a park-wide human impact monitoring system in the early 1970s. Originally, this system utilized backcountry impact cards on which rangers documented impacts using a brief description of both the

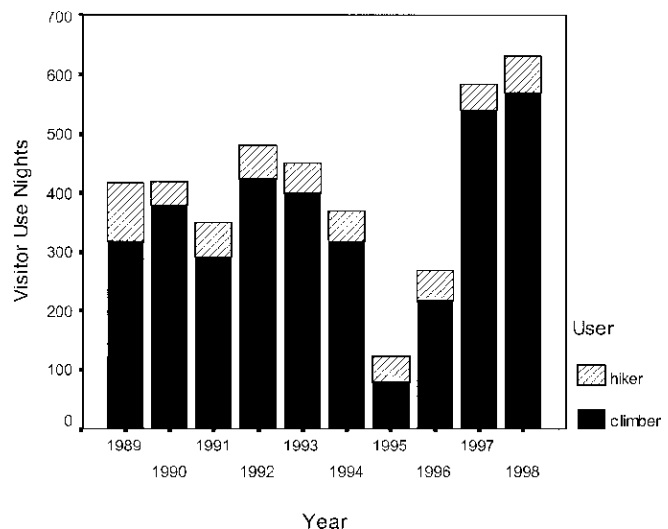


Figure 2—Overnight visitor use of Muir Corridor.

impact and its location. In 1988, 97% of the Park was designated as wilderness and the cards were redesigned to complement the Wilderness Plan. Cards were renamed to wilderness impact cards and the impact description was revised to include an entry of impact type by category (for example, landscape conditions or sanitation) that corresponded to LAC indicators listed in the Wilderness Plan (Samora, 1989). Descriptive locations of impacts were supplemented with Universal Trans Mercator coordinates in 1993 and in 1995, all data from wilderness impact cards was entered into a database linked to the Park's Geographic Information System (GIS). During the 1970s, intensive surveys of designated and informal campsites and social trails (informal trails) were initiated in specific areas of interest. A park-wide program to document quantitative measurements of social trails and campsites was initiated in 1985 to provide a baseline for restoration of human impacts. Measurements were used to rate and rank impacts and to develop supply and materials lists. Dr. Ola Edwards introduced a third method of monitoring human impacts to the park in late 1970s: condition class assessments (Edwards 1985). These methods have been utilized and revised for use in all three broad vegetation zones of the Park: alpine, subalpine, and forest.

Field Survey Methods

Three human impacts survey methods were used: wilderness impact cards, social trail and campsite surveys and condition class surveys. Field personnel generally complete wilderness impact cards when they notice human impacts. Data collected on the card includes date of observation, observer, location of impact is marked on a topographic site map, the category and type of human impact, details and action taken. An example of an impact category is sanitation, the type of impact would be human waste and detail might be 4 piles of human waste/toilet paper. Impact categories correspond to LAC indicators in the Wilderness Plan (Samora 1989). Wilderness impact cards can be used

every day, but in reality, use of cards is subjective, both temporally and spatially.

Social trail and campsite inventories are quantitative surveys that were systematically conducted throughout the zone in 1988, 1995, and 1997. Field personnel walked all accessible areas, within the study zone, in a manner similar to a search for lost people. In 1988, campsites and social trails were mapped using aerial photos, topographic maps and compasses. In 1995 and 1997, a Global Positioning System was used with orthophoto quads and locations were directly downloaded into the Park's GIS. Campsite centers were mapped using a single, differentially corrected and averaged GPS location. A local site map illustrating site shape and landmarks was drawn to assist with future relocation and monitoring and a photograph was taken. Campsite area was calculated by measuring eight radial transects and summing the area of the eight triangles. Slope, aspect and plant community were recorded at each campsite. Social trails were divided into segments both for mapping and measurements. Within each segment, width, depth, length, slope, aspect and vegetation type were recorded. A GPS line feature was recorded for the length of the segment, as well as GPS points (180 minimum) at the beginning and end of each segment. Point collection at the beginning and end of each segment was added to improve accuracy of mapping due to data collection difficulties caused by the steep terrain. Points were differentially corrected and averaged to produce an acceptable measurement. After each survey period, all campsites were obliterated by removing rock walls and restoring the desert pavement (most campsites were in fellfields or talus areas), thus, campsites documented during each time period were recently established.

Condition class surveys were conducted systematically throughout the study area in 1998. First, a 25-m by 50-m grid was superimposed over the area using the Park's GIS. Each point became the center point for a 0.1 ac circular plot in which site condition class, vegetation type, slope, aspect, bareground cover, vegetation cover, and elevation were recorded. Notations were also made if the sample plot included a campsite, social trail, or litter within its perimeter. If a point fell on permanent snow or ice, that was noted and condition class was not recorded. GPS coordinates of sample points were loaded into military GPS receivers for locating plots. Plot centers were also plotted on aerial photos to provide crews with field maps. Military GPS receivers are available to government agencies and were used because they provide access to a more accurate GPS signal than civilian GPS receivers. The latter require post-processing and entry into a GIS to provide acceptable locations. Our goal was to relocate systematically located points so as not to bias our results and to enable future relocation of the same sample points. Plot centers that fell on permanent snow or ice were disregarded, as were points that were inaccessible or hazardous to field crews. A total of 336 plots were inventoried. Five condition classes were used: 0 or pristine, 1 or little change, 2 or significant change, 3 or severe change, and 4 or habitat destroyed (see Table 1).

Table 1—Description of condition classes.

Condition class	Description
0 Pristine	No signs of human use of the area
1 Little change	Small and temporary indications of use caused by people or animals, such as litter, trampled vegetation, scuffed soil, footprints but no lasting damage such as plant loss, erosion, or broken stems
2 Significant change	Human impacts are easily recognizable, but limited in severity or distribution; examples include uprooted plants, clearing of forest litter thus creating a trail or campsite, clearing of pebbles or rocks in fellfields or compacted soil, but not erosion; area of individual impacts should be small (< 0.8 sq. ft. or 1 ft. in diameter) and covering a small portion of the sample area (<10-15%)
3 Severe change	Few severe impacts or many moderate impacts with an extensive distribution so that the sample area is fragmented; severe impacts include walled campsites in an alpine area, eroded social trails (greater than 1" deep), very large compacted sites; extensive, moderate impacts could cover up to 50% of the sample area
4 Habitat destroyed	This level of impact is reached when 50% or more of the site is covered by permanent impacts such as plant or soil loss or erosion.

Data Analysis

All spatial data were entered into Mount Rainier National Park's Geographic Information System. Descriptive and quantitative data was entered into dBase or SPSS databases. Historical patterns of impacts were reviewed by looking at the number and distribution of wilderness impact cards and the distribution of campsites recorded in the social trail and campsite surveys. Numbers of wilderness impact cards were examined for 1989 to 1998 and compared with visitor use nights spent within the wilderness zone. Current impact levels were assessed using the 1998 wilderness impact cards, the 1998 condition class survey, and the 1997 social trail and campsite survey. Geographic distribution of impacts, recorded by each of the three methods, was displayed and visually compared using ArcView. Types of impacts and severity of impacts were also compared between the three methods. Number and severity of impacts was compared between vegetation types for campsite surveys and condition class surveys. Correlation of condition class with elevation and slope was explored using correlation analysis.

Results

Wilderness Impact Cards

Over the past 10 years, 245 wilderness impact cards have been submitted for the Muir Corridor (fig. 3). Comparison of annual visitor use nights (fig. 2) and number of wilderness impact cards submitted (fig. 3) does not reveal similar trends. Visitor use peaks occurred in 1992, 1993, 1997 and 1998. The submission of wilderness impact cards peaked in 1991, 1993 and 1998, relatively low numbers of impact cards were submitted in 1992 and 1997. This is surprising in that numbers of impacts might be expected to increase during years with higher visitation. In the summer of 1998, 53 cards were submitted within five impact categories: smoke, ground disturbance, human waste, litter, and trampled vegetation (fig. 4). There was just one card that listed smoke from multiple stoves as an impact to air quality. Ground disturbance generally referred to the construction of a campsite by clearing rocks, pebbles, and vegetation and often construction of a rock wall to serve as a windbreak. Trampled vegetation was usually noted in the lower portion of the study area where vegetation is lush than the higher elevation fellfields. Time to complete this survey is difficult to estimate because it was a collateral duty, but cards were completed over 19 days. Spatially, the 1998 impacts are concentrated in the northern portion of the study area (fig. 5).

Social Trail and Campsite Inventories

Social trail and campsite surveys were initiated in 1986. In 1987 and 1988, 86 campsites and 74 social trails were documented. All campsites were obliterated in 1989, stabilization and restoration of social trails was initiated (Rocheft 1989) and a campsite-monitoring program was established. Since no camping off snow-covered surfaces was allowed, all campsites discovered were illegal and obliterated. In 1995, 21 new campsites were inventoried and destroyed. In 1997, another

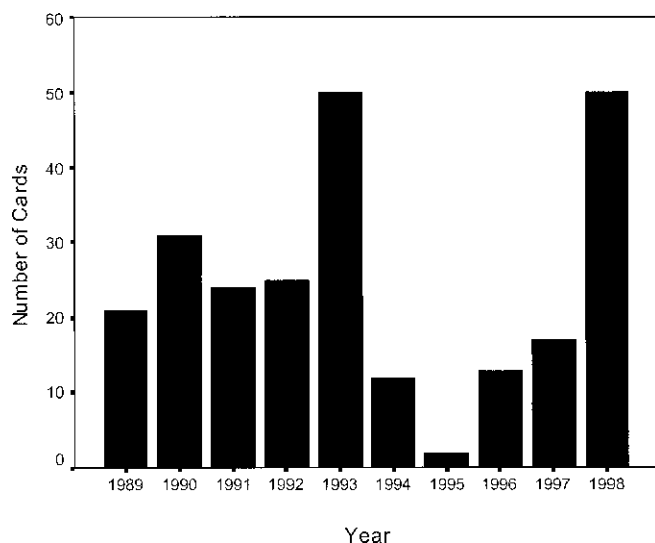


Figure 3—Number of wilderness impact cards submitted since 1989.

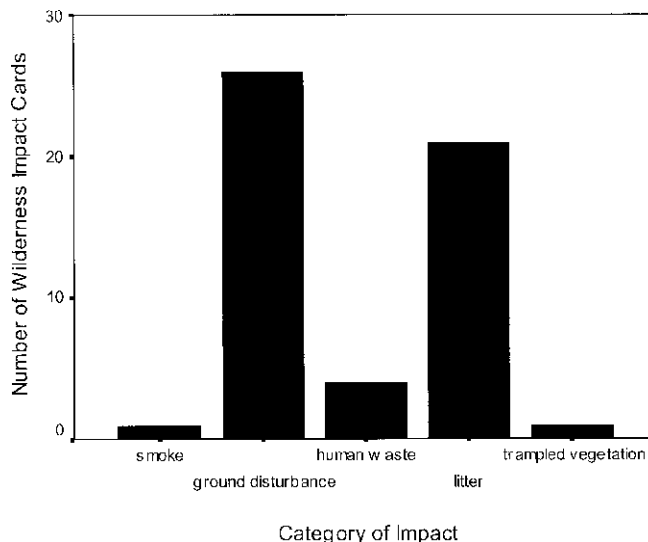


Figure 4—Category of impacts recorded by wilderness impact cards in 1989.

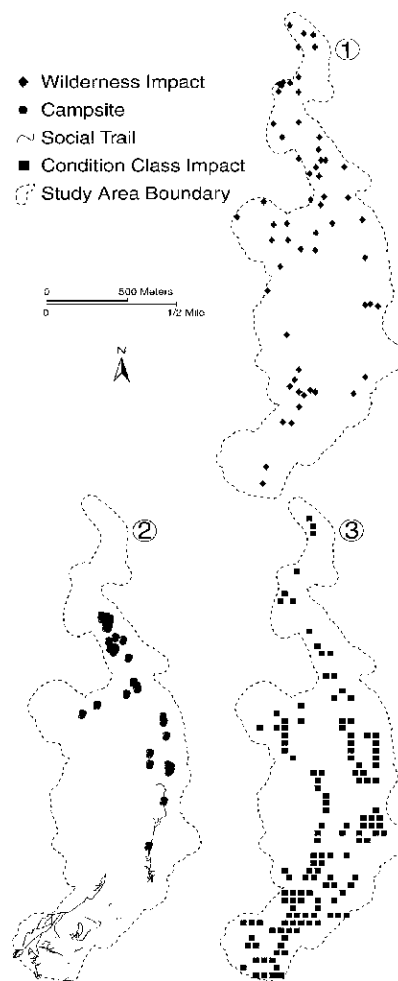


Figure 5—Spatial distribution of human impacts, in Muir Corridor, as recorded by 3 methods: (1) wilderness impact cards (1998), (2) social trail and campsite inventories (1997), and (3) condition class surveys (1998).

43 campsites were discovered and obliterated and all social trails were still visible. In most instances, campsites had been developed on new sites and were not re-establishments of existing sites. The time required to complete these surveys was as follows: 39 workdays (10 hour days) to document all campsites in 1987, 8 workdays to inventory campsites in 1995, 16 workdays to inventory campsites in 1997 and 22 workdays to inventory social trails in 1987. Most campsites, each year, were found in rocky areas such as fellfields and talus slopes (Table 2, fig. 6). Average campsite surface area decreased from 1987 to 1997. The size reduction may reflect the fact that campsites found in 1987 could have been established ten to twenty years before documentation and may have been enlarged with repeated use. Campsites found in 1995 could only have been established since 1988, while those documented in 1997 could have been a maximum of two years old.

A total of seventy-four social trails were documented in Muir Corridor. All trails were found in the southern portion of the study area. Social trails ranged in length from 4.3 m to 580.6 m in length (14 ft. to 1904.4 ft.), the average trail length was 59.3 m (194.5 ft.). Surface area of social trails ranged from 73.6-sq. m. to 633.3-sq. m. (88 to 757 sq. ft.), with a total surface area of 3502.7-sq. m. (4187 sq. ft.). Most social trails meandered through all vegetation types present.

Condition Class Surveys

The 25-m by 50-m grid placed over the 425-acre study area produce 1419 intersections or potential sample points. Sites were only sampled, however, if they were on snow-free sites that were safe to walk to (that is, not too steep). Only 334 points qualified for these criteria. This survey required 31 person days over the course of 10 workdays with a crew of two to four people. Sixty percent of the sample sites were rated as condition class 0 or 1 (fig. 7). The remaining 40% of the sites were in classes 2, 3, or 4—categories that are out of standard in the Wilderness Plan. Contrary to the wilderness impact card surveys, the most severely damaged sites were in the southern portion of the study area. While the campsite-monitoring program documented campsites (the equivalent of a condition class 2, 3, or 4) only in talus and fellfields, condition class surveys also documented heather and sedge areas in condition class 2, 3, or 4. Calculation of Pearson correlation coefficients revealed a significant negative correlation between condition class and elevation ($r = -0.228$, $p = 0.0$, $n = 333$) indicating that condition class generally decreased with increasing elevation. Slope was also negatively correlated with condition class ($r = -0.269$, $p < 0$, $n = 332$) indicating that condition class decreased with increasing slope. This probably reflects the fact that flat areas are more accessible and more attractive to people for walking, sitting and camping.

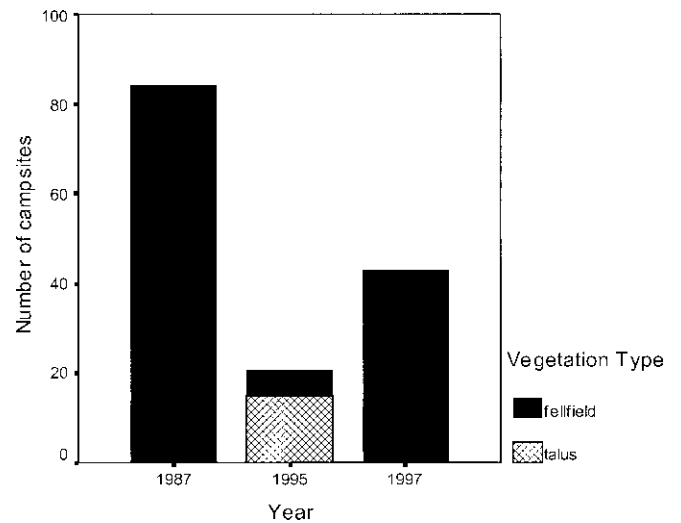


Figure 6—Number of campsites recorded by year and vegetation type.

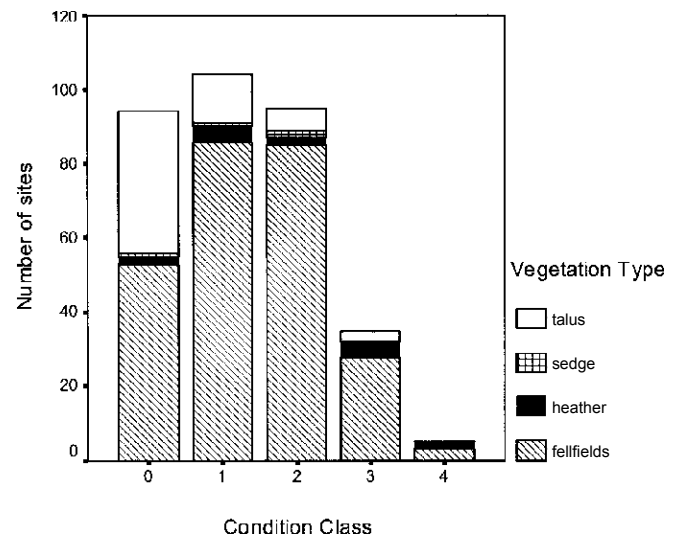


Figure 7—Condition classes recorded in 1998 survey of Muir Corridor.

Summary and Recommendations

Wilderness impact cards recorded point impacts such as campsites, litter and human waste. The cards were often used to record noncompliant personal encounters such as lack of a permit or camping away from snow-covered surfaces. The cards were limited by their inconsistent

Table 2—Characteristics of illegal campsites in Muir Corridor.

Year	No. of campsites	Vegetation type	Mean size (std. dev.) sq. m.	Size range sq. m.
1987	84	fellfield	27.1 (132.3)	0.9 - 1221.7
1995	5	fellfield	6.1 (1.7)	4.6 - 8.7
1995	14	talus	10.8 (4.4)	6.6 - 25.1
1997	42	fellfield	7.5 (4.6)	0.7 - 28.6

implementation and the absence of data fields related to ecological parameters such as vegetation or substrate. All data collected was qualitative, sizes of impacts were never recorded and minor levels of damage such as trampled vegetation was never recorded. Inconsistent implementation of cards was evident both temporally and spatially. There was no correlation between annual overnight use and number of wilderness impact cards (figs. 2 and 3). Since most of the impacts recorded were related to illegal campsites, we expected a correlation between these two factors. In addition, spatial distribution of wilderness impact card submissions showed no impacts in the southern area of the study zone. Based on complete impact surveys (social trails and campsites) and condition class surveys, we know that this area is heavily impacted. We feel this bias in the data reflects the area interests of the climbing rangers and the fact that 50% of the cards were filled out by one person (out of eight). Field personnel did not utilize wilderness impacts cards for recording social trail impacts. This may indicate a limitation of the card format or deficiency in training of field personnel who used these cards.

Social trail and campsite surveys provided the most complete impact inventories because all accessible areas were searched, and all recognizable impacts to soil and vegetation were recorded. In addition, since impacts were measured, this method provided the best quantitative description of impacts to soils and vegetation. Data collection was time-consuming, but if restoration efforts were needed, soil volumes, plant species and plant material volumes could be calculated from the data collected. However, observers did not generally record litter and human waste if it was not located next to a campsite or social trail. Neither this method nor wilderness impact cards recorded diffuse impacts—impacts where patches of vegetation or soil loss were smaller than a campsite or trail.

Condition class surveys described the broadest spectrum of impacts—from diffuse impacts to severely eroded social trails. This method was also adequate for recording litter and human waste. It provided a sample of the study area and probably will not locate all impacts in a study area. However, it was probably the best method for long-term monitoring because sample points were located with a GPS, and the sampling system could easily be modified to collect data of specific interest. In our study, observers recorded vegetation type, slope aspect, microtopography, cryptobiotic crusts and heather reproduction, in addition to condition class assessments.

In summary, we feel that the three survey methods we utilized represent a hierarchy of methods that could be utilized for impact assessments in wilderness or natural areas. Wilderness impact cards could be revised to incorporate ecological data fields and then used as an initial survey method to identify areas that might require intensive surveys or as a means of estimating field personnel needs. Data collection could be improved by systematic or complete surveys of study areas. Complete surveys of study areas in a concentrated time period may provide better assessment of field contact needs than sporadic surveys conducted over an entire field season.

Social trail and campsite surveys provided the best quantitative data, but were the most time-consuming survey method. If restoration of impacted areas is a priority, this

may be the survey method of choice. However, in the alpine area that we surveyed, this method still missed many of the impacted sites that the condition class surveys documented. This may be a reflection of the rocky fellfield substrate. People seem to disperse more readily over the flat areas versus walking through a lush subalpine meadow where temporary trails are easily visible, by trampled vegetation, for the next visitor to follow. While dispersed use did result in small discrete bare areas (condition class 2), these impacts did not readily fall into a category of campsite or social trail so they were not recorded in this method.

Condition class surveys were the optimal methods for assessing the overall ecological integrity of a large area and were relatively fast. Although in our study, only vegetation and soil characteristics were documented, data fields could be added for assessment of aquatic or wildlife resources. This method showed the largest distribution of human impacts within the study area and provided a baseline grid for future monitoring of site conditions. In summary, selection of the best method for any wilderness area is only possible if objectives for surveys and management are clearly articulated.

Future Directions

Currently, Mount Rainier National Park is investigating additional techniques to monitor wilderness and environmental conditions at the Park. During the summer of 1999, we plan to evaluate the use of digital, ortho-corrected, high resolution aerial photographs for mapping environmental conditions such as vegetation, wetland-hydrography features and impacted areas such as bare ground and social trails. We hope this method will be useful for monitoring areas that are difficult to access or infrequently used areas that are currently pristine. If aerial photos reveal impacts, intensive surveys may be detailed to those areas. The Park is also cooperating with the Remote Sensing Group at DOE Pacific Northwest National Laboratory to investigate the use of remote sensing techniques to study relatively small geographic areas. This study will investigate the use of new, higher resolution satellite technology, as well as remote sensing from low-elevation aircraft, to produce sub-meter resolution remote sensing products. It is our opinion that these technologies will enhance, not replace, wilderness monitoring techniques already in use at Mount Rainier. These new methods may give the Park additional tools to monitor larger areas with reduced costs, but also with less then precise results. Results from this type of monitoring may be use provide a trigger for the Park to engage in a more intensive monitoring program such as the social trail and campsite surveys or condition class surveys.

References

- Catton, T. 1996. *Wonderland: An administrative history of Mount Rainier National Park*. Seattle, WA, National Park Service, Cultural Resources Program, 723 pp.
- Edwards, O. M. 1980. *The Alpine Vegetation of Mount Rainier National Park: Structure, Development, and Constraints*. Ph.D., University of Washington. 280 pp.
- Edwards, O. M. 1985. *Human impact on the alpine ecosystem of Mount Rainier*. Final report to Mount Rainier National Park, contract number CX-9000-3-E060. 88p.

- Johnson, D. R., Foster, K. P., and Kerr, K. L. 1991. Mount Rainier National Park 1990 visitor survey. National Park Service, Cooperative Park Studies Unit, College of Forest Resources, University of Washington.
- Rocheft, R. M. 1989. Paradise meadow plan, Mount Rainier National Park. Ashford, WA, Mount Rainier National Park.
- Samora, B. A. 1989. Wilderness management plan, Mount Rainier National Park. Ashford, WA, Mount Rainier National Park. 121 p.
- Schreiner, E.; Morehead, B. 1979. Human impact inventory and management in Olympic National Park backcountry. IN: Ittner, R.; Potter, D. R.; Agee, J. K.; Anschell, S. 1979, Recreation impact on wildlands. Conference Proceedings, October 27 - 29, 1978, Seattle, WA. USDA Forest Service and National Park Service, USDI. R-6-001-1979: 203- 212.
- Stankey, G. H.; McCool, S. F.; Stokes, G. L. 1984. Limits of acceptable change: a new framework for managing the Bob Marshall wilderness complex. *Western Wildlands* 10(3): 33-37.
- National Park Service, 1997. Verp: a summary of the visitor experience and resource protection (VERP) framework. USDI, National Park Service, Denver Service Center. 35 p.
- Wilcox, John. personal communication 1999.

Erosion of Mountain Hiking Trail Over a Seven-Year Period in Daisetsuzan National Park, Central Hokkaido, Japan

Akemi Yoda
Teiji Watanabe

Abstract—Erosion of mountain hiking trails was investigated in Daisetsuzan National Park over a seven-year period. The amount and rate of erosion were different in the two typical landscape components. Cross-section diagrams revealed that trail depth became deeper in snowy vegetated areas than in wind-beaten bare ground areas. The existence and timing of runoff from snowmelt seemed to be important to differential erosion. Trail slope is another factor contributing to erosion. Needle ice or saturation of surface soil appeared to cause side wall erosion. Installation of ropes along the trails made hikers stay on the trail, helping to mitigate erosion.

It is important to observe changes in nature over time for understanding human impacts and for devising effective management methods for conservation. Many studies centering on the degradation of plants and soils due to recreational impacts have been conducted in Europe and the United States. Some studies paid special attention to trail degradation (Bayfield 1973; Bratton and others 1979; Gellatly and others 1986; Price 1985). Others described trail conditions and tried to identify factors contributing to erosion. Experimental studies have been conducted in order to detect human impact. For example, Quinn and others (1980) worked on mechanics of trampling, Cole (1987) observed vegetation recovery after experimental trampling, and Coleman (1981) and Garland and others (1985) investigated relationships between trail deterioration and contributing factors. Yoda (1991) used cross-section diagrams to identify how, when and what part of the paths were eroded and to distinguish the human and natural causes of erosion in Daisetsuzan National Park, northern Japan. Yamada (1993) also used the cross-section diagrams to visualize erosional characteristics of the Mt. Hakusan trail, central Japan.

However, not many studies have been conducted with long-term observations, which are necessary for design of effective and efficient management actions. Long-term studies include Lance and others (1989), who observed trail widening over a five year period and the experiment by Gellatly and others (1986) and Cole (1987) trying to detect the recovery of soil properties and vegetation. Bell and Bliss

(1973) reported on the establishment of plant cover over 31 years in alpine tundra and subalpine meadow. In Japan few studies deal with long-term human impacts on nature with the exception of Watanabe and Fukasawa (1998).

Based on data observed in 1990 and 1997, this study compares the degree of trail erosion in the two major landscape components of snowy vegetated areas and wind-beaten bare ground areas. It also discusses the tendencies of erosional characteristics and the causes of erosion in these landscape components.

Study Area

Topography and Geology

Daisetsuzan National Park is situated in central Hokkaido, northern Japan (fig. 1). This area was designated as a national park in 1934. Its area is about 2,300 km², making it the largest national park in Japan. It is composed of volcanic mountains including Mt. Asahidake (2,290m), the highest peak in Hokkaido. The summit area is covered with the ejecta from Quaternary volcanic activities (Hokkaido Development Agency 1966). In most places the ground surface is covered with volcanic ash and pyroclastic materials.

A variety of periglacial landforms including permafrost, earth hummocks, palsas, patterned ground, solifluction lobes and block fields are spread throughout the summit area above the timberline (Fukuda and Sone 1992; Sone 1992; Takahashi 1990).

Climate

The mean annual temperature observed on the top of Mt. Kurodake (1,984 m) from October 1989 to September 1990 was -2.3°C. The lowest temperature was -21.8°C in January, and the highest one was 18.7°C in July. From October to June is a harsh season, with severe cold and snowfall. The monthly mean temperature was below zero from October 1989 to April 1990, and the study area was completely under snow until early May in 1990. Winter snow usually starts disappearing in May with some snow patches remaining year round.

Vegetation

The timberline is located at about 1,650m in this area (Okitsu and Ito 1984). Japanese stone pine (*Pinus pumila*), Japanese mountain-ash (*Sorbus matsumurana*) and other alpine vegetation occupy the alpine belt. Some species found

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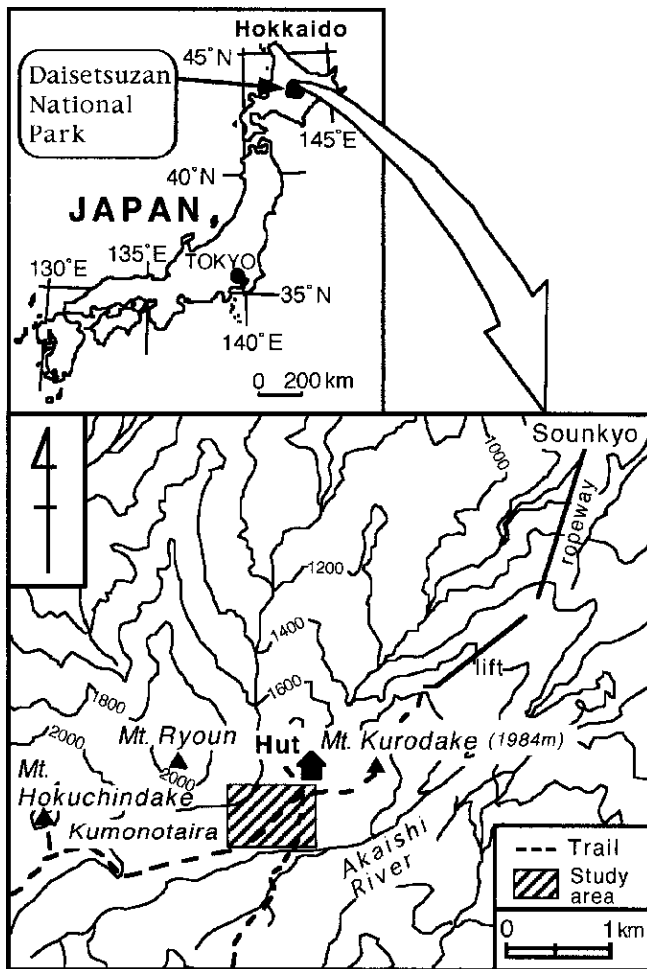


Figure 1—Study area.

on the Kumonotaira plateau, where trails were surveyed, are typical circumpolar arctic plants (Sakai and Otsuka 1970). According to Ito and Sato (1981) the main natural vegetation of wind-beaten bare ground areas are dwarf scrub such as alpine rosemary (*Arctica nana*), alpine-azalea (*Loiseleuria procumbens*), and alpine blueberry (*Vaccinium uliginosum*). Snowy vegetated areas are mainly covered with communities of wedge-leaved primrose (*Primula cuneifolia*), Japanese mountain avens (*Sieversia pentapetala*) and Aleution mountain-heather (*Phyllodoce aleutica*).

Number of Visitors

The number of visitors to Daisetsuzan National Park increased from 410,000 in 1960 to 5,240,000 in 1987 (Oguchi and others 1989). In 1997, 42,814 people visited Mt. Kurodake (data collected by the Kamikawa Forest Office). Because of the severe climate, most visitors come during the summer season when there is little snow (mid June to September).

Trail Management

At the start of the hiking season, park staff installs ropes along the trails to keep hikers on the trails. They put away

ropes at the end of visiting season because of heavy winter snowfall. Each year they install ropes at slightly different places, if replacement of the walking paths is necessary. The few staff members are usually busy watching for the theft of alpine plants in this area, and they do not have enough time to fix degraded trails. Presently no measures have been taken to maintain or fix degraded trails with the exception of less costly small-scale but ad hoc works.

Methodology

The study area includes two typical landscape components: wind-beaten bare ground and snowy vegetated areas (fig. 2). Snowy vegetated areas are covered by either shrub trees or snowy bed community vegetation. Since erosion in the wind-beaten bare ground areas seemed to vary from that in snowy vegetated areas, cross-sectional profiles of the trails were measured in both landscape components. Initially in September 1990, nine cross sections in the wind-beaten bare ground area (cross-sections 1-6 and 10-12) and ten cross sections in the snowy vegetated area (cross-sections 7-9 and 13-19) were measured. These sections were remeasured in June, July or August of 1997.

For accurate repeat measurement at the same site, a pair of aluminum angle stakes was installed at each site on both sides of the trail as a fixed point to observe secular change (fig. 3). To measure the profiles, a fishing line is stretched between the pair of angles, and a tape-measure with a weight attached perpendicular to the ground provides depth (in centimeters) between the line and the trail surface. Depth was measured at 10-centimeter intervals along the line. Thus, the profile of the trail surface was obtained. The amount of erosion or accumulation is recognized by the

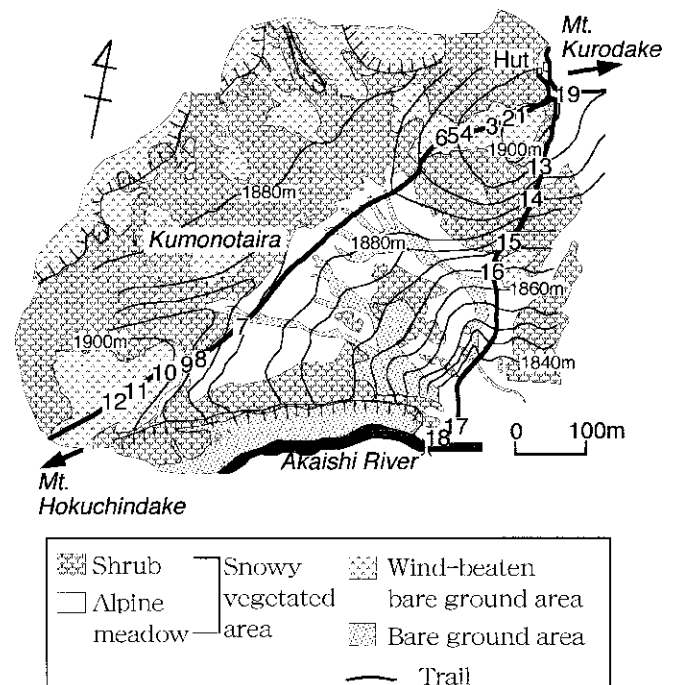


Figure 2—Landscape components and locations of cross sections 1-19.

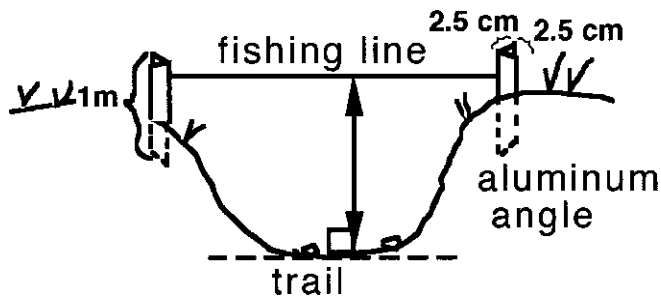


Figure 3—Method for measuring a trail surface.

change in area of each cross section, as reported by Cole (1983).

Patterns and timing of snowmelt were also examined in the field to determine the timing and duration of runoff over the trail surface. The size of each snow patch was measured using a tape-measure and a compass in the field. Measurements were repeated at one month intervals.

Results

Degree of Erosion

All ten cross sections from the snowy vegetated area revealed more erosion than accumulation. The most active erosion occurred at cross-section 7, in the snowy vegetated area, near the head of the gully at the bottom of the trails (fig. 4). Relative height of the gully head at cross-section 7 on the left attained about 80 cm, and the area eroded was about 7,200 cm² over the seven years.

This was not the case for cross sections in the wind-beaten bare ground area. For example, cross-section 6 had 300 cm² of erosion and 1,600 cm² of accumulation in the trail (fig. 5). The deepest erosion was found on the left portion of cross-section 12, becoming about 30 cm deeper in 1997 than in 1990 (fig. 6). Here the eroded area of the trail was 2,200 cm², but the area of soil accumulation on the right side was 2,000 cm².

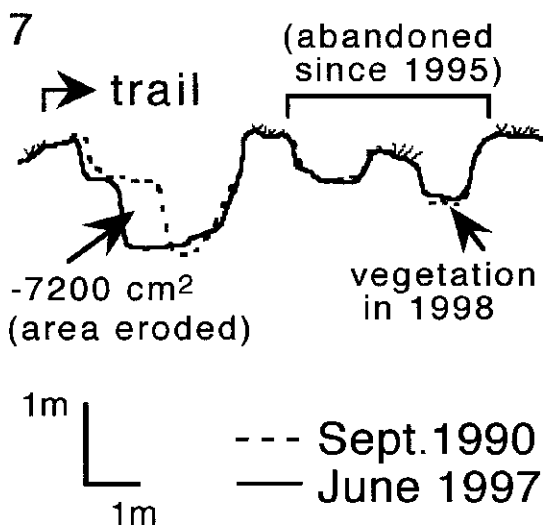


Figure 4—Change in the trail surface at cross-section 7.

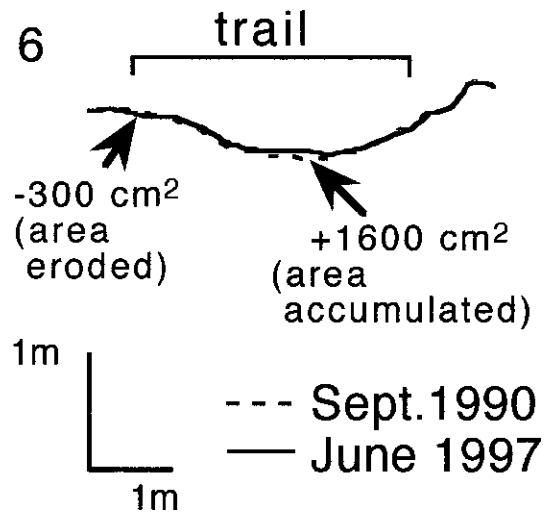


Figure 5—Change in the trail surface at cross-section 6.

Characteristics of Erosion

The erosion was mainly divided into three types in the snowy vegetated area: gully type, valley-shape type and side-wall collapse type. Gully development was observed at section 17 on the left (fig. 7). It was subject to heavy runoff, as snow melts from a nearby patch until late July, becoming 22 cm deeper during the seven years. Other gullies have not been developed at such a rapid pace. Valley-shaped cross sections, such as cross-sections 13 or 14, where people walk on the bottom of the trail showed erosion along the trail bottom (fig. 8). Side-wall collapse was observed at cross-sections 7, 8, 13 and 14 (figs. 4, 9 and 8). In the wind-beaten bare ground area, half of the cross-sections showed small erosion on the trail surface (cross-sections 1, 2, 3, 4, and 11) (fig. 10).

Trail surfaces in the snowy vegetated area had a V- or U-shaped cross section (gully or valley-shape type), if we envision the original ground surface (fig. 11). The wind-beaten bare ground area, on the other hand, had a rather flat cross section. The average ratio of depth to width of the

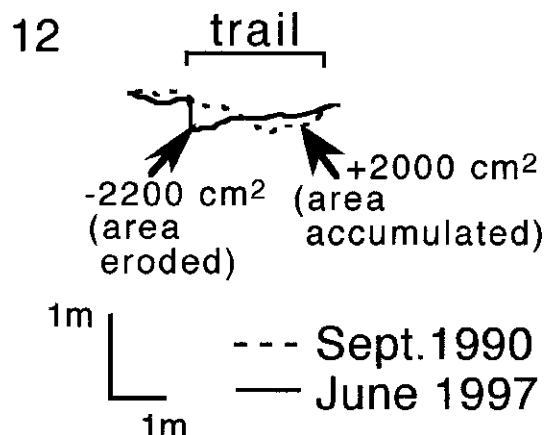


Figure 6—Change in the trail surface at cross-section 12.

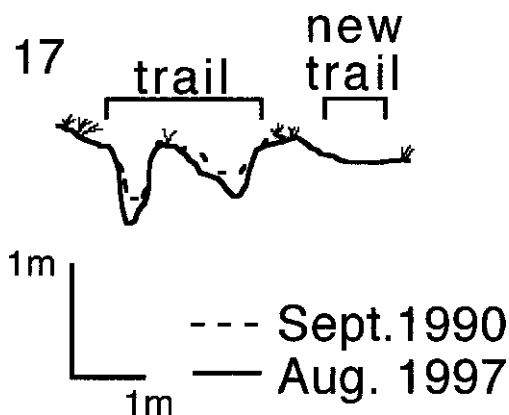


Figure 7—Change in the trail surface at cross-section 17.

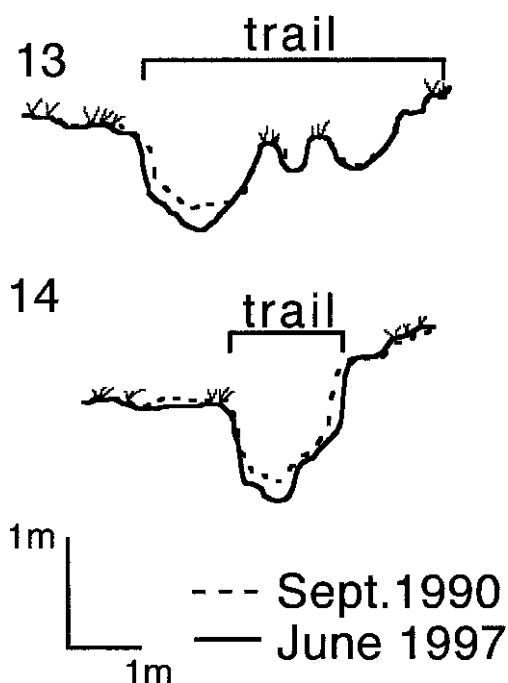


Figure 8—Changes in the trail surface at cross-sections 13 and 14.

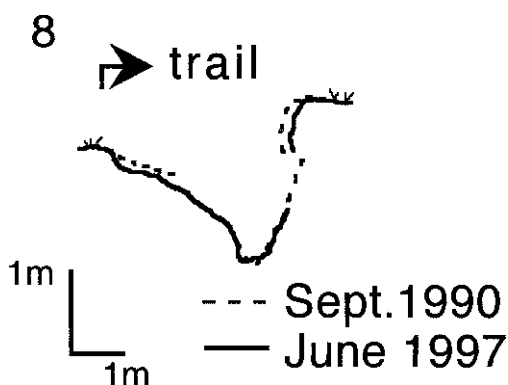


Figure 9—Change in the trail surface at cross-section 8.

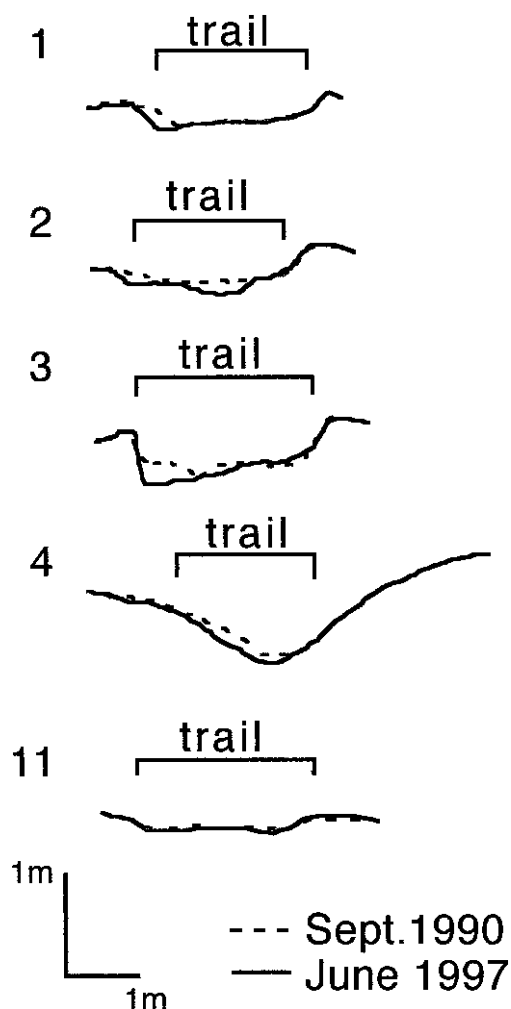


Figure 10—Cross-sections 1, 2, 3, 4, and 11.

eroded trails in snowy vegetated areas was 0.39 (range from 0.06 to 1.00), whereas the average ratio in wind-beaten bare ground areas was 0.18 (range from 0.05 to 0.50). Table 1 shows that trail erosion is more severe in snowy vegetated areas than that observed in wind-beaten bare ground areas. The average amount of erosion in snowy vegetated areas is about 1.9 times larger than in wind-beaten bare ground areas.

Timing of Snowmelt

Snowmelt began nearly one month earlier in wind-beaten bare ground areas than in snowy vegetated areas. The snow began to disappear in the middle of June in 1997 in the wind-beaten bare ground area on the ridge, where cross-sections 1-6 and 10-12 were situated (fig. 12). Subsequently, snow melting went on to the down slope and snow drift areas. Snowmelt down to the Akaishi river in 1997 lasted until the end of July.

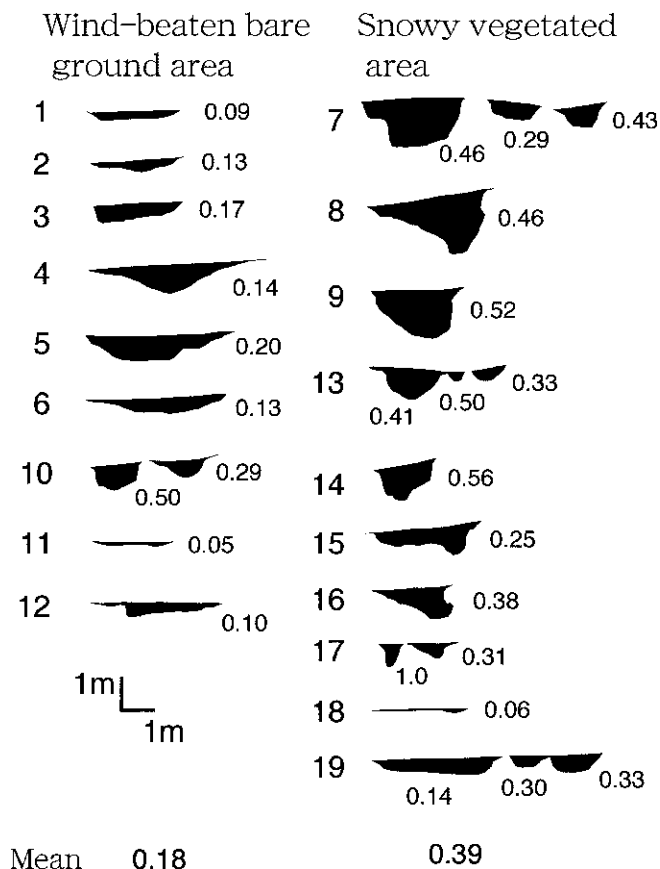


Figure 11—Comparison of the changes in the cross sectional profiles of the trail between the wind-beaten bare ground area and the snowy vegetated area. The figure denotes the ratio of the depth to the width of the eroded trail.

Mitigating Erosion

Installing trail side ropes helped mitigate soil erosion at some sites. Cross-section 10 is a multiple trail (fig. 13). A rope was installed between two paths to designate one path as abandoned. The presently used path on the right had a total of 2,400 cm² of erosion and 500 cm² of accumulation. The path on the left, abandoned for at least ten years, had soil accumulation of 300 cm² on the bottom.

Installing ropes is also useful for vegetation recovery. The abandoned trail in the middle and on the right at cross-section 7 in figure 4 has not been used for three years. There was no erosion, and monocotyledon and other plants had begun to grow on the abandoned trail surface. On the other hand, a new trail was developed on the right side of cross-section 17 due to rope installation (fig. 7). Vegetation was trampled and completely destroyed on the new trail.

Discussion

The study area is covered with loose volcanic materials such as easily erodable pumice and lapili. At cross-section 7 the most active erosion occurred near the gully head. In 1997 the cross-section was located 25 cm downstream from the gully head with the most active erosion due to the gully head retreat. Subsurface layers having different vulnerabilities to erosion will lead to changing erosion rates. Thus, long-term monitoring of erosion is important for forecasting and preventing sudden trail collapse.

Once established gully development continues due to surface runoff. For example, the most developed gully, at cross-section 17, was subject to continuous runoff from a nearby snow patch until the end of July 1997. At this site materials from the ground surface down to a depth of at least 1 m are primarily composed of sandy silt. With other environmental conditions remaining steady the trail surface will

Table 1—The slope angle and amount of erosion observed in trail cross-sections from the two different landscape components (wind-beaten bare ground area and snowy vegetated area) during 1990-1997.

Wind-beaten bare ground area (cross-section number)	Slope angle of the trail (°)	Amount of erosion (cm ²)	Snowy vegetated area (cross-section number)	Slope angle of the trail (°)	Amount of erosion (cm ²)
1	0.5	1200	7	11.5	10900
2	3.0	1800	8	6.0	3900
3	3.0	2600	9	8.0	4400
4	7.5	2000	13	14.0	3600
5	4.0	4700	14	10.0	3700
6	4.0	300	15	10.0	3100
10	4.5	2400	16	10.5	1900
11	2.0	800	17	1.0	2600
12	3.0	2200	18	0.5	700
			19	0.5	2900
average	3.5	2000	average	7.2	3770

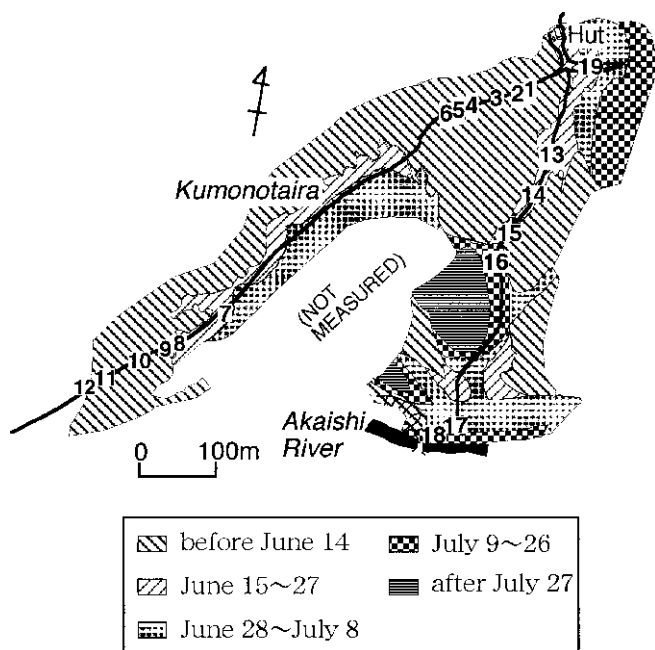


Figure 12—Timing of snowmelt in 1997.

erode continuously until a more erosion resistant layer is encountered. Many trails are now functioning as runoff streams for snowmelt and rain water. Therefore, the timing and the amount of snowmelt at each site are important factors in determining trail erosion rates.

The snowy vegetated area is usually developed on the middle or foot of the slope, and it is subject to greater and larger duration of surface runoff from a nearby snow patch. Although the slope angle is rather small, the amount of erosion at cross-section 17 was 2,600 cm². In this area surface runoff is a large contributor to erosion.

However, slope angle is another potential factor contributing to erosion. Coleman (1981) suggests that path slope is one of the variables contributing to depth of gulying on a

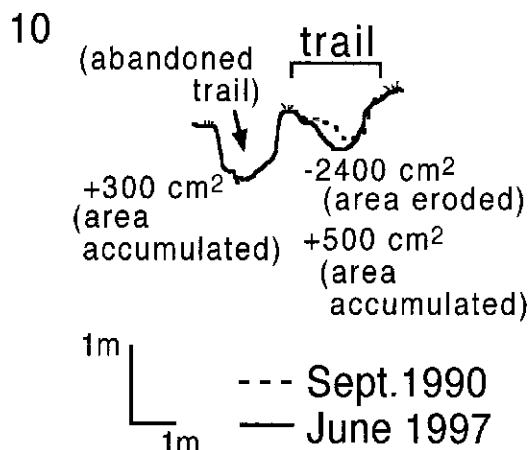


Figure 13—Change in the trail surface at cross-section 10.

trail. The wind-beaten bare ground area is located on the ridge and has a generally small slope. Cross-sectional sites surveyed in the wind-beaten bare ground area have slope angles from 0.5 to 7.5 degrees (table 1). The snowy vegetated areas have slope angles ranging from 0.5 to 14.0 degrees. Average erosion in the snowy vegetated area is about 1.9 times larger than that in the wind-beaten bare ground area (table 1).

The snowy vegetated area tends to be more vulnerable to erosion than the wind-beaten bare ground area in the study area. However, further study is needed to clarify the erosional contribution of each contributing factor (path slope and duration/amount of surface runoff) in the two landscape components.

Observations also suggest other erosion contributing factors. Needle ice erosion or detachment of saturated surface soil seems to have caused the side collapse of cross-sections 7, 13 and 14. This collapse is active during the snowmelt and freeze-thaw season and after heavy rainfall.

Observing trail erosion over the past seven years clearly demonstrates that the snowy vegetated area is more vulnerable to erosion than the wind-beaten bare ground area. This study also determined sites where active erosion is occurring and where immediate remediation measures should be taken.

Acknowledgments

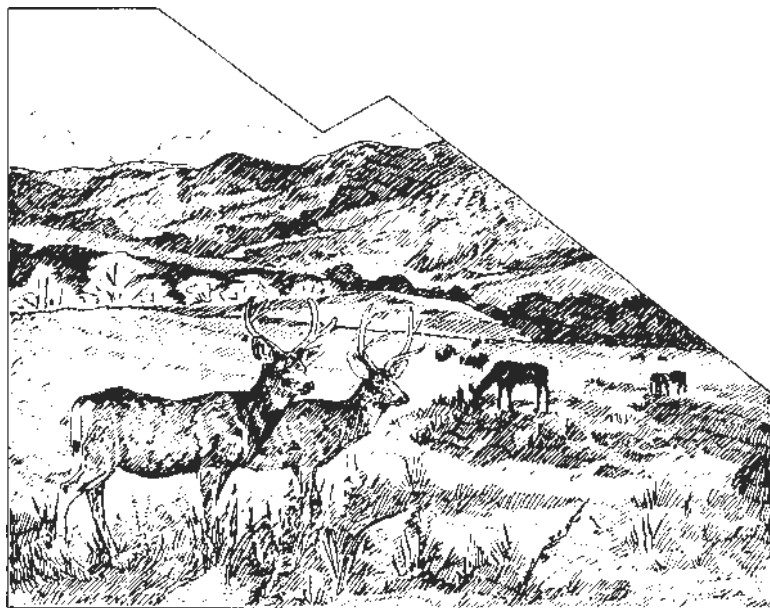
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References

- Bayfield, Neil G. 1973. Use and deterioration of some Scottish hill paths. *Journal of Applied Ecology*. 10: 635-644.
- Bratton, Susan P.; Hickler, Matthew G.; Graves, James H. 1979. Trail erosion patterns in Great Smoky Mountains National Park. *Environmental Management*. 3(5): 431-445.
- Bell, Katherine L.; Bliss, Lawrence C. 1973. Alpine disturbance studies: Olympic National Park, USA. *Biological Conservation*. 5(1): 25-32.
- Cole, David N. 1983. Assessing and monitoring backcountry trail conditions. USDA Forest Service Research Paper, INT-303. 10p.
- Cole, David N. 1987. Effects of three seasons of experimental trampling on five montane forest communities and a grassland in western Montana, USA. *Biological Conservation*. 40: 219-244.
- Coleman, Rosalind. 1981. Footpath erosion in the English Lake District. *Applied Geography*. 1: 121-131.
- Fukuda, Masami; Sone, Toshio. 1992. Some characteristics of alpine permafrost, Mt. Daisetsu, central Hokkaido, northern Japan. *Geografiska Annaler*. 74A(2-3): 159-167.
- Garland, Gerald G.; Hudson, Christine; Blackshaw, Jane. 1985. An approach to the study of path erosion in the Natal Drakensberg, a Mountain Wilderness Area. *Environmental Conservation*. 12(4): 337-342.
- Gellatly, Anne F.; Whalley, W. Brian; Gordon, John E. 1986. Footpath deterioration in the Lyngen Peninsula, North Norway. *Mountain Research and Development*. 6(2): 167-176.

- Hokkaido Development Agency. 1966. Geological map of Japan. Taisetsuzan, Abashiri-43.
- Ito, Koji; Sato, Ken. 1981. Explanatory notes for actual vegetation map of the Daisetsu mountains. Synthetic survey of the Daisetsu mountains, Hokkaido. Hokkaido prefectural government. 23p. (in Japanese).
- Lance, Art N.; Baugh, Ian D; Love, John A. 1989. Continued footpath widening in the Cairngorm Mountains, Scotland. *Biological Conservation*. 49: 201-214.
- Oguchi, Takeo; Kobayashi, Akihiro, Ishikawa, Yukio. 1989. Studies on the conservation of Daisetsuzan National Park (I). *Journal of Environmental Science Laboratory, Senshu University*. 1: 103-125.
- Okitsu, Susumu; Ito, Koji. 1984. The relation of forest limit to the W115 in mountains of Hokkaido. *Japanese Journal of Ecology*. 34(3): 341-346. (in Japanese with English abstract).
- Price, Martin F. 1985. Impacts of recreational activities on alpine vegetation in western north America. *Mountain Research and Development*. 5(3): 263-277.
- Quinn, N. W.; Morgan, R. P. C.; Smith, A. J. 1980. Simulation of soil erosion induced by human trampling. *Journal of Environmental Management*. 10: 155-165.
- Sakai, Akira; Otsuka Koji. 1970. Freezing resistance of alpine plants. *Ecology*. 51(4): 665-671.
- Sone, Toshio. 1992. Permafrost environment of the Daisetsu Mountains, Hokkaido, Japan. *Permafrost and Periglacial Processes*. 3: 235-240.
- Takahashi, Nobuyuki. 1990. Environmental-geomorphological study on the Holocene mire development in the Daisetsuzan Mountains, central Hokkaido, northern Japan. *Environmental Science, Hokkaido University*. 13(1): 93-156.
- Watanabe, Teiji; Fukasawa, Kyoko. 1998. Seven-year deterioration of a hiking trail and measures to mitigate soil erosion, mount Kurodake, Daisetsuzan National Park, Hokkaido, northern Japan. *Geographical Review of Japan*. 71A(10): 753-764. (in Japanese with English abstract).
- Yamada, Shuji. 1993. Footpath erosion in Mt. Hakusan. *Bulletin of the Environmental Research Center, University of Tsukuba*. 17: 65-72. (in Japanese).
- Yoda, Akemi. 1991. The erosion of mountain hiking trails near Mt. Kurodake hut in the Daisetsuzan mountains National Park, Central Hokkaido, Japan. Master's thesis. Graduate School of Environmental Science, Hokkaido University, Japan.

3. Wilderness Restoration



Soil Amendments and Planting Techniques: Campsite Restoration in the Eagle Cap Wilderness, Oregon

David N. Cole
David R. Spildie

Abstract—Results of the first three years of revegetation research on closed wilderness campsites are described. Experimental treatments involved soil scarification, an organic soil amendment (a mix of locally collected organic materials and peat moss and an inoculation of native undisturbed soil), an organic matter and composted sewage sludge treatment and surface application of commercial mulch (Bionet). Half of the experimental plots received native seed and transplants; the other half did not. Seeding and transplanting were highly successful. The organic and compost soil amendment greatly increased seedling growth and increased transplant growth somewhat. Scarification increased seedling establishment of volunteer seedlings.

On federal lands designated by Congress as Wilderness, management objectives stress protection of natural conditions. Despite this emphasis on protection and preservation, wilderness areas are typically open to recreation use, and resultant impacts can be severe, particularly on campsites. Most campsite impact is accepted as necessary if recreation use is to be allowed. However, in some situations, campsite impacts are deemed to be either excessive or inappropriate in that particular location. In these situations, wilderness managers close sites to camping so they can return to conditions approximating those that existed prior to disturbance. Where recovery rates are slow, managers often employ various restoration treatments in an attempt to accelerate successional processes (for example Lester 1989). These efforts are often costly, in terms of time and money, and frequently are not very successful (for example Moritsch and Muir 1993).

In many wilderness ecosystems, little is known about factors that limit the rate of natural recovery or about the effectiveness of techniques designed to accelerate recovery. Consequently, we designed a study to assess the effectiveness of several common restoration treatments on closed campsites in high subalpine forests in the Eagle Cap Wilderness, in northeastern Oregon. Specific objectives were to assess the influence of (1) amending soils with organic matter, composted sewage sludge and a native soil inoculum, (2) transplanting and seeding with local, native species,

and (3) applying a surface mulch on the establishment, survival and growth of vegetation.

Study Sites

The study is being conducted in the Lakes Basin portion of the Eagle Cap Wilderness, Wallowa-Whitman National Forest, northeastern Oregon. This area, located between 2,170 m and 2,320 m, contains a number of subalpine lakes. Located 12-15 km from the closest trailheads, the Lakes Basin attracts large numbers of wilderness campers. Campsite impacts around these lakes are substantial and numerous (Cole 1981, 1993). Early efforts to close and restore campsites began in the 1970s. These efforts were largely unsuccessful. Campsites that have been closed to use have experienced little recovery over a period of more than a decade (Cole and Hall 1992).

Six campsites were selected for restoration in 1995. All are in a subalpine forest consisting of *Abies lasiocarpa*, *Picea engelmannii*, *Pinus contorta* and *Pinus albicaulis*. The most common groundcover plants, in undisturbed places, are *Vaccinium scoparium*, *Phyllodoce empetrifomis* and *Carex rossii*. Soils are derived from a granitic substrate. All are within 70 m of lakes and, therefore, have been illegal campsites for more than 15 years. However, all of these sites received some camping use over this period, had virtually no groundcover vegetation and had not been revegetated in the past. These sites have probably exhibited high levels of impact (soil compaction, lack of vegetation and minimal soil organic horizons) for at least 50 years. Prior to restoration efforts, these campsites were typically about 200 m² in size, with about 100 m² completely devoid of vegetation.

Methods

Each campsite was divided into two whole plots, one with and one without a surface mulch application. Each whole plot was subdivided into six plots, which received combinations of the two factors: soil amendments (organics/inoculum; organics/inoculum/compost; or nothing) and planting (transplanted/seeded; or nothing). All 12 of these 1.5 m by 1.5 m plots were scarified. An additional plot, the control, received no treatment at all. The six campsites provide six replicates.

Treatments

Scarification utilized shovels, picks, pitchforks, hoes and hand kneading to break up compaction and clods to a depth

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of about 15 cm. We tried to avoid turning over the soil, but substantial mixing of soil horizons was unavoidable in our resolve to develop a crumb texture. On several sites, numerous tree roots were cut and removed during scarification. This intensity of scarification exceeds that commonly undertaken. Treatments that received the organics/inoculum treatment were covered with a mix of peat moss and well-decomposed, locally collected organic matter to a depth of about 2.5 cm. The dry peat moss was mixed with water before application. This material was then mixed with mineral soil to a depth of 7.5 cm. Inoculum came from the rooting zone of local transplants that were being transplanted onto the site. About 1.2 liters of soil were mixed with about 20 liters of water to make a slurry. Three liters of this slurry were sprinkled over each plot and raked into the soil. Compost treatments had organic matter and inoculum added in an identical manner. In addition, we added 2.5 cm of composted sewage sludge (Ekocompost from Missoula, Montana), lightly watered and raked into the top 10 cm of organic and mineral soil.

Half of the plots were seeded and transplanted. Seeding involved (1) collecting seed locally from several species with mature seed, (2) division of available seed into equal quantities for each seeded plot, (3) pinch-broadcasting seed over the plot, and (4) raking seed into the upper 2.5 cm of soil. Seeded species varied between campsites and included *Antennaria lanata*, *Aster alpigenus*, *Danthonia intermedia*, *Juncus parryi*, *Penstemon parryi*, *Phleum alpinum*, *Sitanion hystrix* and *Sibbaldia procumbens*. Locally available seed was unusually limited due to the unusually short growing season in 1995. One of the campsites (at Crescent Lake) was not seeded due to a lack of mature seed in the vicinity.

Transplanting involved (1) digging up enough transplants in the vicinity to plant equal numbers of each species in each plot, (2) digging a hole and placing transplants in the hole, along with Vita-start (vitamin B-1) to reduce transplant shock, and (3) giving each transplant 0.6 liters of water. Plots that were not planted were given an equivalent amount of water. Most transplant plugs were between 5 and 25 cm in diameter, and most plots received five to six plugs. Although most plugs contained only one species, some contained more than one. Transplanted species varied between campsites and included *Abies lasiocarpa*, *Achillea millefolium*, *Antennaria alpina*, *Antennaria lanata*, *Aster alpigenus*, *Calamagrostis canadensis*, *Carex rossii*, *Danthonia intermedia*, *Gaultheria humifusa*, *Hypericum formosum*, *Juncus parryi*, *Luzula hitchcockii*, *Oryzopsis exigua*, *Phyllodoce empetrifomis*, *Pinus contorta*, *Polemonium pulcherrimum*, *Sibbaldia procumbens*, *Spiraea betulifolia* and *Vaccinium scoparium*. All seeding and transplanting occurred in the central 1 m² of each plot. Measurements were also confined to this central area, leaving a 1 m buffer between the measured portion of each treated plot.

Half of the plots were covered with a biodegradable erosion control blanket made of straw interwoven with cotton string and jute (Bionet). The blanket was held in place with rocks. Where there were transplants, string was cut to allow the transplants to penetrate the strands of straw. Each campsite was closed to use by blocking main access points with string and an obvious sign. No evidence of camping use has been observed since campsites were closed to use.

In 1996, when it appeared that soils were extremely dry, plots were watered several times. When this was done, all plots were given an equal amount of water. No supplemental watering was done in later years. In all four years of the study, the late snowpack was unusually deep, suggesting that early season conditions were much less droughty than normal. In 1996, the first growing season after restoration, when plots were occasionally irrigated, the summer was dry but cool. In 1997, the summer was cool and wet. In 1998, the summer was hot and dry, and plants were not given supplemental water. In 1998, at Aneroid Lake, located at a similar elevation in an adjacent drainage, the mean maximum temperatures in July were 5-7 degrees Celsius higher than in 1996 or 1997. At Mt. Howard, the closest site with precipitation data, July-August precipitation was 18 cm in 1997, compared with 7 cm or less in 1996 and 1998.

Measurements

For each transplant, we measured areal extent of canopy cover (using a 1-m square PVC frame with a 5-cm by 5-cm grid) and maximum height. Measurements were taken immediately after transplanting (September 1995) and in September of 1996, 1997 and 1998.

Seedling establishment was assessed beginning in early July 1996. Every two weeks from early July to early September (four times), all established seedlings were mapped. Each seedling was identified by species, and a colored toothpick was placed next to it to denote date of establishment. This made it possible to assess period of establishment and death, if mortality occurred. In 1997, seedlings that germinated in 1996 were identified on the basis of their size, location and species. New seedlings (the 1997 cohort) were identified in the surveys conducted every two weeks. In some plots, seedlings were so numerous that they were assessed in subplots. In 1998, seedling assessment occurred twice, in mid-July and early September. Again, we attempted to differentiate between new seedlings (the 1998 cohort) and older plants.

Ten individuals of a seeded species were randomly selected on each plot, and their height was measured in September of 1996, 1997, and 1998. In 1996 and 1997, another four individuals of the same species were carefully excavated. Their root and shoot biomass was measured, following cleaning and drying. In 1998, we measured the height of the tallest individual of the seeded species, which we had found to be well-correlated with biomass. This avoided the need for further destructive sampling. In 1997 and 1998, height and biomass measurements were taken only on seedlings that germinated and established in 1996.

Transplant areal extent and seedling locations were digitized to allow spatial analysis. Treatment effects were analyzed using standard statistical techniques, primarily t-tests and analyses of variance, with post-hoc Duncan's multiple comparisons.

Results

In September 1995, a total of 206 plugs were transplanted onto 36 of the 78 plots. These plugs contained 280 individual

transplants (either separate species, separate individuals or separate vertical stems that might be separate individuals). By September 1996, 96% of these plugs and 92% of the individual transplants were still at least partially alive. By September 1997, 96% of the original plugs were still living, and the number of live individual transplants (382) exceeded the number apparent at the time of planting. Transplant mortality was greater during the hot, dry summer of 1998. Nevertheless, by September 1998, 90% of the original plugs and 86% of the original transplants were still living (table 1). In 1998, the number of live individual transplants (318) still exceeded the number of individuals apparent at the time of planting.

The mean canopy cover of surviving transplants (areal extent) decreased 5 cm² (5%) between September 1995 and September 1996. By September 1997, mean canopy cover was 113 cm²; by September 1998, mean canopy cover was 151 cm². In three years since transplanting, then, the mean cover of surviving transplants increased 52 cm² (55%). The total cover provided by all transplants was 33% greater in 1998 than in 1995, when transplanting occurred. Mean transplant height declined 4% (from 7.6 cm to 7.3 cm) during the first year following transplanting. By September 1997, mean transplant height was 10.2 cm. By September 1998, mean transplant height was 66% greater than at the time of transplanting. In 1997, 12% of the transplants flowered. Forty percent of transplants flowered in 1998.

During the summer of 1996, almost 20,000 seedlings germinated and established on the 78 1 m² plots. Most of these seedlings (>70%) germinated from the seed we had broadcast. However, volunteers germinated from seed that reached the site through natural dispersal processes or, perhaps, from the soil seedbank. In 1996, most of the volunteers were perennial species; in 1997 and 1998, most volunteers were annual species. Germination and establishment continued throughout the two-month assessment period (early July to early September). However, about two-thirds of the seedlings established (cotyledons were well-developed) in the early August period—about one month after snow had left most plots. Germinants were probably emerging from the soil about two weeks prior to the point at which we considered them established.

In 1997, the 1996 seedling cohort generally emerged early, by mid- to late July. The 1997 seedling cohort established throughout the season, but primarily in early August. The 1997 cohort of seeded species was about one-third as abundant as the 1996 cohort (fig. 1). The 1997 cohort of volunteer perennials was about one-half as abundant as the 1996 cohort. However, annuals were about four times more abundant in 1997 than in 1996. In 1997, the total number of seedlings that either re-emerged (the 1996 cohort) or became established (the 1997 cohort) on the 78 1 m² plots exceeded 25,000.

The 1998 cohort of seeded species was small, only about 2% as abundant as the original 1996 cohort. The 1998 cohort of volunteer species was similar in quantity to the 1997 cohort, about one-half of the 1996 cohort. Annual species

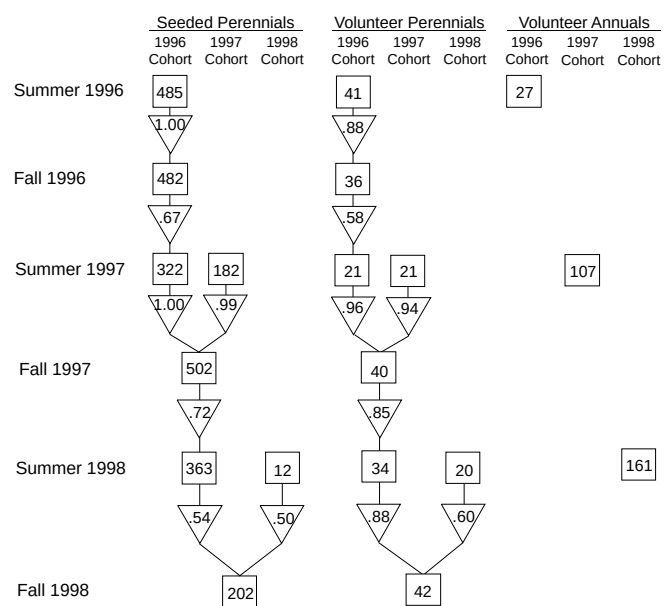


Figure 1—Seedling density (#/m²—in squares) and survival rates (in triangles) of seeded species and volunteers from cohorts of seedlings that established in 1996, 1997, and 1998.

Table 1— Mean (standard error) plug and transplant survival, growth and flowering, 1995–1998.^a

Treatment	Plug survival	Transplant survival	Change in transplant area	Change in transplant height	Transplant flowering
	----- Percent ^b -----		(cm ²) ^c	(cm) ^c	Percent
Soil amendment					
None	88(4)	84(4)	30(12)a	4.4(0.8)a	42(4)
Organics	89(4)	87(4)	49(25)ab	5.7(0.7)ab	76(32)
Organic/compost	92(4)	89(4)	76(12)b	7.6(1.2)b	35(4)
Mulch treatment					
None	90(3)	85(3)	36(15)	6.0(0.7)	60(22)
Mulched	89(3)	89(3)	67(19)	5.8(0.8)	42(4)
Total	90(2)	86(2)	52(12)	6.0(0.5)	51(11)

^aMeans with different letters are significantly different ($\alpha = 0.05$).

^bPlugs and transplants (individual species within a plug) planted in 1995 still alive in 1998.

^cChange between time of planting in 1995 and 1998.

were extremely abundant. In 1998, the total number of seedlings that either re-emerged or became established on the 78 1 m² plots was about 16,000, 36% less than in 1997. Seedling density varied greatly between the six campsites.

Seedling survival varied between years and seasons, as well as between seeded species and volunteers (fig. 1). We could not assess mortality prior to seedling establishment (the emergence of well-developed cotyledons). However, once established, there was virtually no mortality (< 1%) of seeded species during the summer of 1996, when plots were occasionally watered, and the summer of 1997, which was wet. For volunteer perennial species, mortality rates were slightly higher, 12% in the summer of 1996 and about 5% in 1997. For both seeded and volunteer species, mortality during the winter of 1996-1997 was about 35-40%. Overall, about 65% of the seedlings that established in 1996 were still alive in fall of 1997. Mortality within the 1996 cohort was more than offset by germination and establishment of additional seedlings in 1997. About 17,000 perennial seedlings were alive on the 78 1 m² plots in September 1996; about 18,000 perennial seedlings were alive in September 1997.

Winter mortality rates were lower during the winter of 1997-1998—28% for seeded species and 15% for volunteer perennials. However, mortality rates during the hot, dry summer of 1998 were high, particularly for seeded species and for the small 1998 cohort. Close to one-half of the seedlings that re-emerged or established in 1998 had died by September 1998. About 10,000 perennial seedlings were alive on the 78 1 m² plots in September 1998. The proportion of perennial seedlings that were volunteers increased from 16% in 1996 to 35% in 1998.

The spatial pattern of the initial cohort of seedlings was analyzed with GIS. The 1996 cohort of seedlings was neither regularly nor randomly distributed. They were aggregated to a significant degree. Seeded species were more aggregated than volunteers. This suggests that aggregation resulted from both the seeding process and the availability of “safe sites.” Seedling density was greater outside transplant plugs than within, but seedlings were attracted to the transplants (that is, they were located closer to transplants than expected). Seeded and volunteer species did not differ in the extent to which they were less abundant under transplants or more abundant close to transplants. This suggests that conditions close to transplants favor seedling establishment, while conditions underneath transplants discourages establishment. It is unclear whether these spatial patterns result from transplant effects on seed dispersal-entrapment patterns, soil conditions, microclimatic conditions or competitive interactions.

Treatment Effects

Survival of transplants was high (about 86%) regardless of treatment (table 1). Between 1995 and 1998, increase in canopy cover (areal extent) of transplants was significantly greater on plots with the organic and compost soil amendments than on scarified plots that received no soil amendments (fig. 2). Increase in height was also significantly greater on organic and compost plots (table 1). Compared to plots without a surface application of mulch, mulched plots experienced a greater increase in canopy cover but a smaller increase in height. Neither of these differences was

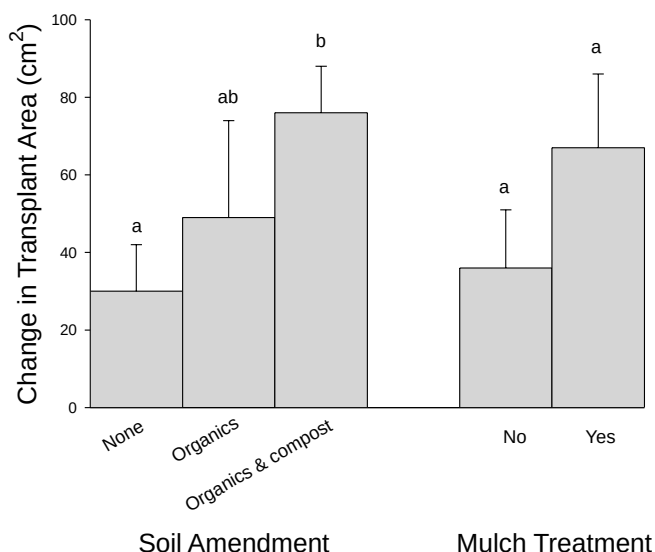


Figure 2—Effect of soil amendments and mulch on increase in canopy cover of transplants, 1995 to 1998. Vertical bars indicate one standard error; different letters indicate significantly different means ($\alpha = 0.05$).

statistically significant, however. Transplant flowering was highest on plots that received organic soil amendments but no compost, as well as on plots without a mulch treatment. This response was highly variable within treatments, however, and differences are not statistically significant.

The effect of scarification on seedling density was assessed by comparing the nonscarified control plot with plots that were scarified but received no other treatment. On these two sets of plots, all established seedlings are volunteers. In 1998, scarified plots had a significantly greater seedling density (mean of 31 seedlings/m²) than control plots (7 seedlings/m²) (fig. 3). Seeding had a tremendous influence on seedling density, with seeded plots having over five times as many seedlings as unseeded plots, three years after seeding. Volunteers were equally abundant on seeded and nonseeded plots. Plots that were amended with organic matter and compost had significantly more seedlings than unamended plots. However, the surface mulch treatment did not have a significant effect on total seedling density, the density of seeded species or the density of volunteers.

Explanations for differences in seedling density varied between the treatments. Scarification was advantageous to seedling establishment. The scarified plots included in the analysis were not seeded, and their mortality rate was not significantly lower than that on controls. Seeding was advantageous because it provided a more abundant source of propagules. Despite higher mortality rates on seeded plots (table 2), seedling density remains higher on seeded plots. The organic/compost amendment was advantageous because mortality rates during the hot, dry summer of 1998 were lower than on other plots (table 2). Propagule availability and establishment rates were no greater on organic/compost plots. After the moist 1996 and 1997 summers, seedling density was not greater on the organic/compost plots.

More than one-third (36%) of the seedlings that re-emerged or established during the hot, dry summer of 1998 died,

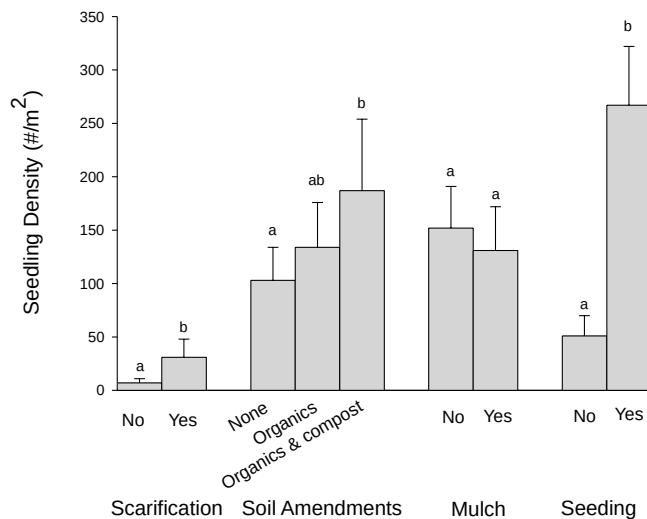


Figure 3—Effect of scarification, soil amendments, mulch and seeding on seedling density, September 1998. The scarification treatment reported received no soil amendment, mulch or seeding. Vertical bars indicate one standard error; different letters indicate significantly different means ($\alpha = 0.05$).

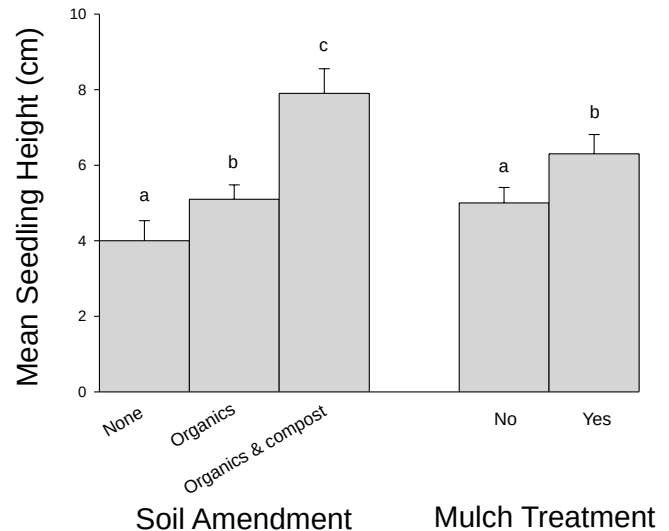


Figure 4—Effect of soil amendments and mulch on increase in mean height of 3-year-old seedlings, September 1998. Vertical bars indicate one standard error; different letters indicate significantly different means ($\alpha = 0.05$).

Table 2—Seedling density in September 1998 and seedling mortality and flowering during 1998.^a

	Seedling density (#/m ²)	Seedling mortality ----- Percent -----	Seedling flowering
Soil amendment			
None	103(31)a	33(7)a	7(4)a
Organics	134(42)ab	23(4)ab	5(2)a
Organics/compost	187(67)b	13(3)b	15(4)b
Mulch treatment			
None	152(39)	24(4)	9(3)
Mulched	131(28)	21(4)	9(3)
Seeding treatment			
None	51(19)a	17(3)a	9(3)
Seeded	267(55)b	32(6)b	9(2)
Scarification treatment			
None	7(4)	36(18)	0(0)
Scarified ^b	31(17)b	23(4)	20(16)
Total	141(28)	23(3)	9(2)

^aMeans with different letters are significantly different ($\alpha = 0.05$).

^bScarified treatment received no soil amendment, mulch or seed.

although the mean seedling mortality rate for the plots was just 23%. Nine percent of surviving seedlings flowered in 1998. Flowering rates were significantly higher on the plots that were amended with organic matter and compost than on unamended plots or plots amended with just organic matter and soil inoculum (table 2).

Seedling growth was influenced by both soil amendments and mulching (fig. 4). In September 1996, the mean height of seedlings of a selected seeded species was 1.7 cm. Seedling height was significantly greater on plots that received either the organics amendment (mean of 1.8 cm) or the organics and compost amendment (1.9 cm) than on plots without

amendments (1.5 cm). Seedling height was also significantly greater on mulched plots (2.0 cm) than on plots without mulch (1.5 cm). By September 1997, the mean height of seedlings established in 1996 had increased to 3.5 cm, and it was possible to confidently guess a plot's soil treatment simply by observing seedling robustness. Mean height increased to 5.7 cm in 1998. In 1997 and 1998, seedling height on the plots with the organics and compost amendment was significantly greater than on plots receiving organics or no soil amendments at all. Plots amended with organics had taller seedlings than unamended plots (fig.4). Seedling height was also significantly greater on the mulched plots than those without mulch, in both 1997 and 1998.

For the selected seeded species, the mean biomass of seedlings that established in 1996 increased from 12 mg in 1996 to 190 mg in 1997. Their root:shoot ratio increased from 0.34 in 1996 to 0.52 in 1997. In both 1996 and 1997, seedlings on plots amended with organics and compost had significantly more biomass than seedlings on other plots. Root:shoot ratios did not differ significantly with soil treatment, although they were higher on the organics (0.65) and organics and compost plots (0.55) than on plots without soil amendments (0.37). Greater biomass of individual seedlings might partially explain the lower seedling mortality rates on the plots with organic and compost amendments. Mulching had no effect on biomass in either year.

Vegetation cover was negligible before restoration. Immediately after transplanting and seeding, mean total cover was 3.7%. Mean cover increased to 9.0% in 1996, 10.6% in 1997 and 13.2% in 1998 (table 3). Most cover was initially provided by the transplants. By 1997 and 1998, however, more cover was provided by plants that germinated from seed. Although initially most of the plants that germinated from seed were seeded species, the proportion of volunteers has increased every year. Planting was the treatment that had the most dramatic effect on cover. Mean cover exceeded

Table 3—Canopy cover of transplanted, seeded, and total vegetation, September 1998.^a

	Canopy cover		
	Transplanted	Seeded	Total
	----- Percent -----		
Soil amendment			
None	5.0(1.2)	5.1(2.5)a	9.2(2.8)a
Organics	6.0(1.7)	6.8(1.5)a	12.4(2.6)ab
Organics/compost	7.4(2.0)	11.8(2.6)b	18.1(3.6)b
Mulch treatment			
None	5.7(1.2)	6.9(1.7)	11.8(2.3)
Mulched	6.6(1.5)	8.9(2.1)	14.6(2.7)
Planting treatment			
None	0(0)a	4.8(1.1)a	4.5(1.1)a
Planted	12.3(1.3)b	11.0(2.3)b	21.9(2.7)b
Scarification treatment			
None	0(0)	<0.1(0)	<0.1(0)
Scarified	0(0)	1.2(0.6)	0.8(0.5)
Total	6.2(0)	7.9(1.3)	13.2(1.7)

^aMeans with different letters are significantly different ($\alpha = 0.05$).

20% on planted plots but was less than 5% on plots that were not planted. The plots amended with organic matter and compost had twice the mean cover of plots that were not amended. Plots that were mulched and scarified also had more cover than plots that were not treated, but differences were not statistically significant.

Discussion and Conclusions

Overall, the restoration techniques we employed were highly effective. Virtually all of the transplants survived the procedure. Most were growing vigorously and many were flowering three years after planting. Soil amendments (organic matter, soil inoculum and composted sewage sludge) contributed greatly to the vigor of transplant growth. The surface mulch had no clear effect, either positive or negative.

Although seedling density varied greatly between campsites, the mean seedling density of perennial species on seeded plots was 267 seedlings/m² three years after seeding. Seedlings were growing vigorously. Mean seedling biomass increased more than ten fold during the second growing season, and mean seedling height increased 68% during the third growing season. Scarification, seeding and the organic and compost soil amendment were all effective in increasing seedling density. Scarification and seeding had more effect on seedling establishment, while the soil amendment had more effect on seedling survival during the hot, dry summer of 1998. The organics and compost treatment was most effective in enhancing seedling growth. This effect of soil amendments only became apparent in the second growing season. The surface mulch and organics treatments had less pronounced positive effects on seedling growth, which only became apparent in the third growing season.

The high level of seedling establishment and survival on all seeded plots during the first two growing seasons and the relative ineffectiveness of the mulch treatment were two

surprising results. Both results might be explained by the unusual climatic conditions that persisted over the first three summers of fieldwork. In all three years, late snow combined with frequent summer rainfall meant that soil moisture levels were probably relatively high. With abundant soil moisture, seedling germination, establishment and survival might have been unusually high, even without some of the microclimatic amelioration that a surface mulch can provide.

The third growing season was hot and dry, and seedling survival declined dramatically. Seedling mortality was much lower where soils were amended with organic matter and compost, perhaps a result of increased soil water-holding capacity and seedlings with better developed root systems. This suggests that supplemental watering may be critical to effective restoration during years with hot, dry summer weather. Supplemental watering may be needed for several years, particularly where soils have not been amended with organic matter.

It is quite possible that much of our success with seeding was the result of the supplemental watering done when seedlings were germinating during the initial growing season following seeding. The unusually high intensity of scarification we employed may also partially account for our success.

Projected recovery rates vary greatly between treatments. Mean vegetation cover on undisturbed stands close to campsites is about 55% (Cole 1982). Plots with the organic and compost amendment that were scarified, planted and mulched had a mean cover of 35% in 1998. This amounts to more than 60% recovery in just three years. Projecting past recovery rates into the future, plots receiving this most beneficial treatment would experience complete recovery of cover in about five years. On planted plots without soil amendments, recovery would require about 10 years. On plots that are scarified but neither amended nor planted, recovery would require more than 100 years. Without scarification, recovery would take even longer.

Although more than 60% recovery of plant cover in three years seems successful, composition has not recovered as rapidly as cover. On restored sites, graminoids constitute more than 50% of the vegetation cover compared to about 25% on undisturbed sites (Cole 1982). On undisturbed sites, the two low shrubs, *Vaccinium scoparium* and *Phyllodoce empetrifomis*, account for 28% and 11% of the vegetation cover, respectively. On restored sites, they account for 7% and 4% of the cover, respectively. Compositional recovery will require many decades unless transplanting, particularly of shrubs, is done at densities that mimic undisturbed conditions.

We recommend that closed sites remain signed and roped off at least until vegetation cover on restored sites approximates pre-disturbance conditions. Even with effective restoration techniques, it will likely require hundreds of years to eliminate the undesirable and unnecessary campsite impacts in the Lakes Basin and confine impacts to the levels and places deemed acceptable. This suggests the importance of avoiding damage in the first place, by implementing effective management programs wherever regular recreation use occurs.

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References

- Cole, D. N. 1981. Vegetational changes associated with recreational use and fire suppression in the Eagle Cap Wilderness, Oregon: some management implications. *Biological Conservation*. 20: 247-270.
- Cole, D. N. 1982. Wilderness campsite impacts: effect of amount of use. Research Paper INT-284. Ogden, UT: U. S. Department of Agriculture, Forest Service, Intermountain Research Station. 34 p.
- Cole, D. N. 1993. Campsites in three western wildernesses: proliferation and changes in condition over 12 to 16 years. Research Paper. INT-463. Ogden, UT: U. S. Department of Agriculture, Forest Service, Intermountain Research Station. 15 p.
- Cole, D. N. and T. Hall. 1992. Trends in campsite condition: Eagle Cap Wilderness, Bob Marshall Wilderness, and Grand Canyon National Park. Research Paper INT-453. Ogden, UT: U. S. Department of Agriculture, Forest Service, Intermountain Research Station. 40 p.
- Lester, W. J. 1989. Revegetation efforts at North Cascades National Park Service Complex. In: Hughes, H. G.; Bonnicksen, T. M. (eds.). *Restoration '89: the new management challenge*. Proc. First Annual Meeting of the Society for Ecological Restoration; 1989 January 16-20; Oakland, CA. Madison WI: Society for Ecological Restoration: 261-270.
- Moritsch, B. J. and P. S. Muir. 1993. Subalpine revegetation in Yosemite National Park, California: changes in vegetation after three years. *Natural Areas Journal*. 13:155-163.

Restoration of Multiple-Rut Trails in the Tuolumne Meadows of Yosemite National Park

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Abstract—This study presents the techniques used in a restoration project in Tuolumne Meadows on the old Glen Aulin trail in Yosemite National Park from 1990 to 1994 and the results of follow-up monitoring in the summer of 1998. The project restored the natural hydrology and soils to a 4,200-foot section of abandoned trail which had two to six one-foot deep ruts. The project utilized hundreds of volunteer work hours and showed that restoration of subalpine meadows is possible.

Yosemite National Park has more than 800 miles of trails that guide people through its 1,200 square miles of Sierra Nevada wilderness. The park receives four million visitors per year and is mandated to preserve its natural and cultural resources while providing for public enjoyment. The division of Resources Management is charged with evaluating past management decisions and mitigating actions that adversely affect the resource.

Problem Statement

Yosemite's trails have protected much of the wilderness, while still allowing visitor access, because most visitors stay on designated trails (Chapman 1993). Unfortunately, some trails have caused significant local damage especially in meadows. This problem is most acute on the trails between Tuolumne Meadows and the five High Sierra Camps (HSC). These trails are used by at least 5,000 people and 600 head of stock, which take supplies to the High Sierra Camps each summer. The combination of stock and human use has created many long sections of deeply rutted, multiple-tread trails.

When impacts to wilderness are considered, trail damage is often overlooked. Problems such as soil erosion, trail widening and multiple treads result in significant amounts

of vegetation and soil loss in wilderness areas (Scott 1998). Yosemite's resource managers have made mitigating trail impacts in subalpine meadows a high priority.

The original trail to the Glen Aulin HSC was located west of Delaney Creek in Tuolumne Meadows at an elevation of 8,600 feet. The trail cut directly across Tuolumne Meadows from Soda Springs to where the Tuolumne River starts dropping down toward the Tuolumne River Canyon. This 4,200-foot segment of trail developed between two and six ruts, some of which were a foot deep. This cumulatively denuded nearly a half acre (0.2 hectares) of subalpine meadow plant community. By unnaturally channeling water and therefore drying out areas, it negatively impacted an additional one acre (0.4 hectares) adjacent to the trail.

In 1960, this trail was rerouted into the trees to prevent further damage to the meadow. For the next 30 years, the deeply rutted, multiple-tread trail still received sporadic use and channeled water. In 1990, the multiple ruts through the meadow were still clearly visible. This paper describes the restoration process, which began in the summer of 1991, and included the efforts of NPS employees and hundreds of volunteers. The paper then explores the results of a 1998 study evaluating restoration success and species composition.

Restoration Techniques

In the summer of 1991, realizing that the trail was not naturally restoring itself, the Yosemite Ecological Restoration staff started investigating ways to fund the restoration of the old Glen Aulin Trail. The Yosemite Fund had already supported restoration projects in Yosemite through annual grants since 1987. The restoration of multiple-rut trails fit nicely into three general criteria for projects that the restoration staff tried to tackle in the early 1990s: 1) It was negatively impacting the natural resource; 2) It was an eye sore; and 3) It was ideal work for youth and voluntary labor.

This trail was chosen, from among several other rutted trails, because the 30 year-old reroute was firmly established and removed use from the area in need of restoration.

Objective: to improve the microenvironment of the old trail area by restoring the natural hydrology and soils to the impacted areas to the extent that the original meadow species would reestablish themselves.

Restoration staff measured the linear feet of trail ruts and used this measurement to estimate the cubic feet of fill needed to bring the ruts back up to grade. It was determined that fill could be obtained from nearby ephemeral drainages.

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Propagated plants were needed, so staff collected seed from reed grass (*Calamagrostis brewerii* [CABR]) and oat grass (*Danthonia intermedia* [DAIN]) from adjacent areas. Sedges (*Carex fillifolia* [CAFI] var. *erostrata*) and rushes (*Juncus parryi* [JUPA]) were collected and divided into plugs for replanting the following year. An estimate was made on the number of person hours it would take to restore the entire 4,200-foot section of trail. It was clear that it would take more than one summer to complete the project.

Realizing that thousands of hours of physical labor in this beautiful area would be needed, the Restoration Staff teamed up with several groups that often provide volunteers to the National Park Service. The Student Conservation Association (SCA) sends six-person high school crews to national parks for four weeks each summer. One SCA crew worked on the Glen Aulin trail in 1992, 1993, and 1994. Because the same leader returned every year, the quality of their work was very high, and they needed minimal guidance. Youth Conservation Corp (YCC) places groups of 12 high school age individuals who are paid minimum wage in the national parks for eight weeks each summer. The Sierra Club (SC) offers work trips where adults pay to come and work in parks for about one week. Finally, Yosemite Association (YA) places groups of 15 adults in the park for one week. Through a partnership between the National Park Service, Yosemite Association, Yosemite Institute and Yosemite Concession Services, the volunteers are provided campsites and meals for a week in exchange for labor. These groups accomplish a varied amount of work based on the proximity of the worksite to their campsite and their degree of acclimatization to the subalpine environment.

1992 Restoration (Segment One)

Work began in a dry meadow section where the old trail departed from the present trail. A YCC crew began by removing old rock check dams that were built with the intention of reducing erosion. Next, the YCC crew salvaged the topsoil and the few scattered plants that had established themselves in the rutted trail tread. An SCA crew scarified the bottom of the ruts to loosen up the soil. Finally, fill material was added to the ruts to bring them up to the level of the surrounding meadow.

The fill material was collected from a nearby ephemeral drainage. These borrow pits were trenches dug wide enough for a string of mules to walk into. Two workers standing on

either side of the trench shoveled fill material into dirt boxes carried by the mules. The NPS packers used these mules to move 64 cubic yards (376 mule loads) of fill in 1992. As a result, a 1,350-foot section of trail was brought back up to grade, and a one to two-inch layer of the salvaged meadow topsoil was spread on top.

In late September, a restoration staff member and two wilderness rangers began replanting 2,200 propagules, 150 meadow pieces and 25 lodgepole pine seedlings. A gas-powered water pump and 1,000 feet of fire hose were used to water the propagules during transplanting. Water helped to get the plants firmly into the ground since damp soil tamps down much tighter. Plants that workers forgot to tamp were frost-heaved out of the ground by 1993. Planting was done in late September to avoid having to re-water (plants experience less water stress when daytime temperatures are cooler and they are nearing winter dormancy) (Rocheffort, 1990). Propagules are particularly susceptible to water stress. This method has been found to be successful in other areas (Olympic National Park, Scott 1998).

Labor for the 1992 section (table 1):

- a) Six-person Student Conservation Association high school work group worked for three weeks bringing 850 feet of trail back up to grade (715 hours).
- b) Twelve-person Youth Conservation Crew work group worked for one week bringing 400 feet of trail back up to grade (400 hours).
- c) Three NPS employees replanted 2,200 nursery propagules (150 hours).
- d) One packer and five mules for one week (40 hours).

1993 Restoration (Segment Two)

In 1993, a total of 1,050 feet of trail through wet meadow was restored. Of this total, 330 feet was brought to grade in 1992. Work progressed slowly because this section of old trail had five distinct ruts, each over a foot deep. Although some vegetation was growing in the ruts, it was primarily pioneer species not found in the undisturbed meadow.

In an area one foot wider than the distance across all the ruts, all living plant material was dug up in pieces averaging 12 inches on a side and 10 inches deep. This left a trench averaging 12 feet wide with purposely undulating sides to reduce unnatural straight lines. Where topsoil existed, it was removed and stockpiled next to the trail. The trench was

Table 1—Finances for the Glen Aulin Trail restoration project.

Finances	1991	1992	1993	1994	Total
Restoration staff	1,000	3,107	3,518	856	8,481
Student Conservation Association		8,000	10,760	12,395	31,155
Youth Conservation Corps		4,500			4,500
Supervision of Student Conservation Association and YA			2,400	1,800	4,200
Packing (mules)		Free	Free	248	248
Tool/vehicles		2,230	500	2,373	5,103
Nursery plants		4,000			4,000
Totals	\$1,000	\$21,837	\$17,178	\$17,672	\$57,687

then filled with imported fill soil collected from ephemeral drainages in the same method as 1992. It was then capped with the stockpiled topsoil. The meadow sod pieces were then planted randomly to imitate the natural mosaic in the adjacent plant community. The large pieces of salvaged sod were replanted in late August but did not suffer as much water stress due to their size and the higher natural soil moisture of this area.

Labor for the 1993 section (table 1):

- a) A six-person Student Conservation association group worked for four weeks restoring 750 feet of trail. A 330-foot section of this trail was revegetated in an area already brought up to grade by the YCC crew in 1992 (1,134 hours).
- b) Twelve Sierra Club volunteers restored 100 feet of trail in six days (480 hours).
- c) Sixteen Yosemite Association volunteers restored 80 feet of trail in four days (480 hours).
- d) Six NPS Restoration staff members restored 120 feet in three days (180 hours).

1994 Restoration (Segment Three)

The final 2,000 feet of trail led out of the dry meadow and into the meadow-forest ecotone. The techniques used in 1993 were repeated. When transplanting sod, buckets of water were hand-carried from Delaney Creek. On one sloped section, old down trees were partially buried in the old trail to hold the soil in place. Since the entire old trail was brought to exact grade, the buried tree trunks were to slow surface runoff until the vegetation grew back. Some sections of trail needed very little work, while other areas needed large amounts of fill material.

The 1992 and 1993 borrow pits had filled in as a result of ephemeral drainages that drop their sediment load during spring runoff. The 1994 borrow pits were caved in and contoured by work leaders at the end of the summer so that they appeared to be natural low spots. These areas have been naturally filled in since the 1994 season.

Labor for the 1994 section (table 1):

- a) Six-person Student Conservation Association crew restored 1,380 feet in four weeks (1,134 hours).
- b) Twelve Yosemite Association volunteers restored 250 feet in four days (408 hours).
- c) Seventeen Yosemite Association volunteers spent one day finishing 370 feet of trail (544 hours).

1996, 1997 Touch Up

Many of the YA volunteers return to Tuolumne and volunteer almost every summer, and they often spend one day each year working on the Old Glen Aulin Trail. In 1997, crews restored a section of old trail adjacent to Delaney Creek because the January 1997 100-year flood washed away some of the fill soil. Seeing the long-term success of projects they worked on keeps volunteers coming back.

The 5,180 hours of volunteer time were essential to the completion of the project. NPS staff worked with many of these volunteers and were able to instill stronger wilderness ethics and graphically show how much work is required to restore subalpine meadow areas.

1998 5-Year Evaluation

In the summer of 1998, Ecological Restoration staff evaluated the restoration success on the Old Glen Aulin Trail. This study was exploratory and descriptive in nature. Its objectives were to evaluate percent cover and species richness within the restored multiple trail ruts and in the adjacent undisturbed meadow. The study highlights factors leading to the success and failure of plant reestablishment in the restored trail area.

Methods

This study compared 40 pairs of adjacent quadrats on the Old Glen Aulin Trail. Within each pair, one quadrat was in the restored area and one was adjacent to the restored area. The quadrats outside the restored area may not represent "pristine meadow" but do represent areas that have been relatively free of compaction and wear by stock and hikers. Of the 40 pairs of quadrats sampled, 15 were in wet meadow, 13 were in dry meadow, and 12 were in meadow-forest ecotone. Meadow-forest ecotone has an understory of predominantly dry meadow species and a sparse canopy created by lodgepole pines (*Pinus contorta* ssp. *murrayana*). The study utilized quadrats that were one meter by 1/10 meter in order to maximize width and sample across the multiple-rut trail. Each quadrat was divided into 10 even sections in order to estimate percent cover of each species most accurately. Two Ecological Restoration staff visually estimated the percent cover of every species. The quadrats were somewhat randomly placed but always between 30 and 50 meters apart. After each quadrat in the old trail (restored area) was assessed, the quadrat was flipped twice, end on end, to sample the adjacent undisturbed meadow, two meters away from the old trail. The direction of placement was alternated, beginning with the right of the old trail and alternating to the left of old trail with every count.

Results

Five years after restoration, the entire restored area has maintained the grade of the natural meadow. The overall mean percent cover in the disturbed area was 43.2 %, while the mean percent cover in the undisturbed area was 55.9% (fig. 1). Seventy species were observed in the undisturbed meadow quadrats, while 64 species were counted inside the restored trail. This is excellent recovery for a high-elevation site but it is more informative to look at what happened in the three different plant communities.

Wet Meadow—The wet meadow areas had numerous deep ruts because people and stock tried to walk on higher, drier areas next to the trail, thus creating a new rut. Unlike many subalpine areas in Yosemite, this wet meadow actually has a true topsoil, meadow loam, which is easily compacted and highly erodible. The wet meadow species tend to have low resistance to trampling, but a high degree of resilience. Some pioneer species like *Juncus balticus* (JUBA) did become established in the ruts, but the natural meadow species could not establish due to a lack of topsoil.

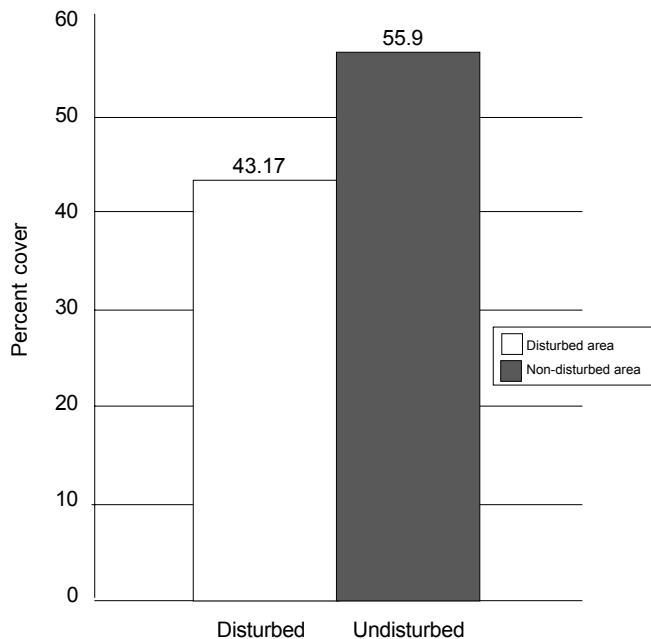


Figure 1—Comparison of overall percent cover in adjacent quadrats in the old restored trail and in the adjacent meadow.

The 15 quadrats in undisturbed areas had 68% cover (SE= 4.7) and an average species diversity of 11.1 (SE=1.1) species per quadrat in 1998. CAFI was present in 75% of the quadrats and accounted for 12% vegetative cover. CABR, *Antennaria corymbosa* (ANCO), DAIN, *Muhlenbergia filiformis* (MUFI) were also present in between 50% and 75% of the quadrats. These five codominant species accounted for 47% of the vegetation. The total species diversity across all 15 quadrats was 52 species.

The 15 disturbed quadrats had an average of 57% cover (SE=6.0) and an average species diversity of 10.9 (SE=1.3) species per quadrat in 1998. CAFI, DAIN, ANCO were present in 75% of the quadrats. MUFI and CABR were present in between 50 and 75% of the quadrats. These five species accounted for 52% of the vegetation. Total species diversity across all 15 quadrats was 43 species.

Plant reestablishment on these sites was extremely successful because the vegetative cover in the disturbed areas reached 83% of the cover levels in the undisturbed area and species diversity per quadrat was almost identical (fig. 2). The dominant species were the same in both disturbed and undisturbed quadrats. *Juncus covielli* (JUCO), *Elymus elymoides* (ELEL), *Poa cusickii* (POCU), *Ivesia lycopodioides* (IVLY) and *Gentiana newberri* (GENE) were present in the undisturbed but absent in the disturbed areas. *Juncus nevadensis* (JUNE), *Gayophytum diffusum* (GADI), *Gentianella amarella* (GEAM), *Lupinus lepidus* (LULE), and *Madia minima* (MAMI) were present in the disturbed areas but absent in the undisturbed.

The authors hypothesize that high moisture levels and high species diversity gives this plant community a high level of resilience. Since the wet meadow started with five codominant species, and 20 other species with significant cover, any type of weather year would facilitate reestablishment of at least a few of these species. The meadow loam soil

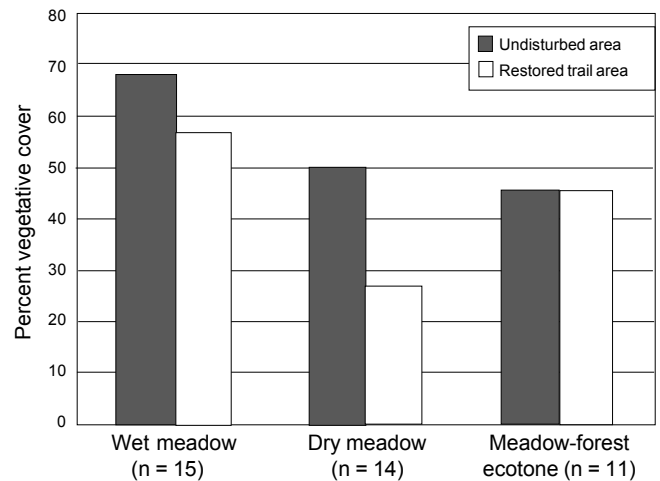


Figure 2—Comparison of percent cover by plant community.

is rich in nutrients and allows new plants to quickly establish a strong root system and therefore survive the dry end of summer.

Dry Meadow—Dry meadows have coarse, sandy, granitic soils. Because of the large particle size, the soil has a low moisture-holding capacity and a low cation exchange capacity (CEC) causing it to be a poor growing medium (Brady and Weil, 1996). The plants in dry meadows generally have an initial resistance to trampling but once destroyed are slow to reestablish. In the dry meadow segments of the restoration project there were two to four ruts, each one-foot deep. Tread compaction was not a large problem, due to the angular, sandy soil's resistance to compaction. Most soil loss occurred where the trail slope exceeded four percent. In these sections, water was channeled in the slightly lower (2-5 cm) trail tread caused by hooves and feet, and once into the tread it gained momentum and scoured the area down 10 to 20 cm more.

The 13 undisturbed dry meadow quadrats averaged 47% cover (SE=6.2) and had an average species richness of 5.3 (SE=1.1) per quadrat. CAFI was present in every quadrat and accounted for 50% of the total vegetation. ANCO, *Lewisia nevadensis* (LENE) and MUFI were each present on 33% of the 13 quadrats and accounted for 9 % of the vegetation. Total species diversity on all 13 quadrats was 23.

The 13 disturbed dry meadow quadrats averaged only 22.7% (SE=5.6) plant cover and had an average species richness of 4.3 (SE=1) per quadrat. CAFI was present in 70% of the quadrats and accounted for only 37% of the vegetation. MUFI was present on approximately 50% of the quadrats. No other species were present on more than 25% of the sites. JUPA was strongly established on two quadrats. Total species diversity was 25.

Plant reestablishment is happening slowly on these dry sites. CAFI was the single dominant species on the undisturbed quadrats and is slow to reestablish. JUPA, MUFI and ANCO are reestablishing on two or three quadrats. Species diversity was actually greater (25) in the disturbed area than in the undisturbed (22).

These sites get 14 hours of direct sunlight during the summer and have soils with low moisture holding capacity.

The existing plants spread vegetatively, but slowly. The area rarely gets sufficient afternoon thundershowers for a first-year seedling to survive the dry August and September months. This community has only one dominant species, CAFI, as opposed to having several codominant species. The authors hypothesize that the combination of high evapotranspiration stress, and having only one dominant species hinder this community's ability to reestablish itself. A codominant community has several species that can proliferate in a wide range of summer moisture conditions. Because the dry meadow community is largely made up of CAFI, percent vegetative cover did not expand during summers with moisture conditions not favorable to CAFI growth.

Meadow-Forest Ecotone—These areas have between 10% and 30% lodgepole pine canopy cover. This is important due to less transpirative loss from herbaceous plants and because pine litter inhibits herbaceous vegetation growth. The trail originally disturbed a five-foot wide swath, which did not develop into distinct ruts, but wore down the entire swath by 10 or more centimeters. Since the old trail had been designed to accommodate stock, it wound between trees but generally stayed outside the canopy.

The 12 undisturbed quadrats averaged 49.6% cover (SE=6) and an average of 5.6 species (SE 0.7) per quadrat. CAFI was growing on 80% of the quadrats and accounted for 30% of the total vegetation. CABR accounted for 23 % of the total vegetation even though it occurred only in 3 of the 12 quadrats. ANCO, MUFI and *Agrostis variabilis* (AGVA) were sporadically present. There were a total of 33 species present on all quadrats.

The 12 disturbed meadow-forest ecotone quadrats averaged 48.3% cover (SE=6.0) and had an average species diversity of 7.5 (SE 0.7) per quadrat. CAFI was found on 66% of those sites but accounted for only 15% of the cover. AGVA and GADI were present on more than 50% of the quadrats and made up less than 10% of the cover. ANCO, LULE, PHAL, ACLE and DAIN were present on 25% to 50% of the quadrat and made up 36% of the vegetation. In summation, there were eight common species, but none of them was dominant.

Neither percent cover nor species diversity was statistically different in the undisturbed versus disturbed quadrats in the meadow-forest ecotone (fig. 2). The authors believe that although this may indicate 100% recovery, there were slight problems with the sampling method. The disturbed quadrats were almost always outside of the canopy, whereas their counterparts often ended up under the canopy. Under the canopy the pine needle layer was thicker which inhibits herbaceous plant establishment. CAFI, the one plant that was often present under the canopy may have been present prior to tree establishment. As a result, the disturbed trail area will probably have higher percent cover and species diversity than the undisturbed sites.

Project Summary

Yosemite's Ecological Restoration Program restored a 4,200-foot section of multiple rutted trail by focusing on returning the topography and soils to natural conditions. This was accomplished by importing 100 cubic meters of fill material from nearby ephemeral drainages. The fill was used to bring the ruts to the level of the surrounding meadow and capped with salvaged topsoil. Although some seeds and propagules were used during the first year, transplants and natural regeneration were relied on in later years. These methods were equally successful but less costly. Over 100 volunteers helped replant over three linear miles of transplants salvaged from the islands between the ruts. Utilizing volunteers educates them about NPS preservation efforts and develops a connection between the volunteer and the resource. This trail restoration project cost about \$14 per linear foot of trail restored.

After only three years most visitors could not tell there had been a trail in this area. After five years, the percent cover in the restored area was at 77% of that in the adjacent undisturbed meadow. Both the disturbed and undisturbed quadrats had a wide variety of native species. While all trail segments are recovering, plants are reestablishing faster in the wet meadow areas than in either the dry meadow areas or the meadow-forest ecotone. We were fortunate to have a string of above average moisture years.

In this subalpine meadow environment, a 30-year trail closure failed to facilitate plant reestablishment. This four-year project of restoring topography, surface hydrology and soils both educated volunteers and resulted in significant gains in plant establishment in just five years.

References

- Brady, N. C.; Weil, R. R. 1996. The Nature and Properties of Soils: Eleventh Edition. Upper Saddle River, NJ: Prentice-Hall Inc.
- Chapman, D. B. 1993. The Role of Trail Maintenance and Construction in Wilderness Restoration: John Muir Wilderness Case Studies. Conference Handbook, A Renaissance in Wilderness Stewardship, Tucson, Arizona, U.S. Department of Agriculture, Forest Service.
- Rocheftort, R. M. 1990. Restoration Handbook, Mount Rainer National Park, park document. 54 p.
- Scott, R. 1998. Wilderness Management and Restoration in High Use Areas of Olympic National Park, Washington, USA. In: Watson, Alan E.; Aplet, Greg; Hendee, John C., comps. Personal, Societal and Ecological Values of Wilderness: Sixth World Wilderness Congress Proceedings on Research, Management, and Allocation, Vol. I; 1997 October; Bangalore, India. Proceedings RMRS-P-4. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 144-147.

The Influence of Wilderness Restoration Programs on Visitor Experience and Visitor Opinions of Managers

Joseph P. Flood
Leo H. McAvoy

Abstract—Wilderness campsites heavily damaged by recreational use pose a significant management challenge that threatens the integrity of the wilderness resource and the quality of the visitors' experience. This study, conducted in the Mission Mountains Wilderness of northwestern Montana, surveyed 293 visitors to determine what influence heavily damaged campsites and site restoration activities have on the quality of the visitors' experience, and to assess visitor opinions of the managers who implement or do not implement restoration. Visitors noticed campsite damage that reduced the quality of their experience as well as their opinions of managers. However, the quality of the visitors' experience and their opinions of managers improved significantly after they observed restoration activities.

The rationale for wilderness recreation management is to protect natural conditions and to provide opportunities for solitude or primitive and unconfined recreation experiences (Hendee and others 1990). When people visit a wilderness area today, they commonly see damage to campsites caused by recreation use and evidence of management actions to address these impacts. For example, when vegetation is severely trampled at a campsite and the soil begins to erode, it can influence the quality of the wilderness visitor's experience. A standard management action is to restore vegetation in heavily damaged campsites in wilderness. Recreational impacts at campsites in wilderness and how they influence the visitor experience is a concern to managers responsible for maintaining natural conditions.

Little is known however, about the perceptions of visitors regarding restoration, the appropriate levels of restoration or the role that managing agencies should play. The purpose of this study is to, (1) use visitor surveys to determine what influence site restoration programs have on the experience of wilderness users, and (2) to assess visitor opinions of the management agencies who implement site restoration.

Although many management actions are implemented to address social and ecological problems in wilderness,

management solutions to these problems and how they influence the quality of the visitors' experience have not been consistent or well documented. Reasons include the size of the wilderness preservation system and the fact that the federal agencies responsible for these areas often use different management approaches.

There are approximately one hundred and four million acres (42,105,263 hectares) of congressionally designated wilderness in the United States. The four federal agencies responsible for managing wilderness are the U.S.D.A. Forest Service, U.S.D.I. National Park Service, U.S.D.I. Bureau of Land Management and the U.S.D.I. U.S. Fish and Wildlife Service. The following excerpt from the 1964 Wilderness Act illustrates the challenges faced by the agency managers "these lands shall be administered for the use and enjoyment of the American people in such a manner as will leave them unimpaired for future use and enjoyment as wilderness."

When areas were first being designated as wilderness, managers believed that the best way to build a political constituency for wilderness was to increase the number of people who visited these areas. By the late 1960s, backpacking gained popularity, and many wilderness campsites were beginning to be severely damaged by an increasing number of visitors. At the same time, different types of wilderness visitors (horse users and hikers are one example) were beginning to experience conflicts. Managers began hearing more complaints from hikers about the damage to trails and campsites caused by horses, mules and other hikers. The federal agencies responsible for managing wilderness have typically reacted to these changes, such as damage to vegetation and crowding, rather than developing a proactive set of solutions that would prevent unacceptable levels of damage at campsites (Flood 1993).

Throughout the 1970s, managers struggled with the intent of the Wilderness Act and cautiously began to develop methods to better understand people's motives for entering wilderness. Because motives are often different for different visitors, managers began to realize they would have to implement measures to protect the resource from further impacts caused by the increasing numbers of people visiting wilderness. The need for comprehensive planning frameworks to improve wilderness management was apparent. During the early 1980s, the Limits of Acceptable Change (LAC) planning framework was developed to address changes in wilderness conditions and to better involve the public in the wilderness management planning process. The LAC process begins with the premise that change is inevitable, then moves on to determine how change will be inventoried, assessed and managed through indicators and standards (Stankey and others 1985; Stokes 1990).

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As a result of people recreating in wilderness over many years, resource impacts often exceed the standards set for a specific area. Examples include the number of campsites allowed in a lake basin, travel corridor or around a lake and, the degree of damage to a particular campsite. Generally, management plans provide a list of potential management actions to address exceeded standards. One option is to implement restoration. Because a wilderness manager's goal is to preserve natural conditions in wilderness according to the Wilderness Act and management plans, some managers have responded by implementing restoration programs.

What motivates visitors to spend time in wilderness and how they evaluate onsite conditions is a growing concern for managers and researchers (Aldo Leopold Wilderness Research Institute 1997). The use of theories to identify visitor intentions, how these intentions lead to benefits sought during the visit, and how onsite conditions influence the quality of the visitors' experience are not well understood. Over the past 20 years, wilderness research has used theories and models from the fields of psychology and sociology that measure human behavior and reshaped them to fit a wilderness paradigm. Previous studies (Cole 1996; Hall and Shelby 1993; Peterson 1974) have used the expectancy model to explore the role of expectations and actual perceived conditions in the satisfaction of wilderness experiences. The expectancy theory is used to signify how much visitor expectations influence the wilderness experience, especially with regard to visitors' perception of onsite conditions.

Being able to predict public visitor expectations of onsite conditions and their support for management practices could help resource managers develop successful strategies to maintain wilderness quality. Understanding the motivation and expectation of visitors is key to determining whether onsite conditions match desired outcomes. Results from recent studies indicate that wilderness user groups generally support management policies to regulate site improvements (Cole and others 1997; Shindler and Shelby 1993).

How visitors react to campsites affected by recreational use and how these campsites may influence their experience are not well understood. A need exists to identify and understand how onsite conditions in wilderness influence the visitors' experience and their opinions of managers. In three Western wilderness areas, Lucas (1987) found that visitors were more disturbed by environmental damage than by seeing other people. Research findings suggest that visitors who are sensitive to environmental damage either readjust their expectations to conform to the changing nature of the experience or are displaced to areas with fewer people and fewer damaged areas (Anderson 1980). Two types of visitors were identified: those who are displaced from an area and never return and those who return but use the resource differently. These individuals may go to other, less affected areas or make a readjustment of their expectations. They are motivated to reach their destination, even if their standards for impact and crowding are exceeded (Anderson and Brown 1984).

Previous experience in wilderness influences how people sort, evaluate and store information about a wilderness experience. In a study by Watson and Cronn (1994), the most experienced day-use visitors (those who first visited the area more than 10 years ago) reported significantly more resource impact problems than the less experienced groups.

This information suggests visitors can provide valuable information about wilderness conditions and visitor perceptions of management actions.

Although the primary goals of wilderness management are to maintain the free operation of natural processes and to preserve qualities such as wildness and solitude (Martin and others 1989), managers are also faced with the difficult task of administering areas "for the use and enjoyment of the American people." The difficulty lies in the fact that recreational use inevitably results in some changes to ecological and social conditions. Although the majority of wilderness areas are still relatively pristine, disturbances to campsites are highly concentrated at popular destinations and result in serious problems of visual impact. Thus, while some damaged campsites may not threaten the ecological integrity of an entire area, extensive soil erosion may produce serious localized resource damage and thus has the potential to influence the quality of visitors' experience (Cole 1993).

When restoration is the selected management action to restore damaged campsites, a series of trade-offs confront managers and wilderness visitors. While some campsites are closed for restoration, visitors may temporarily lose some freedom of choice, but the restored conditions may ultimately improve visitor experiences. The results from several studies suggest that visitors and managers evaluate bare ground, where vegetation has been destroyed, as the least acceptable impact at a wilderness campsite (Lucas 1980; Martin and others 1989; Shelby and Harris 1985). Managers and visitors are also more likely to identify campsite impacts as more severe and unacceptable the deeper they travel into the wilderness. However, more recent findings suggest that the number of campsites, rather than the amount of bare ground at campsites, should be considered when choosing indicators for evaluating campsite conditions (Cole 1993; Marion and others 1993).

An increasing number of studies are being conducted to better understand the attitudes of wilderness visitors toward wilderness management actions. In a study of six areas located in the Alpine Lakes, Mount Jefferson and Three Sisters Wilderness Areas in Washington and Oregon, researchers were surprised to find a high number of visitors who noticed campsite impacts that detracted from their experience. The research results also found a high level of support for current management actions and programs (Cole and others 1997). According to McCool and Lime (1988), "understanding visitor attitudes toward management actions and their benefits, consequences, costs, and values can help managers more effectively provide quality recreational experiences." Visitor attitudes are particularly important to managers where there is conflict among users or feelings of dissatisfaction about existing conditions (Lucas 1987).

Wilderness managers concerned about the steady deterioration of the wilderness resource often provide information and education about wilderness to current and future wilderness visitors. Wilderness education can be one method to influence visitor behavior. When people are provided information about "what" to expect prior to their visit, they are given an opportunity to make better decisions based on better information about biophysical and social conditions (Roggenbuck 1992; Schreyer and Roggenbuck 1978; Watson and Niccolucci 1992).

It would be helpful if managers knew how visitors might respond to different levels of damage at campsites, especially heavily damaged campsites, and how visitors might respond to restoration. It is also very important for managers to understand how different management actions can influence the quality of wilderness visitors' experience. Knowing this information will assist managers in providing quality recreation opportunities for wilderness visitors.

This study investigates how campsite restoration programs and heavily damaged campsites influence the quality of wilderness visitors' experience and their opinions of managers. The research questions are: 1) how do restoration activities and heavily damaged campsites influence the quality of the visitor experience; and 2) how do restoration activities, or lack of restoration activities to address damaged campsites, influence the visitors' opinions of managers?

Method

Description of Study Area

The study site for the research was located in northwestern Montana in the Mission Mountains Wilderness (MMW). The 73,877-acre (29,910 hectare) MMW is part of the National Wilderness Preservation System managed by the U.S. Forest Service. The MMW is located in Region One of the Flathead National Forest on the Swan Lake Ranger District.

Participants

Visitors to the Mission Mountains Wilderness (MMW) in northwestern Montana were recruited for participation in the study during the summer-use season of 1998. Visitors 18 years and older were asked to participate as they exited the wilderness. Visitors were both day-use and overnight visitors. Of those who participated in the study, 70 percent were day-use visitors, while approximately 30 percent were overnight visitors.

The majority (95%) of the visitors to the MMW came from nearby towns and cities (less than 100 miles). They primarily came from Missoula, Kalispell, Big Fork, Polson and other surrounding small towns and, 5% were out-of-state visitors. The average group size was 3 people per group. Many of the groups were family members or close relatives.

Materials

Data were collected using an exit survey developed from pre-existing visitor use surveys, conducted by the Forest Service in the Snow Lakes and Desolation Wilderness Areas. These surveys were used as models for the exit survey used in the Mission Mountains Wilderness. A pilot test of the instrument was conducted at the Glacier Lake trailhead (10 surveys were completed by wilderness visitors) to ensure reliability and validity.

Procedure

Visitors to the MMW were asked to fill out exit surveys during the 1998 summer-use period, from June 15 to September 15. The four sampling locations were areas where

active restoration was occurring. The selection of sampling locations for the four trailheads was based on overall use estimates compiled by the Forest Service for the past 10 years. The number of visitors who filled out exit surveys was proportional to the recorded use estimates. The initial targeted number of survey respondents was 300, which corresponds to approximately 10 percent of the estimated 3,000 visitors for the 1998 season. The number of sampling days assigned to each of the four trailhead contact points is proportional to the use estimates for the four trailheads. Both the data collection sites and days that the data were collected were randomly chosen. Visitors were contacted at the four trailheads on the sampling days and asked to complete the 10-minute exit survey onsite. Those who agreed to complete the survey were briefed about the purpose of the study. A total of 293 exit surveys were completed by MMW visitors. Six people contacted during the study period refused to fill out the survey.

Results

The results of the surveys indicated that a large percentage of visitors to the Mission Mountains Wilderness do notice heavily impacted campsites, which diminishes their experience. Conversely, visitors who observed restoration activities during their visit felt it had a positive effect on their experience and on their opinions of managers. Table 1 shows the number of visitor surveys completed at each trailhead location and the amount of time visitors spent in the wilderness during their visit.

Visitors in the study were asked to list the three most important reasons for taking this trip into the MMW. In the analysis, 87 related responses were grouped into four major reasons. The number one reason visitors listed for visiting the MMW was to engage in recreational activities. These activities included fishing, hiking, camping, using stock animals, rafting, huckleberry picking and swimming. The second was to experience solitude and spiritual renewal. These activities included freeing themselves from society and crowds, getting life into a better perspective, rest and relaxation, achieving a sense of solitude and renewing one's spiritual values. The third reason was nature appreciation. These activities included experiencing the natural scenic beauty, observing wildlife, better understanding the ecology, exploring, communing with nature and experiencing clean rivers, lakes and air. The fourth reason was to spend time with family and friends. These activities included being together with family and friends, introducing their children and grandchildren to wilderness, companionship and sharing Montana with friends and family.

The remaining survey questions asked visitors who observed restoration to rate how this may influence the quality of their future visits, whether it improved or detracted from their experience and how they felt the quality of their experience was influenced by observing heavily impacted sites versus restoration; they were also asked to rate their opinion of managers who implement restoration compared to managers who do little or nothing to address heavily impacted areas in wilderness.

The results in table 2 indicate that visitors felt restored campsites will increase the quality of their future visits. A total of 218 (72%) of the visitors indicated that the restored

Table 1—Visitor survey locations.

Trailhead location	Day-use visitors		Overnight visitors	
	n	hours/visit	n	days/visit
Glacier Lake	130	6	41	3
Cold Lake	47	7	16	3
Cedar & Piper Lakes	8	8	22	4
Crystal Lake	8	7	21	3
Total	193	-	100	-
Mean	-	7	-	3.25

Table 2—How will restored campsites influence the quality of your future visits?

Influence quality	n	M	SD	Median
	292	7	1.9	7

Ranked on a scale of 1 (greatly reduced), 5 (neutral), 9 (strongly increased).

campsites will increase or greatly increase the quality of their future visits.

The results in table 3 indicate that restoration did not detract from, but improved the quality of visitors' wilderness experience. Because we were interested in determining if restoration detracted from visitors' experience, a "neutral" or "had no effect" rating was not considered to detract from the experience. The results indicated that restoration activities had "not significantly detracted" from the quality of visitors' experience. One hundred and eighty-two visitors (63%) responded "strongly agree" or "agree" when asked if observing restoration improved the quality of their experience. Two hundred and sixty-five visitors (91%) responded strongly disagree, disagree or were neutral when asked if restoration detracted from the quality of their experience.

The results in table 4 indicate that visitors who intended to visit a natural setting and found damaged campsites felt this reduced or greatly reduced the quality of their experience. A majority of the visitors (71%) indicated that heavily impacted areas reduced or greatly reduced the quality of

Table 3—Influence of restoration on the quality of the visitor's experience.

Quality of visitor experience	n	M	SD	Median
Improved quality	291	3.8	0.98	4
Detracted quality	292	2.2	0.99	2

Ranked on a scale of 5 (strongly agree), 4 (agree), 3 (neutral), 2 (disagree), 1 (strongly disagree).

Table 4—Influence of campsite conditions on the quality of visitor experience.

Campsite conditions	n	M	SD	Median
Heavily impacted sites	292	3.1	2.1	3
Presence of restoration	292	6.4	2.2	7

Ranked on a scale of 1 (greatly reduced), 5 (neutral), 9 (strongly increased).

their experience, whereas 67% of the visitors indicated that observing restoration increased or greatly increased the quality of their experience.

Visitor responses listed in table 5 indicate that visitor opinions of the managers were less positive after observing heavily impacted areas where little or no effort was made at restoring conditions. The results also indicate that the presence of restoration activities had a positive influence on visitors' opinions of managers. Two hundred and six visitors (71%) reported that their opinions of managers was negative or extremely negative when they observed impacted areas where little or nothing had been done to restore the impacts. This is compared with 226 visitors (74%) who reported that their opinion of managers who implemented restoration was positive to extremely positive. A total of 117 (40%) visitors rated their opinions of managers who implement restoration as extremely positive.

Further analysis compared short-time and long-time visitor responses to the same sets of questions. Responses from short-time visitors (0-5 years visiting the MMW) were compared with the responses of long-time visitors (20-plus years visiting the MMW). Table 6 shows the mean response scores for the groups. Notable differences were not apparent when comparing these two groups. Although both had a positive opinion of managers who implement restoration, they also indicated that their opinion of managers was reduced when little or no restoration effort was made to address impacts to vegetation and soil caused by recreational use.

In addition to the survey questions, respondents were asked to comment on the restoration program or the management of the MMW. Qualitative measures were used in

Table 5—Influence of restoration or "lack of restoration" on the visitor's opinions of managers.

Opinions of managers	n	M	SD	Median
Little or no restoration	288	3.3	1.8	3
Presence of restoration	292	7.2	2.4	8

Ranked on a scale of 1 (extremely negative), 5 (neutral), 9 (extremely positive), 0 (don't know).

Table 6—Comparing short-time and long-time visitor responses to restoration or impacted areas.

Comparing visitor responses	Short time n = 168	Long time n = 52
Survey items	M	M
Influence on future visits	6.8 ^a	6.9 ^a
Influence of restoration on experience	6.2 ^a	6 ^a
Restoration detracts	2.1 ^b	2.2 ^b
Restoration improves	3.7 ^b	3 ^b
Opinion of managers yes restoration	7.1 ^c	7.2 ^c
Opinion of managers no restoration	3.2 ^c	3 ^c

^aResponses were ranked on a scales of 1 (greatly reduced), 5 (neutral), 9 (strongly increased).

^bRanked on a scale of 5 (strongly agree), 4 (agree), 3 (neutral), 2 (disagree), 1 (strongly disagree).

^cRanked on a scale of 1 (extremely negative), 5 (neutral), 9 (extremely positive), 0 (don't know).

the interpretation of the visitor comment section of the survey (Miles and Huberman 1994). Of the total 293 completed surveys, 108 respondents provided written comments. Of these, 58 were directed toward the restoration program. Among the 58 comments, 55 (95%) specifically supported the restoration efforts. The data analysis of these comments included a compilation of visitor comments, generating themes and categories of responses and summarizing them.

The visitor comments provided information about reactions to campsite impacts and restoration activities. Many of the comments highlighted the importance restoration plays in educating visitors about campsite impacts and the need to restore them. The most common phrase used by visitors to explain how they reacted to observing restoration was that they believed the area was "well cared for." Many respondents stated that their positive opinion of managers was the result of the managers' long-term commitment to restoration in the MMW. Many visitors indicated that when they visit other wilderness areas, they do not see impacted areas being restored or cared for at the level observed in the MMW.

Discussion

According to the Wilderness Act, one of the primary goals of wilderness management is to protect natural conditions. Historically, information and education have been two potential solutions to problems related to resource impacts in wilderness. These solutions are generally unobtrusive and hold long-term benefits for visitors and the wilderness resource. It seems that restoration, after information and education is another favored measure to achieve the desired resource conditions in wilderness, at least in the Mission Mountains Wilderness, while providing opportunities for quality visitor experiences.

Restoration is sometimes selected as a preferred management action to restore impacted areas back to their natural conditions. These research findings provide evidence that restoration can be an effective strategy to restore heavily damaged campsites. Although restoration is presumed to be the appropriate action to address damaged campsites, many managers are concerned that the long-term obtrusiveness of restoration activities may outweigh the benefits of restoring onsite conditions. The survey questions used in this study were designed to illuminate the influence restoration had on visitors' experience as well as their opinions of managers.

Because wilderness management is a newly evolving science, it is imperative that managers examine the influence of management actions on wilderness visitors' experience. Management decisions need to err on the side of wilderness and the experience of the wilderness visitor. If a manager's goal is to provide opportunities for quality wilderness experiences, restoration should play a more significant role as the number of visitors and impacts continue to grow. As positive as the results of this study are, replicated studies in other wilderness areas are needed. Additional information will assist wilderness managers who are weighing the potential costs and benefits of restoration.

Whereas many managers are faced with an increasing number of resource impacts, they are also challenged by reduced budgets. When managers face tough decisions, it can be difficult to determine the best use of available funds.

The results from this study inform wilderness managers about the benefits of selecting restoration as a preferred management action. Restoration has a high potential to restore damaged campsites. And, in the case of this study, it appears to improve the quality of visitors' experience, as well as their opinions of managers.

In the future, wilderness managers may select restoration more frequently as a potential solution to address impacts in wilderness. Even when visitors know a temporary loss of freedom will occur as a result of restoration activities, the potential for improved onsite conditions seems to be enough to convince them of the long-term benefits of restoration. Results from this study suggest that restrictions associated with restoration activities were perceived as short-term and less restrictive by visitors. Also, these visitors may be more likely to support restoration as a management approach than long-term restrictive measures, such as limits on types of use or overnight camping closures, to protect and restore resource conditions.

As we enter the next millennium, wilderness managers will continue to make tough management decisions. Many will be based on a "what you get for the dollars spent" philosophy. If the goal of research is to assess a situation and provide guidance in achieving new direction, concrete examples of how selected management actions can achieve the best benefits are needed. The results of this research indicate that restoration does influence visitors' experiences and opinions of managers in a positive way. Given these results, wilderness managers have evidence that supports restoration activities. There are strong similarities between the results from this study and previous studies by Cole and others (1997). Together, they support management decisions to address heavily damaged areas in wilderness.

For many visitors, wilderness is not just a nice place to visit. It is a place for significant contemplative experiences and has the power to enhance the quality of one's life. Restoring heavily damaged areas in wilderness does not have to be an anomaly, but an affirmation about what is right, and what good wilderness management should be.

References

- Aldo Leopold Wilderness Research Institute (1997). Strategic plan for research. USDA Forest Service, Intermountain Research Station.
- Anderson, D. H. (1980). Displacement of visitors within the Boundary Waters Canoe Area Wilderness. Unpublished Doctoral dissertation, Colorado State University, Fort Collins.
- Anderson, D. H. & Brown, P. J. (1984). The displacement process in recreation. *Journal of Leisure Research*, 16 (1), 61-73.
- Cole, D. N. (1993). Campsites in three western wildernesses: proliferation and changes in condition over 12-16 years. (Gen. Tech. Rep. INT- 463). Ogden, UT: USDA, Forest Service, Intermountain Research Station.
- Cole, D. N. (1996). Wilderness Recreation use trends, 1965 through 1994. (Gen. Tech. Rep. INT- RP-488). Ogden, UT: USDA, Forest Service, Intermountain Research Station.
- Cole, D. N., Watson, A. E., Hall, T. E., & Spildie, D. R. (1997). High-use destinations in wilderness: social and biophysical impacts, visitor responses, and management options. (Gen. Tech. Rep. INT- RP-496). Ogden, UT: USDA, Forest Service, Intermountain Research Station.
- Flood, J. (1993). Contracting is a successful option in wilderness restoration. *Handbook for a renaissance in wilderness stewardship* (pp. 36-37). National Interagency Wilderness Conference. Tucson, AZ.

- Hall, T. & Shelby, B. (1993). Wilderness monitoring. Mount Jefferson, Mount Washington, and Three Sisters Wildernesses. (Unpublished Tech. Rep. on file, pp. 93). Eugene, OR: USDA, Forest Service, Willamette National Forest.
- Hendee, J. C., Stankey, G. H., and Lucas, R. C. (1990). Wilderness management, (Rev. ed.). Golden, CO: Fulcrum Publishing.
- Lucas, R. C., (1980). Use patterns and visitor characteristics, attitudes, and preferences in nine wilderness and other roadless areas. (Gen. Tech. Rep. INT-253). Ogden UT: USDA, Forest Service, Intermountain Research Station.
- Lucas, R. C. (1987). Proceedings—national wilderness research conference: issue, state-of-knowledge, future directions. (Gen. Tech. Rep. INT - 220). Ogden UT: USDA, Forest Service, Intermountain Research Station.
- Marion, J. L., Roggenbuck, J. W., & Manning, R. E. (1993). Problems and practices in backcountry recreation management: a survey of National Park Service managers. (Natural Resource Rep. NPS/NRVT/NRR-93/12). Denver, CO: U.S. Department of Interior, National Park Service, Natural Resources Publication Office.
- Martin, S. R., McCool, S. F. & Lucas R. C. (1989). Wilderness campsite impacts: do managers and visitors see them the same? *Environmental Management*, 13 (5), 623-629.
- McCool, S. F. & Lime, D. W. (1988). Attitudes of visitors toward outdoor recreation management policy. Proceedings—national wilderness research conference: issues, state-of-knowledge, future directions (pp. 401-411). Ogden UT: USDA, Forest Service, Intermountain Research Station.
- Miles, M. B. & Huberman, M. A. (1994). *Qualitative data analysis: An expanded sourcebook*, 2nd ed. Thousand Oaks, CA: Sage.
- Peterson, G. L. (1974). A comparison of the sentiments and perceptions of canoeists and wilderness managers in the Boundary Waters Canoe Area. *Journal of Leisure Research*, 6 (3), 194-206.
- Public law 88-577, The Wilderness Act (1964) (pp. 890-896) in the US Statutes At Large.
- Roggenbuck, J. W. (1992). Use of persuasion to reduce resource impacts and visitor conflicts. In M. J. Manfredo (ed.), *Influencing human behavior: theory and applications in recreation, tourism, and natural resource management* (pp. 149-208). Champaign, IL: Sagamore Publishing.
- Schreyer, R. & Roggenbuck, J. W. (1978). The influence of experience expectations on crowding perceptions and social psychological carrying capacities. *Leisure Sciences*, 1 (4), 373-394.
- Shelby, B. & Harris, R. (1985). Comparing methods for determining visitor evaluations of ecological impacts: site visits, photographs, and written descriptions. *Journal of Leisure Research*, 17 (1), 57-67.
- Shindler, B. & Shelby, B. (1993). Regulating wilderness use: an investigation of user group support. *Journal of Forestry*, 41-44.
- Stankey, G., Cole, D., Lucas, R., Peterson, M., & Frissell, S. (1985). The limits of acceptable change (LAC) system for wilderness planning. (Gen. Tech. Rep. INT-176). Ogden, UT: USDA, Forest Service, Intermountain Forest and Range Experiment Station.
- Stokes, G. (1990). The evolution of wilderness management. *Journal of Forestry*, 88 (10), 15-20.
- Watson, A. E. & Niccolucci, M. J. (1992). Defining past-experience dimensions for wilderness recreation. *Leisure Sciences*, 14 (2), 89-103.
- Watson, A. E. & Cronn, R. (1994). How previous experience relates to visitors' perceptions of wilderness conditions. *Backcountry Recreation Management Trends*, 31 (3), 43-46.

Effectiveness of a Confinement Strategy in Reducing Pack Stock Impacts at Campsites in the Selway-Bitterroot Wilderness, Idaho

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Abstract—In 1993, a management program was initiated in the Seven Lakes Basin in the Selway-Bitterroot Wilderness to bring high levels of campsite impact into compliance with management standards. The core of the strategy involved confining use, particularly by stock groups, and restoring certain campsites and portions of campsites. In just five years, campsite impacts were reduced substantially. Disturbed and bare area decreased on campsites, as did tree scarring and mineral soil exposure. Vegetation cover increased. The only impact parameter that continued to get worse was tree root exposure. Continuation of this program would likely reduce the extent of disturbance to less than one-third of the disturbance that existed before implementation. Fiscal costs were substantial, but experiential costs were judged to be minimal. This program should provide a good model for other wildernesses with campsite problems in certain destination areas.

One of the goals of wilderness recreation management is to avoid ecological impacts and provide opportunities for high-quality wilderness experiences. Another goal—which often conflicts with the former—is to provide access for these experiences and to avoid restriction and regulation, which can make experiences seem “confined.” Conflict between these two goals usually results in some compromise of both.

Ecological impacts are most problematic in campsites in popular destination areas. Proliferation of new campsites, leading to unnecessarily high campsite densities, has been a common trend over the past few decades (Cole 1993). Specific impacts include damage to overstory trees, loss of vegetation, changes in species composition, soil compaction, loss of organic horizons and exposure of mineral soils (Cole 1983). The worst sites are almost as compacted as pavement, have been without groundcover vegetation for half a century or more and have numerous scarred trees with roots exposed in tree wells.

Ecological impacts are particularly severe where pack stock use is common because, everything else being equal, groups traveling with horses and mules create more intense

impacts than groups traveling on foot (Cole 1983, Weaver and Dale 1978, Whinam and Comfort 1996). The trampling impacts of stock are qualitatively similar to those of hikers, but more severe (Cole and Spildie 1998, Deluca and others 1998). Impacts associated with grazing and confinement of stock are qualitatively unique to stock use and can be the most severe impacts of all (McClaran and Cole 1993). However, empirical studies of pack stock impacts are rare. Particularly lacking is information on pack stock management strategies and stock use handling techniques.

Increasingly, wilderness management plans are based on a goal-achievement framework, such as Limits of Acceptable Change (LAC) (McCool and Cole 1997). Such plans establish standards for maximum acceptable impact levels. If monitoring indicates that these standards are not being met, management actions that can eventually meet standards are required. Standards are often written for campsite impacts, and many wilderness plans report that there are certain destination areas—often lake basins—where campsite impact standards, such as number of campsites per square mile, are exceeded. The prevalence of this situation suggests the need to identify effective programs for bringing campsite impacts up to standard.

The goal of this paper is to describe a case study conducted in the Seven Lakes Basin in the Selway-Bitterroot Wilderness, Idaho, a destination area in which standards for both campsite density and intensity of campsite impact were violated. The management actions, implemented in 1992, were designation of a small number of stock campsites, closure of some sites to all use and intensive restoration of many sites and trails. This is a version of a containment strategy, one of the most effective approaches to minimizing ecological impact in a heavily used destination (Marion 1995). Specifically, this paper presents data on how the number of campsites and intensity of impact changed between 1993 and 1998 as a result of the implemented management program. We also outline the costs of this management program and discuss the management implications of our findings.

Seven Lakes Basin and Its Management Program

The study area (Seven Lakes Basin) consisted of two adjacent subalpine lake basins (Seven Lakes itself and the Maude-Lottie Lake basin) in the southcentral part of the 540,000 ha Selway-Bitterroot Wilderness, Idaho. The total area of Seven Lakes Basin is about 500 ha. The basin

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contains 11 lakes and is located at an elevation of 1,860–2,000 m. It can be accessed within one day from the Wilderness Gateway trailhead but requires a climb of about 1,000 m in the last 10 km of the 19 km trail. We excluded the area around the two northeasternmost lakes (Rock Lake and Surprise Lake) and the two southwesternmost lakes (unnamed) from our study because few management actions have been taken at these lakes and because they are either physically separated from the others or seldom visited. Use levels in the basin are moderate. Records show that there are virtually never more than four other groups in the basin at one time. Most visitors are fishermen who camp near a lake for several days (many with pack stock). Many groups with pack stock camp at lower meadows, where there is more feed, and visit the basin on day rides.

The Selway-Bitterroot Wilderness Limits of Acceptable Change Plan established standards for the Wilderness that were not being met at Seven Lakes. In this class III area (on a scale with class IV accepting the most impact), campsite density should not exceed three campsites per square mile. There should be no extremely impacted campsites and no more than one moderately impacted campsite per square mile. Monitoring showed that previous recreation use, particularly by groups with pack stock, has left 26 substantially impacted campsites in the area (fig. 1). Campsite density was as high as 13 campsites per square mile, the number of extremely impacted campsites reached five per square mile, and the number of moderately impacted sites reached four per square mile. To reach standards, as many as 10 campsites per square mile needed to be closed and

restored, and up to five extremely impacted campsites and three moderately impacted sites needed either improvement or restoration.

To make progress toward the goals established in the Wilderness LAC plan, a restoration plan was developed for the Seven Lakes Basin in 1992. This plan established more realistic short-term (“interim”) standards, although ultimately the original LAC standards were to be met. These interim standards called for reducing density to no more than eight campsites per square mile and reducing the number of intensively impacted campsites, while leaving at least one campsite open for stock use at each of the major lakes. These objectives were to be met by implementing management actions. The most important of which were (1) the designation of three day-use stock containment areas and six overnight stock containment areas, where stock are to be tethered between designated trees with a high line, rope or electric corral, (2) the prohibition of stock containment on other campsites or other parts of designated campsites, and (3) the prohibition of all camping on four campsites. Tying stock directly to trees or in places where tree roots can be damaged was prohibited. Stock numbers were limited to a maximum of 10 animals.

Regulations on where to camp and contain stock were communicated to the public on a brochure, signs on bulletin boards at the trailhead and at the entry point to the lake basin on all trails, in local newspapers and by frequent visits of wilderness rangers to the area. Compliance was enforced through special orders and heavy ranger presence. Some trails in the basin were reconstructed; about 1 km of trail

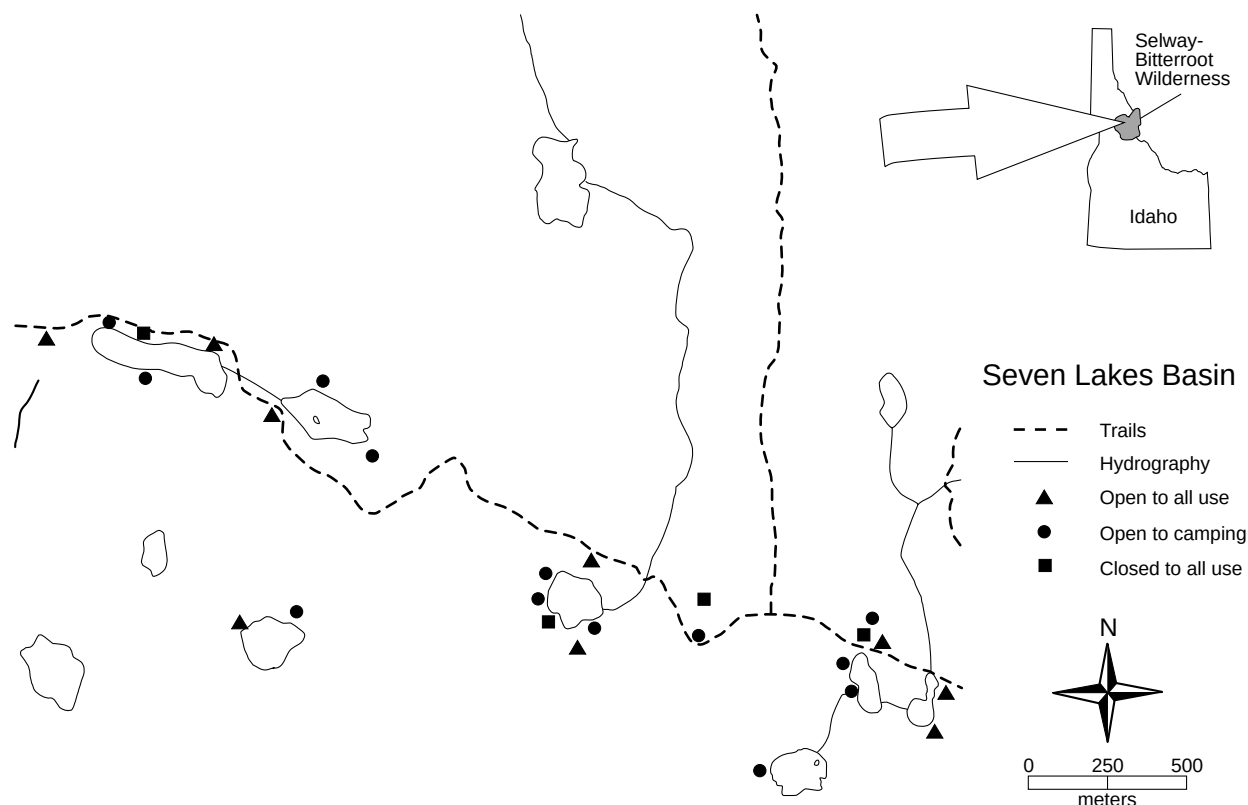


Figure 1—Location of campsites, differentiated by closure category, in the Seven Lakes Basin study area.

was rerouted, and another km of trail was closed and rehabilitated. Two bridges were built. Forty-seven former stock-holding areas were closed to stock containment. These areas were generally adjacent to clumps of trees with roots and mineral soil exposed by decades of tying horses to trees. These 47 areas were on 12 campsites that were closed to stock use, six campsites that remained open to stock use and one former campsite where day-use containment only of stock is allowed. Designated high line trees were signed at each of the six open stock campsites with a designated stock-holding area and the three day-use stock-holding areas. These campsites, where stock use is still allowed, were signed, as were four campsites that were closed to all use. Most closed areas were intensively restored. Seeds were collected, and about 2000 seedlings of three species, intermediate oatgrass (*Danthonia intermedia*), partridgefoot (*Luetkea pectinata*), and beargrass (*Xerophyllum tenax*), were propagated at the University of Idaho Forestry Nursery and packed up to the basin. Soils were scarified, organic matter was added to soils, and large rocks were used as "icebergs" (placed to protrude from the ground, making the site undesirable for camping). Stumps were flush-cut and tree wells were filled with soil. Pitch and charcoal were applied to trees to minimize evidence of tree scarring. Propagated seedlings, locally collected seed and local transplants were used to revegetate areas. Finally, some areas were covered with a mulching material. Campsite impact conditions were monitored over the period.

This work was largely accomplished by two people who shared one seasonal wilderness ranger position, about four weeks of work per year. A six-person Student Conservation Association (SCA) crew assisted for two years (for a total of 2,400 person hours), and a seven-to-eight person Appalachian Mountain Club crew assisted for two years (total of 768 person hours). Other volunteers, including many from the IDAWA project, a partnership between the Forest Service and the Iowa Department of Education, contributed 720 person hours of work.

Field Methods

In this study, we assessed conditions on designated stock campsites (sites on which stock use continues), former stock campsites (sites on which stock use is no longer allowed), and backpacker campsites (sites on which stock use has seldom occurred). Designated stock campsites contained three different types of areas: (1) a camping area; (2) former stock-holding area(s); and (3) a designated stock-holding area. We assessed impacts, using different methods, on each of these three types of areas. In addition, we monitored change on an adjacent, undisturbed control site. Former stock campsites, on which either all camping or camping with stock is prohibited, had a camping area and former stock-holding area(s). Backpacker campsites had a camping area only.

Camping Areas

Although old campsite monitoring data were available, protocols had changed enough through the years that some could not be used for comparisons. Consequently, we reinventoried each camping area, the area around tents and

cooking areas in July 1993. We used a rapid inventory approach, referred to in Cole (1989) as the Bob Marshall method, where it was first developed. This procedure involves rapid estimates of vegetation cover, exposed soil cover, tree damage, root exposure, level of development, cleanliness, number of social trails, camp area and barren core area. In addition, radial transects were used to more precisely estimate the camp area (area evidently disturbed by use) and the barren core area (area completely devoid of vegetation in the most heavily used part of the camp). These procedures, with the exception of the radial transects, were repeated on each camping area in August 1998.

Former Stock-Holding Areas

Forty-seven distinct former stock-holding areas were found on 19 different campsites used by stock. Six of the 19 campsites remain open to stock use, one is now a day-use stock-holding area, and 12 have been closed to stock use. Where stock use continues, containment is confined to designated areas and is not allowed on the former stock-holding areas. In each of the former stock-holding areas, we established a permanent center point (marked with a buried nail). We measured the distance and direction from this point to (1) the first vegetation and (2) the edge of obvious disturbance along a variable number of radial transects. The number and location of transects are the minimum needed to capture the shape of the campsite (Marion 1991). The distances to the first vegetation define bare area, while distances to disturbance define the disturbed area of the former stock-holding area. Within the perimeter of the disturbed area—assuming straight lines between adjacent transect end points—we assessed impacts to trees. For each tree greater than 2.5 cm d.b.h., we noted whether the tree was alive, dead and standing, a cut stump or a stump of undefined origin; measured the areal extent of scarring to the bole (considering linear slices to have a width of 0.5 cm if they were narrower than this); and measured the linear extent of exposed roots at least 2.5 cm wide. Percent cover of live vegetation and exposed mineral soil was visually estimated in 1-m square quadrats, using 10% cover classes or to the nearest percent if cover was less than 10%. Quadrats were located along one to four transects running between the center point and end points (located at the edge of the disturbed area and permanently marked with buried nails) in cardinal directions. Number of transects, quadrats per transect and total quadrats varied with the size and configuration of the area. On the smallest area, only one quadrat was assessed. The maximum number of quadrats was 12. When aggregated to the campsite level, seven campsites had between one and four quadrats in their former stock-holding areas; seven had between 5 and 13 quadrats, and five had between 14 and 23 quadrats.

Designated Stock-Holding Areas

The nine designated stock-holding areas have two trees designated as high line trees. Stock should be confined in a rectangle between these two trees. We established a transect between the two trees, permanently marking the center point of this transect. Then we established four permanent

points at the corners of the rectangle, 4 m perpendicular from each of the endpoints of the initial transect (adjacent to the designated tree). Disturbed area and bare area were assessed by measuring the distance from the center point to first vegetation and edge of disturbance, along eight transects running in cardinal directions. Tree impacts were assessed, as done in former stock-holding areas within the rectangle. Sixteen 1-m square quadrats were located along four 8-m long transects, located equidistant and perpendicular to the initial transect between trees. Quadrats were located 1 m apart, on the same side of the transect. Vegetation and mineral soil cover were visually estimated in each quadrat.

To adjust for natural changes in vegetation and exposed soil cover due to climatic variation, we established control sites in undisturbed places in the vicinity of each designated area. Sites were selected on the basis of similarity to the designated area in terms of topography, rockiness, tree canopy cover and understory species. Controls were 8-m squares with permanent markers (buried nails) at each corner. Sixteen quadrats were located 1 m apart on the same side of four 8-m long transects located 2 m apart. Vegetation and mineral soil cover were visually estimated in each quadrat.

All measurements were repeated in August 1998. Buried nails were relocated using reference information, such as distances and azimuths from obvious landmarks such as large trees, unusual species, rocks and so on. A magnetic pin locator facilitated this process. In the results presented in this paper, we have combined the three designated day-use stock campsites with the six designated stock campsites.

Data Analysis

Data analysis was complicated by the fact that the number of impacted areas varied greatly between campsites. Some campsites had as many as five separate former stock-holding areas, a designated stock-holding area and a camping area. All former stock-holding areas on the same campsite were aggregated into a single set of measures per campsite. A single set of measures for the total campsite involved aggregating designated and former stock-holding areas, along with the camping areas. For disturbed area (area obviously disturbed by trampling), bare area (area of the central area completely devoid of vegetation), tree scarring (total area of tree scarring on all trees) and root exposure (total length of exposed root on all trees), aggregation involved simply adding all the values together. For vegetation and exposed mineral soil cover (expressed as percent of the campsite), however, it was necessary to weight the percent cover of each area by the proportion of total disturbed area in that area. This procedure was used when aggregating former stock-holding areas as well.

This provided data for camping areas, former stock-holding areas, designated stock-holding areas and the total campsite for each campsite for 1993 and 1998. Means and standard errors are presented for each of the six impact variables, along with an estimate of change, expressed as a percent of 1993 values. Change was the 1993 value minus the 1998. Minus change values represent deterioration in conditions, except in the case of vegetation cover. The number of campsites that improved, deteriorated or stayed the

same was assessed. If values did not change by more than 10% of their original condition, they were considered unchanged.

Since we censused all impacted sites in the basin, there was no need to use inferential statistics to assess confidence in our estimates of change between 1993 and 1998. Changes reported did occur, subject to measurement error. We did use t-tests and analysis of variance to assess the extent to which magnitude of change was significantly influenced by (1) whether the site had been restored and (2) whether it was open to all use, open to backpackers only or closed to all use. The latter two categories were sometimes difficult to distinguish. Originally, 10 campsites were slated for complete closure, and signs indicating site closure were established at seven closed campsites. By 1998, only two of these sites still had closure signs; another two sites were so intensively iceberged and revegetated that further camping was very unlikely. Consequently, we decided that only four campsites were still clearly closed to all camping (fig. 1).

Results

Described below are both the benefits and costs of the management program implemented at Seven Lakes Basin. Initially, we present data on changes between 1993 and 1998 in amount of impact. We have data on three types of areas within campsites—designated stock-holding areas, former stock-holding areas and camping areas—as well as for the campsite as a whole. For each of six different impact parameters, we present data for each of these types of area and the entire campsite. We describe what conditions were like in 1993 and how they changed between 1993 and 1998 (table 1). We present change as a percent of the condition in 1993, as well as the number of campsites that improved, deteriorated or stayed the same, within 10% of the original value. Then we compare changes on sites that were or were not restored (table 2), as well as on sites that were open to all users, open only to hikers or closed to all camping (table 3). Finally, we describe basin-wide changes in conditions.

Benefits of the Management Program

Changes in Disturbed Area on Campsites—The area obviously disturbed by visitors provides perhaps the best overall indication of the areal extent of recreation impact. Mean disturbed area for the entire campsite was 135 m² in 1993 (table 1). On the popular sites with designated stock-holding areas, the disturbed area was typically quite large. Both areas where stock-holding had been allowed and camping areas were typically smaller. For all campsites, disturbed area declined almost 40% by 1998—to a mean of 85 m². Sixteen campsites improved substantially, while only five deteriorated over this period. Moreover, the sites that improved most were those that were most disturbed in 1993 (fig. 2). Of the different types of area, only the disturbed area of designated stock-holding areas increased between 1993 and 1998 (4%). The disturbed area of former stock-holding areas and camping areas decreased between 40% and 50% in just five years.

For all campsites, the decline in disturbed area on sites that received assisted restoration (for example, iceberging,

Table 1—Means and standard errors for impact parameters in 1993 and 1998, percent change and number of areas that improved, deteriorated, or were unchanged.^a

	1993	1998	% change	Improved	Deteriorated	Unchanged
Disturbed Area (m ²)						
Designated	79 (28)	82 (16)	−4	2	6	1
Former	87 (24)	46 (17)	47	16	1	2
Camp	48 (10)	25 (6)	48	12	3	9
Total	135 (34)	85 (21)	37	16	5	5
Bare Area (m ²)						
Designated	14 (7)	29 (10)	−1	1	6	2
Former	26 (7)	14 (7)	46	12	1	6
Camp	25 (7)	7 (3)	71	15	0	9
Total	47 (10)	27 (9)	43	16	3	7
Tree Scarring (cm ²)						
Designated	17 (12)	38 (21)	−121	0	3	6
Former	571 (238)	390 (196)	32	3	4	11
Camp	—	—	—	—	—	—
Total	497 (208)	351 (172)	29	3	5	13
Root Exposure (cm)						
Designated	400 (136)	1067 (329)	−166	0	7	2
Former	1674 (428)	1603 (433)	4	5	6	7
Camp	—	—	—	—	—	—
Total	1607 (404)	1831 (479)	−14	5	9	7
Vegetation (%)						
Designated	39 (7)	24 (4)	38 ^b	0	8	1
Former	28 (5)	47 (6)	−68 ^b	13	3	3
Camp	51 (7)	49 (7)	3 ^b	4	20	0
Total	39 (5)	46 (6)	−19 ^b	13	7	6
Soil Exposure (%)						
Designated	9 (4)	18 (7)	−100	1	8	0
Former	10 (3)	6 (2)	39	10	8	1
Camp	10 (3)	10 (4)	−9	6	16	2
Total	11 (3)	9 (3)	18	9	8	9

^aPercent change is the 1993 value minus the 1998 value divided by the 1993 value. Sites were considered changed if 1998 values were $\pm 10\%$ or less of 1993 values.

^bIn contrast to all other impact parameters, negative change in vegetation cover indicates improvement rather than deterioration.

scarifying and planting) was somewhat greater than on sites that were not restored (fig. 3), but the differences were not statistically significant (table 2). Note that figure 3 shows change as a percentage of original conditions, while table 2 shows change in the original units of measure, not adjusted for original conditions. The same was found on former stock-holding areas. We do not show data for designated stock-holding areas because they were never restored. In contrast, camping areas on sites that were not restored improved

significantly more than sites that were restored. This might be explained by the fact that many of the restored camping areas were popular sites that would take longer to recover than less impacted sites. As expected, on sites that were closed to all use, disturbed area declined more than on sites that remained open to camping (fig. 4, table 3). What was surprising was that sites open to all use—both horses and hikers—often recovered as much, if not more, than sites closed to all use. This probably reflects the within-site

Table 2—Change in impacts on restored (R) and non-restored (NR) campsites.^a

	Former stock-holding areas			Camping areas			Total campsite		
	R	NR	p	R	NR	p	R	NR	p
Disturbed Area (m ²)	43(12)	36(25)	.39	14(7)	45(19)	.04	55(23)	45(31)	.40
Bare Area (m ²)	19(5)	−8(10)	.01	17(6)	21(9)	.35	27(9)	8(5)	.03
Tree Scarring (cm ²)	35(66)	56(56)	.44	—	—	—	23(58)	31(31)	.46
Root Exposure (cm)	387(291)	−1032(653)	.02	—	—	—	−25(358)	−621(398)	.16
Vegetation (%) ^b	−19(6)	−1(6)	.03	9(5)	5(7)	.33	−6(5)	0(7)	.24
Soil Exposure (%)	1(5)	8(3)	.09	−1(5)	−3(3)	.44	0(5)	6(7)	.22

^aTable reports mean (standard error) change between 1993 and 1998 (1993 values minus 1998 values) and results of t-tests.

^bIn contrast to other impact parameters, negative change in vegetation cover indicates improvement rather than deterioration.

Table 3—Change in impacts between campsites that are open to all use, open to camping, and closed to all use.^a

	Former stock-holding areas			p	Camping areas			p	Total campsite			p
	Open to all use	Open to camping	Closed to all use		Open to all use	Open to camping	Closed to all use		Open to all use	Open to camping	Closed to all use	
Disturbed Area (m ²)	62(18)	21(16)	45(22)	.24	36(21)	13(7)	34(21)	.40	76(46)	26(12)	79(43)	.39
Bare Area (m ²)	11(10)	12(8)	15(14)	.97	21(9)	15(6)	26(18)	.70	10(7)	22(7)	40(31)	.31
Tree Scarring (cm ²)	170(147)	-28(21)	-19(22)	.22	—	—	—	—	93(87)	-28(21)	-19(22)	.35
Root Exposure (cm)	-239(806)	175(298)	331(345)	.77	—	—	—	—	-826(520)	175(298)	331(345)	.16
Vegetation (%) ^b	9(8)	16(9)	20(7)	.72	-16(12)	-4(3)	-2(8)	.41	-7(5)	7(6)	19(6)	.08
Soil Exposure (%)	-3(6)	7(6)	5(3)	.48	-10(11)	0(2)	8(6)	.21	-9(7)	4(3)	21(12)	.02

^aTable reports mean (standard error) change between 1993 and 1998 (1993 values minus 1998 values) and results of analysis of variance. Sites open to camping are closed to stock.

^bIn contrast to other impact parameters, negative change in vegetation cover indicates improvement rather than deterioration.

recovery from confining stock to only one designated area on each campsite.

The total area of disturbance in the Seven Lakes Basin was 3518 m² in 1993. In 1998, total disturbance was just 2205 m². This represents a 37% decrease in the extent of impact in just five years. If the current management program continues, all closed campsites and former stock-holding areas should recover completely, probably within a decade or two. At that time, the total area of impact would be just 1262 m²—or only 36% of what it was before the management program was implemented.

Changes in Bare Area on Campsites—The bare area represents the size of the heavily used part of camp that is devoid of vegetation. Sites with large bare areas are highly problematic, since re-establishment of vegetation is often difficult. Mean bare area of the entire campsite declined from 47 m² in 1993 to 27 m² in 1998, a decrease of nearly 43% (table 1). Sixteen campsites improved, while only three deteriorated during this time period. Only on designated stock-holding areas did bare area increase between 1993 and 1998 (1%). Former stock-holding areas improved on 12 of the 19 sites. Camping areas improved most. Mean bare

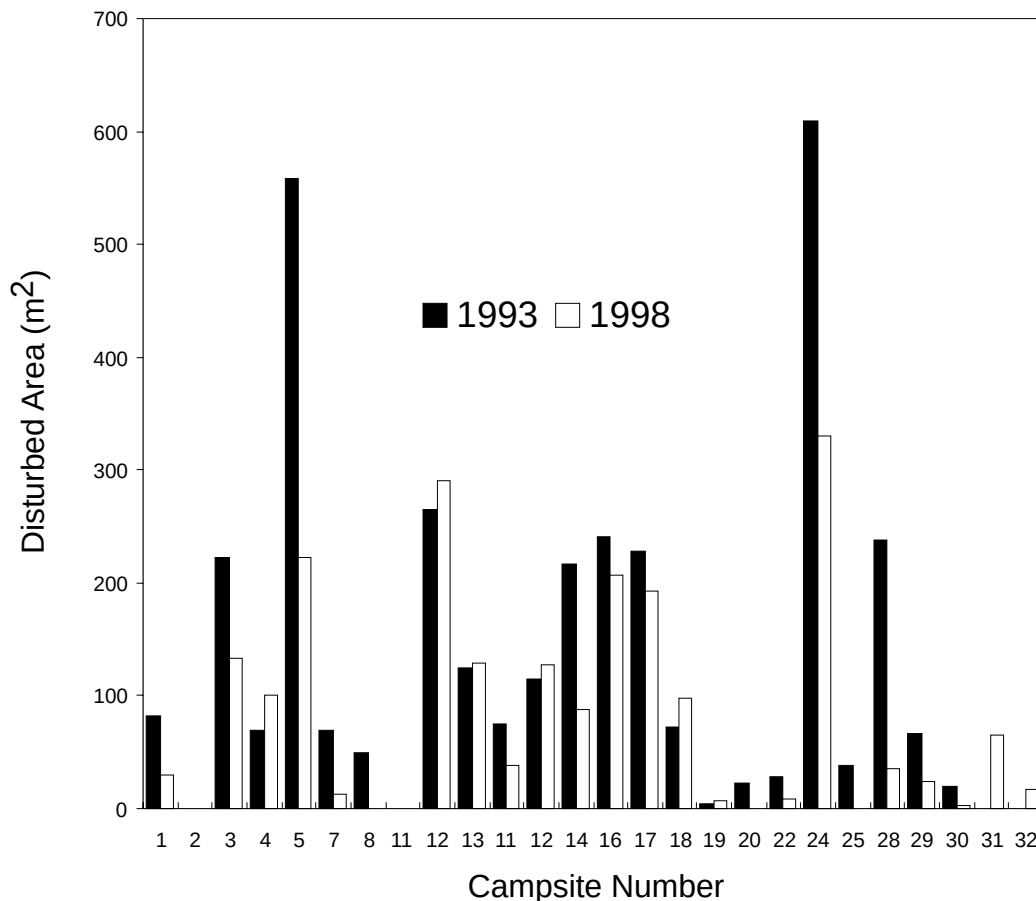


Figure 2—Change in the disturbed area of each of the 26 campsites between 1993 and 1998.

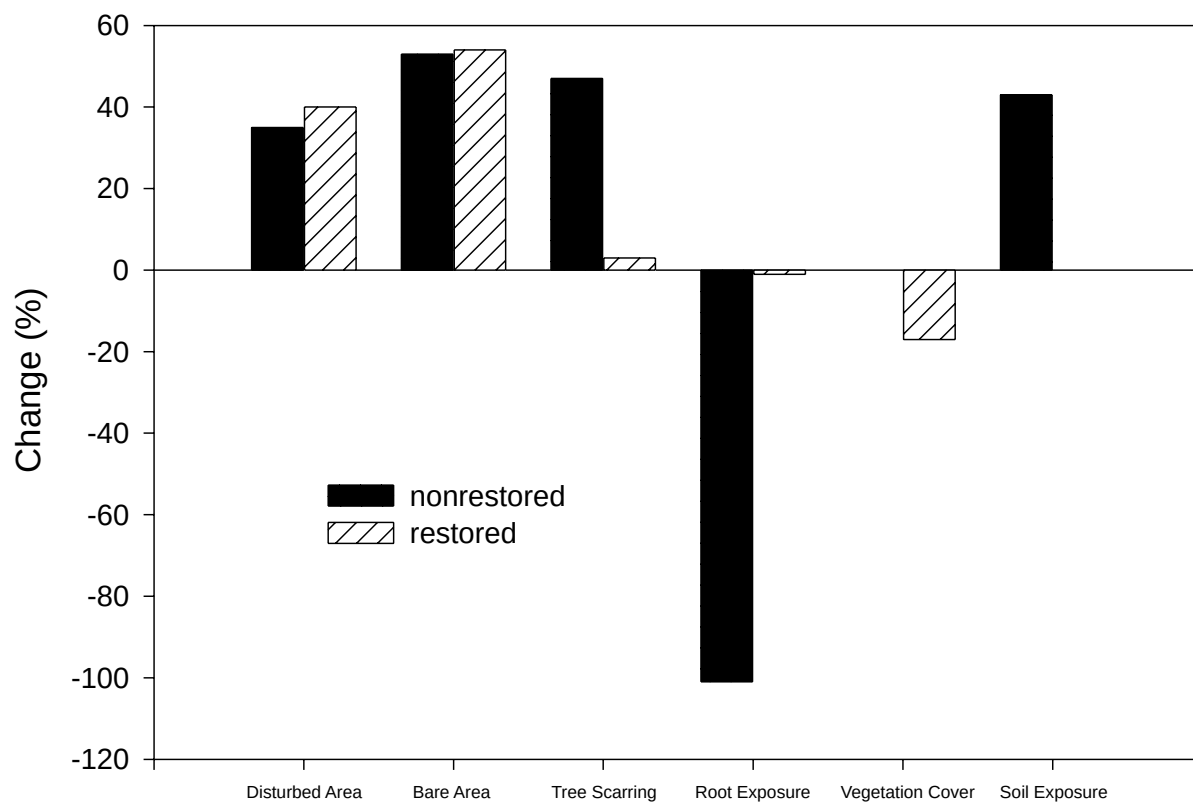


Figure 3—Percent change (1993 value minus 1998 value, divided by 1993 value) in impact parameters for non-restored and restored campsites. Negative values indicate deterioration, except in vegetation cover.

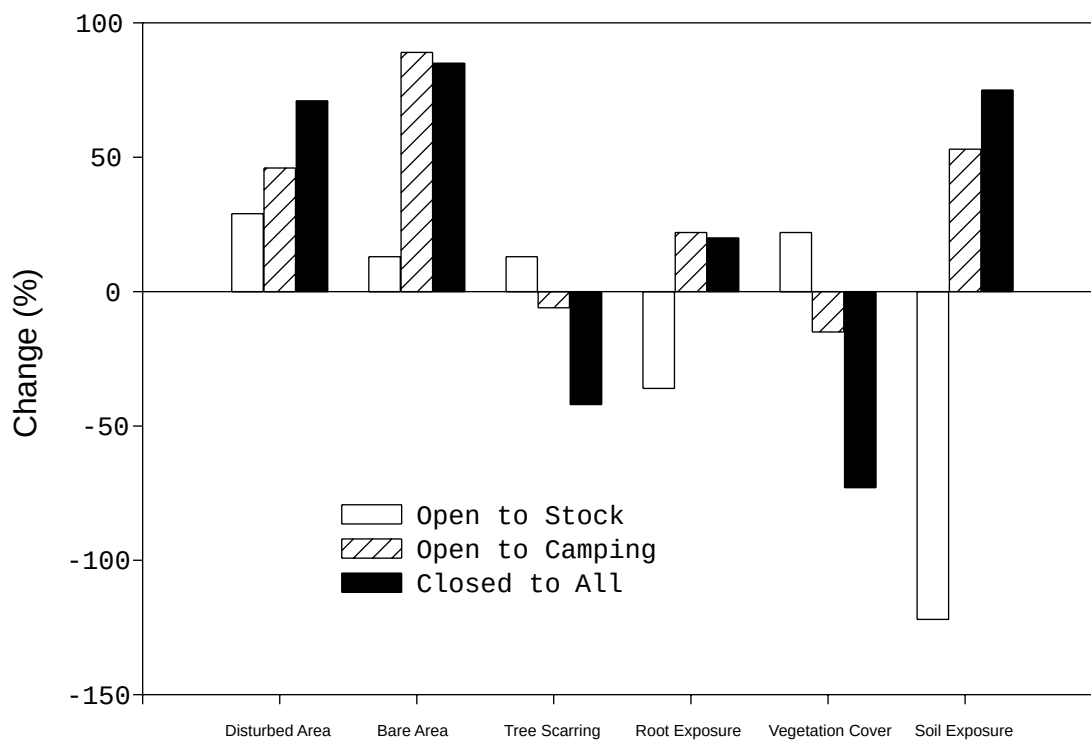


Figure 4—Percent change (1993 value minus 1998 value, divided by 1993 value) in impact parameters for campsites that are open to all use, open to camping, or closed to all use. Negative values indicate deterioration, except in vegetation cover.

area declined 71%, and 15 of 24 sites improved, while none deteriorated.

For all campsites, the decline in bare area on sites that received restoration (27 m²) was significantly greater than the decline on nonrestored sites (8 m²), (fig. 3, table 2). This was also true of the former stock-holding areas, which improved significantly due to restoration efforts. This was expected since these areas were closed to all use and targeted for restoration efforts. Restoration efforts were not significantly more effective in reducing the bare area of camping areas. At the campsite level of analysis, bare area declined more on sites that were closed to all use than on those that were open to camping or open to all use. The same was true for the former stock-holding areas and camping areas; however, none of the differences were significant (table 3). This demonstrates the positive effect of complete closure of a site on the recovery of bare ground impacts.

The total bare area in the Seven Lakes Basin declined from 1222 m² in 1993 to 699 m² in 1998, a considerable decrease of 43% in five years. If current management continues and closed sites and former stock-holding areas recover completely, the total bare area should decrease to approximately 289 m², or just 24% of the impact prior to implementation.

Changes in Tree Scarring on Campsites—Tree scarring can affect the vigor and is a long-lasting visual impact. Although these wounds will slowly heal, there is no known restoration procedure to mitigate these impacts. Consequently, trees that are scarred do not recover from this impact, even if campsites are closed. The designated stock-holding areas showed a substantial increase in tree scarring (over 120%) between 1993 and 1998 (table 1). Most of the increased damage occurred on three of the nine designated campsites. Conditions improved on three of 18 campsites with former stock-holding areas; but 11 of these campsites were stable. Improvement of tree damage was most likely due to the removal of scarred and severely damaged trees during the restoration process. A different metric was used for determination of tree damage on camping areas, and these were not included in the final analysis. For the entire campsite, mean tree scarring declined and conditions were unchanged on most campsites. However, more campsites deteriorated than improved. This suggests that the rate of increase in tree damage has declined as a result of the confinement strategy, although further damage continues to be a problem on a few of the campsites.

Restoration efforts had little effect on the amount of change in tree scarring on the designated areas, former stock-holding areas, or the campsites (fig. 3, table 2). Degree of campsite closure also did not have a statistically significant effect on tree scarring (table 3). Ironically, there was more improvement on the sites still open to stock because several scarred trees were cut down on these sites (fig. 4). The long-lasting nature of tree impacts means they are unlikely to improve substantially in the time-frame of this study.

Change in Root Exposure on Campsites—The exposure of roots in tree wells and other areas of stock confinement is an impact that is visually obtrusive and ultimately associated with tree mortality. This impact is unnecessary because stock need not be tied to trees. Moreover, rapid

improvement is possible because exposed roots can be easily eliminated by filling in tree wells with soil and organic matter. As expected, given the concentration of pack stock use in designated stock-holding areas, these areas showed marked deterioration. Root exposure increased from 400 cm to 1067 cm, or 166% in five years (table 1). Seven of the nine designated areas deteriorated. Mean root exposure on former stock-holding areas improved slightly between 1993 and 1998, but only five of the 18 improved. For all campsites, root exposure increased slightly. Five campsites improved and nine deteriorated.

On former stock-holding areas, restoration efforts resulted in a reduction in root exposure, while root exposure increased on non-restored former stock-holding areas (table 2). Root exposure, at the scale of the entire campsite, increased less dramatically on restored sites (fig. 3) but differences were not statistically significant. Campsites open to all use deteriorated, while closed sites and sites open only to backpackers improved (fig. 4). This difference was particularly pronounced on former stock-holding areas. However, there was so much site-to-site variability that differences were not statistically significant (table 3).

Changes in Vegetation Cover on Campsites—Mean campsite vegetation cover increased 19%—from 39% in 1993 to 46% in 1998 (table 1). Vegetation increased on 13 of the campsites and decreased on six campsites. Despite this general improvement, vegetation cover decreased on designated stock-holding areas from 39% in 1993 to 24% in 1998. Vegetation cover also decreased slightly (3%) on camping areas. The improvement in conditions occurred on former stock-holding areas where vegetation cover increased 68%, from 28% in 1993 to 47% in 1998. The improvement more than compensated for deterioration of current stock-holding and camping areas.

On former stock-holding areas, restoration efforts resulted in significantly more improvement in vegetation cover (table 2). Since these areas are now closed to all use, the re-establishment of vegetation through seeding and transplanting was quite effective. Restoration efforts were not very effective within camping areas. At the scale of the entire campsite, vegetation cover increased more on restored sites (fig. 3), but differences were not statistically significant. Vegetation cover declined on campsites open to stock and increased on campsites not open to stock. Improvement was greatest on sites closed to all use (fig. 4) ($p = 0.08$, table 3). These results suggest that closure to all use, in combination with active restoration, is the most effective prescription for vegetation recovery. However, recovery can occur simply by eliminating use by stock.

Changes in Soil Exposure on Campsites—Designated stock-holding areas received concentrated pack stock use, so it was not surprising that mineral soil exposure increased 100%, from 9% in 1993 to 18% in 1998 (table 1). With the cessation of pack stock use in the former stock-holding areas, mean soil exposure percentages decreased by 39% during the study period, with ten of the 19 areas improving. Since most camping areas continued to receive use, soil exposure increased 9%. For all campsites, soil exposure values decreased 18%, with nine campsites improving, eight deteriorating and nine unchanged. Again, improvement of the former stock-holding areas more than compensated for the deterioration in other parts of the campsite.

Soil exposure declined on campsites closed to stock use, while it increased on campsites open to stock use (table 3). However, restored campsites actually recovered less than those campsites that were not restored (table 2). This unexpected result reflects the fact that, (1) restoration of a duff layer was not among the techniques employed at Seven Lakes and (2) the campsites that were restored were the most severely impacted.

Costs of the Management Program

Despite the decided ecological benefits of the management program, there were also more limited experiential costs and substantial financial costs.

Experiential Costs—The need for more signage within the Selway-Bitterroot Wilderness (as you enter the basin and at designated sites and the two remaining signed closed sites) degrades the aura of wildness in this place. However, so do the extensive trail systems and heavy impacts. The management programs reduced freedom of recreation use, but not substantially. Camping is now restricted to about 17 campsites in the basin, and people with stock can now use only six sites to camp at in the basin. Moreover, the number of stock allowed and the number of people allowed per group, when camping, is only 10. All stock users—day and over-night—must not tie stock to trees; they must tie up at one of the nine designated stock-holding areas. Stock users and backpackers who want to travel in groups larger than 10 or who do not want to contain their stock between designated trees can simply camp outside the basin and make day visits. Since there are no limits on amount of use, no lakes where camping is not allowed, and no groups excluded from visiting the basin (other than those who do not meet entrance criteria for the entire Selway-Bitterroot Wilderness), we conclude that experiential costs are minor.

Financial Costs—Cost estimates were kept, although many hidden costs were undoubtedly missed, and certain costs were ballpark estimates. When both Forest Service and contributed costs are combined, we estimate conservatively that total costs for the first five years of the program exceeded \$135,000 (table 4).

Clearly, the financial costs of this program have been considerable, and they will be ongoing, but to a lesser degree. Nevertheless, through innovative use of volunteers and removing the fixed costs of seasonal employees, the “new” costs to the Forest Service were only about \$8,500 per year for the first five years and probably will be no more than \$1,000 per year into the future. This cost, while very high for measly wilderness budgets, is very reasonable when one considers the magnitude of remediation that was necessary after decades of unrestricted, high-impact use on the basin. If there were about 20 similar problem destination areas in the entire Selway-Bitterroot Wilderness, additional remediation costs to the Forest Service would be about \$87,000 per year for the first ten years of work. In the greater scheme of things, this is not a large amount of money, especially if it can bring most conditions into compliance with LAC standards in the fourth largest Wilderness in the continental United States. However, avoiding creation of excessively impacted places in the first place would be much less costly.

Table 4—Financial costs of the Seven Lakes restoration program, 1992-1998, broken out by costs to the Forest Service and costs contributed by volunteers.

	Forest Service	Contributed ^a	Totals
Planning	16,375	—	16,375
Restoration			
Seed collecting/cleaning	250	—	250
Seed propagation	500	2,000	2,500
Restoration work			
SCA crew	20,000	24,000	44,000
AMC crew	2,000	7,680	9,680
IDAWA crew	—	15,000	15,000
Volunteers	10,500	20,000	30,500
Wilderness rangers	8,000	—	8,000
Pack support	6,500	—	6,500
Materials	1,000	—	1,000
Monitoring	750	—	750
Signage	500	—	500
Forest Service cost	66,375	—	—
Contributed cost ^a	—	68,680	—
Total cost	—	—	135,055

^a The cost of work contributed by volunteers estimated at \$10/hour.

Discussion

Campsite impacts in the Seven Lakes Basin are comparable to those of many other moderately used wilderness destinations, where many visitors travel with pack stock. In many such destinations, existing conditions are not in compliance with management standards, and these places need to be more intensively managed. There are often many more campsites than necessary, and campsites are often heavily impacted. Pack stock impacts are particularly pronounced. For example, on Seven Lakes campsites in 1993, about 50% of the bare area and disturbed area were on sites used exclusively for confining stock. In addition, virtually all root exposure is a result of tying horses to trees. This suggests that camping with pack stock at least doubles the amount of disturbance that would be caused by camping without pack stock. Many wildernesses with these problems have had difficulty devising effective management programs to deal with them, particularly where horseback groups have been highly vocal and wield considerable political power. The Seven Lakes Plan provides a case study for evaluating the benefits and costs of a management plan that attempts to deal effectively with these problems, without curtailing use or significantly reducing freedoms.

Our results clearly show that the Seven Lakes Basin restoration program has been highly successful in reducing impacts associated with camping. Considerable progress has been made in the five years since the plan was initiated. Campsite densities have decreased slightly, and magnitude of impact has decreased on virtually all campsites and has decreased greatly on many sites. In just five years, disturbed area has decreased 37%, and bare area has decreased 43%. Disturbed area and bare area have declined at least 10% on 16 of the 26 campsites. Tree scarring has declined, although primarily from masking scars with pitch and charcoal. Vegetation cover has increased and mineral soil exposure

has decreased. Only root exposure has gotten worse. Moreover, if the management program is continued, the greatest positive changes are still to come. Disturbed area and bare area are likely to decline in a few decades to just 36% and 24%, respectively, of what they were in 1993.

Most of these positive changes come from confining where camping can occur, particularly by groups with pack stock. Improving conditions on former stock-holding areas have more than compensated for the increased impact on newly designated stock-holding areas. The closure of some campsites to all use and efforts to reduce the size of open campsites, through both closure and restoration of portions of large sites, have also been highly effective. Reductions in maximum group size have undoubtedly contributed to success. For these benefits to continue or increase in the future, the programs need to remain in effect.

The fiscal costs of this program are significant. As detailed in the results section, the five-year costs exceeded \$135,000, although the Forest Service was able to reduce out-of-pocket costs by more than 50% by using volunteer groups extensively. Experiential costs appear meager. They were borne primarily by stock groups because these groups were the primary source of impact problems. Stock groups had some of their freedom of site selection removed, although each lake still had one legal stock site. They were also not allowed to graze in the basin; they had to bring in feed. Finally, their ability to travel in large groups—more than 10 head of stock—was curtailed. However, they were still allowed near-complete freedom virtually everywhere else in the 500,000 ha Selway-Bitterroot Wilderness. They could graze and camp wherever they wanted, even in large groups, immediately outside the basin and visit it during the day. In other words, stock groups were still allowed access, and their behavior remained largely unrestricted.

In conclusion, the Seven Lakes Basin management program has been more successful than expected in its first five years. Because recovery always takes more time than impact, we expected little positive change in the first five years. In the moderately resilient environments of the northern Rocky Mountains, positive benefits occurred in the first five years. In a few decades, the program will probably reduce campsite impacts by at least two-thirds. This illustrates that the confinement strategy can be highly effective, particularly with types of use that have more potential to cause impact, such as stock groups. The Seven Lakes Basin restoration program provides a good model for the vast majority of wildernesses that have problem areas in which campsite impact is unacceptably high.

Our study also shows that it is possible to take management actions to bring conditions back into compliance with LAC standards, even without limiting use or seriously restricting user behavior. The concern is that it is very costly—from the perspective of extremely limited funds for wilderness management. Costs can be kept to reasonable levels by using volunteers extensively and by not tackling too many different destinations at one time. This problem also illustrates the need to prevent problems in the first place, rather than attempt to correct them after they have

already occurred, particularly with the types of use that can to cause substantial disturbance. This is one reason for rethinking the common principle of not taking restrictive actions until it is clear that nonrestrictive actions have failed (Cole 1995). It is important to anticipate where impact is likely to occur and to take effective, preventive actions, even if they need to be restrictive.

Finally, in addition to being costly, restoring recreation impact will be a slow and never-ending process. At Seven Lakes, the management program can now shift into more of a maintenance mode, perhaps selecting another problem area to restore. However, in the maintenance mode, restrictions must be kept in force, and frequent ranger presence is still needed to obtain reasonable compliance. Given the minimal budgets for on-the-ground wilderness management, even the maintenance mode will stretch available resources.

References

- Cole, David N. 1983. Campsite conditions in the Bob Marshall Wilderness, Montana. Research Paper INT-312. Ogden, UT: U.S. Department of Agriculture, Intermountain Research Station. 18 p.
- Cole, David N. 1989. Wilderness campsite monitoring methods: a sourcebook. General Technical Report INT-259. Ogden, UT: U.S. Department of Agriculture, Intermountain Research Station. 57 p.
- Cole, David N. 1993. Campsites in three western wildernesses: proliferation and changes in condition over 12 to 16 years. Research Paper INT-463. Ogden, UT: U.S. Department of Agriculture, Intermountain Research Station. 15 p.
- Cole, David N. 1995. Wilderness management principles: science, logical thinking or personal opinion? *Trends*. 32 (1), 6-9.
- Cole, D. N.; Spildie, D. R. 1998. Hiker, horse and llama trampling effects on native vegetation in Montana, USA. *Environmental Management*. 53: 61-71.
- DeLuca, T.H.; Patterson, IV, W.A.; Freimund, W.A.; Cole, D.A. 1998. Influence of llamas, horses, and hikers on soil erosion from established recreation trails in western Montana, USA. *Environmental Management*. 22: 255-262.
- Marion, Jeffrey L. 1991. Developing a natural resource inventory and monitoring program for visitor impacts on recreation sites: a procedural manual. Natural Resources Report NPS/NRVT/NRR-91/06. Denver, CO: U.S. Department of Interior, National Park Service. 59 p.
- Marion, Jeffrey L. 1995. Capabilities and management utility of recreation impact monitoring programs. *Environmental Management*. 19, 763-771.
- McClaran, Mitchel P.; Cole, David N. 1993. Packstock in wilderness: use, impacts, monitoring, and management. General Technical Report INT-301. Ogden, UT: U.S. Department of Agriculture, Intermountain Research Station. 33 p.
- McCool, Stephen F. and Cole, David N., comps. 1997. Proceedings—limits of acceptable change and related planning processes: progress and future directions. General Technical Report INT-371. Ogden, UT: U.S. Department of Agriculture, Intermountain Research Station. 84 p.
- Weaver, T.; Dale, D. 1978. Trampling effects of hikers, motorcycles and horses in meadows and forests. *Journal of Applied Ecology*. 15: 451-457.
- Whinam, J.; Comfort, M. 1996. The impact of commercial horse riding on sub-alpine environments at Cradle Mountain, Tasmania, Australia. *Journal of Environmental Management*. 47: 61-70.

Would Ecological Landscape Restoration Make the Bandelier Wilderness More or Less of a Wilderness?

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Abstract—The purpose of this paper is to foster discussion on the basic issue of whether it is appropriate or not to intervene in designated wilderness areas that have been “trammed by man” and, as a result, no longer retain their “primeval character and influence.” We explore this wilderness management dilemma (whether we can or should actively manage wilderness conditions to restore and protect wilderness and other values) by asking seven questions relating to a wilderness area that is no longer “natural.” (For the purposes of this discussion, “natural” is defined by words and phrases used in the 1964 Wilderness Act: “a community of life untrammed by man”; “land retaining its primeval character and influence”; and or existing in an “unimpaired condition.”) Debate on this issue is not new, but is intensifying, since most wilderness areas in the continental United States are not pristine and ecosystem research has shown that conditions in many are deteriorating. To facilitate dialog on this wilderness management topic we focus on a case-study of a proposed large-scale project to restore piñon-juniper woodlands in the Bandelier Wilderness, New Mexico.

Many ecosystems in the Bandelier Wilderness (23,000+ acres in Bandelier National Monument, New Mexico) exhibit human-caused damage and unsustainable trends because of a land use history that includes federally sanctioned overgrazing and fire suppression over the past century. This situation has caused park managers and wilderness advocates to ask several important philosophical and practical questions that must be carefully addressed to manage wilderness in general, and the Bandelier Wilderness in particular:

- Does a Park’s enabling legislation (or the National Park Service Organic Act) reign supreme and, if so, at what cost to other resource values, including wilderness values, recognized later in a Park’s history?
- Should federal land managers intervene if wilderness ecosystems are degraded and unsustainable due to the historic activities of motorized societies?
- Can we restore the “natural range of variability” and will it be sustainable?

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- If restoration is possible, what should our goal (target conditions) be in wilderness?
- If current wilderness conditions warrant urgent management attention, are drastic restorative measures justified?
- Is it appropriate to conduct large-scale ecosystem restoration work in wilderness?
- If we start manipulating wilderness to reach an “unimpaired condition” goal, when and where will management intervention end?

Bandelier Wilderness Case Study

A case-study is used to explore, but not definitively answer, these questions. Through these questions, we hope to initiate dialog that will result in informed decisions for the long-term management of Bandelier and other wilderness areas in the National Wilderness Preservation System (NWPS).

Question 1: Does a Park’s enabling legislation (or the National Park Service Organic Act) reign supreme and, if so, at what cost to other resource values, including wilderness values, recognized later in a park’s history?

Question 1 is the easiest to address since the answer is contained within the 1964 Wilderness Act (P.L. 88-577). The act simultaneously limits and permits management action to protect both park and wilderness values (which are arguably the same). In addition, the act makes it clear that wilderness designation does not supercede a park’s enabling legislation or the National Park Service (NPS) Organic Act, but is supplemental to it. Section 4(a)(3) states that: “Nothing in this Act shall modify the statutory authority under which units of the national park system are created. Further, the designation of any area of any park, monument, or other unit of the national park system as a wilderness area pursuant to this Act shall in no manner lower the standards evolved for the use and preservation of such park, monument, or other unit of the national park system in accordance with the Act of August 25, 1916, the statutory authority under which the area was created,....” Section 4(a)(3) makes it clear that the NPS has the legal responsibility to meet its mission requirements and other mandates even in wilderness areas. These provisions are similarly stated for other wilderness management agencies (Section 4(a) and (b)).

In Section 4(b), the act gives the NPS (in this case) responsibility for meeting its mission as well as preserving “wilderness

character.” Unfortunately, wilderness character is not clearly defined and, thus, a dilemma arises for the wilderness ecosystem manager. To some, “wilderness character” means that wilderness areas should evolve in whatever direction nature chooses (be free-willed) after the lands have been designated as wilderness, regardless of pre-existing condition or future consequences. This means that all resource managers (including wilderness/ecosystem restorationists) and researchers should not be permitted to do anything in wilderness using motorized equipment. This position is not wholly supported in the act, as in Section 2(a), the act calls for the preservation, protection and administration of wilderness areas “in such a manner as to leave them unimpaired for future use and enjoyment as wilderness....” Section 4(c) of the act gives the wilderness administrator strong direction to accomplish the preservation and protection task without motorized equipment, but it also permits its use if there is justifiable need “to meet requirements for the administration of the area for the purpose of this Act....”

The 1916 NPS Organic Act dictates that the NPS mission is “to conserve the scenery and the natural and historic objects and the wildlife therein and to provide for the enjoyment of the same in such manner and by such means as will leave them unimpaired for the enjoyment of future generations.” Bandelier National Monument (Park, Bandelier), as one of the oldest units in the national park system, was established in 1916 to preserve and protect “prehistoric aboriginal ruins” on the Pajarito Plateau because of their “unusual ethnologic, scientific, and educational” values.

In October 1976, President Gerald Ford signed legislation creating the Bandelier Wilderness, including 23,267 acres. The NPS was initially opposed to this wilderness designation, in part because of a general concern that cultural resources research and management in a “traditional cultural resource park” could be severely constrained. The Bandelier Wilderness was one of the first NPS wilderness areas authorized in New Mexico after passage of the 1964 Wilderness Act. The Bandelier Wilderness, like most wilderness areas in the NWPS, was not pristine when it was created due to a history of harmful EuroAmerican land use practices, yet the public felt strongly that the area belonged in the NWPS (McDonald 1987). Additional wilderness-quality lands were added to the Park in 1977, so that today approximately 71% of the Park is designated wilderness, while more than 90% (about 30,000 acres) is managed as wilderness.

Scientific study in and adjacent to the Bandelier Wilderness since 1987 strongly supports the notion that historic EuroAmerican use of the area has triggered unprecedented change in most Park ecosystems (Allen 1989; Davenport and others 1998); similar changes have occurred throughout much of the Southwest (Allen and others 1998; Bogan and others 1998). For example, federally sanctioned livestock grazing and fire suppression from 1880 through 1932 catalyzed severe accelerated soil erosion across the Park’s extensive mesas that are dominated by piñon-juniper (PJ) woodlands (Gottfried and others 1995; Wilcox and others 1996). These old, relatively shallow soils are the physical matrix for thousands of “aboriginal ruins” that Bandelier National Monument was established to protect beginning in 1916 (Head 1992; unpublished data on file at Bandelier

National Monument). The Bandelier Wilderness contains significant portions of these altered ecosystems and “ab- original ruins.” Over 90% of the Park’s 11,730 acres of PJ woodlands are within designated wilderness – thus, resolution of any resource issues related to PJ woodlands necessarily involves wilderness considerations. In particular, the majority of documented archeological sites at Bandelier occur in PJ woodland settings (Gottfried and others 1995), and recent extensive and detailed surveys indicate that more than 80% of the PJ archeological sites are being damaged by one or more types of erosion impacts (Head 1992; unpublished data on file at Bandelier National Monument). An estimated 2,500 cultural resource sites located in the Bandelier Wilderness are subject to accelerated erosion-caused damage, or risk of complete loss, within the next century.

The NPS, to accomplish its protection and conservation mandate, must respond to known resource threats within the Bandelier Wilderness. Based on extensive experimentation, it appears that the most effective and least damaging management response to the erosion problem in the Bandelier PJ woodlands will likely require use of motorized equipment (see question 3). However, a minimum tool analysis has yet to be completed for this case-study so the extent of motorized equipment use, if any, is uncertain at this time. In any case, the potential to use motorized equipment to control unnatural rates of erosion appears to be permitted under the provisions of the Wilderness Act, as we demonstrated at the beginning of this discussion.

Question 2: Should federal land managers intervene if wilderness ecosystems are degraded and unsustainable due to the historic activities of motorized societies?

The answer to question 2 is a matter of opinion since some agencies and wilderness advocates disagree on the fundamental issue of wilderness ecosystem restoration or management intervention. Let us look at the Bandelier PJ woodlands case-study for some key facts that could influence perspectives on this case.

While some uncertainties persist on the nature of historic ecological changes in PJ woodlands of the Bandelier area, a great deal of research work has been conducted (and continues) on the ecology, hydrology, archeology and land use histories of local woodlands. Synthesizing existing information with published research from other areas, along with consultations with local resource managers and researchers, leads to the following general scenario of changes in the Bandelier PJ woodlands (Allen 1989; Davenport and others 1998; Gottfried and others 1995; Reneau and McDonald 1996).

Woodland soils in Bandelier likely formed, to a large degree, under different vegetation during cooler, moister conditions of the late Pleistocene; in other words, they are over 10,000 years old, and many are over 100,000 years old (McFadden and others 1996). Changes in climate and vegetation in the early Holocene (8,500-6,000 years ago) led to at least localized episodes of soil erosion on adjoining uplands (Reneau and McDonald 1996, Reneau and others 1996). During this time, the dominant climatic and associated vegetation patterns of the modern southwestern United States developed, including PJ woodlands and savannas

(Allen and others 1998). On the basis of local fire history (Allen 1989; Morino and others 1998; Touchan and others 1996), PJ age class (Bandelier National Monument, unpublished data; Julius 1999) and soils data (Davenport 1997; Earth Environmental Consultants 1974; McFadden and others 1996), we believe that Bandelier's PJ woodlands were formerly more open, with well-developed herbaceous understories that: 1) protected the soils from excessive erosion during intense summer thunderstorm events, and 2) provided a largely continuous fuel matrix, which allowed surface fires to spread through the woodland zone from the adjoining ponderosa pine and grassland types.

Native American effects on local woodlands are thought to have been insignificant or highly localized until the late 12th century, when the Ancestral Puebloan (also referred to as the Anasazi) population began to intensively occupy and utilize the Bandelier area (Powers and Orcutt 1999). Cutting and burning of PJ trees for cooking, heating, building and agricultural activities likely led to significant deforestation of upland mesas from about 1150-1550 A.D. Thus, Ancestral Puebloan land use practices favored herbaceous vegetation. Intensive soil disturbance certainly occurred in farmed areas and around habitations, but there was probably little net change in landscape-wide erosion rates due to the small size and dispersed locations of "fields" and villages.

EuroAmerican settlement of the adjoining Rio Grande valley and the introduction of domestic livestock grazing began in 1598. It is unlikely, however, that significant livestock grazing (that is, with substantial effects on the herbaceous understory, fire regime or erosion rates) took place in much of Bandelier until railroads linked the Southwest to commercial markets in the 1880s. Millions of sheep and cattle were placed in the New Mexico landscape at that time (Bogan and others 1998). Livestock grazing was allowed in Bandelier until 1932, and feral burros were similarly allowed to cause grazing impacts until about 1980 (Allen 1989). The resultant high intensity grazing apparently triggered a number of ecological changes in local PJ woodlands. Overgrazing caused sharp reductions in the herbaceous ground cover and associated organic litter, effectively suppressing previously widespread surface fires (in concert with institutionalized fire suppression initiated by the federal government in the early 1900s). Exacerbated by severe drought in the 1950s (Allen and Breshears 1998), the reduced cover of herbaceous vegetation and litter also led to decreased water infiltration and increased surface runoff from the typically intense local rainfall events. Given reduced herbaceous competition and the elimination of surface fires over the past 120 years, fire-sensitive piñon and juniper trees became established in densities unprecedented for at least the past 800 years (Bandelier National Monument, unpublished data; Julius 1999). As these trees grew, they became increasingly effective competitors for water and nutrients. Thus, a positive feedback cycle was initiated that favors tree invasion and decreased herbaceous ground cover in mesa-top settings.

This land use history has caused degraded and unsustainable ecosystem conditions in today's Bandelier Wilderness, particularly the sparsely vegetated and eroding soils that characterize understory patterns in the PJ woodlands. For example, three kilometers of line transect data from Bandelier woodlands in the 1990s document herbaceous plant cover

(basal intercept) of only 0.4 to 9% *versus* exposed bare ground of 38 to 75% (Bandelier National Monument, unpublished data; Gottfried and others 1995), and the intense summer thunderstorms typical of this region result in high rates of runoff and soil erosion (Davenport and others 1998; Reid and others 1999; Wilcox and others 1996a and 1996b). The intercanopy soils of Bandelier's woodlands are apparently eroding at net rates of up to one-half inch per decade (Bandelier National Monument, unpublished data; Earth Environmental Consultants 1974; Wilcox and others 1996a/b). Given soil depths averaging only one to two feet in many areas (Davenport 1997; Wilcox and others 1996a), there will soon be loss of entire soil bodies across extensive areas of the Bandelier Wilderness.

Ecological thresholds have apparently been crossed such that harsh physical processes are now dominant across Bandelier's degraded PJ woodlands (Davenport and others 1998). The loss of organic topsoils, decreased plant-available-water, extreme soil surface temperatures and freeze-thaw activity severely impede herbaceous vegetation establishment and productivity (Davenport and others 1998; Jacobs and Gatewood 1999; Loftin 1999). Reestablishment of herbaceous ground cover under today's desertified mesa-top conditions may also be difficult due to depleted soil seed banks, highly efficient seed predators, particularly harvester ants (Snyderman and Jacobs 1995), and an unnaturally large elk population (Allen 1996b). Herbivore exclosures established in 1975 show that protection from grazing, by itself, fails to promote vegetative recovery in Bandelier's PJ ecosystems (Chong 1992; Potter 1985). Without management intervention, this human-induced episode of accelerated soil erosion appears to be highly persistent and irreversible (Davenport and others 1998).

In conclusion, the present appearance and dominant ecological processes of the Bandelier Wilderness are to a large degree an anthropogenic legacy of the past land use practices of our motorized society. This history includes substantial (though inadvertent) contributions by federal land managers to the current unsustainable situation of accelerated, landscape-wide soil erosion in the PJ woodlands. While a basic tenet of wilderness is that the "imprint of man's work [is] substantially unnoticeable," human impact on essential ecological patterns and processes is profound in the Bandelier Wilderness. If one believes that long-term protection of natural ecosystem function and appearance is important in wilderness, management intervention may be warranted. On the other hand, if one believes that wilderness is defined exclusively by the absence of apparent evidence of human management in the short-term, then management intervention is not warranted in the Bandelier case-study. For additional discussion on this issue see Landres and others in these proceedings.

Question 3: Can we restore the "natural range of variability" and will it be sustainable?

The answer to question 3 lies in scientific study, to define the natural range of variability, and experimentation, to address and test sustainability. Let us look again at the Bandelier PJ woodlands case-study to see what has been discovered.

Since most of the soils of the Park's PJ woodlands are over 100,000 years old (McFadden and others 1996) we can

be sure that the natural range of variability in these ecosystems generally allowed for soil development and stability, rather than the high rates of degradational erosion observed in recent decades. From this fact of long-term soil persistence we can infer that some type of vegetation was protecting the soils from excessive erosion over time, including the last 8000+ years of the Holocene during which a basically modern climatic regime prevailed. We can also determine that herbaceous vegetation must have been the now-missing glue for the soils, given that there is no evidence of formerly closed-canopy woodlands (indeed, the ages of local piñon and juniper trees are largely quite young), and since fire-scar studies show a history of recurrent surface fires that could not have occurred without herbaceous vegetation.

Cessation of domestic livestock grazing in 1932 and removal of feral burros since the 1970s have been insufficient to induce vegetative restoration in degraded woodlands at Bandelier. Ecological thresholds have apparently been crossed, and physical (rather than biological) processes now dominate in Bandelier's PJ woodland areas, precluding recovery to more stable soil/vegetation conditions (Davenport and others 1998). Our research indicates that the Park's PJ woodlands are unlikely to regain any semblance of their pre-1880s condition without management intervention (Davenport and others 1998; Jacobs and Gatewood 1999). Unfortunately, the piñon-juniper ecosystems of the Bandelier Wilderness seem unable to heal themselves.

Fortunately, controlled, progressive experiments within and outside of the Bandelier Wilderness since 1992 (Chong 1993, 1994; Jacobs and Gatewood 1999; Snyderman and Jacobs 1995) have shown (at three years posttreatment) that undesirable losses of soils, herbaceous vegetation and cultural resources can be mitigated through active management, involving use of motorized equipment (chain saws), to thin the smaller trees and leave scattered slash in the form of lopped branches from cut trees. This treatment directly reduces tree competition with herbaceous plants for scarce water and nutrients, and the application of slash residues across the barren interspaces greatly reduces surface water runoff and ameliorates the harsh microclimate at the soil surface, immediately improving water availability for herbaceous plants. This restoration approach has produced a two- to sevenfold increase in total herbaceous cover (at three years posttreatment), relative to both controls and pretreatment conditions (Jacobs and Gatewood 1999), while also increasing the diversity of herbaceous plants. This tree thinning and scattered slash treatment method is labor intensive and requires extensive use of chain saws to limb and flushcut the PJ trees, given the hard, dense wood of these species (especially juniper) and the large number of trees that require treatment.

Other treatment methods to restore herbaceous ground cover were tested. Seeding in the absence of tree thinning was ineffective, and seeding combined with a thinning/slash treatment conferred little additional benefit. Alternative tree thinning techniques are unlikely to be effective, safe or practical. For example, surface fire cannot currently carry through the barren understory of Bandelier's PJ woodlands; girdling and herbicide treatment do not generate the on-the-ground slash necessary for the creation of microclimatic conditions that facilitate vegetation recovery, as dead trees would be left standing; and exclusive use of non-motorized

tools would take too long, given the urgency of the situation, and also place too many people in the wilderness environment for extended periods, causing other unacceptable wilderness impacts.

In the Bandelier case study, through scientific investigation we are confident that a "range of natural variability" (Landres and others 1999) is reasonably defined. We have also found a seemingly effective restoration technique, but the long-term outcome will only be known as time progresses. The treated areas, though initially dominated by biannual forbs, are becoming increasingly populated by native perennial grasses, which represent more natural conditions. Will the restored herbaceous cover be able to reduce erosion rates to natural, sustainable levels? Based on preliminary data, it appears likely. However, the substantial quantities and distribution of the woody slash/mulch used in this restoration approach are not natural and could support large unnaturally intense fires. The potential for widespread fire can be eliminated by limiting the size of treated areas, and dispersing them across the landscape. The resulting mosaics of fuels and vegetation will provide a margin for error and mitigate aesthetic concerns. Prescribed fire will be introduced to eliminate excessive woody fuel loads and prepare treated areas for naturally occurring fires once adequate herbaceous cover is successfully restored. Experiments will begin in AD 2000 to determine the appropriate timing and prescription for the initial reintroduction of fire.

Question 4: *If restoration is possible, what should our goal (target conditions) be in wilderness?*

Achieving agreement on target conditions can be seen as the crux of the wilderness restoration dilemma. Ideally, target conditions (a range of natural ecosystem structures and naturally functioning processes) exist when a wilderness area is set aside. However, established wildernesses are generally far from pristine—that is, they do not fully retain their "primeval character and influence...." The Bandelier Wilderness provides a well-studied example.

The current resource management vision (desired condition) for Bandelier, including the Bandelier Wilderness, is that:

Natural and cultural resources are promoted and preserved within naturally-functioning and sustainable environmental conditions as existed prior to modern human influence (that is prior to landscape-level livestock grazing and wildfire suppression and following Ancestral Puebloan occupation of the area).

This vision of target conditions for PJ woodlands within Bandelier is functional, as opposed to structural or compositional. In this case, our goal is to have biological processes once again control the rate of erosion and natural fires move across the landscape unimpeded, restoring a natural range of variability. The time it will take to reach sustainability and to test our fire maintenance hypothesis is not yet known. As mentioned in question 3, we have the funds and will initiate restoration-focused fire research in PJ woodlands within and outside of designated wilderness beginning in AD 2000.

Please note that we do not say anything about what the Bandelier Wilderness will "look like" in our target condition statement. The type of experience a person may have in the wilderness is also not defined. We believe these are important omissions because, although wilderness involves scenery and

“human experience” management, it is not necessarily or solely defined by them. Others undoubtedly will disagree with us—thus, the dilemma. Another way of looking at this dilemma is to decide whether management intervention is a form of “trammeling.” Do two trammels, however well intentioned, make a right?

Question 5: *If wilderness conditions warrant urgent management attention, are drastic restorative measures justified?*

The answer to this question, like question 2, is a matter of opinion. The key difference between these questions is question 5's focus on urgency. (The question of magnitude is addressed in question 6.)

Our research data show that the high rates of soil erosion recently measured in Bandelier's PJ woodlands are rapidly degrading the Park's shallow soils and damaging thousands of archeological sites, and that this condition is the result of the actions of a motorized society. We know that delaying or taking no action to mitigate the unnaturally accelerated erosion rates in Bandelier's PJ woodlands will have irreversible adverse consequences for the Park's soils and cultural resources. Every rain event reduces the information-yielding potential of the “aboriginal ruins.” For example, in a single storm on June 29, 1995, 1,040 artifacts were transported off-site and captured in a 1m³ sediment trap at the mouth of a 0.1 hectare catchment basin (Bandelier National Monument, unpublished data). To a significant degree, the Park's biological productivity and cultural resources are literally washing away.

While the Bandelier resource loss data are compelling, we recognize that caution must be exercised when interpreting research findings, given the inherent limitations and uncertainties in all scientific endeavors. For the sake of discussion, however, let's assume that the findings in the Bandelier case-study are scientifically sound and we can be confident that the “natural range of variability” in wilderness conditions, as outlined by Landres and others (1999) and Swetnam and others (1999) is adequately known. Do current conditions and their causes justify taking corrective actions? After all: 1) erosion is a ubiquitous geomorphic process; 2) localized episodes of accelerated erosion have occurred naturally in the past (Reneau and others 1996); and 3) it is impractical to preserve the cultural resource sites at Bandelier in stasis. Further, some Native Americans do not want the NPS manipulating the landscape or archeological sites for any reason, even to stabilize ancestral sites. In addition, some wilderness advocates are understandably concerned about a loss of “wildness” if local land managers have too much latitude to manipulate wilderness resources, even to achieve high-minded and defensible goals.

Given this information, there is no question that we must assess the problem and possible solutions cautiously and responsibly. The decision to implement drastic restoration measures must be made with extreme humility. Yet, it is clear that delays in making this decision in the Bandelier Wilderness come at the cost of ongoing resource loss, since we are losing the intercanopy soils due to high erosion rates and the soils are relatively shallow. Many, eventually thousands, of cultural resource sites will also be damaged or lost since at least 80% of the sites within the PJ woodlands have documented erosion problems.

Societal opinion about large-scale wilderness restoration efforts undoubtedly hinges upon a more complete understanding of the issues and thoughtful evaluation of the potential consequences of alternative actions, including “no action,” to the Bandelier Wilderness and its associated cultural resources. The NEPA process will be used as the primary vehicle through which the NPS and the public will formally assess trade-offs and uncertainties to determine if “drastic restorative measures” to protect cultural resources, soils and ecosystems are justified in the Bandelier Wilderness.

Question 6: *Is it appropriate to conduct large-scale ecosystem restoration work in wilderness?*

The NPS Organic Act and other federal laws mandate protection of park and wilderness resources and values when we know they are threatened (refer to question 1 discussion). In response to these laws, resource management activities such as exotic plant control, application of prescribed fire and wildlife reintroductions are routinely and legally accomplished in federal wilderness areas, as wilderness administration and resource management decisionmaking power are vested to the federal wilderness manager through the 1964 Wilderness Act. None of these laws, including the Wilderness Act, specify that a “no action” decision is justifiable based solely on the magnitude or scale of the possible mitigation alternatives. Therefore, NPS resource managers are obligated to: 1) consciously decide on a course of action when we detect a threat no matter how large or significant, and 2) make responsible decisions about the type and scale of our response to all kinds of resource threats.

The actions proposed for restoration of Bandelier's PJ woodlands will likely require the use of motorized equipment (that is, chain saws). If treated, portions of 8,000 acres of Bandelier Wilderness PJ woodlands will contain scattered evidence of modern peoples, in the form of cut marks on small stumps and scattered slash mulch, for about two decades—the estimated time it will take for natural processes like fire and decomposition to consume the small stumps and slash. Does the large scale of the possible Bandelier Wilderness management action make a difference? From a strictly legal perspective, the answer appears to be “No.” This answer does not make the action ethically correct, however.

The Bandelier Wilderness PJ woodlands restoration project is considered relatively large-scale. Yet, although scale does matter because it affects the cumulative magnitude of the potential effects, the size of the proposed action is not the only important consideration and should not be preeminent in our opinion. A central question in the Bandelier case-study might be: Is a large-scale, management-generated impact of relatively short-term duration acceptable in designated wilderness to restore and sustain “naturalness” or “wildness” and to preserve the prehistoric cultural resources for which Bandelier National Monument was established? Based on our mulch treatment tests, evidence of management intervention superficially disappears within 5 to 10 years depending on site conditions. We hypothesize that if fire is reintroduced to accelerate woody material decomposition and degrade the low flush cut stumps, the evidence of management intervention will be substantially undetectable in 20 years. To deal effectively with the threat

of a wildfire consuming the woody materials too soon after treatment, we must treat the woodlands in patches, thus creating a mosaic of conditions and appearances. Perhaps the duration of the evidence of management intervention matters more than the spatial extent or appearance of that evidence.

Obviously, the answer to question 6, like questions 2 and 5, is a matter of individual perspective, values and opinion.

Question 7: *If we start manipulating wilderness to reach an “unimpaired condition” goal, when and where will management intervention end?*

Question 7 must be answered if management intervention is to be seriously contemplated. There is justifiable public concern that federal wilderness managers could abuse the wilderness resource in the name of ecosystem health restoration. Management intervention should not be a license to control nature, harvest resources or create stasis; it should be a means of facilitating natural healing of motorized societies’ impacts to wilderness ecosystems.

We believe that question 7 (along with #’s 3 and 4) can only be addressed through extensive scientific research both to diagnose the health of wilderness ecosystems and to understand the causes of unnatural change. We suggest that management intervention should end when the natural processes present prior to industrial-age humans are once again working in formerly dysfunctional or “impaired” ecosystems. In the Bandelier case-study, based on over 10 years of on-site research, this end point would be achieved when there is sufficient herbaceous cover to carry naturally occurring fires. The herbaceous cover will reduce soil erosion (and cultural resource loss) to natural rates, and fire should maintain the restored herbaceous cover and prevent recurrence of the erosion problem. After restoration, the PJ wilderness ecosystem will be left alone to evolve, driven by natural processes. We submit that this level of restoration would restore important aspects of wildness or “free will” to the Bandelier Wilderness, consistent with the definition of wilderness established in the 1964 Wilderness Act.

Conclusions

One of the penalties of an ecological education is that one lives alone in a world of wounds. Much of the damage inflicted on land is quite invisible to laymen. An ecologist must either harden his shell and make believe that the consequences of science are none of his business, or he must be the doctor who sees the marks of death in a community that believes itself well and does not want to be told otherwise. (Aldo Leopold)

Although there are no simple answers to the wilderness questions presented in this paper, we suggest that a research-based management approach, including identification of a process-oriented goal to achieve an ecologically functional endpoint, sets the stage for making rational decisions about whether and how to intervene when natural conditions do not exist in wilderness areas. As Aldo Leopold pointed out in the quote above (Leopold 1953), we have a choice when we know that the “land is sick.” We can “make believe” that everything will turn out right if nature is left to take its course in our unhealthy wildernesses, or we can intervene to facilitate the healing process.

The Bandelier piñon-juniper woodlands case-study is used in this paper to explore key issues, trade-offs and uncertainties inherent to the wilderness restoration dilemma. While definitive answers are not presented, this case-study is an opportunity for further discussion on an old, thorny and increasingly vital philosophical question: If wilderness managers intervene to restore unnaturally functioning ecosystems, does a designated wilderness area become more or less of a wilderness, as defined under the 1964 Wilderness Act?

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References

- Allen, C. D. 1989. Changes in the landscape of the Jemez Mountains, New Mexico. Ph.D. dissertation, Department of Forestry and Resource Management, Univ. of California, Berkeley. 346 p.
- Allen, C. D., (tech. ed.). 1996a. Fire effects in southwestern forests: Proceedings of the second La Mesa Fire Symposium. USDA Forest Service General Technical Report RM-286, Fort Collins, CO. 216 p.
- Allen, C. D. 1996b. Elk population response to the La Mesa Fire and current status in the Jemez Mountains. pp.179-195 In: C.D. Allen (tech. ed.), Fire Effects in Southwestern Forests: Proceedings of the Second La Mesa Fire Symposium. USDA Forest Service General Technical Report RM-286, Fort Collins, CO. 216 p.
- Allen, C. D., and D. D. Breshears. 1998. Drought-induced shift of a forest/woodland ecotone: rapid landscape response to climate variation. Proceedings of the National Academy of Sciences of the United States of America 95:14839-14842.
- Allen, C. D., J. L. Betancourt, and T. W. Swetnam. 1998. Landscape changes in the southwestern United States: Techniques, long-term datasets, and trends. pp. 71-84 In: T. D. Sisk (ed.), Perspectives on the Land Use History of North America: A Context for Understanding our Changing Environment. U.S. Geological Survey, Biological Science Report USGS/BRD/BSR-1998-0003.
- Bogan, M.A., C.D. Allen, E.H. Muldavin, S.P. Platania, J.N. Stuart, G.H. Farley, P. Melhop, and J. Belnap. 1998. Southwest. pp. 543-592 In: Mac, M.J., P.A. Opler, and P.D. Doran (eds.), National Status and Trends Report, U.S. Geological Survey, Washington, D.C. 964 p.
- Bandelier National Monument unpublished data. Unpublished data on file at Park Headquarters, Resource Management Office, Bandelier National Monument, Los Alamos, New Mexico.
- Chong, G.W. 1992. Seventeen Years of Grazer Exclusion on Three Sites in Pinyon-Juniper Woodland at Bandelier National Monument, New Mexico. Unpublished report on file at Bandelier National Monument. 23 p.
- Chong, G. W. 1993. Revegetation of piñon-juniper woodlands with native grasses. pp. 34-41 In: E. F. Aldon and D. W. Shaw (tech. coord.), Managing Piñon-Juniper Ecosystems for Sustainability and Social Needs. USDA For. Serv. Gen. Tech. Rep. RM-236, Fort Collins, CO.
- Chong, G. W. 1994. Recommendations to improve revegetation success in a piñon-juniper woodland in New Mexico: A hierarchical approach. M.S. thesis, University of New Mexico, Albuquerque, NM.
- Davenport, D. W. 1997. Soil survey of three watersheds on South Mesa, Bandelier National Monument, New Mexico. Unpublished

- final report on file at USGS Jemez Mts. Field Station. 60 pp. + maps, data sheets, field notes, lab test results.
- Davenport, D. W., D. D. Breshears, B. P. Wilcox, and C. D. Allen. 1998. Viewpoint: Sustainability of piñon-juniper ecosystems—A unifying perspective of soil erosion thresholds. *J. Range Management* 51(2):229-238.
- Earth Environmental Consultants Incorporated. 1974. Soil survey and survey of range and ecological conditions on a southern part of Bandelier National Monument. Unpublished report on file at Bandelier National Monument. 29 p.
- Gottfried, G. J., T.W. Swetnam, C.D. Allen, J.L. Betancourt, and A. Chung-MacCoubrey. 1995. Pinyon-juniper woodlands. pp. 95-131 (Chapter 6) In: D.M. Finch and J. A. Tainter (tech. eds.), *Ecology, Diversity, and Sustainability of the Middle Rio Grande Basin*. USDA For. Serv. Gen. Tech. Rep. RM-GTR-268.
- Head, G. N. 1992. The Bandelier archeological survey: 1991 preliminary report. Unpublished NPS report on file at Bandelier National Monument. 30 p.
- Jacobs, B. F., and R. G. Gatewood. 1999. Restoration studies in degraded piñon-juniper woodlands of north-central New Mexico. pp. 294-298 In: Monsen, S. B., R. Stevens, R. J. Tausch, R. Miller and S. Goodrich (eds.), *Proceedings: Ecology and Management of Pinyon-Juniper Communities within the Interior West*. USDA Forest Service Proc. RMRS-P-9. Ogden, UT. 411 p.
- Julius, C. 1999. A comparison of vegetation structure on three different soils at Bandelier National Monument, New Mexico, USA. Unpublished report on file at Bandelier National Monument. 72 p.
- Landres, P., M.W. Brunson, L. Mergliano, C. Sydoriak, and S. Morton. 1999. In: Cole, D. N., McCool, S.F. 2000. *Proceedings: Wilderness science in a time of change*. Proc. RMRS-P-000. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Landres, P., P. Morgan, and F. Swanson. 1999. Evaluating the usefulness of natural variability in managing ecological systems. *Invited Feature on Historical Variability Concepts in Managing Ecosystems*. *Ecological Applications* 9:1179-1188.
- Leopold, L.B. (editor). 1953. *Round river: From the journals of Aldo Leopold*. Oxford University Press, New York. 173 p.
- Loftin, S. R. 1999. Initial response of soil and understory vegetation to a simulated fuelwood cut of a pinyon-juniper woodland in the Santa Fe National Forest. pp. 311-314 In: Monsen, S.B., R. Stevens, R.J. Tausch, R. Miller and S. Goodrich (eds.), *Proceedings: Ecology and Management of Pinyon-Juniper Communities within the Interior West*. USDA Forest Service Proc. RMRS-P-9. Ogden, UT. 411 p.
- McDonald, C. 1987. *The dilemma of wilderness*. Sunstone Press. Santa Fe, New Mexico. 115 p.
- McFadden, L.D., P.M. Watt, S.L. Reneau, and E.V. McDonald. 1996. General soil-landscape relationships and soil-forming processes in the Pajarito Plateau, Los Alamos National Laboratory area, New Mexico. pp. 357-366 In: F. Goff, B.S. Kues, M.A. Rogers, L.D. McFadden, and J.N. Gardner (eds.), *New Mexico Geological Society Guidebook, 47th Field Conference, Jemez Mountains Region*. 484 p.
- Morino, K., C. H. Baisan, and T. W. Swetnam. 1998. Expanded fire regime studies in the Jemez Mts., New Mexico - final report. Unpublished report on file at USGS Jemez Mts. Field Station. 94 p. + data files.
- Powers, R.P. and J.D. Orcutt (editors). 1999. *The Bandelier archeological survey*. Intermountain Cultural Resource Management Professional Paper No. 57. National Park Service, Santa Fe, New Mexico. 651 p.
- Potter, L.D. 1985. Re-evaluation studies of grazing exclosure plots, Bandelier National Monument. Unpublished report on file, Bandelier National Monument. 39 p.
- Reid, K. D., B. P. Wilcox, D. D. Breshears, and L. MacDonald. 1999. Runoff and erosion in a piñon-juniper woodland: Influence of vegetation patches. *Soil Science Society of America Journal* 63:1869-1879.
- Reneau, S. L., and E. V. McDonald. 1996. Landscape history and processes on the Pajarito Plateau, Northern New Mexico. Los Alamos National Laboratory Report LA-UR-96-3035. Los Alamos, NM. 195 p.
- Reneau, S.L., E.V. McDonald, J.N. Gardner, T.R. Kolbe, J.S. Carney, P.M. Watt, and P.A. Longmire. 1996. Erosion and deposition on the Pajarito Plateau, New Mexico: Geomorphic responses to late Quaternary climatic change. pp. 391-398 In: F. Goff, B.S. Kues, M.A. Rogers, L.D. McFadden, and J.N. Gardner (eds.), *New Mexico Geological Society Guidebook, 47th Field Conference, Jemez Mountains Region*. 484 p.
- Snyderman, D., and B. F. Jacobs. 1995. Piñon-juniper restoration studies at Bandelier – 1995. Unpublished report, Bandelier National Monument, Los Alamos, NM.
- Swetnam, T. W., C. D. Allen, and J. L. Betancourt. 1999. Applied historical ecology: Using the past to manage for the future. *Invited Feature on Historical Variability Concepts in Managing Ecosystems*. *Ecological Applications* 9:1189-1206.
- Touchan, R., Allen, C. D. and Swetnam, T. W. 1996. Fire history and climatic patterns in ponderosa pine and mixed-conifer forests of the Jemez Mountains, northern New Mexico. pp. 179-195 In: C.D. Allen (tech. ed.), *Fire Effects in Southwestern Forests: Proceedings of the Second La Mesa Fire Symposium*. USDA Forest Service General Tech. Report RM-286, Fort Collins, CO. 216 p.
- Wilcox, B. P., J. Pitlick, C. D. Allen, and D. W. Davenport. 1996a. Runoff and erosion from a rapidly eroding pinyon-juniper hillslope. pp. 61-77 In: M. G. Anderson and S. M. Brooks (eds.), *Advances in Hillslope Processes*, Volume 1. John Wiley & Sons Ltd., New York. 1306 p.
- Wilcox, B. P., B. D. Newman, C. D. Allen, K. D. Reid, D. Brandes, J. Pitlick, and D. W. Davenport. 1996b. Runoff and erosion on the Pajarito Plateau: Observations from the field. pp. 433-439 In: *New Mexico Geological Society Guidebook, 47th Field Conference, Jemez Mountains Region*, 1996.

Understanding the Factors That Limit Restoration Success on a Recreation-Impacted Subalpine Site

Catherine Zabinski
David Cole

Abstract—Factors that limit successful revegetation of a subalpine site were studied through a combination of soil assays, greenhouse studies, and field manipulations. Campsite soils had higher available nitrogen, lower microbial community diversity, and lower seed bank density than undisturbed soils. In the greenhouse, there was no significant difference in plant growth on disturbed versus undisturbed soils. In the field, seedling establishment patterns did not vary between experimental plots with five different soil treatments (ranging from a control to a compost and inoculum amendment). Addition of seeds and transplants increased seedling density, but not growth. Microclimatic variation may be the overriding limiting factor at this site.

Restoration of sites impacted by recreation in wilderness areas poses challenges for managers and researchers for several reasons. First, restoration of the site must be done rapidly to limit reuse, since a site that looks like a campsite, roped off or not, will likely be used for camping. Second, the goal of wilderness restoration is to revegetate with native species, and specifically, those found in the immediate area, so that revegetated patches will blend into the landscape. We have only limited information about suitable growing conditions for many native plant species, making it difficult to know what amendments will increase revegetation success. Finally, the distribution of designated wilderness areas, primarily in high-elevation and low-productivity habitats, means that wilderness restoration occurs in an environment that is itself challenging for plant establishment (Chambers 1997).

The purpose of our research has been to identify factors that limit successful revegetation of recreation-impacted sites, and to suggest ways to address those limitations. High-elevation sites, our focus for this work, are characterized by moderate to high stress conditions (a short growing season, high exposure and poorly developed soils) that can limit revegetation at all life stages. Recreation impacts produce conditions that represent an even higher stress environment for plant growth (fig. 1). Our research was

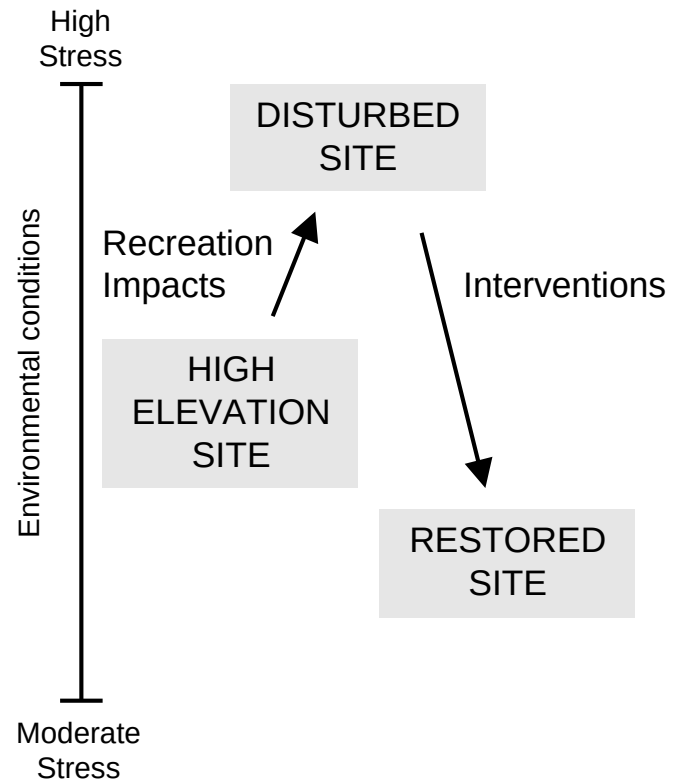


Figure 1—Stress gradient for wilderness sites impacted by recreation.

motivated by reports from wilderness managers of restoration projects in high elevation environments that met with varying success, despite similar restoration protocols. We developed a hierarchical model for addressing problematic restoration projects, which takes into account limiting factors at different plant life stages, including seed availability, seed germination, seedling establishment and plant growth (fig. 2).

Propagule availability may be limiting in undisturbed high-elevation environments to begin with (Bliss 1971), but disturbed areas may have even fewer propagules, because of disturbance to the seed bank or from loss of parent plants that produce seeds or vegetative sprouts. Conditions for seed germination also change with recreation impacts. Changes in the soil resulting from compaction and loss of surface organic matter may limit seed germination and seedling establishment through changes in soil moisture, the availability of safe sites (Urbanska 1997), or microtopographic

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Is propagule availability limiting?



Is seed germination limiting?



Is plant growth limited by

impacted soil conditions

AND/OR

microclimatic conditions

AND/OR

biotic interactions?

Figure 2—Hierarchy for addressing factors limiting to revegetation.

conditions (Harper and others 1965). Plant growth and reproduction may be affected by environmental and soil conditions (Ebert May and others 1982, Chambers and others 1990), and by biotic interactions that have changed with the disturbance.

Our research has used a combination of greenhouse, laboratory and fieldwork to address the following hypotheses:

Question 1. How do recreation impacts affect environmental conditions that may limit plant growth?

H₁: The functional diversity of the soil microbial community does not differ in recreation-impacted sites relative to undisturbed sites.

H₂: Nutrient availability does not differ between recreation-impacted sites and undisturbed sites.

H₃: Changes in the soil due to recreation impacts do not affect plant growth under greenhouse conditions.

H₄: Seed bank density and composition does not differ between recreation-impacted sites and undisturbed sites.

Question 2. What factors limit revegetation in the field?

H₅: Propagule availability is limiting revegetation.

H₆: Germination and seedling establishment are limiting revegetation.

H₇: Establishment and growth are limited by soil conditions.

Study Site

The Heart Lake basin is a subalpine site (1,770 m elevation) in the northern Bitterroot Mountains in western Montana (46°57'N, 114°58'W). Precipitation at the site averages 208 cm annually, and the area is typically free of snow between July and October. The soils are very fine-grained and are classified as Andic Cryochrepts—Loamy Skeletal, Mixed of the Belt geological formation. Vegetation consists of scattered clumps or single trees with a dense shrub layer. Tree species present include *Abies lasiocarpa*, *Pinus contorta*, *P. albicaulis*, *P. monticola*, *Picea engelmannii* and *Tsuga mertensiana*. The understory surrounding the study sites is dominated by *Xerophyllum tenax*, *Vaccinium globulare* and *Menziesia ferruginea*.

There are four main campsites and five smaller sites on the north side of Heart Lake. In the fall of 1995, the largest of the four campsites was closed for use, and we established restoration plots on that site. The campsite is roughly rectangular, 6 m wide and 11 m long. There is one established tree within the site, and scattered clumps of *Xerophyllum tenax*.

Methods

Revegetation Study

Within the site, 25 1-m² plots were established, and each was randomly assigned to one of five soil treatments and one of two revegetation treatments. The soil treatments included a control, which was not treated; scarified only plots; scarified plus inoculum; scarification plus compost; and scarification plus compost and inoculum. Scarification was done by hand with pulaskis and shovels, to a depth of approximately 15 cm. Inoculum was in the form of a slurry composed of soil from an adjacent undisturbed site (c.a. 30 ml) mixed with 2 liters of stream water and incorporated into the top 15 cm of soil. Compost was added in the form of commercially available Ekocompost[®]. One and one-third cubic feet of compost was added to each m²-plot and incorporated into the top 15 cm of soil.

Half of the plots were given a revegetation treatment, which included transplants and seeds. Five four-month-old *Spiraea splendens* seedlings, grown from seeds collected on site, were planted in each of the plots assigned the revegetation treatment. Seeds were collected from within one mile of Heart Lake in 1994 and 1995, and lightly raked into the surface of plots in October 1995. Species selected were common in the area and producing fruit: *Agrostis exarata*, *A. scabra*, *Achillea millefolium*, *Anaphalis margaritacea*, *Aquilegia flavescens*, *Aster occidentalis*, *Bromus caranatus*, *Carex luzulina*, *Delphinium occidentale*, *Epilobium watsonii*, *Juncus ensifolius*, *Polygonum douglasii*, *P. phytolacefolium*, *Polythicum lonchitis*, *Spiraea betulifolia*.

Soil Microbial Community Functional Diversity

Microbial community functional diversity was measured through carbon utilization profiles (Biolog[®]). Soil microbial communities were exposed to 95 unique carbon substrates,

and their ability to metabolize the substrate was indicated by the oxidation of a tetrazolium dye. Quantification of microbial activity across this broad array of substrates is used as an indicator of the functional diversity of the soil microbial community (Seastone and others, in review). This research was summarized in a previous publication (Zabinski and Gannon 1997), and details of methodology can be found there.

Soil Nutrient Status

Fifteen cores, 2.2 cm dia x 10 cm depth, were extracted from random locations across the campsite in September 1994. An additional 15 cores were randomly located in a vegetated area adjacent to the campsite. Soil analysis was completed by Camas Analytical Labs, Inc. Total nitrogen was analyzed with the micro Kjeldahl method on a 0.5 gram sample. Phosphate-phosphorus was determined from a sodium bicarbonate extract of soil and determined by the ascorbic acid method. Iron and potassium was measured by atomic absorption spectroscopy on an ammonium acetate and DPTA extract. When data were normally distributed with equal variances, t-tests were used to test the hypothesis that nutrient levels differ on and off campsites. For non-normal data or data with unequal variances, a Mann-Whitney rank sum test was used.

Soil Effects on Plant Growth

Soil collected from Heart Lake campsites in September 1994 was used to test the effects of soil amendments and species on plant growth. The experimental design was a complete factorial with two plant species common to subalpine habitats, tufted hairgrass (*Deschampsia caespitosa*) and pearly everlasting (*Anaphalis margaritacea*); and five soil treatments—disturbed, disturbed plus compost, disturbed plus inoculum, disturbed plus compost and inoculum, and undisturbed soil. There were 10 replicates of the disturbed soil treatments and eight replicates of the undisturbed soil treatment for each species. Plants were grown for 15 weeks in 12 cm dia pots in the greenhouse, after which shoot tissue was dried and weighed. Analysis of variance was used to test for effects of soil and species and a two-way interaction on biomass accumulation. Post-hoc Tukey's tests were used for pair-wise comparisons between treatments.

Seed Bank Density and Composition

Seed bank density and composition were summarized by monitoring germinants in the greenhouse from soil samples collected from heavily impacted campsites, lightly impacted sites, and undisturbed sites. Details of methodology can be found in Zabinski and others (2000).

Factors Limiting Revegetation in the Field

Plots were monitored at two- to four-week intervals throughout the growing season during the summers of 1996 and 1997. At each sampling time, all of the plants within the m² plot were mapped and identified if possible. Patterns of mortality and establishment were recorded. During 1998,

plots were monitored twice during the growing season, at the beginning of July and the beginning of August. At the August sampling time, height of the most common species was measured in each of the plots, and size of *Spiraea* transplants was recorded.

Results

Effects of Recreation Impacts

For four of the six soil nutrients analyzed, there were significant differences between disturbed and undisturbed sites (table 1). Both nitrate-nitrogen and ammonium-nitrogen were significantly higher on the campsite than on adjacent, undisturbed sites. Total nitrogen was lower, although not significantly, on the campsite, relative to the undisturbed site. Phosphate and potassium were significantly lower on the campsite. There was no statistical difference in iron levels between campsite and undisturbed soils.

Greenhouse Experiment

The test of plant growth on disturbed, undisturbed and amended soils showed that there were significant treatment effects for species ($F_{1,95} = 221.7$, $p < 0.001$) and soil treatments ($F_{4,95} = 33.43$, $p < 0.001$), and a significant interaction term ($F_{4,95} = 9.7$, $p < 0.001$). Pearly everlasting biomass increased with the addition of compost and inoculum, but showed no difference across the other four treatments (fig. 3). Tufted hairgrass biomass increased with the addition of compost or compost plus inoculum, and these two treatments were not statistically different from each other (fig. 4). There was no difference in growth of either species on disturbed versus undisturbed soil. The addition of inoculum by itself did not affect growth relative to the control.

Field Experiment

The number of seedlings in each plot during July 1997 did not differ between soil treatments (one-way ANOVA, $p < 0.77$; fig. 5). There was a large amount of heterogeneity within the five replicates of each treatment. In 1998, there was also no effect of soil treatment on seedling number (one-way ANOVA, $p < 0.21$; fig. 6).

Table 1—Soil nutrient analysis on and off disturbed campsite, measured as ppm.

	Campsite (n = 15)	Undisturbed site (n = 15)	Significance
NO ₃ -N	0.28	0.12	p = 0.011 ^a
NH ₄ -N	9.59	3.87	p < 0.001 ^b
Total N	2,813	3,325	p = 0.08 ^a
PO ₄ -P	12.23	24.31	p = 0.001 ^b
K	136.3	188.8	p = 0.03 ^b
Fe	18.7	23.1	p = 0.26 ^a

^at-test with 23 df.

^bMann-Whitney Rank Sum Test.

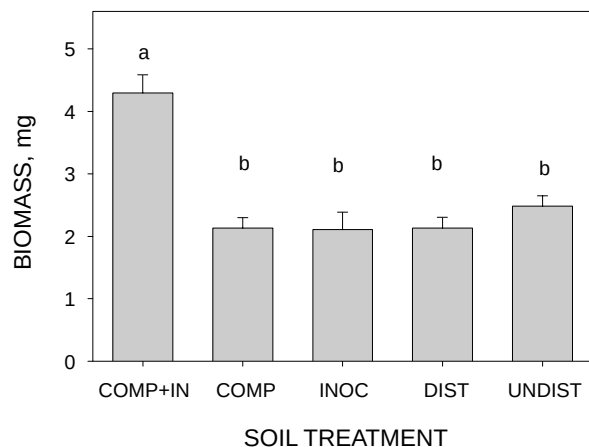


Figure 3—Pearly everlasting biomass vs. soil treatment. Treatments include disturbed soil with compost and inoculum added, compost added, inoculum added, no additions, and soil from an undisturbed site. Error bars represent standard error of the mean.

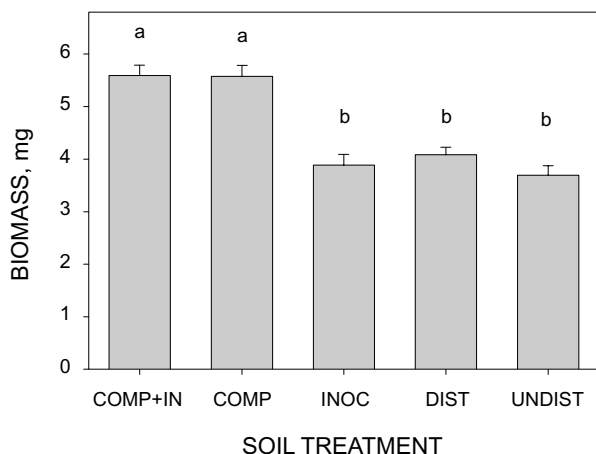


Figure 4—Tufted hairgrass biomass versus soil treatment. Error bars represent standard error of the mean.

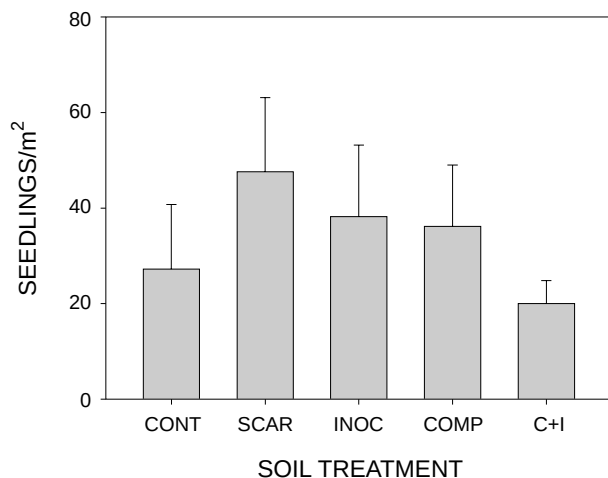


Figure 5—Number of seedlings/plot versus soil treatment, 1997 growing season. Error bars represent standard error of the mean.

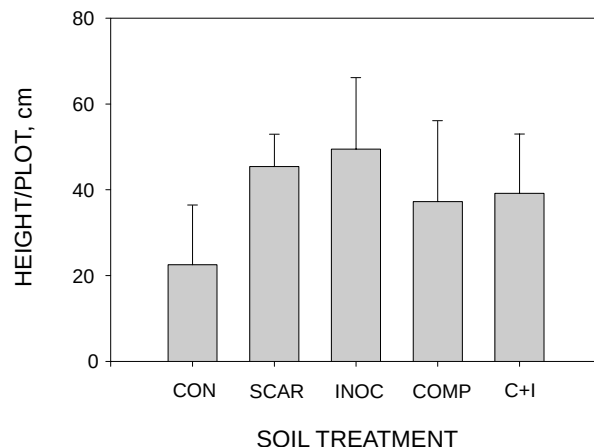


Figure 6—Number of seedlings/plot versus soil treatment, 1998 growing season. Error bars represent standard error of the mean.

In both 1997 and 1998, the number of seedlings was significantly higher with the revegetation treatment than without. In 1997, there was an average of 48 seedlings/plot on the revegetation plots, and 18 seedlings/plot on no-revegetation plots (t-test, $p < 0.005$). In 1998, differences between revegetation and no-revegetation treatments were 53 and 29 seedlings/plot, respectively (t-test, $p < 0.022$).

The average height of sedges in each plot did not differ across soil treatment (one way ANOVA, $p < 0.84$), and ranged from 2.25 cm on scarified-only plots to 2.8 cm on compost plus inoculum plots. The total height of sedges on a plot (sum of longest leaf length for each plant within the m^2 plot) did not differ between soil treatments (one-way ANOVA on ranks, $p < 0.19$; fig. 7).

Pearly everlasting was the second most common plant on the plots, occurring on 12 of the 25 plots. There were no individuals present on any of the control plots, so comparisons were made between scarified and amended plots. There was no significant soil treatment effect on total

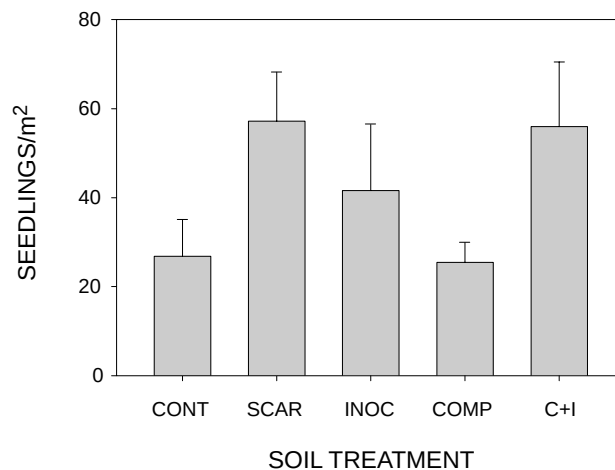


Figure 7—Total sedge height/plot versus soil treatments. Error bars represent standard error of the mean.

height (one way ANOVA, $p < 0.249$). In a comparison of the plots to which compost was added versus plots without compost (scarified only and inoculum added plots), there was a significant increase in pearly everlasting height (4.6 cm on compost treatments versus 2.5 cm on noncompost treatments; t-test, 10 df, $p < 0.046$; fig. 8). Sedge height did not differ on compost versus noncompost amended sites (t-test, 23 df, $p < 0.39$).

Scarification without soil amendments increased seedling establishment relative to controls in the Eagle Cap Wilderness (Cole and Spilbie 2000). In this study, there was a close to significant difference in the total number of seedlings on scarified relative to control plots in 1998 (t-test, 8 df, $p < 0.058$), with an average of 27 seedlings on the control plots and 57 seedlings on the scarified plots. In 1997, there was no difference in seedling number between control and scarified plots (t-test, 8 df, $p < 0.35$). There was no difference in sedge number (t-test, 8 df, $p < 0.51$) or sedge total height (t-test, 8 df, $p < 0.370$) between control and scarified plots.

Discussion

Recreation Impacts on Site Conditions

Soil conditions, including microbial community structure, nutrient availability, and seed bank density, were significantly affected by recreational use. Functional diversity of the microbial community on campsite soils was decreased by 44% relative to soils from undisturbed sites, although total numbers of microbes, as measured by colony-forming units on spread plates, were not different (Zabinski and Gannon 1997). Carbon utilization profiles are a measure of the diversity of carbon substrates that a microbial community can metabolize, and serve as a proxy measurement for soil microbial diversity. Soil microorganisms have important ecological effects on plant establishment and growth (Bever 1994, Chanway and others 1991, Turkington and others 1988), so understanding changes in the microbial community associated with disturbance and restoration amendments could be very important. But until we are able to

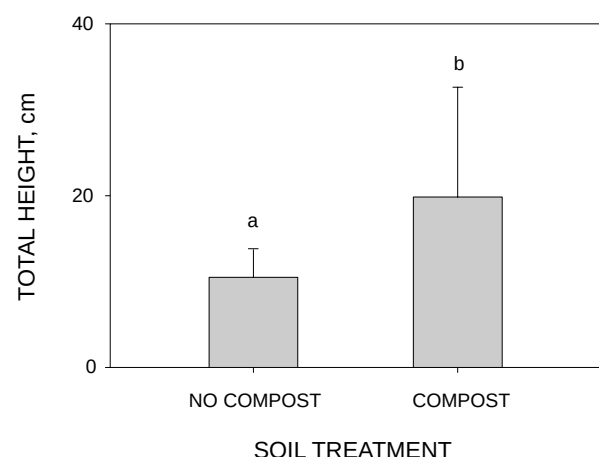


Figure 8—Total pearly everlasting height/plot versus composted/noncomposted soil treatments. Error bars represent standard error of the mean.

elucidate relationships between specific microbial partners and plant species or groups, our ability to predict or ameliorate the effects of disturbance is limited.

Available nitrogen in the form of nitrate and ammonium increased in campsite soils relative to undisturbed soils, while total nitrogen decreased. This apparent discrepancy is most likely due to a decrease in plant uptake of nitrogen on campsite soils, leaving available N, while a decrease in surface organic matter results in a less carbon-bound nitrogen. That there was no difference in plant growth in the greenhouse on disturbed versus undisturbed soils suggests that the changes in soil nutrient availability and microbial community function are not important (for at least the species tested), when plants are grown under benign conditions. The greenhouse study did not address the effect of changes in the physical structure of soil, since the process of moving soil from the field to the greenhouse disrupted patterns of compaction.

Field Revegetation Experiment

Revegetation success overall was low in this study. The number of seedlings establishing on some of the m^2 plots was over 100, but the average was 40 seedlings/ m^2 during 1998. The overall growth of plants was very low. The average sedge height on a plot was near two cm, after three seasons of growth. Several species flowered on the site, including *Polygonum douglasii* and *Aster occidentalis*.

Limiting Factor: Propagule Availability

Our results suggest that propagules may be limiting natural revegetation of this site. The density and composition of the seedbank was affected by disturbance in this subalpine ecosystem. Seedbank density was 441 seeds/ m^2 on heavily impacted sites, 1495 seeds/ m^2 on lightly impacted sites, and 4188 seeds/ m^2 on undisturbed sites (Zabinski and others 2000). Ten of the 22 taxa identified from the seed bank in undisturbed and lightly impacted sites were not present on heavily impacted campsite soils (Zabinski and others 2000). This suggests that natural revegetation from seedbank soils would result in an impoverished suite of species recolonizing the site.

Field study results also suggest that propagule availability may be limiting. The only significant treatment effect in the field experiment was the revegetation treatment, which doubled the number of seedlings present. Three of the field plots that were not seeded had relatively high seedling numbers—ranging from 48 to 73 in 1998. In two of the plots, most of the seedlings are sedges, and in the third plot most of the seedlings are conifers. Both the patchy distribution of seeds and heterogeneity in conditions that affect seedling establishment could explain these results.

Limiting Factor: Seed Germination and Seedling Establishment

Limitations of seed germination and seedling establishment can be more clearly distinguished in the seeded plots within the study. Comparable amounts of seeds were added to each of the seeded plots, but numbers of seedlings on those

plots ranged from 8 to 98 in 1997. On those plots with a very low number of seedlings, seed germination and seedling establishment are limiting. This may be due to biotic interactions such as herbivory, but more likely to environmental conditions. In plots with a high number of seedlings, the low growth response over the three years of this study suggesting that conditions that affect plant growth are limiting to success.

Limiting Factor: Soil Conditions

The soil treatments used in this study were designed to affect the physical, biological, and chemical properties of the campsite soils. Scarification loosens up the compacted soils, at least temporarily. The addition of compost adds large pieces of organic matter that contribute to the physical structure of the soil by providing spaces for water and root penetration. Nutrient availability also increases with compost addition, along with microbial community functional diversity. That there were no significant differences in seedling number across soil treatments suggests that soil conditions are not the primary limiting factor at this site.

Plant growth was significantly increased by the addition of compost in the greenhouse and, for pearly everlasting, in the field. This suggests that if microclimatic conditions are limiting seedling establishment on this site, and if shade cloth or water addition could ameliorate that limitation, plant growth may respond to soil treatments.

Conclusions

Revegetation at Heart Lake is limited by a combination of factors. Propagule availability is limiting across most of the campsite, as evidenced by the increase in seedling number with the addition of seed. Seed germination and seedling establishment were patchy, suggesting that environmental conditions are important in determining the success at that stage. Soil treatments showed no significant effect on seedling number or growth, suggesting that microclimatic differences that vary with patterns of sunlight and water drainage may be the primary limiting factor. This is not to suggest that soil amendments are ineffective, but that until other factors can be ameliorated, soil conditions as affected by the amendments are not limiting to revegetation success.

We will continue this work with the addition of water and a ground cover to reduce surface desiccation. On sites such as this one that are problematic for revegetation purposes, amelioration of microclimatic conditions may be essential for seedling establishment and growth.

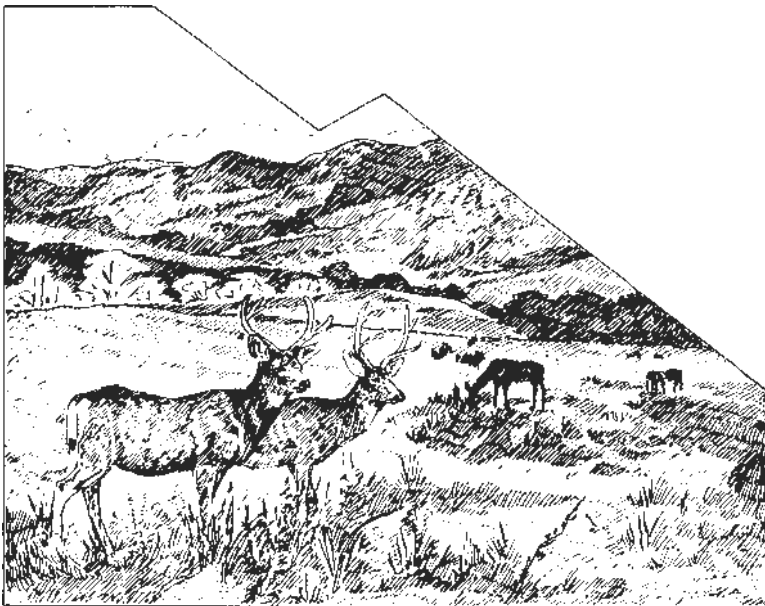
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References

- Bever, J. D. 1994. Feedback between plants and their soil communities in an old field community. *Ecology*. 75:1965-1977.
- Bliss, L. C. 1971. Arctic and alpine plant life cycles. *Ann. Rev. Ecol. Syst.* 2: 405-438.
- Chambers, Jeanne C. 1997. Restoring alpine ecosystems in the western United States: environmental constraints, disturbance characteristics, and restoration success. In: K. M. Urbanska, N. R. Webb, and P. J. Edwards, eds. *Restoration ecology and sustainable development*. Cambridge University Press: 161-187.
- Chambers, J. C.; MacMahon, J. A.; Brown, R. W. 1990. Alpine seedling establishment: the influence of disturbance type. *Ecology*. 71:1323-1341.
- Chanway, C. P.; Turkington, R.; Holl, F. B. 1991. Ecological implications of specificity between plants and rhizosphere microorganisms. *Advances in Ecological Research*. 21:121-1699.
- Cole, David N.; Spildie, David R. 2000. Soil amendments and planting techniques: campsite restoration in the Eagle Cap Wilderness, Oregon. In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. *Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management*; 1999 May 23-27; Missoula, MT. *Proceedings RMRS-P-15-VOL-5*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Ebert May, Diane; Webber, P. J.; May, T. A. 1982. Success of transplanted alpine tundra plants on Niwot Ridge, Colorado. *Journal of Applied Ecology*. 19:965-976.
- Harper, J. L.; Williams, J. T.; Sagar, G. R. 1965. The behavior of seeds in soil. Part 1. The heterogeneity of soil surfaces and its role in determining the establishment of plants from seed. *J. of Ecol.* 53:273-286.
- Seastone Moynahan, O.; Zabinski, C.; Gannon, J. [In review]. Microbial community structure and metabolic diversity in a mine tailings restoration study.
- Turkington, R.; Holl, F. B.; Chanway, C. P.; Thompson, J. D. 1988. The influence of microorganisms, particularly *Rhizobium*, on plant competition in grass-legume communities. In: Davy, A. J.; Hutchings J. J.; Watkinson A. R., eds. *Plant population ecology*. Blackwell Scientific Publications, Oxford, UK: 343-363.
- Urbanska, Krystyna M. 1997. Safe sites—interface of plant population ecology and restoration ecology. In: Urbanska, K. M.; Webb, N. R.; Edwards P. J., eds. *Restoration ecology and sustainable development*. Cambridge University Press, Cambridge: 81-110.
- Zabinski, Catherine A.; Gannon, James E. 1997. Effects of recreational impacts on soil microbial communities. *Environmental Management*. 21:233-238.
- Zabinski, Catherine; Wojtowicz, Todd; Cole, David. 2000. The effects of recreation disturbance on subalpine seed banks in the northern Rocky Mountains. *Canadian Journal of Botany*. 78:577-582.

4. Wilderness Fire and Management



Mixed-Severity Fire Regimes in the Northern Rocky Mountains: Consequences of Fire Exclusion and Options for the Future

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Abstract—Findings from fire history studies have increasingly indicated that many forest ecosystems in the northern Rocky Mountains were shaped by mixed-severity fire regimes, characterized by fires of variable severities at intervals averaging between about 30 and 100 years. Perhaps because mixed-severity fire regimes and their resulting vegetational patterns are difficult to characterize, these regimes have received limited recognition in wilderness fire management. This paper presents examples of mixed-severity fire regimes in Glacier National Park, the Bob Marshall Wilderness Complex and the Selway-Bitterroot Wilderness and discusses how suppression and fire management policies have affected them. It suggests possible management actions to return a semblance of the historical mixed-severity fire regimes to these and other natural areas.

The ecological problems associated with removing frequent low-intensity fires from ponderosa pine ecosystems are well known to forest and wilderness managers, and restoration of fire is being planned or implemented in many of these ecosystems (Bailey and Losensky 1996; Covington and others 1997; Kilgore and Curtis 1987). In contrast, ecosystems historically characterized by infrequent stand-replacement fires may not have been greatly altered by 60 to 90 years of fire suppression, partially because it is often not possible to suppress high-intensity fires (Agee 1993; Johnson and Larsen 1991; Romme and Despain 1989). However, little recognition has been given to possible effects of fire exclusion in ecosystems historically shaped by mixed-severity fire regimes. Mixed-severity regimes produced highly diverse forest communities containing abundant seral, fire-dependent species, including multi-aged stands with large, old fire-resistant trees that are of great importance as wildlife habitat (McClelland 1979). These regimes also helped produce intricate mosaics of even-aged tree groups and contrasting forest communities at the landscape level. Effects of fire exclusion on ecosystems shaped by mixed-severity fire regimes should concern wilderness managers

because these ecosystems are important components of national parks, wilderness and other natural areas of the northern Rocky Mountains (Fischer and Bradley 1987; Smith and Fischer 1997). A recent field inspection of areas historically characterized by mixed-severity fire regimes in the Bob Marshall Wilderness led us to this analysis of the situation.

Defining “Mixed Severity”

Fire plays a complex role in wildland ecosystems, and individual fires can have highly variable effects in space and time. An individual fire's behavior can change dramatically as it moves across the landscape under the influence of daily and longer term changes in temperature, humidity and wind. The fire is also affected by changes in stand structures, fuels and topography. To facilitate communication, planning and management related to wildland fire, Brown (1995) presented a simplified classification of “fire regimes” to characterize the kinds of fires that have occurred over the past several hundred years in different regions or forest types.

The classification is based on fire severity, namely what happens to the dominant vegetation—in this case, trees. If most of the overstory trees die in most fires, that area is said to be characterized by a “stand-replacement fire regime.” Conversely, if most trees survive most fires, it is called a “nonlethal fire regime.” If severity is a mixture of the above—for example, frequent nonlethal fires and infrequent stand replacement fires—it is a “variable fire regime” (Arno and others 1995; Brown 1995). If severity is generally intermediate—many trees dying and many surviving—it is a mixed-severity fire regime. Variable and mixed-severity fire regimes probably intergrade and may be difficult to differentiate based on available evidence; thus, for this discussion, we will lump both into “mixed-severity fire regimes.” Fire frequency is often inversely related to fire severity. Nonlethal fire regimes generally have frequent fires (commonly at intervals of 5 to 30 years), and stand-replacement regimes have infrequent fires (intervals of 100 to 400 years in the northern Rocky Mountains), while mixed-severity fire regimes have fires at intermediate frequencies, with average intervals ranging from about 30 to 100 years. Fire sizes and burning patterns are additional components of fire regimes not dealt with directly in the classification (Brown 1995).

Characteristically, a mixed-severity fire regime will have a number of individual fires that burn at mixed severities. It may also have some stand-replacement fires and some

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nonlethal fires. Individual mixed-severity fires typically leave a patchy, erratic pattern of mortality on the landscape, which fosters development of highly diverse communities (fig. 1). Overall, these fires kill a large proportion of the most fire-susceptible tree species, such as subalpine fir, which tend also to be the shade-tolerant species favored by fire exclusion (Minore 1979). Conversely, mixed-severity fires kill a smaller proportion of the fire-resistant species—including western larch, ponderosa pine, western white pine and whitebark pine, which are long-lived species that are replaced successional by shade-tolerant species with fire exclusion (Arno and others 1997; Hartwell and others, in process; Keane and Arno 1993; O’Laughlin and others 1993).

Historical Conditions

In past centuries, mixed-severity fire regimes characterized large areas of forest ecosystems throughout the western United States (Arno, in process), and specifically in the northern Rocky Mountains (Arno 1980; Arno and others 1993; Barrett and others 1991; Brown and others 1994; Murray 1996; Zack and Morgan 1994). In Northern Rocky Mountain forests, mixed-severity regimes occupied about 50 percent of the area now in national forest lands, nonlethal regimes included about 30 percent of this area, and stand-replacement regimes covered about 20 percent (Quigley and others 1996). A Fire Regime Analysis being conducted by the USDA Forest Service has found similar proportions of these fire regimes nationwide (Hardy, personal communication).

The presence of appreciable quantities of old trees with scars from pre-1900 fires is *prima facie* evidence of historical mixed-severity or nonlethal fire regimes. In the northern Rockies, nonlethal regimes are primarily confined to forests where ponderosa pine was historically dominant. Mixed-severity regimes were found across a broad range of forest types, including some of those dominated by interior Douglas-fir and western larch, western white pine, lodgepole pine and whitebark pine, as well as some relatively moist ponderosa pine types. Other areas of these same forest types (except, possibly, ponderosa pine) were characterized by stand-replacement fire regimes. The kinds of fire occurring in a given forest type depended on fuel and vegetation development patterns, climatic factors, topography, and sometimes the history of Indian burning (Arno and others 1997; Barrett and Arno 1982). Mixed-severity fire regimes covered sizeable areas in the largest national parks and wilderness areas, including Glacier National Park (Barrett and others 1991), the Bob Marshall Wilderness complex (Davis 1977; Gabriel 1976; and observations presented later in this paper), the Selway-Bitterroot Wilderness (Brown and others 1994), the Frank Church-River of No Return Wilderness (Crane and Fischer 1986) and Yellowstone National Park’s northern range (Barrett 1994; Houston 1973).

Forests associated with mixed-severity regimes were often dominated by the early seral, fire-dependent tree species, but also may have had a substantial component of late-successional trees. Individual stands were often uneven-aged and multi-layered. Moderately short fire intervals allowed important seral shrubs and hardwoods to remain



Figure 1—A stand on the Lolo National Forest, Montana, shaped by a mixed-severity fire regime. The tall trees (western larch) were established after various fires between the mid-1400s and the early 1800s. The older larch have survived 4 to 5 fires between the mid-1600s and 1904. A few of the lodgepole pines survived fires in 1889 and 1904, but most of the densely stocked smaller trees (lodgepole pine, subalpine fir, and Engelmann spruce) became established after these latest fires (S. Arno, unpublished data).

abundant (Fischer and Bradley 1987). These included aspen, Scouler willow, serviceberry, chokecherry, and redstem and evergreen ceanothus (Arno and others 1985). Small meadows and grassy openings and a variety of early seral herbaceous plants would also have been abundant (Gruell 1980; Arno and others 1985; Steele and Geier-Hayes 1993; Stickney 1990). As a result of the moderately frequent fires and variable fire severities, stands often formed a complex and intricate mosaic on the landscape. However, young seral stands and young seral components of mixed-age stands were abundant (Baker 1993; Keane and others 1996, 1998a; Romme 1982).

Current Conditions and Future Trends

By the late 1800s, the historical role of fire on the landscape had been reduced in many areas by heavy livestock grazing that removed fine fuels and by disruption of burning by Native Americans (Arno and Gruell 1986; Boyd 1999). By the late 1930s, fire suppression had become effective in reducing the annual extent of fires, even in large wilderness areas in the northern Rocky Mountains (Barrett and others 1991; Brown and others 1994). The Selway-Bitterroot Wilderness has had the most active program to restore natural fires in the region (since 1973), allowing some lightning fires to burn, taking only limited suppression on others and carrying out full suppression on the rest. Despite these outstanding efforts to restore natural fire, the program has still produced a significant reduction in average area burned compared to the pre-fire suppression period (Brown and others 1994). Moreover, the political repercussions of the 1988 Yellowstone fires have further limited the application of natural fire programs throughout the western United States (Parsons and Landres 1998). The natural fire program in the Selway-Bitterroot Wilderness has produced a somewhat higher proportion of stand-replacement and a lower proportion of nonlethal and mixed-severity burning than was characteristic of the pre fire-suppression period in the same area (Brown and others 1994, 1995a).

A study of fire regimes in Glacier National Park concluded that fire suppression had been very effective in areas that previously had a mixed-severity fire regime, but much less effective in areas of stand-replacement fire regimes (Barrett and others 1991). A detailed study of the entire inland portion of the northwestern United States also concluded that areas historically under a nonlethal or mixed-severity fire regime have now shifted toward stand replacement regimes (Morgan and others 1998; Quigley and others 1996). By the late 1990s, mixed-severity fire regimes have been reduced to about 30 percent of the landscape, and nonlethal regimes occupy only about 10 percent, whereas forests typically burning in stand-replacement fires now encompass about 60 percent of the national forest lands (Quigley and others 1996).

The effects of substantial reductions in areas burned in historical mixed-severity fire regimes are predictable and observable (Keane and others 1996). Intensive comparisons of historical (circa 1900) and modern stand structures in unlogged areas near the eastern boundary of the Selway-Bitterroot Wilderness show major declines in ponderosa

pine, western larch and whitebark pine, and corresponding increases in Douglas-fir at lower elevations and subalpine fir at middle and high elevations (fig. 2) (Arno and others 1993, 1995; Hartwell and others, in process). Lodgepole pine has maintained its historical abundance, but young lodgepole communities (which contain numerous early seral undergrowth species) have become less common.

On landscapes such as large wilderness areas, the effects of fire exclusion tend to include greater uniformity in stand ages and in stand composition and structure, together with a declining diversity of undergrowth species (Arno and others 1993; Keane and others 1996). Basal area and numbers of trees per acre may increase dramatically (Arno and others 1997). This results in increased physiological stress and the opportunity for extensive forest mortality caused by epidemics of insects and diseases (Fellin 1980; Monnig and Byler 1992; Biondi 1996). Fire exclusion and related advancing succession also brings increased loadings of dead and living (ladder) fuels across the forest landscape, which increases the likelihood of unusually severe and extensive wildfires (Barrett and others 1991; Barbouletos and others 1998; Quigley and others 1996; Morgan and others 1998). When a large and unusually severe fire occurs in a wilderness environment, it ultimately creates a correspondingly large mass of heavy fuels, starting 12 to 15 years after the fire when much of the dead timber has fallen (Lyon 1984). This becomes incorporated into a new dense fuel bed with small conifers and large shrubs, which can readily support another severe wildfire, or "double burn" (Barrett 1982; Brown 1975; Wellner 1970).

Modeling suggests that the effects of continuing this trend will be higher proportions of large stand-replacement fire in wilderness landscapes (Baker 1992; Keane and others 1996, 1998a). There will be a loss of multi-aged stands of seral tree species. The intricate, fine-grained landscape mosaic of diverse stand structures and compositions will be replaced by a coarser pattern of even-aged stands (fig. 3). Longer fire intervals will cause seral herbaceous and shrub species to decline because they will have difficulty surviving under extended periods of dense conifer coverage—the "stem-exclusion stage" (Oliver and Larsen 1996). In addition to

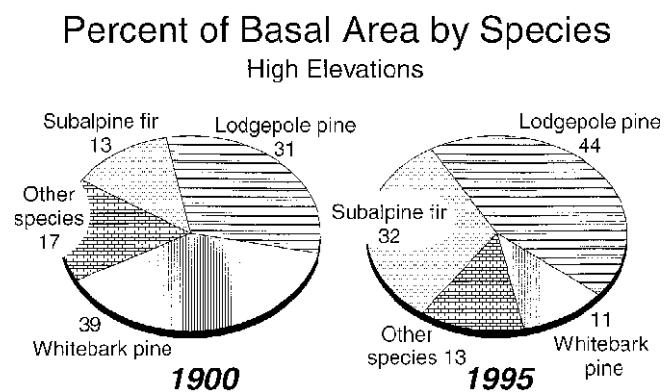


Figure 2—Historical and modern stand structures in an unlogged upper elevation forest zone on the Bitterroot Range, Montana (from Hartwell and others, in process). The historical forest was in a mixed-severity fire regime; the modern forest is influenced by fire exclusion.

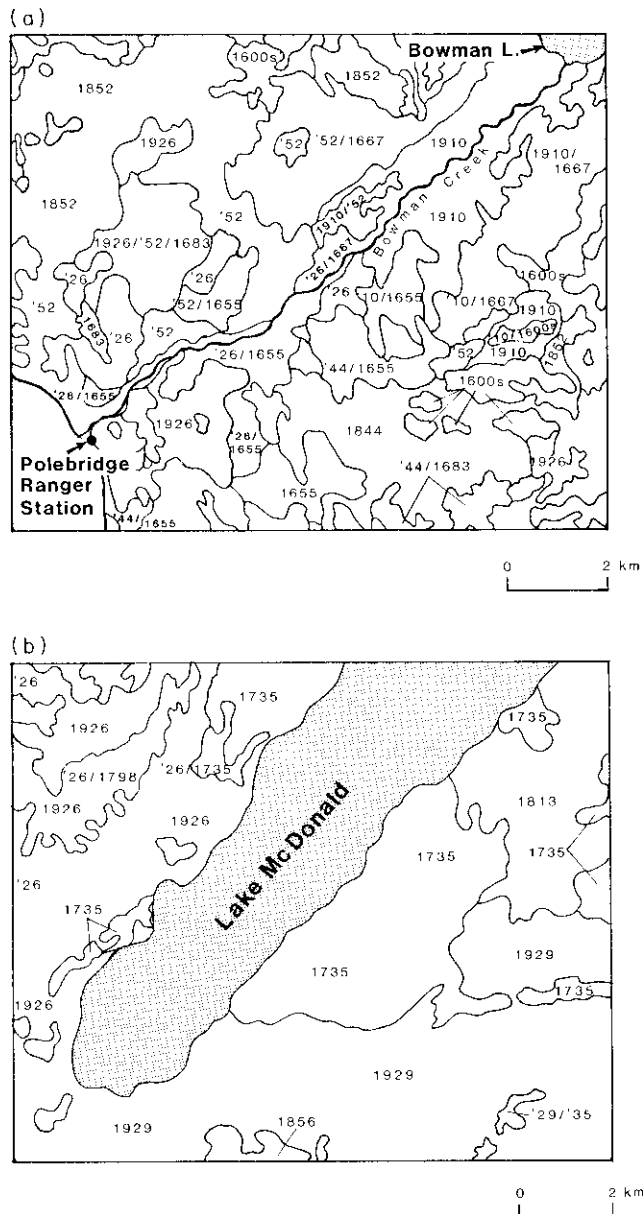


Figure 3—Age-class mosaics resulting from a mixed-severity fire regime (a) and a stand-replacement fire regime (b) in Glacier National Park, Montana. Dates indicate fire years that resulted in establishment of seral western larch and lodgepole pine age classes (from Barrett and others 1991).

ecological impacts, the accompanying pattern of larger and more severe wildfires will pose increasing health risks due to smoke production, as well as risks of fire escaping the wilderness and threatening people and private property (Hill 1998).

An Example From the Bob Marshall Wilderness

On July 11-15, 1998, we conducted field observations in the South Fork Flathead drainage, Bob Marshall Wilderness, at the request of District Ranger Carol Eckert. During

this trip, we also discussed the management implications associated with the area's fire ecology with a group of national forest managers and staff.

Much of the Bob Marshall Wilderness was historically characterized by a stand-replacement fire regime, with fire intervals of 150 to 250 years in a given stand (Keane and others 1994). Today, this fire regime is generally considered to be functioning within its "historical range of variability" (Morgan and others 1994) as a result of periodic wildland fires and some lightning fires allowed to burn under prescription. Our observations are directed to mixed-severity fire regimes that occur in the drier areas of the South Fork Flathead drainage (Gabriel 1976). We observed two kinds of forests that historically experienced mixed-severity fire regimes, based on abundant fire scars found on living trees, multiple age-classes of seral fire-dependent trees and intricate stand mosaics. (In addition, there is a historical nonlethal fire regime associated with nearly pure ponderosa pine stands on dry, gravelly river terraces.) In this area, forest types historically maintained by the mixed-severity regime are ponderosa pine-mixed conifer and larch/Douglas-fir/lodgepole pine. The ponderosa pine-mixed conifer forest type covers a few thousand acres in the South Fork Valley, below 5,000 feet. The larch/Douglas-fir/lodgepole pine type is widespread and extends up to about 5,500 feet.

The ponderosa pine-mixed conifer type contains large ponderosa pines 200 to 600 years of age, but few less than 60 years old. They are being replaced by younger Douglas-fir, Engelmann spruce and lodgepole pine. In the areas we examined, past fires occurred at intervals of about 25 to 40 years, with the most recent burns, dated from increment borings, having occurred in about 1929. One living ponderosa pine a mile south of Big Prairie Ranger Station is about 410 years old and has well-formed scars from at least seven different fires. We also found several lodgepole pines, often growing among scattered ponderosa pines, that have three fire scars dating between the mid-1800s and the early 1900s. Although these stands appear to have once been open and parklike, today they are generally dense with young Douglas-fir, lodgepole pine and other conifers, and they contain substantial quantities of duff (including deep mounds at the base of old trees) and down woody fuels. Under current conditions, a summer wildfire that escaped suppression could easily become a large, stand-replacing burn. Successional studies indicate that such a fire would probably give rise to new stands of lodgepole pine and Douglas-fir, with little if any ponderosa pine (Arno and others 1985). These post-fire stands would probably have a dense, even-aged structure, as well as abundant fire-killed downed trees, favoring continuance of a stand-replacement fire regime in the future (Scott 1998).

The larch/Douglas-fir/lodgepole pine mixed-severity type extends up tributary drainages, where it adjoins the stand-replacement fire regime types. In response to historic mixed-severity fires, stands generally have a multi-aged structure. Many of the larch, Douglas-fir and some of the lodgepole pines have one to three scars from past fires. In one stand near White River Butte, we found a large larch with scars from four different fires and a fallen old-growth Douglas-fir with scars from five fires.

In the ponderosa pine-mixed conifer type, it has generally been 70 to 100 years since the last fire, two to four times as

long as the average historic fire interval. Current fire-free intervals in the larch/Douglas-fir/lodgepole pine type are probably approaching two times the length of historic mean intervals. These lengthened intervals are not necessarily unprecedented in any one stand; however, because current intervals since the last fire in most stands are near or beyond the upper end of the historical range of fire intervals, associated fuel accumulations provide the opportunity for unusually large, stand-replacing fires. Lodgepole pine is a common forest component in the mixed-severity fire regimes, and it is susceptible to mass mortality as a result of bark beetle epidemics when it reaches ages of 80 years in dense stands (McGregor and Cole 1985). Landscapes of beetle-killed lodgepole pine are at high risk of large, stand-replacing fires (Brown 1975). Frequent fires of the past provided a natural mosaic of diverse stand structures, which reduced chances of large, stand-replacing fires in the mixed-severity fire regime (Barrett and others 1991).

Unusually severe fires in mixed-severity and nonlethal fire regimes have been linked to effects of fire exclusion (Agee 1993; Barbouletos and others 1998; Barrett 1988; Steele and others 1986). The North Fork Flathead Valley in Glacier National Park, an area characterized by a mixed-severity fire regime, experienced the unusually large and severe Red Bench Fire in 1988, after the fire-free interval had more than doubled due to successful fire exclusion (Barrett and others 1991). In 1994, Park managers used prescribed natural fire and confine-and-contain strategies on two nearby wildfires to accomplish 14,000 acres of mixed-severity burning in an adjacent area within this fire regime (Kurth 1996; Van Horn, personal communication).

Possible Restoration Strategies

Any effort to restore fire to a more natural role in wilderness must recognize a great paradox: Direct human intervention—suppression of natural fires—has greatly altered fire frequency and fire severity, important processes that historically shaped wilderness ecosystems. Moreover, this intervention will surely continue. Wilderness management (like wildland forest management in general) still operates largely under a fire suppression strategy. Although the concept of eventually returning fire to a more natural role is often accepted by land managers, wilderness fire programs are greatly restricted by concerns about liability for fires escaping wilderness, public safety, smoke pollution and possible complaints from the public. In 1963, a panel of scientists called upon by the Secretary of Interior concluded that the exclusion of natural fire is not consistent with maintenance of ecosystems in national parks—or, by extension, in wilderness (Leopold and others 1963). Although this advice did result in prescribed burning on a small scale in some areas, it has had little effect on landscape-scale management in most national parks or wilderness areas (Parsons and Landres 1998). Restoring natural fires in wilderness requires much stronger support on behalf of the fire manager. Today, if a manager chooses to use or allow fire, he or she is exposed to considerable risk (Czech 1996; Mutch 1997). Conversely, choosing to put out any and all natural fires is relatively risk-free. Ironically, each natural fire suppressed within or near wilderness may be construed

as an act of “trammeling” inconsistent with the concept of wilderness as a place where the forces of nature act without human interference (The Wilderness Act: Public Law 88-577, 1964).

Restoration of fire in nonlethal and mixed-severity regimes requires special care because fuel and stand structures in many areas are outside the historic range of variability (Morgan and others 1994; Quigley and others 1996). Some naturally ignited fires burning under these altered conditions might adversely impact natural biodiversity (Covington and others 1997; Harrington 1996). Depending on the situation, we have listed the following four approaches, which might be useful for restoring a semblance of the conditions historically associated with the mixed-severity fire regime in wilderness. These approaches could apply to restoring any natural fire regime, but may be especially pertinent to mixed-severity regimes because a range of fire intensities and effects is acceptable. Any effort to restore natural fire processes requires careful fire management planning (Brown and others 1995b), education within all cooperating agencies and the public and a willingness to accept some degree of risk. All alternatives for restoring fire would be aided by developing low-risk fuel conditions—for example, thinning combined with fuel removal or prescribed burning—in strategic locations along the boundary of the park or wilderness. Such treatments are, however, likely to be expensive and politically sensitive.

1. Allow all or most lightning fires to burn.

Since suppression of lightning fires has been the major factor creating the current situation, a plausible goal could be to fully restore lightning fires as an ecological factor. However, this may not be desirable where the current buildup and continuity of fuels allow lightning fires to become unusually severe and threaten adjacent areas. Still, restoration for the effects of the historical fire regime is essential if wilderness areas are going to support natural ecosystems. It will be challenging to allow most lightning fires to burn. A valuable asset to this approach would be an improved ability to predict fires or fire seasons likely to become severe so that only those situations will require suppression. Such prediction will require modeling of potential fire consequences, using tools such as FOFEM (Reinhardt and others 1997) and FARSITE (Finney 1998). Overall, the goal of this approach is to maximize the use of lightning ignitions to return fire to its natural role; realistically, however, it may be more expedient to use some prescribed fires, as explained below.

2. Reignite suppressed lightning fires.

Conceptually, it is an act of human interference to suppress a lightning fire in a wilderness area. Therefore, when a land manager finds it necessary to suppress a natural fire, we propose the following strategy to “restore” that fire as soon as conditions permit. Determining acceptable prescriptions for reigniting suppressed fires is the key to this approach. This strategy may be especially useful in the initial round of fire reintroduction. If the reignition criteria are too stringent, the resulting fire may be ineffective and insignificant. If the burning conditions are favorable, but a sudden, extreme weather event results in a costly suppression effort or property damage, the manager needs to be buffered from accepting calculated risk, provided proper procedures were followed. Ignition shortly before a season-ending rain or

snow storm is a possibility; but accomplishing significant fire size in this case might require ignition at many points.

Although the reignition of lightning fires may seem easy to justify on philosophical grounds (for returning a natural process), it poses a unique problem. It is nearly impossible to attain the hypothetical burned area and severity pattern lost when the fire was suppressed because it was out of prescription. There is rarely enough time before snowfall or season-ending rain events to recoup this acreage or to recreate the pattern of fire severity. Therefore, some other means may be necessary to burn the area left unburned when the fire was suppressed. A possible procedure might be to simulate the behavior of the suppressed fire in a spatially explicit fire-growth model such as FARSITE to compute the total area that the fire might have burned using the daily weather that actually occurred. Then, near the end of the season or the following year, this area could be burned using conventional prescribed fire methodologies.

3. Reignite suppressed fires from past years.

This approach has appeal as a way to reverse effects of past fire suppression in a manner consistent with letting nature take its course. However, if the fuels are beyond the historical range of variation or if ignitions are made under cooler or wetter conditions, the result may not mirror the natural role of fire. If the fire is relit in the same location where a fire was originally suppressed, there is at least some hope of simulating a historical natural fire. Whether the actual fire fulfills this promise is problematical.

4. Use prescribed fire as a preparation for restoring lightning fires.

In this discussion, prescribed burning is defined as systematic manager ignition of certain areas under conditions prescribed to accomplish desired effects—in this case, to reduce excessive fuels and return to a semblance of historical ecological conditions. Many alternatives are available for use in obtaining the desired result including varying the season of burning, time of day, prescribed weather, fuel moistures, ignition method and ignition pattern (Brown and others 1995b; Kilgore and Curtis 1987). Managers of some of the large national parks in the Canadian Rocky Mountains are using prescribed burning largely as a substitute for lightning fires (Woodley 1995). For larger U.S. National Parks and wilderness, we propose prescribed burning as a way to return fuel conditions that will allow lightning fires to again play a more natural role.

Using prescribed burning to restore conditions that can allow natural processes to proceed again is logical. Nevertheless, it does involve subjective decisions as to when, where, under what conditions and so forth. Some will see this as inappropriate in wilderness, so a strong case needs to be made for why it may be the only option in some cases. This will raise issues related to methods. The uncertainty of outcome in allowing natural ignition to meet planned objectives is the “risk.” Use of prescribed fire minimizes this risk by management choosing time, place and conditions.

Concluding Remarks

Restoration strategy number 1—allowing nearly all lightning fires to burn—is probably not attainable and perhaps not ecologically desirable under current conditions. It could

be viewed as the long-term goal for large national parks and wilderness areas. Strategies 2 through 4 all involve prescribed fire applications, methods opposed by many wilderness advocates as inappropriate and unacceptable in wilderness. They argue that any human decisions on when or where fires burn constitute management of natural processes, which counters the intent of the 1964 Wilderness Act. They fear that prescribed burning (by managers) would be used to intentionally manipulate wilderness conditions; that fire would become a manipulative tool rather than a natural process in wilderness (Nickas 1999). As a counterpoint, we maintain that human activities and constraints, such as fire suppression and the artificially confining boundaries of wilderness ecosystems, have already significantly affected these areas and limited how we can manage them. The use of prescribed fire applications provides a critical tool to mitigate such impacts, as long as the ultimate goal of restoring natural processes is not compromised. We fear that the apparent willingness of some wilderness supporters to accept continued fire suppression and fire exclusion rather than the interim use of prescribed fire in wilderness will further exacerbate the problems of accumulating fuels and loss of structural diversity. On the other hand, we recognize the concern that wilderness would lose much of its value if it becomes more of a human-determined landscape. Land managers have the responsibility to document and justify the need for management ignitions on a case-by-case basis.

At the other end of the philosophical spectrum, some people argue for some form of mechanical manipulation to restore more natural or manageable conditions, so that fire can be used or allowed to burn. This may be pertinent in the immediate vicinity of human developments or areas of cultural or historic value, such as backcountry ranger stations, where removing ladder fuels could greatly reduce risk and allow lightning fires to burn instead of being suppressed. However, we argue that mechanical manipulation should be considered inappropriate in general for lands managed as wilderness.

All the options for returning fire to wilderness require better information on fuels, vegetation inventories, successional dynamics, fire effects and so forth (Keane and others 1998b). On the other hand, we are degrading these ecosystems rapidly in some cases, and we cannot afford to “do nothing” and thereby continue the damaging process of fire exclusion. “No action” is a conscious decision with a definite impact. We need to build the case to get started, area by area, monitor what we do, learn from it, and adapt. This is adaptive management.

In summary, restoration of fire is critical to assure long-term sustainability of mixed-severity (and nonlethal) fire regime ecosystems. Most likely, success in achieving goals (and they must be clearly articulated) will come from some combination of the above 4 strategies tailored to fit each wilderness area. Plans for restoring a semblance of the natural fire regime need to be made and then acted upon expeditiously.

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References

- Agee, J. K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Wash., D.C. 493 p.
- Arno, S. F. 1980. Forest fire history in the Northern Rockies. *Jour. For.* 78:460-465.
- Arno, S. F. In process. Fire regimes in western forest ecosystems. Chapter V in: Effects of wildland fire on ecosystems: flora and fuel. Rocky Mountain Research Station, General Technical Report.
- Arno, S. F.; Gruell, G. E. 1986. Douglas-fir encroachment into mountain grasslands in southwestern Montana. *Journal Range Management* 39:272-275.
- Arno, S. F.; Reinhardt, E. D.; Scott, J. H. 1993. Forest structure and landscape patterns in the subalpine lodgepole pine type: a procedure for quantifying past and present conditions. USDA Forest Service, General Technical Report INT-294. 17 p.
- Arno, S. F.; Scott, J. H.; Hartwell, M. G. 1995. Age-class structure of old growth ponderosa pine/Douglas-fir stands and its relationship to fire history. Research Paper INT-481, USDA Forest Service. 25 p.
- Arno, S. F.; Simmerman, D. G.; Keane, R. E. 1985. Forest succession on four habitat types in western Montana. USDA Forest Service, General Technical Report INT-177.
- Arno, S. F.; Smith, H. Y.; Krebs, M. A. 1997. Old growth ponderosa pine and western larch stand structures: Influences of pre-1900 fires and fire exclusion. USDA Forest Service, Research Paper INT-495. 20 p.
- Bailey, D. W.; Losensky, B. J. 1996. Fire in western Montana ecosystems: a strategy for accomplishing ecosystem management through the effective use of prescribed fire in the Lolo National Forest. Lolo National Forest, Missoula, MT. 145 p.
- Baker, W. L. 1992. Effects of settlement and fire suppression on landscape structure. *Ecology* 73:1879-1887.
- Baker, W. L. 1993. Spatially heterogeneous multi-scale response of landscapes to fire suppression. *Oikos* 66:66-71.
- Barboulletos, C. S.; Morelan, L. Z.; Carroll, F. O. 1998. We will not wait: why prescribed fire must be implemented on the Boise National Forest. In: Pruden, T. L.; Brennan, L. A., editors. Fire in ecosystem management: shifting the paradigm from suppression to prescription. Tall Timbers Fire Ecol. Conference Procedure No. 20. Tall Timbers Research Station, Tallahassee, FL: 27-30.
- Barrett, S. W. 1982. Fire's influence on ecosystems of the Clearwater National Forest: Cook Mountain fire history inventory. Clearwater National Forest, Fire Management, Orofino, ID. 42+ p.
- Barrett, S. W. 1988. Fire suppression's effects on forest succession within a central Idaho Wilderness. *Western Journal Applied Forestry* 3:76-80.
- Barrett, S. W. 1994. Fire regimes on andesitic mountain terrain in northeastern Yellowstone National Park, Wyoming. *Int. Journal Wildland Fire* 4(2):65-76.
- Barrett, S. W.; Arno, S. F. 1982. Indian fires as an ecological influence in the Northern Rockies. *Journal Forestry* 80:647-651.
- Barrett, S. W.; Arno, S. F.; Key, C. H. 1991. Fire regimes of western larch-lodgepole pine forests in Glacier National Park, Montana. *Canadian Journal Forestry Research* 21:1711-1720.
- Biondi, F. 1996. Decadal-scale dynamics at the Gus Pearson Natural Area: evidence for inverse (a)symmetric competition? *Canadian Journal Forestry Research* 26:1397-1406.
- Boyd, R., editor. 1999. Indians, fire, and the land in the Pacific Northwest. Oregon State University Press, Corvallis, OR. 313 p.
- Brown, J. K. 1975. Fire cycles and community dynamics in lodgepole pine forests. In: Baumgartner, D. M., editor. Management of lodgepole pine ecosystems, symposium proceedings. Cooperative Extension Service, Washington State University, Pullman, WA: 429-456.
- Brown, J. K. 1995. Fire regimes and their relevance to ecosystem management. In: Proceedings of Society of American Foresters National Convention; 1994 Sept. 18-22; Anchorage, AK. Society of American Foresters, Bethesda, MD: 171-178.
- Brown, J. K.; Arno, S. F.; Barrett, S. W.; Menakis, J. P. 1994. Comparing the prescribed natural fire program with presettlement fires in the Selway-Bitterroot Wilderness. *Int. Journal of Wildland Fire* 4:157-168.
- Brown, J. K.; Arno, S. F.; Bradshaw, L. S.; Menakis, J. P. 1995a. Comparing the Selway-Bitterroot fire program with presettlement fires. In: Brown, J. K.; Mutch, R. W.; Spoon, C. W.; Wakimoto, R.H. tech. coords. 1995. Proceedings: symposium on fire in wilderness and park management. USDA Forest Service General Technical Report, INT-320:48-54.
- Brown, J. K.; Mutch, R. W.; Spoon, C. W.; Wakimoto, R. H., tech. coords. 1995b. Proceedings: symposium on fire in wilderness and park management; USDA Forest Service, General Technical Report, INT-320.
- Covington, W. W.; Fule, P. Z.; Moore, M. M.; Hart, S. C.; Kolb, T. E.; Mast, J. N.; Sackett, S. S.; Wagner, M.R. 1997. Restoring ecosystem health in ponderosa pine forests of the Southwestern United States. *Journal Forestry* 95(4):23-29.
- Crane, M. F.; Fischer, W. C. 1986. Fire ecology of the forest habitat types of central Idaho. USDA Forest Service, General Technical Report, INT-218. 86 p.
- Czech, B. 1996. Challenges to establishing and implementing sound natural fire policy. *Renewable Resources Journal* 14(2):14-19.
- Davis, K. M. 1977. The fire history of Coram Experimental Forest. M. S. Thesis., University of Montana, Missoula, MT. 134 p.
- Fellin, D.G. 1980. A review of some relationships of harvesting, residue management, and fire to forest insects and disease. In: Environmental consequences of timber harvesting in Rocky Mountain coniferous forests. USDA Forest Service, General Technical Report, INT-90:335-414.
- Finney, M. A. 1998. FARSITE: Fire Area Simulator—model development and evaluation. USDA Forest Service, Research Paper RMRS-RP-4. 47 p.
- Fischer, W. C.; Bradley, A. F. 1987. Fire ecology of western Montana forest habitat types. USDA Forest Service, General Technical Report, INT-223. 95 p.
- Gabriel, H. W., III. 1976. Wilderness ecology: The Danaher Creek drainage, Bob Marshall Wilderness, Montana. Ph.D. Disserta., University of Montana, Missoula, MT. 224 p.
- Gruell, G. E. 1980. Fire's influence on wildlife habitat on the Bridger-Teton National Forest, Wyoming. Vol. 1—photographic record and analysis. USDA Forest Service, Research Paper INT-235. 207 p.
- Hardy, C. C. 1999. personal comm. Research Forester, Rocky Mountain Research Station, Missoula, MT.
- Harrington, M. G. 1996. Prescribed fire applications: restoring ecological structure and process in ponderosa pine forests. In: Hardy, C. C.; Arno, S. F., editors. The use of fire in forest restoration. USDA Forest Service, General Technical Report INT-341: 41.
- Hartwell, M. G.; Alaback, P.; Arno, S. F. [In press.] Comparing historic and modern forests on the Bitterroot Front. In: Smith, Helen Y., ed. The Bitterroot Ecosystem Management Research Project: What we have learned. Symposium proceedings; 1999 May 18-20; Missoula, MT. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Hill, B. T. 1998. Western National Forests: Catastrophic wildfires threaten resources and communities. U.S. General Accounting Office, Congressional Testimony: GAO/T-RCED-98-273. 23 p.
- Houston, D. B. 1973. Wildfires in northern Yellowstone National Park. *Ecology* 54:1111-1117.
- Johnson, E.A.; Larsen, C.P.S. 1991. Climatically induced change in fire frequency in the southern Canadian Rockies. *Ecology* 71:194-201.

- Keane, R. E.; Arno, S. F. 1993. Rapid decline of whitebark pine in western Montana: evidence from 20-year remeasurements. *Western Journal of Applied Forestry* 8(2):44-47.
- Keane, R. E.; Garner, J. L.; Schmidt, K. M.; Long, D. G.; Menakis, J. P.; Finney, M. A. 1998b. Development of input spatial data layers for the FARSITE fire growth model for the Selway-Bitterroot Wilderness complex, USA. USDA Forest Service, General Technical Report RMRS-GTR-3. 66 p.
- Keane, R. E.; Morgan, P.; Menakis, J. P. 1994. Landscape assessment of the decline of whitebark pine (*Pinus albicaulis*) in the Bob Marshall Wilderness complex, Montana, USA. *Northwest Science* 68(3): 213-229.
- Keane, R. E.; Ryan, K. C.; Finney, M. A. 1998a. Simulating the consequences of altered fire regimes on a complex landscape in Glacier National Park, USA. In: Pruden, T. L.; Brennan, L. A. editors: Tall Timbers Fire Ecology Conference No. 20, Tall Timbers Research Station, Tallahassee, FL: 310-324.
- Keane, R. E.; Ryan, K. C.; Running, S. W. 1996. Simulating effects of fire on northern Rocky Mountain landscapes with the ecological process model FIRE-BGC. *Tree Physiology* 16:319-331.
- Kilgore, B. M.; Curtis, G. A. 1987. Guide to understory burning in ponderosa pine-larch-fir forests in the Intermountain West. USDA Forest Service, General Technical Report INT-233. 39 p.
- Kurth, L. J. 1996. Examples of fire restoration in Glacier National Park. In: Hardy, C. C.; Arno, S. F. editors. The use of fire in forest restoration. USDA Forest Service, General Technical Report INT-341:54-55.
- Leopold, A. S.; Cain, S. A.; Cottam, C.; Gabrielson, I. N.; Kimball, T. L. 1963. Report of the Advisory Board on Wildlife Management in the National Parks. *Sierra Club Bulletin* 48(3):4-11.
- Lyon, L. J. 1984. The Sleeping Child Burn—21 years of postfire change. USDA Forest Service, Research Paper, INT-330. 17 p.
- McClelland, B. R. 1979. Habitat management for hole-nesting birds in forests of western larch and Douglas-fir. *Journal Forestry* 77:480-483.
- McGregor, M. D.; Cole, D. M. editors. 1985. Integrating management strategies for the mountain pine beetle with multiple-resource management of lodgepole pine forests. USDA Forest Service General Technical Report INT-174. 68 p.
- Minore, D. 1979. Comparative autecological characteristics of North-western tree species—a literature review. USDA Forest Service, General Technical Report, PNW-87.
- Monnig, E.; Byler, J. 1992. Forest health and ecological integrity in the Northern Rockies. USDA Forest Service, Northern Region, Forest Pest Management Report 92-7. Missoula, MT. 18 p.
- Morgan, P.; Aplet, G. H.; Haufler, J. B.; Humphries, H. C.; Moore, M. M.; Wilson, W. D. 1994. Historical range of variability: a useful tool for evaluating ecosystem change. *Journal of Sustainable Forestry* 2:87-111.
- Morgan, P.; Bunting, S. C.; Black, A. E.; Merrill, T.; Barrett, S. 1998. Past and present fire regimes in the interior Columbia River basin. In: Fire management underfire (adapting to change); proc. of the 1994 Interior West Fire Council Meeting. International Association of Wildland Fire, Fairfield, WA:77-82.
- Murray, M. P. 1996. Landscape dynamics of an island range: interrelationships of fire and whitebark pine (*Pinus albicaulis*). Ph.D. Disserta. University of Idaho, Moscow, ID. 121 p.
- Mutch, R. W. 1997. Need for more prescribed fire: but a double standard slows progress. In: Bryan, D. C. editor. Environmental regulation and prescribed fire: legal and social challenges. Conference Proceedings, Florida Division of Forestry. Tallahassee, FL:8-14.
- Nickas, G. 1999. Wilderness fire. *Wilderness Watcher* 10(1):4-5.
- O'Laughlin, J.; MacCracken, J. G.; Adams, D. L.; Bunting, S. C.; Blatner, K. A.; Keegan, C. E.; III. 1993. Forest health conditions in Idaho: executive summary. Idaho Forest, Wildlife and Range Experiment Station Report No. 11., University of Idaho, Moscow, ID. 37 p.
- Oliver, C. D.; Larson, B. C. 1996. Forest stand dynamics. Wiley, New York. 520 p.
- Parsons, D. J.; Landres, P. B. 1998. Restoring natural fire to wilderness: how are we doing? In: Pruden, T. L.; Brennan, L. A. Fire in ecosystem management: shifting the paradigm from suppression to prescription. Tall Timbers Fire Ecology Conference Procedure No. 20. Tall Timbers Research Station, Tallahassee, FL:366-373.
- Quigley, T. M.; Haynes, R. W.; Graham, R. T. technical editors. 1996. Integrated scientific assessment for ecosystem management in the Interior Columbia Basin. USDA Forest Service, General Technical Report PNW-382. 303 p.
- Reinhardt, E.; Keane, R. E.; Brown, J. K. 1997. First order fire effects model: FOFEM 4.0 User's Guide. USDA Forest Service, General Technical Report INT-GTR-344. 65 p.
- Romme, W. H. 1982. Fire and landscape diversity in subalpine forests of Yellowstone National Park. *Ecological Monographs* 52:199-221.
- Romme, W. H.; Despain, D. G. 1989. Historical perspective on the Yellowstone fires of 1988. *BioScience* 39:695-699.
- Scott, J. H. 1998. Fuel reduction in residential and scenic forests: a comparison of three treatments in a western Montana ponderosa pine stand. USDA Forest Service, Research Paper RMRS-5. 19 p.
- Smith, J. K.; Fischer, W. C. 1997. Fire ecology of the forest habitat types of northern Idaho. USDA Forest Service, General Technical Report INT-363. 142 p.
- Steele, R.; Arno, S. F.; Geier-Hayes, K. 1986. Wildfire patterns change in central Idaho's ponderosa pine-Douglas-fir forest. *Western Journal Applied Forestry* 1(1):16-18.
- Steele, R.; Geier-Hayes, K. 1993. The Douglas-fir/pinegrass habitat type in central Idaho: succession and management. USDA Forest Service, General Technical Report INT-298. 83 p.
- Stickney, P. F. 1990. Early development of vegetation following holocaustic fire in Northern Rocky Mountain forests. *Northwest Science* 64:243-246.
- Van Horn, F. 1999. personal comm. Fire Mgt. Officer, Glacier National Park, West Glacier, MT.
- Wellner, C. A. 1970. Fire history in the northern Rocky Mountains. In: The role of fire in the Intermountain West, Proc. School of Forestry, University of Montana, Missoula, MT:42-64.
- Woodley, S. 1995. Playing with fire: vegetation management in the Canadian Parks Service. In: Brown, J. K.; Mutch, R. W.; Spoon, C. W.; Wakimoto, R. H. technical coordinators. Proceedings: symposium on fire in wilderness and park management. USDA Forest Service, General Technical Report, INT-320:30-33.
- Zack, A. C.; Morgan, P. 1994. Fire history on the Idaho Panhandle National Forests. USDA Forest Service, Idaho Panhandle National Forests, Coeur d'Alene, ID. 44 p.

Returning Fire to the Mountains: Can We Successfully Restore the Ecological Role of Pre-Euroamerican Fire Regimes to the Sierra Nevada?

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Abstract—This paper examines the resultant conditions of Sequoia and Kings Canyon National Park's burn program relative to knowledge about past fire regimes in this ecosystem. Estimates of past fire-return intervals provide management direction and were used to develop approximations of area burned prior to Euroamerican settlement. This information was used to develop simple methods to compare fire management achievements against historic benchmarks. Two analyses were used to evaluate the results of the burn program relative to pre-settlement conditions. These were a reconstruction of annual "area burned" within major vegetation classes and an analysis of "fire return-interval departures" (FRID), with and without management fires, over the past 30 years. Given the current information base about fire regimes, the "area burned" analysis indicated the burn program continues to fall behind, relative to forest change, while the FRID analysis suggested the program has had a substantial impact on areas with the greatest ecological need for burning.

Striking changes in structural and functional components of Sierran ecosystems have occurred since 1860, largely due to alternations in the pre-Euroamerican settlement fire regime (Leopold and others 1963; Kilgore 1973; Vankat and Major 1978). Shifts in the fire regime have been attributed to multiple causes, including intense grazing that removed fine fuels important for fire spread, loss of Native American populations as an ignition source and, more recently, 20th century fire suppression efforts (Caprio and Swetnam 1995; Kilgore and Taylor 1979). Today unnaturally heavy fuel accumulations occur in many of Sequoia and Kings Canyon National Park's fire-dependant forest ecosystems along with associated increases in forest stand densities (Kilgore 1972, 1973; Vankat and Major 1978). With these shifts have come changes in fire regime characteristics, with large stand-destroying burns (>1 ha) occurring in plant communities (i.e. mixed-conifer forest) where such burns were exceedingly rare or unknown in the

past. Because National Park Service policy states that parks will protect natural resources, life, and property from unnatural wildfires and restore and maintain natural fire regimes to perpetuate natural processes and values, an active fire management program has been implemented within the parks.

The fire management program in Sequoia and Kings Canyon National Parks (SEKI) began using prescribed fire extensively in 1968 (Bancroft and others 1985), when the first large prescribed burn on NPS lands in the Western states was ignited (Kilgore 1971). The overall fire management goals of this active program have been to restore and maintain fire as a natural process to the maximum extent possible. Specific program objectives have generally focused on fuel reduction, although they have recently been undergoing modification to include ecological function and the preservation and restoration of the structural components of plant communities (Keifer and others 2000). Since 1921, when written historic fire records began, 60,370 ha have burned in the Parks, with 34,776 ha (58%) having been some form of management fire (either a human-ignited prescribed burn or a lightning-ignited burn given various names over the years—"let burns," "prescribed natural fire" and, most recently, "wildland fire used for resource benefit"). Today, the Parks are one of the leading NPS units using fire for resource benefits.

However, although SEKI is a leader in utilizing fire, there continues to be considerable debate about whether the program has been successfully restoring the ecological role of fire within park ecosystems. We offer here a quantitative evaluation of fire management program achievements over the past 30 years in reducing fuels and restoring fire as an ecological process relative to historic benchmarks based on pre-Euroamerican conditions. We used two approaches to evaluate the effectiveness: (1) the area-burned approach extends the ideas of several authors (Graber and Parsons 1998; van Wagtenonk 1995) by applying information on fire-return intervals (FRI) derived from fire history studies (such as those calculated by Parsons (1995) or Parsons and Botti (1996) for sequoia groves) to derive an estimate of what the annual average area burned prior to 1860 might have been, (2) our second analysis used a geospatial model of fire-return interval departures (FRID) from pre-Euroamerican conditions (Caprio and others 1997, in press) to evaluate quantitative and spatial aspects of the SEKI burn program. Actual 1998 FRID values were compared to 1998 FRID values for a hypothetical landscape where management burns had not been carried out.

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Study Area

Sequoia and Kings Canyon National Parks are located in the south central Sierra Nevada and encompass some 349,676 ha (864,067 ac) extending from the Sierra crest to the western foothills on the eastern edge of the San Joaquin Valley. Topographically, the area is rugged, with elevations ranging from 485 to 4,392 m (1,600 to 14,495 ft). The Parks are drained by the Kern, Kaweah, Kings and San Joaquin Rivers. The elevation gradient from the foothills to the higher peaks is steep on both the east and west margins of the Sierra, with rapid transitions between vegetation communities. Three broad vegetation zones dominate the Parks (slightly over 200,000 ha are vegetated by forest, shrub or grassland communities)—*foothills* (485 to 1,515 m) composed of annual grasslands, oak and evergreen woodlands and chaparral shrubland, *conifer forest* (1,515 to 3,030 m) with ponderosa (*Pinus ponderosa* Dougl.), lodgepole (*P. contorta* Dougl. var *Murrayana* Englm.), giant sequoia (*Sequoiadendron giganteum* [Lindl.] Buchholz), white fir (*Abies concolor* Lindl. & Gord.) and red fir (*A. magnifica* Murr.) forests, and *high country* (3,030 to 4,392 m) composed of subalpine forests with foxtail pine (*P. balfouriana* Jeff.), whitebark pine (*P. albicaulis* Englm.), alpine vegetation and unvegetated landscapes. A variety of classification schemes have been defined for vegetation within the Parks (Rundel and others 1977; Stephenson 1988; Vankat 1982).

The climate is Mediterranean, with cool, moist winters and warm summers with rainfall limited to sporadic summer thunderstorms associated with monsoonal flow from the Southwest. Precipitation increases as elevation increases, to about 102 cm (40 in) annually, from 1,515 to 2,424 m on the west slope of the Sierra, decreasing as one moves higher and to the east (Stephenson 1988). Substantial snow accumulations are common above 1,515 m during the winter. Total annual precipitation during the period of record has varied from 30 to 130 cm at Ash Mountain in the foothills and from 38 to 214 cm in Giant Forest at a mid-elevation location.

European settlement of the area began in the 1860s with extensive grazing, minor logging and mineral exploration. Sequoia National Park and Grant National Park (now part of Kings Canyon National Park) were founded in 1890 with the intent of protecting sequoia groves from logging. Over time, significant new areas have been added to the Parks, including the Kern Drainage (1926), while much of the upper portion of the upper Kings drainage was set aside as Kings Canyon National Park (1940 and 1965) (Dilsaver and Tweed 1990; Farquhar 1965).

Methods

Burn Area Analysis

We applied summarized FRI data (RI_{avg} and RI_{imax}) to each of the 12 major vegetation classes currently defined for the Parks (Caprio and Lineback, in press). RI_{avg} was based on mean FRI, while RI_{imax} was a more conservative estimate based on mean maximum intervals. Both were based on dendrochronological sampled fire histories for the period from 1700 to 1860. Because of the importance of aspect in affecting fire behavior and spread (Agee 1993; Pyne and others 1996), we refined FRI estimates and vegetation

classes to include this influence and provide a more realistic estimate of area burned. Several fire history investigations have reported such shifts in FRI by aspect (Allen and others 1995; Laven and others 1980; Taylor and Skinner 1998). Most FRI data summarized in Caprio and Lineback (in press) were generally representative of south aspects, with the exception of the estimate for red fir forest (data from Pitcher 1987 and Caprio 1998). This information has recently been supplemented by recent fieldwork in SEKI, comparing differences in FRI between north and south aspects. It also suggests striking differences, with FRI about three-times greater in mid-elevation conifer forest on south aspects relative to similar north aspects (Caprio, unpublished data). To be conservative, we only doubled the values on south aspects relative to north aspects.

Area estimates for north and south aspects for the 12 major vegetation classes were delineated using GIS (fig. 1), with south aspects defined topographically as aspects from 105-184° and north as 185-104° (Caprio and Lineback, in press). Due to lags in surface heating N/S delineation was skewed to the west. Aspects were interpreted and digitized from a topographic map of the Parks (1:25,000), with areas greater than 250 contiguous hectares mapped. Smaller landscape units were not included to remove the influence of micro-topographic features imbedded within a dominant aspect. Lastly, using both the RI_{avg} and RI_{imax} FRI estimates, an estimate of area burned annually prior to Euroamerican settlement was determined by dividing the area within a vegetation class and aspect by the FRI for that category then summing these across all vegetation categories.

FRID Analysis

Resource managers at Sequoia and Kings Canyon National Parks have been developing an “ecological needs model” that conservatively categorizes vegetation types based on departures from pre-Euroamerican settlement fire return intervals (FRID) (Caprio and others 1997, in press). Landscape units defined in this model may be further categorized to allow integration of information about burn status—such as whether an area is unburned, undergoing restoration burns or is in a maintenance condition—within the FRID values.

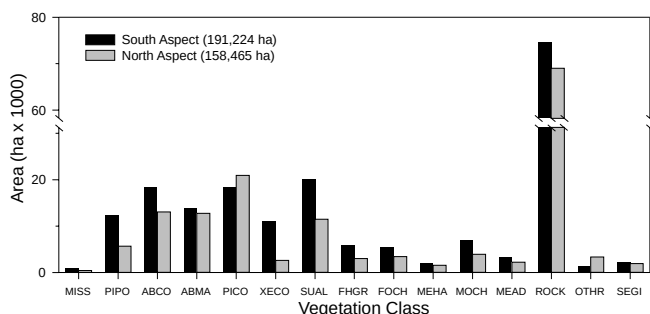


Figure 1—Area of each vegetation class by aspect used to calculate burn area values. See table 2 for explanation of vegetation class codes (nonvegetation types not listed in table 2 are MISS = missing; ROCK = rock; OTHR = other).

$$\text{Fire Return Interval Departure (FRID)} = \frac{\text{TSLF} - \text{RI}_{\max}}{\text{RI}_{\max}}$$

in which,

$\text{RI}_{\max}$ = maximum average return interval for the vegetation class (maximum values provide a conservative estimate)

and,

TSLF (time since last fire) = time that has passed since the most recent fire based on historic fire records or using a baseline date of 1899, derived from fire history chronologies, of when areas last burned.

The departure index ranged from negative one to 16, given our data set with a starting TSLF of 1899 (this date was used as a conservative estimate of when the last fire burned) and a minimum $\text{RI}_{\max}$ value of six (formula is modified from Caprio and others (1997) to give departure values as positive numbers). We reclassified the index values into four rating categories that were likely to capture current forest conditions and the need for burning based on historic FRI (table 1).

Our analysis compared the differences between FRID values across the landscape relative to what they would have been if no management burns had occurred between 1968 and 1998. We defined management burns for this analysis as being either management-ignited prescribed fire (MIPF) or prescribed natural fire (PNF). Maps and data were developed using ArcInfo/GRID and ArcView (ESRI 1997) for the “actual” 1998 FRID and the alternative 1998 “no management ignitions” FRID. Comparison of these two sets of geographic data allowed quantitative and spatial comparisons to be made about the Parks burn program. In addition, hypothetical annual FRID values with “no fire occurrence” since 1899 were calculated for a period beginning in 1900. This provided a baseline that allowed us to contrast the impact of various fire scenarios relative to a no-fire landscape. Our FRID analysis did not include an aspect component since this element had not yet been integrated into the geospatial model on which FRID is calculated. The model used vegetation classes that were combined across aspects.

Results

Burn Area Analysis

Average area burned annually from 1921 to 1968 under full fire suppression was 325 ha relative to 1,504 ha burned annually following the initiation of management burning (fig. 2). Significant fire years, with greater than 1,000 ha burned, only occurred three times prior to 1969 (1926, 1948, 1950), compared to 16 times since 1969. Overall, 60,370 ha have burned in the Parks with 34,776 ha (58%) being some form

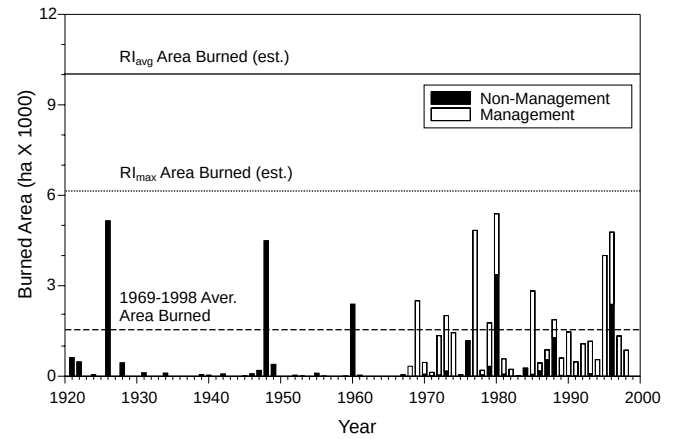


Figure 2—Area burned annually within SEKI since 1921 by management and nonmanagement fires. Comparison of area burned over the last 30 years relative to estimates area burned prior to Euroamerican settlement is shown by horizontal lines. The greatest annual area burned by management ignited fires occurred in 1977 while the greatest number of hectares burned in any given year since 1921 occurred in 1980.

of management fire. Since 1969, 45,111 ha have burned, with 34,776 ha (77%) of this from management fires.

Total area burned annually prior to Euroamerican settlement, without separating aspects, was estimated to be 11,697 ha using RI_{avg} and 7,142 ha using RI_{max} . When aspect differences in FRI were considered, reconstructed estimates for the combined average area burned annually in the Parks was $10,006 \text{ ha} \cdot \text{yr}^{-1}$ using RI_{avg} and $6,113 \text{ ha} \cdot \text{yr}^{-1}$ using RI_{max} (table 2 and table 3). The vegetation types with the greatest contribution to area burned annually were ponderosa-mixed conifer (PIPO), white fir-mixed conifer (ABCO) and red fir (ABMA). Vegetation classes that were minor contributors to the annual area burned included: montane chaparral (MOCH), lodgepole pine forest (PICO), foothill chaparral (FOCH), subalpine forest (SUCO) and meadow (MEAD). Annual contribution was dependant on both total area occupied by a vegetation type and the length of the FRI. While the area occupied by ponderosa-mixed conifer was only about 42% of the area of lodgepole pine forest, the vegetation class with the largest area in the Parks, it burned about 25 times more frequently. The result was the greatest average area burned annually of all the vegetation classes.

The reconstructed estimates of area burned annually also indicated that about three times more area burned on south aspects than on north aspects. Aspect differences in annual area burned were greatest for xeric conifer forest and ponderosa pine-mixed conifer forest (5.7 and 4.2 times more area burned on south than north aspects, respectively). Minimal differences were suggested for red fir, lodgepole pine forest and sequoia-mixed conifer forest (only 1.7, 1.8, and 2 times more area on south versus north aspects).

FRID Analysis

Our FRID analysis produced detailed geospatial output that provided both quantitative information and maps of FRID categories that were important tools for visually interpreting changes in FRID. Comparison of the maps

Table 1—Fire return interval departure (FRID) index for each ecological need category.

Extreme	High	Moderate	Low
•5	<5 and •2	<2 and •0	<0

Table 2—Burn area values based on mean fire return intervals (RI_{avg}). Return interval values were based on Caprio and Lineback (in press).

Vegetation class	Code	RI_{avg} north	RI_{avg} south	Ha/Yr north	Ha/Yr south	Ha/yr
Ponderosa-mixed conifer forest	PIPO	8	4	662.9	2,816.4	3,479.3
White fir mixed conifer forest	ABCO	20	10	716.7	1,712.9	2,429.6
Red fir forest	ABMA	30	15	477.9	811.8	1,289.7
Lodgepole pine forest	PICO	204	102	100.3	183.8	284.2
Xeric conifer forest	XECO	60	30	59.1	338.6	397.7
Subalpine conifer forest	SUAL	374	187	30.6	107.3	137.8
Foothill hardwoods and grasslands	FHGR	22	11	155.8	495.9	651.8
Foothill chaparral	FOCH	60	30	54.1	187.0	241.2
Mid-elevation hardwood forest	MEHA	14	7	119.4	270.4	389.8
Montane chaparral	MOCH	60	30	65.6	230.1	295.7
Meadow	MEAD	80	40	32.1	72.3	104.5
Giant sequoia groves	SEGI	20	10	100.3	204.9	305.2
Total				2,574.8	7,431.5	10,006.4

Table 3—Values based on mean maximum fire return intervals (R_{max}). Return interval values were based on Caprio and Lineback (in press).

Vegetation class	R_{max} north	R_{max} south	Ha/yr north	Ha/yr south	Ha/yr combined
Ponderosa-mixed conifer forest	12	6	441.9	1,877.6	2,319.5
White fir mixed conifer forest	32	16	447.9	1,070.6	1,518.5
Red fir forest	50	25	286.7	487.1	773.8
Lodgepole pine forest	326	163	62.8	115.0	177.8
Xeric conifer forest	100	50	35.4	203.2	238.6
Subalpine conifer forest	1016	508	11.3	39.5	50.7
Foothill hardwoods and grasslands	34	17	100.8	320.9	421.7
Foothill chaparral	120	60	27.1	93.5	120.6
Mid-elevation hardwood forest	46	23	36.3	82.3	118.6
Montane chaparral	150	75	26.2	92.0	144.5
Meadow	130	65	19.8	44.5	64.3
Giant sequoia groves	32	16	62.7	128.1	190.8
Total			1,595.0	4,554.3	6,113.3

showed attributes of current and no-management burn FRID and information about how and where they differed. Striking differences were obvious by visual inspection of the actual 1998 FRID map to the 1998 FRID map where all management fires had been removed (fig. 3).

Baseline estimates of FRID, if no fires had occurred in the Parks since 1899 (fig. 4), showed change in FRID through time, with “break points” when FRID values jumped between categories. This baseline provided values against which to assess “actual” burn area values. In addition, understanding the temporal location of the break points was important in interpreting changes in FRID through time. Specific shape and location of the break points depended on how the four FRID categories (low, moderate, high, extreme) are defined and spatial area of the various vegetation classes.

We made comparisons of three potential FRID outcomes: actual 1998 FRID, hypothetical 1998 FRID if no fires had occurred since 1899 and 1998 FRID excluding management burns (fig. 5). The difference between the hypothetical and the actual 1998 FRID showed change due to all fires that have occurred since 1921. The difference between the hypothetical 1998 FRID and the 1998 no management FRID showed the impact of all suppressed fires since 1921. To evaluate the burn program over the past 30 years we used the difference between the actual 1998 FRID and the 1998

FRID without management fires. The difference provided an estimate of change in 1998 FRID due to management burns. This comparison of 1998 data indicates that the SEKI burn program has reduced area in the extreme category by 28% and increased area in the low category by 23% (table 4). Only moderate or little change was observed in the moderate and high 1998 FRID category. These data show the current state of all areas burned since 1968 and do not reflect information about the specific category of the areas burned. Visual interpretation shows that areas with greatest changes in FRID values are the Grant Grove-Redwood Mountain area, Cedar Grove, Sugarloaf Valley and both the Swanee area of the Marble Fork and much of the Middle Fork of the Kaweah River. Some areas (Redwood Mountain, Middle Fork of the Kaweah and Swanee), where burns had been carried out in the 1970s and 1980s with no subsequent burning, are now reverting back to higher FRID categories.

Discussion

Burn Area Analysis

Aspect differences in area burned annually (table 2 and table 3) are greater than expected based on simple FRI and

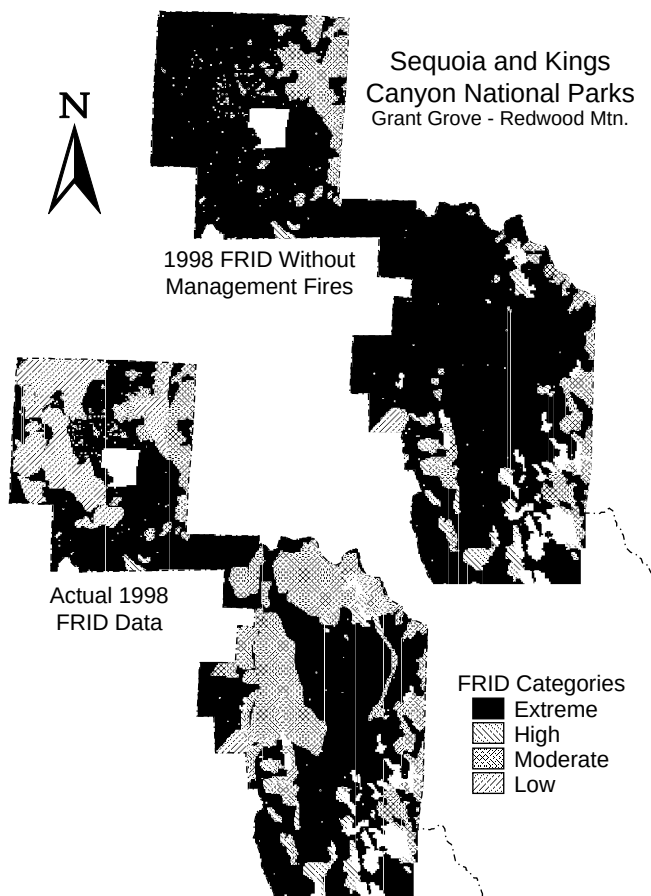


Figure 3—Graphical representation of the impact of management burning on the landscape for the Grant Grove/Redwood Mountain area of the parks. Maps show the differences in 1998 FRID values when management fires are excluded (top) or included (bottom) in the analysis.

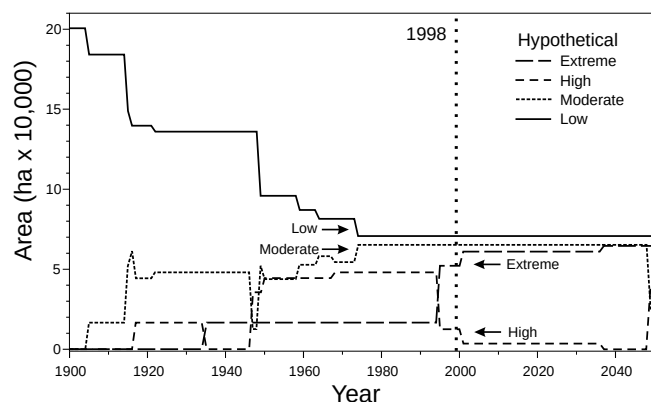


Figure 4—Change in FRID category values through time (since 1899) if complete fire suppression had been achieved since 1899. These values provide a baseline to compare current values and recent changes in FRID. Specific rates of change through time and inflection points depend on FRI for specific vegetation class. Actual FRID category values for 1998 are shown along the vertical dotted line and show a greater than expected area in the “low” category and a lower than expected area in the “high” category.

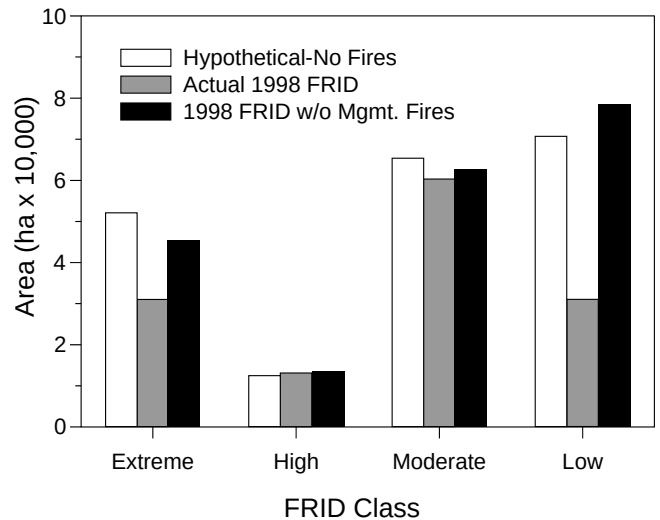


Figure 5—Area in the four FRID classes under three management scenarios. These include no fires since 1899 (complete fire suppression), actual 1998 FRID values, and 1998 FRID values if no management burning had occurred. The difference between the actual 1998 FRID and 1998 FRID without management fires represents the impact of the fire management program for the last 30 years on FRID values. The greatest changes are in the “high” and “low” categories.

total area categorized as south aspect (191,224 ha) versus north (158,465 ha) (the few flat areas are categorized as south aspect). Overall differences appeared to be due to changes in FRI and aspect by vegetation class. Most importantly, vegetation types with the highest fire frequency are located on south aspects. For example, ponderosa pine-mixed conifer forest, with the shortest average FRI, is more prevalent on south aspects (11,266 vs 5,303 ha; fig. 1), along with xeric conifer (10,158 vs 3,544 ha), although FRI are longer for the latter and do not have as great an influence on the final differences.

The analysis identified high priority landscape units for potential fire restoration. These data indicate that prior to Euroamerican settlement, the general area with the highest amount of acreage burned in the Parks, on a year-to-year basis, was lower-elevation conifer forest on south aspects. Thus, as a result of fire exclusion over the past 140 years, these areas probably exhibit the greatest degree of vegetation change. This suggests they are areas where fire managers should concentrate efforts in restoring fire (such as ponderosa pine-mixed conifer forest found on south aspects). Once restoration is completed, maintenance of fire as a natural ecosystem process in a wilderness setting will be easier, and larger land units could be burned with fewer operational resources.

The values given for annual area burned are mean values. Actual area would be quite variable from year-to-year, ranging from years with little or no area burned to years when very large areas burned. Variation is predominantly a result of interannual fluctuations in weather and ignition sources.

Several potential problems exist with the current FRI data set used in the analysis. While we have high quality information from some vegetation classes, particularly on

Table 4—Area in the 1998 hypothetical FRID (no fires since 1899) and the actual 1998 FRID, the percent change, and area and percent change due to all non-management and management fires respectively.

FRID class	Hypoth. (ha)	Actual (ha)	(% Δ)	Non-mgmt Δ(ha)	(% Δ)	Mgmt Δ(ha)	(% Δ)
Extreme	52,069	31,208	-40.0	-6,509	-12.3	-14,374	-27.7
High	12,443	13,267	6.6	1,144	9.3	-325	-2.7
Moderate	65,347	60,935	-6.8	-2,769	-4.3	-1,671	-2.5
Low	70,681	95,126	34.6	8,265	11.4	16,150	23.2

south aspects, data are of much poorer quality from other classes and on north aspects. Caprio and Lineback (in press) reviewed the quality of this information and present a geospatial analysis of the Parks fire regime knowledge. Sampling is currently being carried out in the Parks to provide higher quality information about past fire regimes and their range of variation across a broad range of vegetation types and aspects (Caprio 1997, 1998). Our current estimate that FRI were two-times greater on south than north aspects was based on results from other regions in the West and supported by preliminary findings from within-park sampling at mid-elevation sites (Caprio, unpublished data). In addition, our current vegetation map contains discrepancies and lumps some similar vegetation associations. For example, the FRI found in ponderosa pine forest (3-4 years) is the shortest recorded in any vegetation type within the Parks (Caprio, unpublished data; Warner 1980), but the current vegetation classification lumps this type with ponderosa pine-mixed conifer. Similarly, western juniper, pinyon pine and Jeffrey pine communities are all combined into xeric conifer, although fire tolerances among the species are quite different (Wright and Bailey 1982).

Comparison of the two estimates for average pre-Euroamerican settlement area burned annually (fig. 5) show that the burn program has reached neither the RI_{avg} (10,006 ha) nor the more conservative estimate based on RI_{max} (6,143 ha), although area burned during several years (1977, 1980, 1995 and 1996) approached the later (fig. 2). The long-term average of 1,504 ha from 1969 to 1998 fell well below these estimates. A plot of cumulative area burned over time (fig. 6), both pre-Euroamerican and current, demonstrates the trajectory of divergence in annual area burned. Thus, the Parks are continuing to fall behind in area that needs to be burned, if pre-Euroamerican settlement conditions are the objective.

Notably, in no year since 1921 (when written fire records begin) does the area burned approach the RI_{avg} or RI_{max} level found prior to Euroamerican settlement. We believe this is a result of the dramatic vegetation and fuel changes that began in the 1860s and continued with fire suppression activities in the 20th century. Intense grazing at the end of the 19th century—Farquhar (1965) cites historic documents which indicate that over 500,000 sheep were being grazed in the Kings and Kern drainages by the early 1880s—probably broke up contiguous areas of fine fuel and caused temporary compositional shifts (at a minimum) in many plant communities. In addition, pre-Euroamerican settlement fires probably burned for long periods of time during the dry summer/fall months, periodically flaring up and making runs over large areas. While 20th century suppression actions may not have been able to catch all initial starts, they would have

been highly successful at containing burns during quiescent periods, which would effectively limit final fire size. Lastly, there is the possibility that the difference is due to the loss of Native American ignitions, although no direct evidence exists for or against this.

FRID Analysis

FRID analysis is a new GIS data/fire management technique being utilized at SEKI to assist in burn planning and operations. It has been useful in providing ecological input into fire management planning and operations. In addition, a variety of new types of information have been derived from the procedure. Our results reflect one of these analyses, in which actual FRID values were compared to FRID values from several potential historic fire management scenarios. While our analysis centers on past management decisions, this type of analysis could be used to extrapolate outcomes into the future to examine alternative management strategies.

The results of our FRID analysis portray the outcome of 30 years of management burning quite differently from the results of the “area burned” analysis. It suggests the Parks’ burn program is having substantial positive effects on many

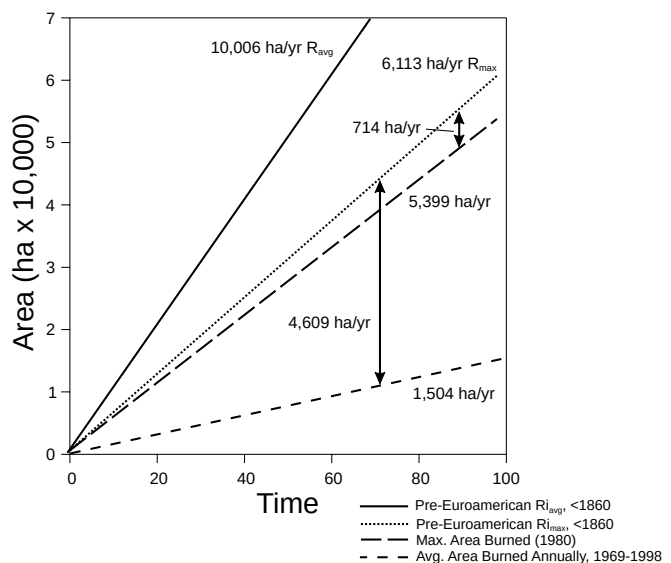


Figure 6—Accrual of area burned over time based on reconstructed pre-Euroamerican fire regimes (RI_{avg} as average FRI and RI_{max} as mean maximum FRI), the average actual area burned between 1969 and 1998, and the year with the maximum area burned between these dates (1996).

areas that have departed most significantly from pre-Euroamerican fire regimes. The difference in the results between the “area burned” analysis and FRID analysis reflects the spatial output of the latter and the fact that departures, which do not accrue annually, are grouped into specific categories with an upper limit of change. However, while large areas of the Parks have been treated, the FRID analysis also highlights areas where initial restoration burns took place, but subsequent restoration burns have not been executed (our current projection is that two-to-four restoration burns may be required to treat areas before burning can be considered to be routine maintenance). In these locations, any restoration gain from the initial burn is being lost as forest conditions revert back toward pre-burn conditions.

However, several problems in using FRID should be considered when interpreting output. They have been reviewed by Caprio and Lineback (in press) and include problems with the underlying vegetation map, aspect differences in fire regimes that have not yet been incorporated into the FRID model, and spatial limitations on the geographic extent of our fire regime knowledge across the Parks that are used to drive the model in a diverse ecosystem. Caprio and Lineback (in press) used several criteria to rate the quality of fire history data spatially across the Parks by vegetation class and aspect.

The two sets of analyses provide a valuable review and a first estimate of long-term targets for a burn program based on actual pre-Euroamerican settlement FRI within specific vegetation classes and aspects. The burn-area analysis gives quantitative guidelines on annual burn area for a land unit as a whole or for specific subcategories, such as vegetation class or aspect. FRID is valuable because it provides an index of the extent to which an area has departed from pre-Euroamerican settlement conditions. Both of these complement other methods used in describing changing fire regimes, such as cumulative frequency distributions or natural fire rotations. Use of these evaluation techniques may be useful for determining long-term success of a burn program and in guiding future direction in either highly managed or wilderness landscapes. However, such an evaluation requires a certain level of knowledge about past fire regimes within an ecosystem to provide an assessment with some accuracy.

Additional research should focus on relationships between the amplitude of FRID and the associated vegetative and fuel response for each vegetation type. If, for example, it is not possible for one reason or another to achieve a three-to-five year fire return interval in ponderosa pine, but it is possible to maintain a 12-year interval, is the latter rate sufficient to achieve desired ecological and fuel objectives within the bounds of normal range of variation?

Constraints

The challenge that remains, however, is how can the large expanse of area indicated by the fire history reconstructions be burned? Greater area can be achieved through the combined effects of using larger, variable-intensity ignitions (Parsons 1995) and increasing the reburning of areas burned in the recent past. The tree-ring fire history record suggests that large areas burned annually because a few common vegetation types burned at frequent intervals. The most

important of these was ponderosa pine-mixed conifer, followed by white fir and sequoia-mixed conifer. Frequent fires could occur in these vegetation types because burns were low-intensity understory fires with rapid fuel recovery; fuels components were probably a matrix of herbaceous species, the subshrub mountain misery (*Chamaebatia foliolosa* Benth.) and litter fall. In ponderosa pine-mixed conifer, reburns of a site would often occur within two or three years of the preceding fire (Caprio and Swetnam 1995; Caprio, unpublished data). In contrast, the burn program at SEKI has carried out very few secondary burns following initial restoration burns, which has hindered efforts to boost area burned over the long term. If a concerted effort were to be made to balance repeat burning with initial restoration ignitions, greater success might be achieved. Currently, considerable time and effort are applied to carrying out initial restoration burns, resulting in limited area burned annually due to the difficulty of implementation. Secondary restoration and, eventually, maintenance burns, where fuel, smoke and potential escape problems are minimal, could successfully accomplish much greater acreage annually.

A variety of constraints are encountered when examining the practicality of carrying out a burn program on the scale intended to replicate pre-Euroamerican settlement conditions. These include limited funding, unnatural fuel loads and forest structure where burning is difficult, air quality issues, availability of qualified personnel and other resources, political boundaries that may require continued use of managed fire, cultural and archeological concerns, occurrence of rare or invasive exotic species, difficulty in maintaining long-term management goals, poor knowledge about past and current ecosystem processes, fire regimes and structural components used for decision-making and inadequate standards to evaluate a burn program (Mitchell 1995; Parsons 1995; Parsons and Botti 1996; Parsons and Landres 1998). In addition, an ecosystem-level burn program must be carried out within a diverse and dynamic landscape with a high degree of biotic complexity. While burning *X* amount of area appears to be a simple goal, in actuality there are a suite of additional ecosystem elements that must be addressed by a fire program. Restoration of natural fire means returning fires to an ecosystem that burns with similar effects, frequencies, intensities and other characteristics of pre-Euroamerican settlement fire (Parsons and van Wagtenonk 1996). It must be understood that spatial and temporal heterogeneity of fire within ecosystems are important and need to be incorporated into a burn program (Parsons and Botti 1996).

Conclusion

Our two analyses provide a quantitative evaluation of the burn program at Sequoia and Kings Canyon National Parks over the past 30 years using new methods. They suggest that while progress has been made, considerable gaps still exist between the accomplishments of our current burn program in burning substantial amounts of area annually and our reconstructed pre-Euroamerican estimates. The difference is important because it indicates we are not maintaining fire as a natural process to the extent that policy prescribes. This goal will be accomplished when contemporary fires burn with similar characteristics to

pre-Euroamerican settlement fires (Parsons and van Wagtendonk 1996). This may be achieved through either natural ignitions or management ignitions where burning with naturally ignited fires is difficult or restricted. The difference also highlights the constraints that will always limit achievements. These may be insurmountable at specific locations, and alternative means of achieving management goals may be required.

Acknowledgments

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References

- Agee, J.K. 1993. *Fire Ecology of Pacific Northwest Forests*. Island Press, Washington, D.C. 493 p.
- Allen, C.D., R. Touchan, and T.W. Swetnam. 1995. Landscape-scale fire history studies support fire management action at Bandelier. *Park Science* 15:18-19.
- Bancroft, L., T. Nichols, D.J. Parsons, D.M. Graber, B. Evison, and J. van Wagtendonk. 1985. Evolution of the natural fire management program at Sequoia and Kings Canyon National Parks. p. 174-180. In: J.E. Lotan, B.M. Kilgore, W.C. Fischer, and R.W. Mutch (tech. coord.). *Proceedings-Symposium and Workshop on Wilderness Fire. Nov. 15-18, 1983, Missoula, MT*. USDA Forest Service Gen. Tech. Rep. INT-182.
- Caprio, A.C. unpublished data. Preliminary results: fire history of the East Fork watershed, Sequoia and Kings Canyon National Parks. Science and Natural Resources Management Division, Sequoia and Kings Canyon National Parks.
- Caprio, A.C. 1997. Fire History. pp. 51-54. In: A.C. Caprio (ed.). *Mineral King Risk Reduction Project: Monitoring, Inventory, and Research. 1996 Annual Report*, USDI NPS, on file at Sequoia and Kings Canyon National Parks. 75 p.
- Caprio, A.C. 1998. Fire History. pp. 56-61. In: A.C. Caprio (ed.). *Mineral King Risk Reduction Project: Monitoring, Inventory, and Research. 1997 Annual Report*, USDI NPS, on file at Sequoia and Kings Canyon National Parks. 91 p.
- Caprio, A.C., C. Conover, M. Keifer, and P. Lineback. 1997. Fire management and GIS: a framework for identifying and prioritizing fire planning needs. In: *Proceedings: 1997 ESRI GIS Conference, San Diego, CA*.
- Caprio, A.C., C. Conover, M. Keifer, and P. Lineback. in press. Fire management and GIS: a framework for identifying and prioritizing fire planning needs. In: *Proceedings of the Conference on Fire in California Ecosystems: Integrating Ecology, Prevention, and Management. Nov. 17-20, 1997, San Diego, CA*.
- Caprio, A.C. and P. Lineback in press. Pre-Twentieth Century Fire History of Sequoia and Kings Canyon National Parks: A Review and Evaluation of Our Knowledge. In: *Proceedings of the Conference on Fire in California Ecosystems: Integrating Ecology, Prevention, and Management. Nov. 17-20, 1997, San Diego, CA*.
- Caprio, A.C. and T.W. Swetnam. 1995. Historic fire regimes along an elevational gradient on the west slope of the Sierra Nevada, California. p. 173-179. In: J.K. Brown, R.W. Mutch, C.W. Spoon, R.H. Wakimoto (tech. coord.). *Proceedings: Symposium on Fire in Wilderness and Park Management: Past Lessons and Future Opportunities, March 30-April 1, 1993, Missoula Montana*. USDA Forest Service, INT-GTR-320, 283 p.
- Dilsaver, L.M. and W.C. Tweed. 1990. *Challenge of the Big Trees: A Resource History of Sequoia and Kings Canyon National Parks*. Sequoia Natural History Association, Inc. Three Rivers, CA. 379 p.
- ESRI. 1997. ArcView. Environmental Systems Research Institute, Inc., Redlands, CA.
- Farquhar, F.P. 1965. *History of the Sierra Nevada*. Univ. of Calif. Press, Berkeley, Calif. 262 p.
- Graber, D.M. and D.J. Parsons. 1998. Twenty-six years of prescribed fire management in Sequoia and Kings canyon National Parks: What has been accomplished in restoring fire and its effects? Abstract In: T.L. Pruden and L.A. Brennan. *Fire in Ecosystem Management: Shifting the Paradigm from Suppression to Prescription. Tall Timbers Fire Ecology Conference Proceedings*, 20:244
- Keifer, M., N.L. Stephenson and J. Manley. 2000. Prescribed fire as the minimum tool for wilderness forest and fire regime restoration: a case study from the Sierra Nevada, California. In: Cole, David N.; McCool, Stephen F. 2000. *Proceedings: Wilderness Science in a Time of Change*. Proc. RMRS-P-000. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Kilgore, B.M. 1971. The Role of Fire in Managing Red Fir Forests. *Trans. 36th North Amer. Wildlife and Nat. Res. Conf.*, p. 405-416.
- Kilgore, B.M. 1972. Fire's Role in a Sequoia Forest. *Naturalist*. 23:26-37
- Kilgore, B.M. 1973. The ecological role of fire in Sierran conifer forests: its application to national park management. *Quaternary Research*. 3:496-513
- Kilgore, B.M. and D. Taylor. 1979. Fire history of a sequoia-mixed conifer forest. *Ecology* 60:129-142
- Laven, R.D., P.N. Omi, J.G. Wyant, and A.S. Pinkerton. 1980. Interpretation of fire scar data from a ponderosa pine ecosystem in the central Rocky Mountains, Colorado. p. 46-49. In: M.A. Stokes and J.H. Dieterich (tech. coord.). *Proceedings of the Fire History Workshop. October 20-24, 1980, Tucson, Arizona*. USDA Forest Service, GTR-RM-81.
- Leopold, A.S., S.A. Cain, C.M. Cottam, J.M. Gabrielson, and T.L. Kimball. 1963. Wildlife management in the national parks. *Amer. Forests* 69:32-35, 61-63.
- McBride, J.R. and D.F. Jacobs. 1980. Land use and fire history in the mountains of southern California. pp. 85-88. In: M.A. Stokes and J.H. Dieterich (tech. coord.). *Proceedings of the Fire History Workshop. October 20-24, 1980, Tucson, Arizona*. USDA Forest Service, GTR-RM-81.
- Mitchell, B. 1995. The Federal Clean Air Act and its impact on fire management programs. p. 74-76. In: J.K. Brown, R.W. Mutch, C.W. Spoon, R.H. Wakimoto (tech. coord.). *Proceedings: Symposium on Fire in Wilderness and Park Management: Past Lessons and Future Opportunities, March 30-April 1, 1993, Missoula Montana*. USDA Forest Service, INT-GTR-320, 283 p.
- Parsons, D.J. 1995. Restoring fire to giant sequoia groves: what have we learned in 25 years? p. 256-258. In: J.K. Brown, R.W. Mutch, C.W. Spoon, R.H. Wakimoto (tech. coord.). *Proceedings: Symposium on Fire in Wilderness and Park Management: Past Lessons and Future Opportunities, March 30-April 1, 1993, Missoula Montana*. USDA Forest Service, INT-GTR-320, 283 pp.
- Parsons, D.J. and S.J. Botti. 1996. Restoration of fire in National Parks. p. 29-31. In: C.C. Collin and S.F. Arno (eds.). *The use of fire in forest restoration*. USDA For. Serv. Gen. Tech. Rep. INT-GTR-341. 86 p.
- Parsons, D.J. and P.B. Landres. 1998. Restoring natural fire to wilderness: how are we doing? p. 366-373. In: L.A. Brennan and T.L. Pruden (eds.). *Fire in ecosystem management: shifting the paradigm from suppression to prescription. Proceedings of the Tall Timbers Fire Ecology Conference, No. 20*. Tall Timbers Research Station, Tallahassee, FL.
- Parsons, D.J. and J.W. van Wagtendonk. 1996. Fire research and management in the Sierra Nevada national parks. p. 25-48. In: W.L. Halvorson and G.E. Davis (eds.). *Ecosystem management in the national parks*. Tucson, AZ: University of Arizona Press.
- Pitcher, D.C. 1987. Fire history and age structure in red fir forests of Sequoia National Park, California. *Canadian Journal of Forest Research*. 17:582-587.
- Pyne, S.J., P.L. Andrews, and R.D. Laven. 1996. *Introduction to Wildland Fire*. John Wiley and Sons, Inc., NY, 769 p.
- Rundel, P.W., D.P. Parsons, and D.T. Gordon. 1977. Montane and subalpine vegetation of the Sierra Nevada and Cascade Ranges. p. 559-599. In: M.G. Barbour and J. Major (eds.). *Terrestrial Vegetation of California*. Wiley & Sons, Inc., NY. 1002 p.

- Stephenson, N.L. 1988. Climatic control of vegetation distribution: the role of the water balance with examples from North America and Sequoia National Park, California. Dissertation, Cornell University, NY. 295 p.
- Taylor, A.H. and C.N. Skinner. 1998. Fire history and landscape dynamics in a late-successional reserve, Klamath Mountains, California, USA. *Forest Ecology and Management* 111:285-301.
- Vankat, J.L. 1982. A gradient perspective on the vegetation of Sequoia National Park, California. *Madroño* 29:200-214.
- Vankat, J.L. and J. Major. 1978. Vegetation changes in Sequoia National Park, California. *J. Biogeog.* 5:377-402.
- van Wageningen, J.W. 1995. Large fires in wilderness areas. p. 113-116. In: J.K. Brown, R.W. Mutch, C.W. Spoon, R.H. Wakimoto (tech. coord.). *Proceedings: Symposium on Fire in Wilderness and Park Management: Past Lessons and Future Opportunities, March 30-April 1, 1993, Missoula Montana*. USDA Forest Service, INT-GTR-320, 283 p.
- Warner, T.E. 1980. Fire History in the Yellow Pine Forest of Kings Canyon National Park. p. 89-92. In: M.A. Stokes and J.H. Dieterich (tech. coord.). *Proceedings of the Fire History Workshop. October 20-24, 1980, Tucson, Arizona*. USDA Forest Service, GTR-RM
- Wright, H.A. and A.W. Bailey. 1982. *Fire Ecology: United States and Southern Canada*. John Wiley & Sons, Inc., New York, 501 p.

Continuing Fire Regimes in Remote Forests of Grand Canyon National Park

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Abstract—Ponderosa pine forests in which frequent fire regimes continue up to the present would be invaluable points of reference for assessing natural ecological attributes. A few remote forests on the North Rim of Grand Canyon National Park come close to this ideal: never-harvested, distant from human communities and fire suppression resources, and with several low-intensity fires in the past century—a highly unusual recent fire regime in the Southwest. Recent fires appear to have played a crucial role in preventing the increases in forest density that characterize most southwestern pine forests. The study sites are not unaffected by the ecological changes associated with settlement, but they do present an important reference resource for study and management of ponderosa pine ecosystems.

Ponderosa pine forests in which historically frequent fire regimes continue up to the present would be invaluable points of reference for assessing natural attributes of ecological structure and function. Grand Canyon National Park contains one of the largest old-growth forests in the Southwest, where tree harvesting has not occurred and grazing has been eliminated for over 60 years. Although most fire disturbance regimes in the park have been disrupted since European settlement, a few remote sites may retain near-natural conditions. The majority of the forested area lies on the North Rim, part of the Kaibab Plateau, which supports ponderosa pine, mixed conifer and spruce-fir forests at elevations ranging from 7,500 to 9,165 feet on well-drained limestone soils.

The North Rim is remote from modern human communities, but Altshul and Fairley (1989) document the long human history of the region. The lower elevations of the rim were densely populated by Native Americans prior to 1250-1300 A.D. Six tribes—the Paiute, Hopi, Havasupai, Hualapai, Navajo, and Zuni—have ancestral and current connections to the canyon and rim habitat. An expedition led by the Spaniards Dominguez and Escalante in 1776 marked the first European presence on the Arizona Strip, the land north of the Colorado River that includes the

Kaibab Plateau. It took another 78 years before the first European settlement was begun by Mormon explorers and pioneers in 1854. Fighting with Utes and Navajos kept settlers out of the Arizona Strip until 1869. With the establishment of peace, there was a rapid expansion of livestock grazing, logging and mining activity.

Frequent fire regimes were disrupted in forested highlands of the Arizona Strip as early as 1870 in the Mt Trumbull area (Fulé and others, unpublished data). As elsewhere in the Southwest, early livestock grazing was excessive (Altshul and Fairley 1989) and removed fine herbaceous fuels, stopping fire spread. The establishment of the Grand Canyon Forest Reserve in 1893 and creation of Grand Canyon National Park (GCNP) in 1919 brought organized fire detection and suppression crews. On the North Rim, Wolf and Mast (1998) found complete fire exclusion by about 1920 in ponderosa pine and mixed conifer forests. Even in the high-elevation spruce-fir forests, with historically longer fire-return intervals and more severe fires, protracted fire exclusion has led to the development of increasingly dense, homogeneous stands (White and Vankat 1993). Park management policy has changed in recent decades to favor restoration of natural ecological processes, including fire (GCNP 1992), but the presently dense forests and heavy fuel loads hinder effective re-introduction of fire on much of the North Rim. Fuel problems are not only an ecological concern: the difficulty of fire management on the North Rim has been cited by Pyne (1989) as a major factor impeding the Park's plan for wilderness designation of the area (Morehouse 1996).

Among the challenges faced by managers are: 1) lack of knowledge about natural ecological conditions as a point of reference for restorative management (Moore and others 1999), 2) uncertainty about the appropriate mix of prescribed fire and tree thinning for treating accumulated fuels (Nichols and others 1994), and 3) a host of off-site issues including air quality, developing management procedures suitable for wilderness areas and working in a highly charged political environment.

Our study focuses on the first of these questions: characterizing fire regimes on several sites that may be among the least impacted by recent fire exclusion in the Southwest. The northwestern points and plateaus of the North Rim have the most frequent lightning ignitions in the Park (GCNP fire records). As part of a broader study on fire and forest structure in the Park, we selected three representative sites: Powell Plateau, a mesa separated from the Kaibab Plateau "mainland," Fire Point, the westernmost extension of the rim, and Rainbow Plateau, a peninsula to

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the southeast (fig. 1). The forests are dominated by old trees, with lush understory vegetation and substantial evidence of recent fires. The sites are not without recent human impact: these areas were grazed prior to construction of the North Rim boundary fence in 1938 (Schroeder, personal communication) and fire suppression was practiced here through much of the 20th century (Pyne 1989). However, we anticipated that the limited water for livestock and difficult access for firefighters might have minimized disruption of the fire regime. Our goals were to quantify the fire regime, describe any post-settlement changes and assess management implications.

Methods

The study sites totaled nearly 1,700 acres. The Powell Plateau site covered 780 acres, ranging from 7,400 to 7,660 feet in elevation. The Fire Point site was 333 acres, 7,570 to 7,770 feet in elevation. The Rainbow Plateau site was 550 acres, 7,550 to 7,658 feet in elevation. Soils have not been mapped in detail for these sites, but North Rim soils in general are predominantly of the Soldier series, derived from Kaibab limestone. Average annual precipitation on the North Rim is 23 inches, with an average annual snowfall of 129 inches. Temperatures are cooler than on the South Rim, ranging from an average July maximum of 79° F to an

average January minimum of 30° F (GCNP1992; White and Vankat 1993). Vegetation includes ponderosa pine (*Pinus ponderosa*), Gambel oak (*Quercus gambelli*), and New Mexican locust (*Robinia neomexicana*) trees, with an understory of forbs and perennial grasses (Bennett 1974).

Fire-scarred tree sampling was done in June-July, 1998. Partial cross-sections were cut from scarred "catfaces" on trees, logs, and stumps of conifers that appeared to represent the oldest and/or most extensive fire records. Samples were mapped when collected and were well-distributed throughout the study areas. In the lab, samples were mounted, surfaced with progressively finer sandpaper and crossdated (Stokes and Smiley 1968) using characteristic patterns of narrow marker years: 1722, 29, 35, 48, 52, 72, 82 (false ring), 1810, 13, 20, 22, 45, 47, 73, 79, 96, 99, 1902, 04, 51, 63, 77, 96. All dates were independently confirmed by another dendrochronologist. The season of fire occurrence (Baisan and Swetnam 1990) was estimated from the relative position of each fire lesion within the annual ring.

Fire history data were analyzed with the FHX2 software (Grissino-Mayer 1995). Analysis at each site began with the first year with an adequate sample depth (Grissino-Mayer and others 1994). Fire return intervals were analyzed statistically in different categories related to the size and/or intensity of past fires. The fire data were filtered to look at progressively greater proportional scarring as a proxy for

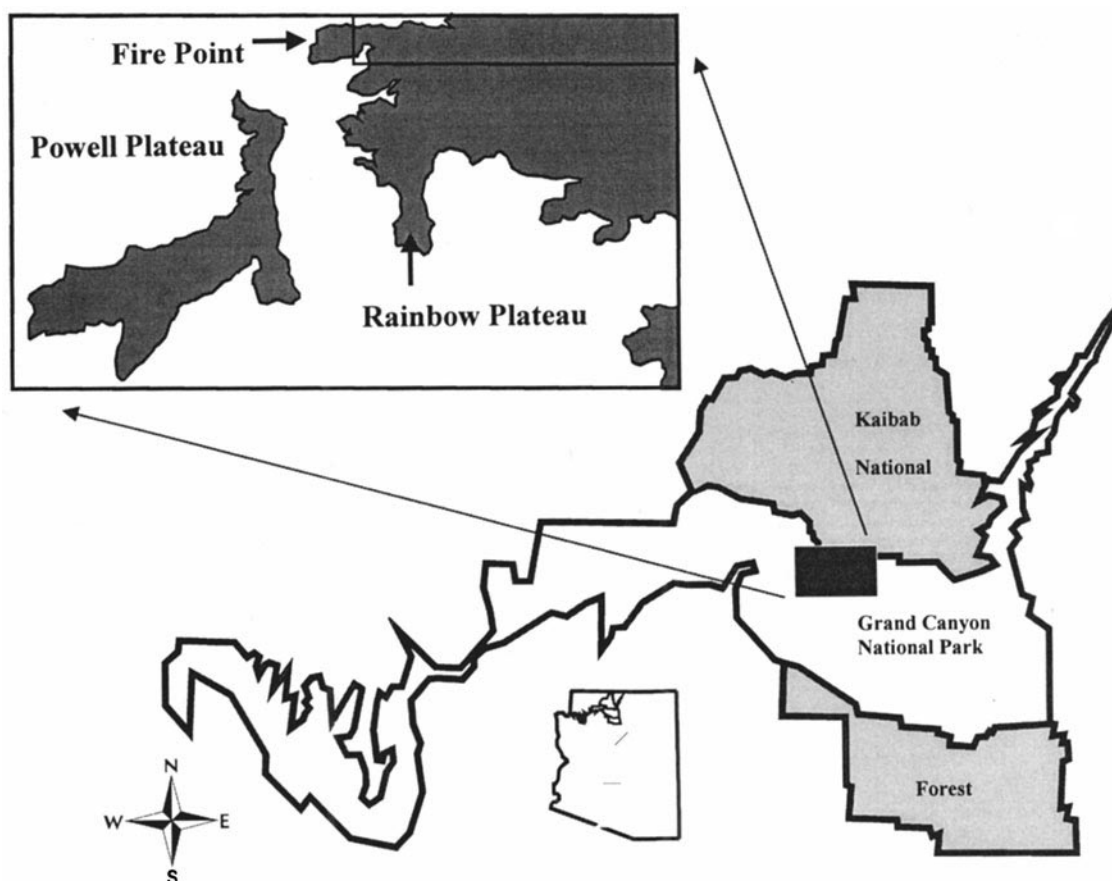


Figure 1—Study sites on the North Rim of Grand Canyon National Park.

fire size (Swetnam and Baisan 1996). First, all fire years, even those represented by a single scar, were considered. Then, we included only those fire years in which 10% or more, and 25% or more, of the recording samples were scarred. The statistical analysis of fire return intervals includes several measures of central tendency: the mean fire interval (MFI, average number of years between fires), the median and the Weibull median probability interval (WMPI).

The relationship between climatic fluctuations and fire occurrence was compared by superposed epoch analysis (SEA), using software developed by Grissino-Mayer (1995). A locally developed, ponderosa pine tree-ring chronology served as a proxy for climate. The SEA superimposes fire years and summarizes the climate variable (tree-ring width) for fire years, as well as preceding and succeeding years. The output of the SEA was a comprehensive comparison of the climate, as represented by tree-ring width, for five years before fire years, the fire years themselves, and two years after fire years. The degree to which the climate variable in each analysis year differed from the average climate was assessed with 90%, 95%, and 99% confidence intervals, developed using bootstrapping methods, with 1,000 simulations based on random windows with the actual fire events (Grissino-Mayer 1995).

Accuracy of the fire scar record could be tested because a relatively high number of fires occurred on the study sites in the 20th century, due both to the isolation of the sites and to recent fire management policies. Across western North

America, we usually find sites with good records but no fires (USA—Swetnam and Baisan 1996) or many fires but limited records (Mexico—Fulé and Covington 1997, 1999). The Grand Canyon sites in the present study have both recent fires and written historical data: fire records maintained at GCNP since 1924 provide an unusual opportunity for a quantitative test of the utility of fire scars in reconstructing the temporal and spatial pattern of past fires.

The fire record data were used with caution. The database was patchy in the early years. Many recorded fire sizes and geographic locations were considered approximate, and some evident errors were observed, such as coordinates that placed fires well outside the Park's boundaries. Nonetheless, the database was a valuable independent source of fire history information. After completing the fire scar analysis without reference to the database, we selected records of fires occurring in and around (within 1 km) the study sites for comparison with the fire scar data.

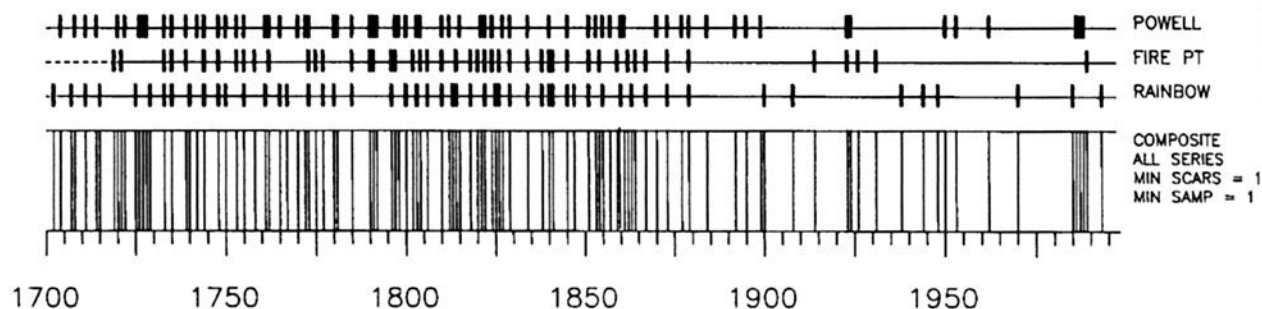
Results and Discussion

Fire Regimes

Composite fire history graphs for all fires on all three sites show that fires were frequent through 1879 and fires continued to occur sporadically up through the present (fig. 2). Prior to 1879, the Weibull Median Probability Interval

Fire History 1700 to 1997

All Fires



Fires scarring 25% or more of the samples

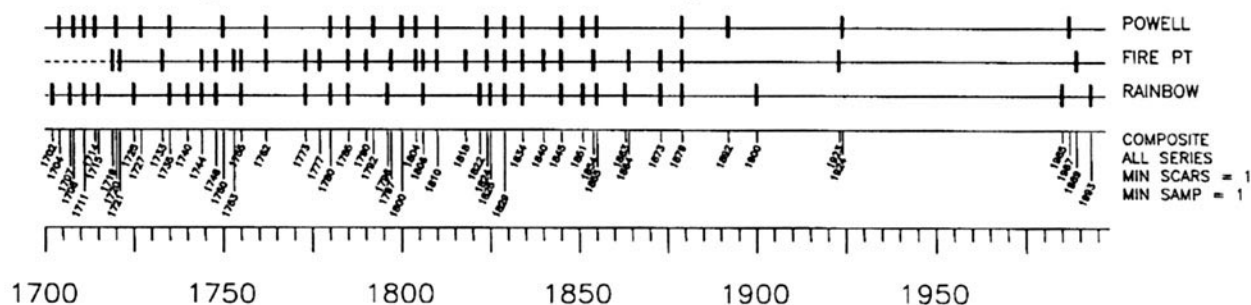


Figure 2—Fire history results are summarized in these graphs, with each horizontal line representing the composite of all sampled trees on a site and the short vertical lines noting the year of fire occurrence. Fire regimes are compared in two categories: all fires, including even those which scarred only a single sample tree (top graph); and fires scarring 25% or more of the sample trees (bottom graph).

(WMPI) at Powell Plateau was less than three years, rising to about 3.9 years on Rainbow Plateau (table 1). Considering only the 25%-scarred fires, the WMPI values were about two to three times higher, suggesting that small fires were more common than larger ones. The fire return intervals fall within the range of values reported at other southwestern sites (Swetnam and Baisan 1996), close to the high frequency end of the distribution. These sites might have been expected to have relatively infrequent fires, as did isolated smaller mesas in Zion National Park, Utah (Madany and West 1982), because the study sites are isolated high-elevation landmasses at the western edge of the canyon rim. The prevailing southwestern winds tend to carry fire out of the sites, while the likelihood of importing fire from lower-elevation lands to the west seems low, due to the reduced chance of low-elevation lightning strikes and discontinuous fuels (although there is a chaparral belt extending for about 1,000 feet below the rim). The fact that high fire frequencies were observed suggests that lightning densities are high on the study sites. Ignitions by Native Americans may have played a role as well (Schroeder, personal communication). The many synchronous fire dates in fig. 2 suggest that presettlement fires spread between the study sites in many years, or that sites were ignited separately in the same years.

Fires occurred primarily in dry years following wet years (fig. 3), assuming that tree-ring widths in the local chronology adequately reflect moisture variability. Similar patterns were observed across the Southwest by Swetnam and Betancourt (1990), who suggested that increased herbaceous production in moist years led to high fuel loading and continuity in subsequent years.

Clearly, fewer fires burned after 1879, especially using the 25%-scarred criterion that filters out the presumably smaller fires that scarred fewer samples (fig. 2). The question is, how much disruption of the fire regime results in ecologically significant changes? Each site has had either two or three large fires since settlement. These post-settlement fires contrast with fire exclusion in the majority of forests in the Southwest (Swetnam and Baisan 1996). Fires were excluded in most of the only other large unharvested southwestern ponderosa pine forest, New Mexico's Gila/Aldo Leopold Wilderness Areas (Swetnam and Dieterich 1985), although

some portions of the Wildernesses have had repeated 20th century fires (Rollins and others, in press).

The timing of postsettlement fires may also be important. Regeneration flushes in the early 20th century, especially 1919, were important in forming dense forests in northern Arizona (Savage and others 1996; Mast and others 1999). Large fires on Powell Plateau in 1892 and 1924, Fire Point in 1923 and Rainbow Plateau in 1900 may have been instrumental in thinning seedlings. The other post-settlement fires were all post-1980, reflecting the change in park policy toward prescribed natural fire. In light of the fire regime data, we will evaluate forest structural information from the same study sites to assess changes from reference conditions.

Comparison to Fire Records

Fire scars were highly accurate in identifying historic fires: every fire on the study sites recorded since 1924 and larger than 20 acres was identified from fire scars. The largest fire from written records missed in the fire-scar reconstruction was a 20-acre prescribed natural fire on the Rainbow Plateau in 1987. Many smaller fires, suppressed at one acre or less in size, did not show up in the fire scar record. The proportion of scarred trees was generally related to fire size. The greatest discrepancy between fire size and scarring proportion occurred with the 1931 Fire Point fire, which burned 160 acres but was recorded only on a single scarred sample. In the case of this fire event, reliance on the 25%-scarred criterion would underestimate fire size.

Mapped fire perimeters from the Emerald prescribed natural fire in 1993 matched well with fire scar data (fig. 4). The fire was recorded on only half of the 12 fire-scarred samples collected from within its boundary, but the six samples were well-distributed. Taking these six samples and applying a reasonable spatial buffer of 1,000 feet around them would fairly closely approximate the geographic boundary and size of the Emerald fire.

The close correspondence between the fire scar data and the Park's fire records builds confidence in the interpretation of presettlement fire regime characteristics. While fire scar methods do have limitations (Johnson and Gutsell 1994), our results suggest that the rationale described by Swetnam and Baisan (1996) for proper use of fire scar data,

Table 1—Fire return intervals at the study sites, in two categories: (1) all fire years, including even those represented by a single scarred sample, and (2) fire years in which 25% or more of the recording trees were scarred.

Site	Mean fire interval (MFI)	Minimum	Maximum	Weibull Median Probability Interval (WMPI)
Powell – All	3.24	1	9	2.97
Powell – 25%	9.45	3	24	8.56
Fire Pt. – All	3.65	1	11	3.42
Fire Pt. – 25%	6.35	2	11	6.25
Rainbow – All	4.00	1	11	3.86
Rainbow – 25%	7.81	3	18	7.53

Climate-Fire Relationship (SEA)

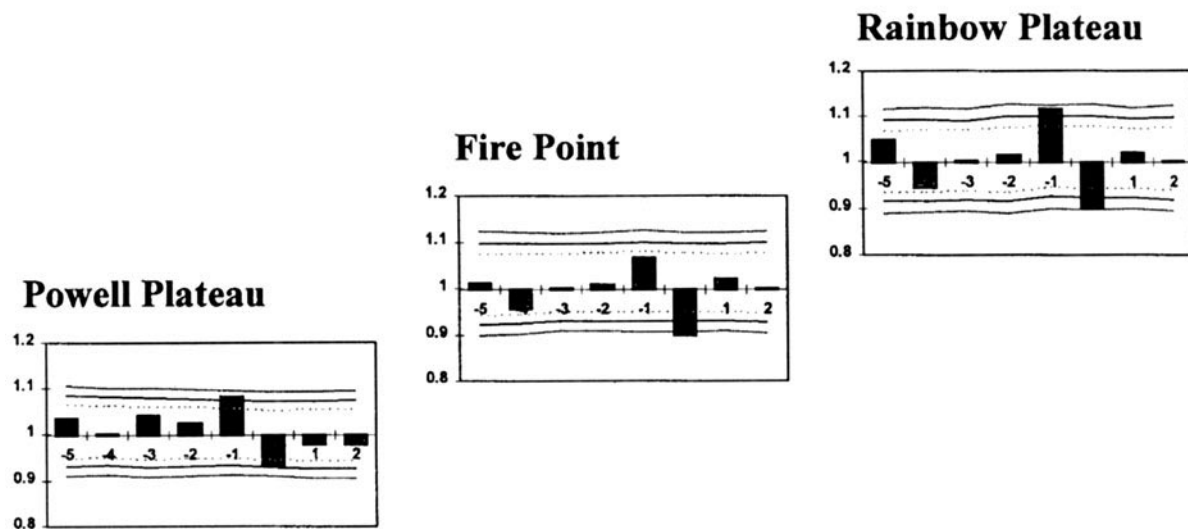


Figure 3—Relationship between local climate (tree-ring width index) and fire occurrence determined by superposed epoch analysis (SEA). The average climate value is scaled to one. Bootstrapping procedures were used to assess the statistical significance of climate departures above the mean ("wet years") and below the mean ("dry years") in the fire years (year 0), the five years preceding fires (-5 through -1), and the two year after fires (1-2). The three lines above and below the x-axis in each graph represent confidence intervals of 90%, 95%, and 99%.

filtering data according to the proportion of scarred samples, is sound.

Management Implications

Presettlement fire regime data serve as a point of reference for ecosystem management, particularly in Park Service wildlands managed primarily for their natural qualities. One logical course of action would be to permit natural ignitions to burn without impediment, using the reference fire regime data as a standard against which to judge the effectiveness of fire restoration.

But if a natural fire policy were to be fully adopted, the Park would have to accept the occurrence of large fires spreading over thousands of acres, dropping below the rim and burning during the summer fire season. The biggest fires, reaching well into the higher elevations of the Kaibab Plateau, would most likely occur during the driest years. When these fires encountered the dense mixed conifer forests above 8,000 feet elevation, intense fire behavior and severe fire effects would occur, in contrast to the effects of the frequent surface fires that prevailed prior to settlement (Wolf and Mast 1998). In the past decade, the two driest years in northern Arizona have been 1989 and 1996. The 1989 Muav wildfire, ignited by lightning, was perceived as a threat of such magnitude that the use of bulldozers was authorized in the Park to construct fireline. The 1996 fire season was the worst on record in the Southwest, with the 50,000 acre Bridger Complex fire burning just north of the Park boundary. Neither in 1989 nor in 1996 would park managers have been able to authorize natural fires to burn,

even though it is in precisely such years that large presettlement fires occurred.

Current management policy is directed toward beneficial use of wildland fire for resource benefits, primarily applied in ponderosa pine forests, where burning is less risky. In 1998, for example, a small lightning-ignited fire was intentionally expanded with aerial ignition over the Rainbow Plateau. When fire use involves management ignitions, park managers are faced with different questions. The fire behavior may be less hazardous, but the fire timing and spread will be controlled more by management than by fuel and weather patterns. Fire use is unlikely to be permitted during dry fire seasons, so fire timing and size would probably remain outside the range of natural variability. Fire use may also pose conflicts with the Park's wilderness proposal, because fire managers rely on helicopters, vehicles, and other equipment to carry out burns.

Smoke remains a significant management challenge at Grand Canyon because of the Class I airshed designation, the importance of scenic vistas for park visitors, and the active role taken by the Park in opposing other off-site pollution sources, such as power plants.

Despite the difficulties in restoring fire to the Park, there is no alternative: fuels will burn eventually. The question is how best to intervene. An ecological restoration experiment that tests thinning of small trees, as well as prescribed burning (Covington and others 1997; Heinlein and others, in press), may offer management alternatives for some areas of the Park.

The resilience of forest ecosystems will be key to the eventual restoration of natural processes. Although presettlement fire

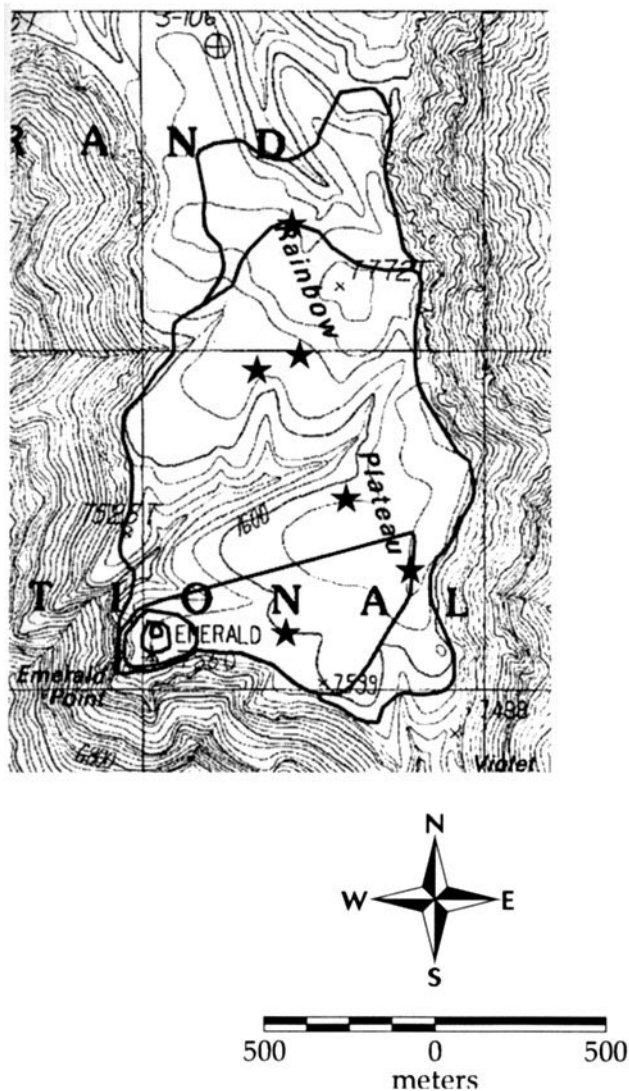


Figure 4—Map of the Emerald prescribed natural fire, which burned from 8/10/93 to 8/24/93, reaching a size of approximately 342 acres. Fire perimeters are shown at approximately 2-day intervals. The stars indicate the locations of six fire-scarred samples that recorded the fire; another six samples within the fire perimeter did not scar in this burn.

frequency was much higher than the post-1879 fire occurrence, and post-settlement fire-free intervals have been substantially greater than the presettlement maximums, the study sites on the northwestern points and plateaus may still be the best existing representatives of natural ponderosa pine forest landscapes in the Southwest. If a few widely spaced fires can have ecological effects reasonably similar to those of the natural fire regime, managers may be able to manipulate modern fire regimes to accommodate constraints without significant damage to ecosystems.

Acknowledgments

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D. Spotskey, M. Schroeder, J. Schroeder, K. Crumbo, N. Bryan, J. Balsam, R.V. Ward, A. Horn-Wilson and D. Bertollette. Northern Arizona University's Ecological Restoration Program students and staff, especially J.P. Roccaforte, supported data collection and sample preparation. This work was funded by a grant from the U.S. Department of the Interior.

References

- Altschul, J.H., and H.C. Fairley. 1989. Man, models, and management: An overview of the archaeology of the Arizona Strip and the management of its cultural resources. USDA Forest Service and USDI Bureau of Land Management Report contract # 53-8371-6-0054, submitted by Dames & Moore, Inc.
- Baisan, C.H., and T.W. Swetnam. 1990. Fire history on a desert mountain range: Rincon Mountain Wilderness, Arizona, USA. *Canadian Journal of Forest Research* 20:1559-1569.
- Bennett, P.S. 1974. The fire ecology of the ponderosa pine ecosystem of the North Rim forests, Grand Canyon National Park. Report on file at Grand Canyon National Park, AZ.
- Covington, W.W., M.M. Moore, P.Z. Fulé, and H.B. Smith. 1997. Grand Canyon Forest Ecosystem Restoration. Proposal for Adaptive Ecosystem Research and Management to the U.S.D.I. Interior Fire Coordinating Committee.
- Fulé, P.Z., Assistant Research Professor, Northern Arizona University, and others: unpublished fire history data from Mt. Trumbull area, AZ.
- Fulé, P.Z., and W.W. Covington. 1999. Fire regime changes in La Michilia Biosphere Reserve, Durango, Mexico. *Conservation Biology* 13(3):640-652.
- Fulé, P.Z., and W.W. Covington. 1997. Fire regimes and forest structure in the Sierra Madre Occidental, Durango, Mexico. *Acta Botanica Mexicana* 41:43-79.
- Grand Canyon National Park. 1992. Fire management plan. On file at Grand Canyon National Park, AZ.
- Grissino-Mayer, H.D. 1995. Tree-ring reconstructions of climate and fire history at El Malpais National Monument, New Mexico. Ph.D. Dissertation, University of Arizona, Tucson.
- Grissino-Mayer, H.D., C.H. Baisan, and T.W. Swetnam. 1994. Fire history and age structure analyses in the mixed conifer and spruce-fir forests of Mount Graham. Final Report, Mount Graham Red Squirrel Study Committee, US Fish and Wildlife Service, Phoenix, AZ.
- Heinlein, T.A., W.W. Covington, P.Z. Fulé, and M.M. Moore. In press. Restoration of fire-adapted forest ecosystems at Grand Canyon National Park. In: Cole, David N.; McCool, Stephen F. 2000. Proceedings: Wilderness Science in a Time of Change. Proc. RMRS-P-000. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Johnson, E.A., and S.L. Gutsell. 1994. Fire frequency models, methods, and interpretations. *Advances in Ecological Research* 25:239-287.
- Madany, M.H., and N.E. West. 1983. Livestock grazing—fire regime interactions within montane forests of Zion National Park, Utah. *Ecology* 64(4):661-667.
- Mast, J.N., P.Z. Fulé, M.M. Moore, W.W. Covington, and A. Waltz. 1999. Restoration of presettlement age structure of an Arizona ponderosa pine forest. *Ecological Applications* 9(1):228-239.
- Moore, M.M., W.W. Covington, and P.Z. Fulé. In press. Evolutionary environment, reference conditions, and ecological restoration: a southwestern ponderosa pine perspective. *Ecological Applications*.
- Morehouse, B.J. 1996. A Place Called Grand Canyon: Contested Geographies. The University of Arizona Press, Tucson.
- Nichols, T., B. Callenberger and G. Kleindienst. 1994. Report of the task force review of the hazard fuel situation on the North Rim of the Grand Canyon. Report on file at Division of Resource Management, Grand Canyon National Park.
- Pyne, S.J. 1989. Fire on the Rim. Weidenfeld & Nicolson, New York.
- Rollins, M., T. Swetnam, and P. Morgan. In press. Twentieth-century fire patterns in the Selway-Bitterroot Wilderness, Idaho/Montana, and the Gila/Aldo Leopold Wilderness Areas, New

- Mexico. In: Cole, David N.; McCool, Stephen F. 2000. Proceedings: Wilderness Science in a Time of Change. Proc. RMRS-P-000. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Savage, M., P.M. Brown, and J. Feddema. 1996. The role of climate in a pine forest regeneration pulse in the southwestern United States. *Ecoscience* 3(3):310-318.
- Schroeder, M. Archeologist, Grand Canyon National Park. Personal communication, 1999.
- Stokes, M.A., and T.L. Smiley. 1968. An introduction to tree-ring dating. University of Chicago Press, Chicago.
- Swetnam, T.W., and C.H. Baisan. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. Pages 11-32 in Allen, C.D. (ed.), Proceedings of the 2nd La Mesa Fire Symposium. USDA Forest Service General Technical Report RM-GTR-286, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.
- Swetnam, T.W., and J.L. Betancourt. 1990. Fire—Southern Oscillation relations in the southwestern United States. *Science* 249:1017-1020.
- Swetnam, T.W., and J.H. Dieterich. 1985. Fire history of ponderosa pine forests in the Gila Wilderness, New Mexico. Pages 390-397 in Lotan, J.E., B.M. Kilgore, W.C. Fischer, and R.W. Mutch (tech. coords.), Proceedings—Symposium and Workshop on Wilderness Fire. USDA Forest Service General Technical Report INT-182, Intermountain Forest and Range Experiment Station, Ogden, UT.
- Wolf, J.J., and J.N. Mast. 1998. Fire history of mixed-conifer forests on the North Rim, Grand Canyon National Park, Arizona. *Physical Geography* 19(1):1-14.
- White, M.A., and J.L. Vankat. 1993. Middle and high elevation coniferous forest communities of the North Rim region of Grand Canyon National Park, Arizona, USA. *Vegetatio* 109:161-174.

Development of Ecological Restoration Experiments in Fire Adapted Forests at Grand Canyon National Park

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Margaret M. Moore
Hiram B. Smith

Abstract—The management of national park and wilderness areas dominated by forest ecosystems adapted to frequent, low-intensity fires, continues to be a tremendous challenge. Throughout the inland West and particularly in the Southwest, ponderosa pine (*Pinus ponderosa*) and mixed conifer forests have become dense and structurally homogeneous after periods of intense livestock grazing, followed by more than 100 years of fire suppression. Prior to the late 1800s, pine-dominated forests at Grand Canyon National Park were structurally diverse, averaging 45 to 90 trees per acre, with frequent, low-intensity fires burning across the landscape every 7 to 11 years. Today, much of the historic landscape heterogeneity has been replaced by dense, contiguous stands averaging 600 to 900 trees per acre. The beneficial reintroduction of fire to these areas is difficult and often results in fire effects that are uncharacteristic of those produced by historic fire regimes. In response, park managers have called for the exploration of restoration approaches using combinations of prescribed fire and understory thinning. The goal of this approach is to achieve more natural and sustainable forest structures while conserving the most fragile elements of the existing ecosystem such as old-growth trees and native herbaceous communities. This paper describes the approach, rationale and preliminary results of a project designed to examine the utility and ecological effects of three, small-scale restoration experiments on a suite of forest structure attributes.

Ponderosa pine (*Pinus ponderosa*) and mixed conifer forests throughout western North America have undergone striking changes since the cessation of a high-frequency, low-intensity fire regime in the late 1800s (Swetnam and Baisan 1996; Barrett and others 1997; Covington and others 1997a). At Grand Canyon National Park increasingly dense forest structures have enveloped meadows,

homogenized landscapes, and now provide a conduit for the development of large-scale, high intensity fires (GRCA 1992; Moore 1994; Nichols and others 1994). Recent attempts to reintroduce fire into these dense forests have been costly to prepare, difficult to control and often produce damaging fire effects, particularly in mixed conifer forests on the North Rim of the Park (Nichols and others 1994). The most notable effect of these high intensity fires is a marked loss of old-growth trees due to excessive crown scorch and pockets of stand replacing fire. These effects are of particular concern since Grand Canyon National Park contains some of the largest remaining tracts of unharvested old-growth forest in the Southwest.

Interagency fire management reviews have called for the exploration of methods to mechanically thin understory trees and remove forest floor fuels before widespread application of prescribed fire (Davis 1981; Nichols and others 1994; Botti and others 1997). In consultation with Park managers, we designed a project to explore alternative approaches to deal with this management problem (Covington and others 1997b). This approach is based on evidence that, for ecosystem restoration to be successful, tree understories must be thinned prior to the application of fire, because underburning is insufficient to thin the dense stands that have developed since fire regime disruption (Sackett and others 1996). Restored tree structures that reflect historic variability as expressed by densities, spatial patterns and species composition should result in more sustainable growing conditions for native plant communities and the habitats they provide (Kaufmann and others 1994).

Fire managers at Grand Canyon are highly proficient and have been proactive in their attempts to return the natural role of fire to the Park's forests. In recent years with the adoption of more flexible fire-use policies, significant acreages (primarily in the ponderosa pine type) are being burned. These management fires do exhibit some benefits such as a measurable decrease in forest floor organic matter and the consumption of ladder fuels. However, due to heavy fuel concentrations, these fires also have the potential to create locally intense fire effects that are likely to be detrimental to native plants and animals (*sensu* Neary and others 1999). These adverse effects which include old growth tree mortality, pockets of crown fire, and intense soil heating are particularly apparent in the mixed conifer forests on the North Rim and prevail even though prescribed burns are applied conservatively and with great skill (Nichols and

In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management; 1999 May 23–27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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others 1994). Attempts to reintroduce fire to the Park's ponderosa pine forests produce fewer adverse effects, but in general, management ignitions in this forest type typically do not adequately thin sapling and pole sized trees, as they have already developed fire-resistant bark (Sackett and others 1996). When fires of sufficient intensity to thin small diameter ponderosa pine trees are prescribed, the result is increased old growth tree mortality and soil heating (Nichols and others 1994; Sackett and others 1996; and Neary and others 1999). In an attempt to surmount these difficult issues, we proposed to test alternative methods that utilize combinations of process restoration (fire) and structure restoration (tree thinning), to restore forest structures that will be more sustainable upon the reintroduction of frequent fires.

The development of any relevant, ecologically sound restoration approach relies upon a solid understanding of the historic range of variability of the ecosystem (Morgan and others 1994; Moore and others 1999). For this project, target conditions were quantified for experimental restoration approaches based upon reconstructions of forest structures and disturbance regimes that existed at the time of fire regime disruption. Reconstructions of forest conditions in the Southwest are quite feasible given a continuous period of fire suppression since the late 1800s, coupled with extremely slow rates of decomposition. This combination of factors has maintained most evidence of the tree structure that existed at the time of fire regime disruption allowing for an accurate reconstruction of tree density, spatial pattern, and species composition (Fulé and others 1997; Huffman and others 1999). It is important to realize that while initial restoration treatments are directed toward a particular point in time, we do not view such a treated condition as a static structure to be maintained in perpetuity. Instead, we view this restored condition as merely a sustainable starting point for the reintroduction of fire that is consistent with historic fire regimes and the forest structures they produced. It will be the effects of future fires burning in the restored fuel matrix that will shape and maintain more natural forest structures and processes.

The development of specific restoration prescriptions has been a lengthy process that was shaped by public comments and feedback from park managers, scientists and environmental groups. In particular, this relates to restoration experiments in mixed conifer forests on the North Rim, which are within areas proposed for wilderness designation. While the overall goals of restoration based management are compatible with wilderness management, there are several short-term effects associated with restoration activities that create temporary conflicts with wilderness values. These may include tree thinning and associated stumps, forest floor fuel manipulation, and the operation of mechanized equipment. The artifacts of restoration activities, such as stumps and slash, are usually erased following several prescribed burning cycles (5–15 years). When compared to the hundreds of years it takes to re-establish old growth tree structures that are lost to high intensity fires, a one to two decade visual effect may prove to be an acceptable alternative. Through consultation with Park managers, we continue to develop and refine approaches that seek to make short-term restoration practices as compatible as possible with wilderness values.

Objectives

The specific objectives of this project are to quantify historic forest structures and fire regimes, and measure current forest structures. Secondly, we will examine the operational utility and effects on forest structure of several restoration treatments using combinations of prescribed fire and understory tree thinning. Since experiments are ongoing, this paper is limited to a discussion of current and reconstructed presettlement conditions, as well as the development of experimental treatments.

Study Area

Grand Canyon National Park is located in north central Arizona, approximately 70 miles north of Flagstaff. Elevations range from 1,650 to 9,165 feet, with vegetation communities that range from desert scrub to subalpine forests. The Park consists of two management units, South Rim and North Rim, bisected by the vast inner canyon carved by the Colorado River. Our project focuses on the forest ecosystems above the rims and includes ponderosa pine - Gambel oak (*Quercus Gambelii*) and mixed conifer forests. Mixed conifer forests at the Grand Canyon contain combinations of ponderosa pine, Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*) and quaking aspen (*Populus tremuloides*). Though none of the Park is designated wilderness, much of the North Rim is proposed wilderness, which requires management actions that will maintain wilderness character.

Methods

In order to measure the treatment effects of restoration experiments, we established small-scale (80-acre) experimental blocks on both rims of the Park. One experimental block was located in the North Rim mixed conifer forest, and two were located within South Rim ponderosa pine—Gambel oak forests. One of the South Rim experimental blocks is located adjacent to the Park boundary on the Tusayan District of the Kaibab National Forest. Each experimental block was divided into four, twenty-acre units that were randomly assigned one of four experimental treatments. Within each unit, we installed twenty, 0.1-acre permanent plots, where we measured tree ages, tree overstory (species, condition, diameter, height, and crown characteristics), seedling structure, evidence of insects and pathogens, forest floor fuel loadings, and herbaceous/shrub community structure. Reconstructions of presettlement forest structure are based on a dendroecological model described in (Covington and Moore 1994; and Fulé and others 1997).

To obtain fire history information, we collected partial cross sections from fire-scarred trees, snags, stumps and logs located throughout the study areas. Samples were surfaced, cross-dated and years were assigned to specific fire events as indicated by fire scarred tree-rings (Stokes and Smiley 1968). Fire event determinations were verified by a second dendroecologist and fire events were then compiled to illustrate the dynamics of the historic fire regime at each study site (Fulé and Heinlein, this volume).

Results and Discussion

Presettlement and Contemporary Forest Structures and Fire Regimes

Prior to fire regime disruption, presettlement ponderosa pine/Gambel oak forests on the South Rim experimental blocks (EB1 and EB2) averaged 45.2 and 48.4 trees per acre with a total basal area of 40.5 to 60.8 ft²/acre (table 1). There have been no widespread fires on these sites since 1887. However, prior to 1887, fires occurred on average, every 7 to 9 years (Covington and others 1998). Current forest structures have changed substantially, both in terms of tree density and basal area. The same areas now average 580.3 to 897.4 trees per acre with a total basal area of 98.5 to 102.5 ft²/acre (table 1). Current tree densities are significantly different between experimental blocks and this trend is likely attributable to contrasting land-use histories. For example, EB1, which is located on the Kaibab National Forest, was logged in the early 1900s, continuously grazed by domestic livestock and is a fuelwood harvesting area. In contrast, no logging, fuelwood harvesting or livestock grazing has occurred on EB2.

Forest structures have also changed substantially within the North Rim mixed conifer experimental block (EB3). Prior to fire regime disruption, this site averaged 93.1 trees per acre with a total basal area of 101.1 ft²/acre (table 1). There have been no widespread fires on this site since 1879. Prior to 1879, fires occurred every 7 to 11 years (Covington and others 1998). Today, the same area contains 571.1 trees per acre with a total basal area of 188.7 ft²/acre (table 1). In

addition to these large density increases, species composition has shifted from a ponderosa pine dominated mixed conifer forest to a white fir dominated forest (table 1).

Experimental Treatments

In light of the ecological trends occurring in Grand Canyon forests, we propose to test the effects of three treatments and a control on a suite of ecological variables, including fire behavior and fire effects on tree structure, herbaceous plant community structure and forest floor fuels. Within the Park, thinning activities will exclusively target trees less than 5 inches dbh, with all cut material remaining on-site. Thinned material on the Kaibab National Forest experimental block (EB1) will be sold to a fuelwood contractor. Hand tools are proposed for thinning the North Rim site (EB3), while chain saws are proposed for thinning the South Rim sites (EB1 and EB2). In addition, fences have been constructed around EB1 to exclude future livestock grazing. Detailed descriptions of experimental treatments are as follows:

Control—No thinning or prescribed fire treatments will take place. Forest structure will be monitored over time to track the continued effects of fire exclusion. Deliberate protection from wildfire or prescribed burning will continue on control sites in perpetuity.

Prescribed Fire—This treatment will test the effects of using prescribed fire without any manipulation of understory trees or forest floor fuels. This approach represents current management practices being applied at Grand Canyon National Park (GRCA Fire Management Plan 1992).

Table 1—Comparison of presettlement and current overstory tree density and basal area. Presettlement forest structures are based on reconstructed conditions at the time of fire regime disruption. Historic fire regimes were disrupted in 1887 on experimental block 1 and 2. The last widespread fire on experimental block 3 occurred in 1879. Data is derived from 80, 0.1-acre plots per experimental block.

Study area/tree species	Presettlement forest				Current forest			
	Trees per acre		Basal area (Ft ² /Ac)		Trees per acre		Basal area (Ft ² /Ac)	
	Mean	(S.E.)	Mean	(S.E.)	Mean	(S.E.)	Mean	(S.E.)
Experimental Block # 1								
Tusayan								
<i>Pinus ponderosa</i>	26.9	4.3	55.1	10.8	738.8	196.5	77.4	9.83
<i>Juniperus osteosperma</i>	4.3	2.2	2.2	1.3	25.1	9.64	4.4	2.4
<i>Pinus edulis</i>	0.1	0.1	0.1	0.1	3.2	1.5	0.2	0.2
<i>Quercus Gambelii</i>	17.1	4.7	3.4	1.6	130.3	34.7	16.5	17.5
Total:	48.4	6.8	60.8	10.8	897.4	202.2	98.5	9.9
Experimental Block # 2								
Grandview								
<i>Pinus ponderosa</i>	25.3	4.6	35.9	8.1	402.6	85.4	84.3	10.9
<i>Juniperus osteosperma</i>	0.6	0.4	0.3	0.2	6.6	2.9	0.6	0.4
<i>Pinus edulis</i>	0.3	0.3	0.1	0.1	2.1	1.3	0.1	0.1
<i>Quercus Gambelii</i>	19	7.7	4.2	2.1	169.1	60.6	17.5	4.6
Total:	45.2	7.9	40.5	7.8	580.4	103.4	102.5	9.6
Experimental Block # 3								
Swamp Ridge								
<i>Pinus ponderosa</i>	50.6	7.2	73.6	10.1	79.8	12.2	89.9	15.2
<i>Abies concolor</i>	19.6	4.5	22.1	6.3	370.1	47.1	67.7	10.1
<i>Populus tremuloides</i>	20.3	7.1	3.4	1.2	63.5	15.8	18.1	6.1
<i>Pseudotsuga menziesii</i>	2.5	1.3	1.9	1.1	55.4	16.2	10.9	3.7
<i>Picea engelmannii</i>	0.1	0.1	0.1	0.1	2.9	1.3	2.1	1.2
Total:	93.1	8.4	101.1	10.8	571.7	59.8	188.7	12.8

Minimal Thinning—This treatment will involve thinning the minimum number of trees necessary to reintroduce prescribed fires without significant mortality to old-growth trees. The focus of thinning activities is to break up ladder fuels and remove small trees in close proximity to presettlement-era trees. The extent of thinning around targeted old-growth trees will vary, and will be based on existing research results focused on the effects of accumulated fuels and associated fire effects (Ryan and Reinhardt 1988; Ryan and others 1998). Additionally, thick accumulations of duff and litter will be raked from around the base of all presettlement trees to minimize adverse effects from smoldering combustion (Ryan and Frandsen 1991; Swezy and Agee 1991; Sackett and others 1996). Prescribed fire will then be applied to consume existing fuels and to promote additional thinning of fire intolerant postsettlement trees (Thomas and Agee 1986; Mutch and Parsons 1997). Additional prescribed fires are planned in perpetuity, based on reconstructed fire regimes (Covington and others 1998).

Full Restoration Treatment—This treatment will use a diameter-limit understory tree thinning to reconfigure stand structures and spatial patterns to more closely resemble those that existed at the time of fire regime disruption (Covington and others 1997). This alternative focuses on the conservation of presettlement-era trees and the maintenance of specific spatial structures. This will be accomplished through the thinning of small diameter trees (up to 5 inches at dbh), raking of accumulated forest floor fuels around the base of presettlement trees and the reintroduction of repeated frequent, low-intensity fires. Fires will be prescribed following reconstructed intervals and seasonalities.

Discussion

With the exception of several remote areas in the northwest corner of the Park (Fulé and others, this volume), forests throughout the Grand Canyon have become much denser, have heavier fuel loads than in the past and are at tremendous risk for stand-replacing fires. While there is general agreement among ecologists and ecologically-trained managers that restoration strategies must be developed and implemented, the selection of an appropriate management approach remains the subject of much debate (Parsons and others 1984; Bonnicksen and Stone 1985; Stephenson 1999). A process restoration approach that involves the reintroduction of fire and a structure restoration approach that requires tree thinning prior to the application of fire, are the two methods we will explore. The process restoration method, in the form of prescribed natural and manager ignited fires, has been implemented in many national parks and wilderness areas (Parsons and Landres 1998). Ecologically and economically, this approach has been most beneficial in areas that contain large contiguous tracts of forest that have evolved with infrequent, high-intensity stand replacement fire regimes (primarily mesic mixed conifer and subalpine forests). In other areas, such as sequoia groves (*Sequoiadendron giganteum*) in the Sierra Nevada, modern fire suppression has not impacted historic stand structures to the point where the reintroduction of fire is detrimental (Stephenson 1999).

Unfortunately, these same approaches have proven to be of limited utility in the lower-elevation ponderosa pine dominated ecosystems found throughout the interior West (Sackett and others 1996). Large increases in stand densities and successional advances by shade-tolerant tree species coupled with steady accumulations of forest floor fuels make the reintroduction of prescribed fire in these areas a very complex, costly and potentially damaging endeavor.

Compared to other ongoing restoration projects in the Southwest (Covington and others 1997a), the experiments being applied within the Park experimental blocks (EB1 and EB2) are more conservative in their approach. The major differences are the extent of structural manipulation prior to the reintroduction of fire and the amount of time it will take to restore presettlement structures. This project proposes to mechanically thin trees smaller than 5 inches dbh and will rely more on the thinning effects of fire to achieve forest structures that resemble target conditions. In both the ponderosa pine and mixed conifer areas, the vast majority of trees occur within the 1-5 inch diameter class (fig. 1), but for several reasons, the effects of this thinning will likely differ. In the South Rim ponderosa pine/Gambel oak forests there will be a substantial number of residual postsettlement ponderosa pines, particularly in the 5-9 inch diameter class, that may be difficult to thin with prescribed fires (Sackett and others 1996). In the North Rim mixed conifer forest, there will also be a sizeable number of residual trees in the 5-9 inch diameter class (fig. 1). However, the majority of these trees are fire-intolerant white fir, Douglas-fir, and Engelmann spruce, all of which are easily killed by low-intensity fires (Thomas and Agee 1986; Ryan and Reinhardt 1988; and Mutch and Parsons 1997). In summary, we expect a more rapid return to presettlement conditions in the North Rim mixed conifer forest and a longer process in the South Rim ponderosa pine/Gambel oak areas.

The application of structure restoration, in combination with prescribed fire, should be viewed as a flexible management tool that can be used in many situations to accomplish a variety of goals. For example, fire managers could initially use operational-scale restoration thinning to accomplish components of their current workload, including the preparation of control-lines for large prescribed burns, securing boundaries with adjoining land owners, reduction of fuels in old-growth areas and the protection of administrative sites. The completion of this preliminary work would provide a monitored, incremental step toward the expansion of structure restoration that would be of immediate utility. Once implemented and refined, managers could further apply the methodology with greater confidence and efficiency.

Regardless of the rationale presented in this paper, restoration through management intervention is controversial and remains a relatively untested concept in national parks. There are critics who argue that a hands-off management approach is entirely appropriate and most closely aligned with Park Service mandates. Even among those who agree that intervention is needed, there are many additional issues surrounding the choice of an appropriate method that must be resolved. This research project is intended to more fully inform debate on these issues by providing information on the effects of both process and structure/process restoration on vegetation structure, fuel loadings, operational utility and social reaction.

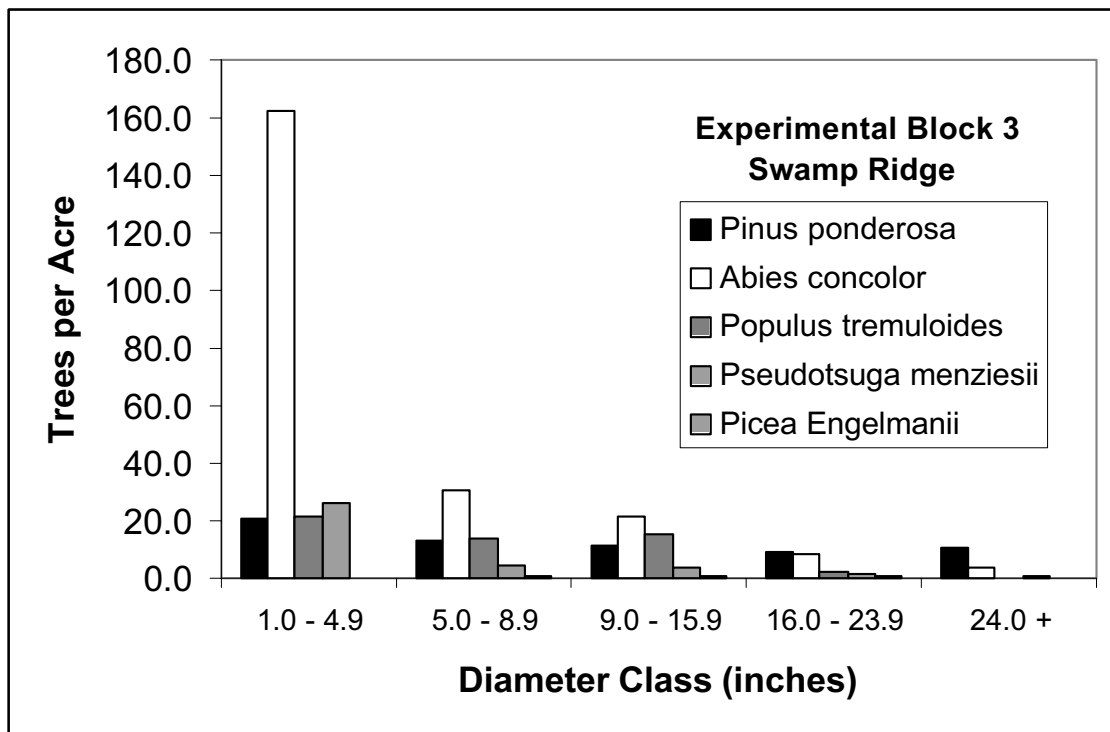
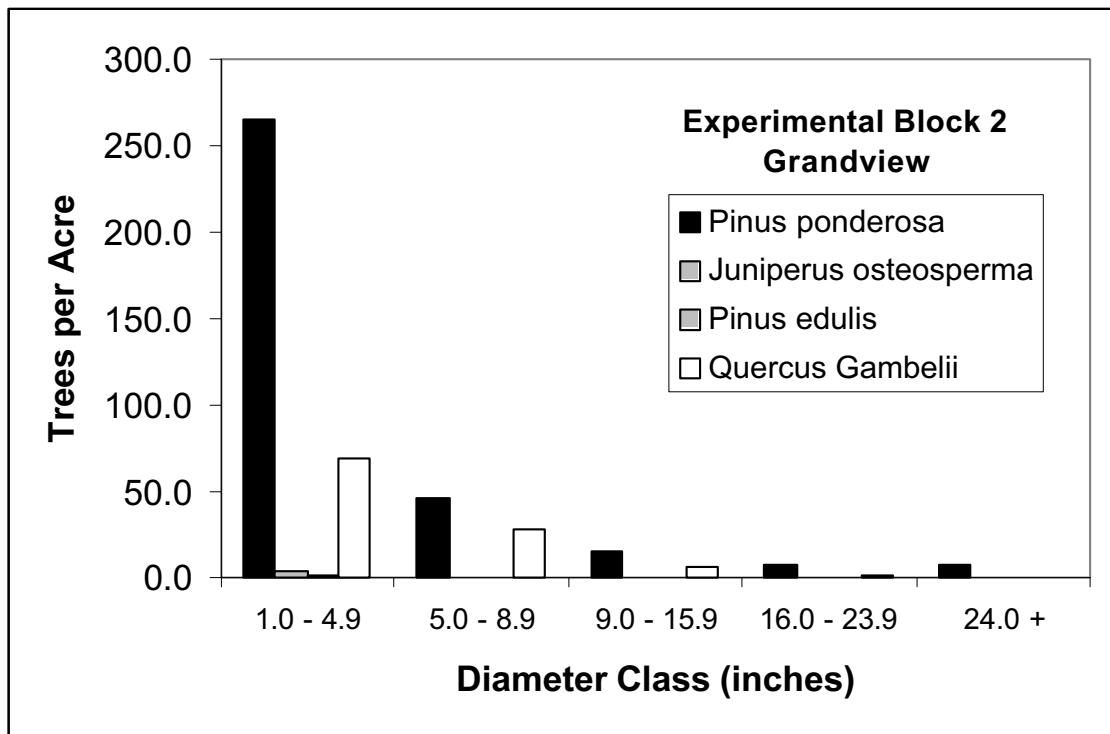


Figure 1—Current diameter distribution of live trees within Grand Canyon National Park experimental blocks. n = 80, 0.1-acre plots per experimental block.

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References

- Barrett, S.W., S.F. Arno, and J.P. Menakis. 1997. Fire episodes in the Inland Northwest (1540-1940) based on fire history data. General Technical Report INT-GTR-370. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 17 p.
- Bonnicksen, T.M., and E.C. Stone. 1985. Restoring naturalness to national parks. *Environmental Management*. 9(6):479-486.
- Botti, S., J. Douglas, W. Josephson, and S. Tryon. 1997. North Rim of the Grand Canyon fire management program review. Report on file at Division of Resource Management, Grand Canyon National Park.
- Covington, W.W., P.Z. Fulé, M.M. Moore, S.C. Hart, T.E. Kolb, J.N. Mast, S.S. Sackett, and M.R. Wagner. 1997a. Restoration of ecosystem health in southwestern forest ecosystems. *Journal of Forestry*. 92(1):39-47.
- Covington, W.W., M.M. Moore, P.Z. Fulé, and H.B. Smith. 1997b. Grand Canyon forest ecosystem restoration. Proposal for adaptive ecosystem research and management to the U.S.D.I. Interior Fire Coordinating Committee.
- Covington, W.W., M.M. Moore, P.Z. Fulé, and H.B. Smith. 1998. Grand Canyon forest ecosystem restoration. Report on pre-treatment measurements of experimental blocks. Report on file at Division of Resource Management, Grand Canyon National Park.
- Davis, K.D. 1981. Report to Western Region Regional Director on fire hazards at Grand Canyon National Park. Report on file at Division of Resource Management, Grand Canyon National Park.
- Fulé, P.Z., W. Wallace Covington, and M.M. Moore. 1997. Determining reference conditions for ecosystem management in southwestern ponderosa pine forests. *Ecological Applications* 7(3):895-908.
- Fulé, P.Z., Heinlein, T.A., Covington, W.W., and M.M. Moore. Continuing fire regimes in remote forests of Grand Canyon National Park. This volume.
- Grand Canyon National Park. 1992. Fire Management Plan. Plan on file at Division of Resource Management, Grand Canyon National Park.
- Huffman, D.W., M.M. Moore, W.W. Covington, P.Z. Fulé, and J.E. Crouse. 1999. Evaluation of ponderosa pine forest reconstruction techniques using historical data. Abstract in Bulletin of the Ecological Society of America, 1999 Annual Meeting, Spokane, WA.
- Kaufmann, M.R., R.T. Graham, D.A. Boyce Jr., W.H. Moir, L. Perry, R.T. Reynolds, R.L. Bassett, P. Mehlhop, C.B. Edminster, W.M. Block, and P.S. Corn. 1994. An ecological basis for ecosystem management. General Technical Report RM-246. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station and Southwestern Region. 22 pp.
- Morgan, P., G.H. Aplet, J.B. Haufler, H.C. Humphries, M.M. Moore, and W.D. Wilson. 1994. Historical range of variability: A useful tool for evaluating ecological change. *Journal of Sustainable Forestry* 2:87-111.
- Moore, M.M. 1994. Tree encroachment on meadows of the North Rim of Grand Canyon National Park. USDI, NPS, CA 8000-8-0002. Report on file with Grand Canyon National Park and Cooperative Studies Unit, Northern Arizona University, Flagstaff, AZ.
- Moore, M.M., W.W. Covington, and P.Z. Fulé. 1999. Evolutionary environment, reference conditions, and ecological restoration: a SW ponderosa pine perspective. *Ecological Applications* (In press)
- Mutch, L.S., and D.J. Parsons. 1997. Mixed conifer forest mortality and establishment before and after prescribed fire in Sequoia National Park, California. *Forest Science*. 44(3):341-355.
- Neary, D.G., C.C. Klopatek, L.F. DeBano, and P.F. Ffolliott. 1999. Fire effects on belowground sustainability: a review and synthesis. *Forest Ecology and Management*. 122:51-71.
- Nichols, T.B., B. Callenberger, and G. Kleindienst. 1994. Report on the task force review of the hazard fuel situation on the North Rim of the Grand Canyon. Report on file at Division of Resource Management, Grand Canyon National Park.
- Parsons, D.J., D.M. Graber, J.K. Agee, and J.W. van Wagtenonk. 1986. Natural fire management in national parks. *Environmental Management* 10(1):21-24.
- Parsons, D.J., and P.B. Landres. 1998. Restoring natural fire to wilderness: how are we doing? in T.L. Pruden and L.A. Brennan (eds.). *Fire in ecosystem management: shifting the paradigm from suppression to prescription*. Tall Timbers Fire Ecology Conference Proceedings, No. 20. Tall Timbers Research Station, Tallahassee, FL.
- Ryan, K.C., and W.H. Frandsen. 1991. Basal injury from smoldering fires in mature *Pinus ponderosa* Laws. *International Journal of Wildland Fire* 1(2):107-118.
- Ryan, K.C., and E.D. Reinhardt. 1988. Predicting postfire mortality of seven western conifers. *Canadian Journal of Forest Research* 18:1291-1297.
- Ryan, R.C., D.L. Peterson, and E.D. Reinhardt. 1988. Modeling long-term fire-caused mortality of Douglas-fir. *Forest Science* 34(1):190-199.
- Sackett, S.S., S.M. Haase, and M.G. Harrington. 1996. Lessons learned from fire use for restoring southwestern ponderosa pine ecosystems. Pages 53-60 in Covington, W., and P.K. Wagner (tech. coords.), *Conference on adaptive ecosystem restoration and management: restoration of cordilleran conifer landscapes of North America*. USDA Forest Service General Technical Report RM-GTR-278, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.
- Stephenson, N.L. 1999. Reference conditions for giant sequoia forest restoration: structure process, and precision. *Ecological Applications*. (In press)
- Stokes, M.A., and T.L. Smiley. 1968. *An introduction to tree-ring dating*. University of Chicago Press.
- Swetnam, T.W., and C.H. Baisan. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. Pages 11-32 in Allen, C.D. (ed.), *Proceedings of the 2nd La Mesa Fire Symposium*. USDA Forest Service General Technical Report RM-GTR-286, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.
- Swezy, D.M., and J.K. Agee. 1991. Prescribed-fire effects on fine-root and tree mortality in old-growth ponderosa pine. *Canadian Journal of Forest Research* 21: 626-634.
- Thomas, T.L., and J.K. Agee. 1986. Prescribed fire effects on mixed conifer forest structure at Crater Lake, Oregon. *Canadian Journal of Forest Research* 16: 1082-1087.

Restoring Natural Fire Regimes to the Sierra Nevada in an Era of Global Change

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Abstract—A conceptual model of fire and forest restoration and maintenance is presented. The process must begin with clearly articulated goals and depends upon derivation of science-driven models that describe the natural or desired conditions. Evaluating the extent to which contemporary landscapes depart from the model is a prerequisite to determining the need for restoration. Model landscapes that include the historical range of variability are commonly used as target conditions in setting restoration objectives. Restoration is a corrective step that ultimately must be replaced by a maintenance process. In a world of changing climate, structural targets of historical conditions will become progressively less meaningful to ecosystem maintenance. Future fire management needs to focus more on fire as a process, in particular as it pertains to proper ecosystem functioning. One area in need of much further research is the critical role of gap formation in forest regeneration.

Forests of the Sierra Nevada in California, like other western coniferous forests, have had ecosystem processes greatly disturbed by fire management practices of the 20th century. This impact has been repeatedly documented through historical studies of fire frequencies revealed in the annual growth rings of fire-scarred trees. These dendrochronology studies show a high frequency of fire prior to Euroamerican settlement, with fires in many mid-elevation forest stands occurring at intervals of roughly every 5–25 years (fig. 1). The fact that these estimates are based upon trees that have persisted through repeated fires demonstrates that the pre-Euroamerican fire regime was one of low intensity/severity fires over a significant portion of the landscape. Beginning in the latter half of the 19th century, fire frequency declined and throughout the 20th century, fires have been largely excluded from these forests (fig. 1). This is in striking contrast to other Californian ecosystems such as lower elevation shrublands, where suppression has not diminished fire on the landscape (Keeley and others 1999).

Several factors contribute to highly successful fire exclusion in coniferous forests. Surface fuels are often separated from canopy fuels, reducing the tendency for crown fires

(Kilgore and Sando 1975), and making fire suppression easier. Also, the fire season is moderately short, generally restricted to a period of three to four months plus humans contribute less to fire ignitions than lightning (Parsons 1981), which is confined to weather patterns often conducive to rapid fire suppression.

Fire exclusion has perturbed forest structure in several critical ways. It has allowed woody fuels and duff to accumulate to unnaturally high levels, it has greatly reduced the size and frequency of gaps necessary for regeneration of certain dominant trees, and it has apparently led to an alteration of forest age structure (GAO 1999; Stephenson 1999). These changes have created two potential problems: Fire hazard has been greatly increased, and forest ecosystem elements and processes have been altered in ways that may represent artifacts of human interference.

In response to these problems, over 30 years ago Sequoia and Kings Canyon national parks initiated a program aimed at restoring fire to these ecosystems, through prescribed burning (for example, the 1969 fire in fig. 1) and other fire management policies (Botti and Nichols 1979; Bancroft and others 1985; Graber 1985; Parsons 1990; Parsons and van Wagendonk 1996). The purpose of this paper is to articulate the steps necessary to restoring fire to these ecosystems and to contrast this approach to the needs for sustainable management into the future.

Model of Forest Restoration and Maintenance

A conceptual model of fire restoration goals and objectives was presented by Parsons and others (1985) and more recently elaborated upon at a recent National Park Service workshop (fig. 2). This decision tree in figure 2 is an attempt to more clearly articulate the goals and methodology in restoration of Sierran forests. Each stage is elaborated upon below, but in brief; this process begins with precise goals and the derivation of science-driven models describing the structural and functional attributes of landscapes and ecosystems that meet those goals. Scientists and resource managers then work cooperatively to evaluate the extent to which contemporary conditions approximate the model. The conclusion for much of the Sierra Nevada landscape is that, due to nearly a century of fire exclusion, restoration is a necessary management response. An important part of having a model landscape is that it provides a clear target for restoration efforts, particularly in the setting of objectives. Afterwards, evaluation of restoration efforts is critical and requires careful monitoring, which may point out shortcomings in the restoration execution, or in the setting of target conditions or even in the formulation of the model landscape.

In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management; 1999 May 23–27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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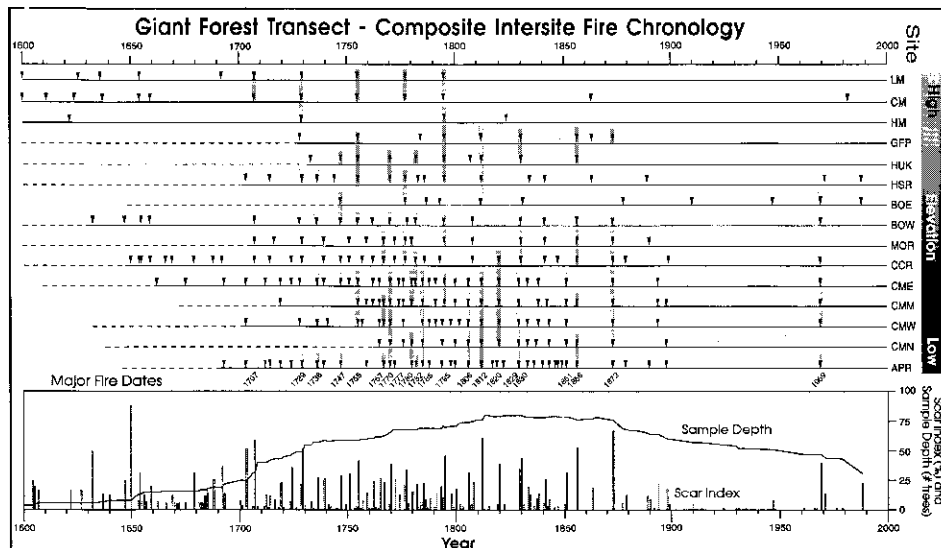


Figure 1—Example of fire scar dendrochronology data used to calculate fire return intervals—composite site fire chronology for 15 sites and 91 samples (76 logs/snags and 15 trees) in the Kaweah Drainage, Sequoia National Park (from Caprio and Swetnam 1995). Triangles indicate fire scars and each horizontal line is a composite of all fires recorded by two or more trees at a site (0.5-2 ha). Dashed lines reflect the interval prior to first fire scar.

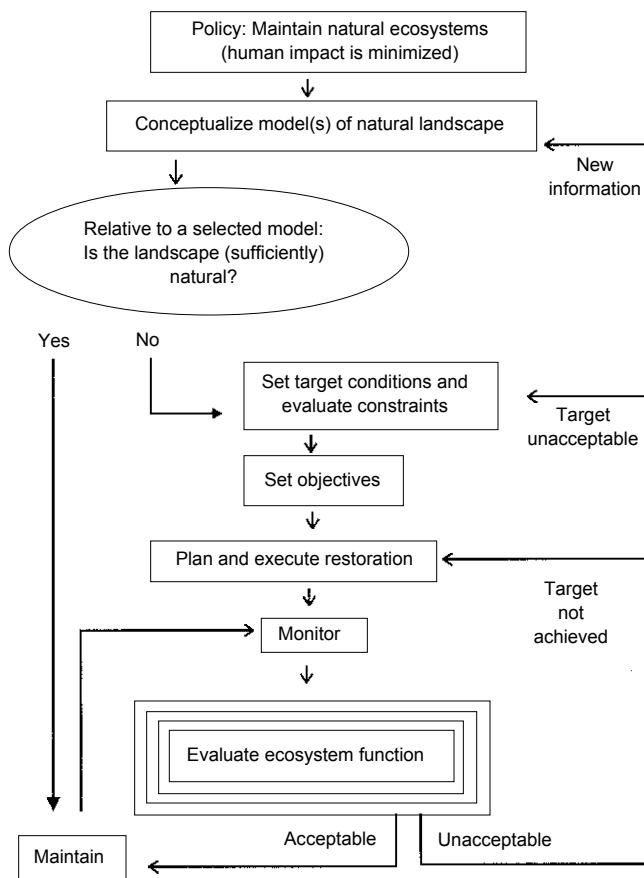


Figure 2—Decision tree diagram for implementing restoration and maintenance of fire in National Park ecosystems, based on collective efforts of the “Fire Management Objectives Workshop,” Rancho Cordova, CA; 3-6 November 1998.

Ultimately, it is expected that restoration is an interim process, one that leads to maintenance of the desired condition.

Step 1: Goals

An important National Park Service goal (fig. 2) is to restore and maintain natural ecosystems (NPS 1988; Wagner and Kay 1993). This is complicated by differences of opinion on defining “natural” (for example, Kilgore 1985) and, even within agencies such as the National Park Service there is a lack of consistency in how the term is defined (Bancroft and others 1985). We maintain that the underlying feature connecting most definitions of natural is a lack of human influence, for example, areas that allow “the unimpeded [by humans] interaction of native ecosystem processes and structural elements” (Parsons and others 1985). Some argue that no part of the landscape is truly natural because humans have at least indirectly affected all parts of the biosphere (for example, Shrader-Frechette and McCoy 1995). We do not dispute this, but in a relative sense there are regions that are less affected than others are. Therefore, *natural is defined here as environments where human impacts are minimized*. This is, of course, relative to one’s frame of reference, and thus a natural environment to an urbanite may be far too heavily affected by humans to be considered natural to a person steeped in the wilderness experience. One advantage of replacing a qualitative notion of naturalness with such a quantitative concept is that a level of naturalness can be empirically determined. One caveat relevant to the goal of minimizing human impact is the realization that *achieving this goal* often requires human intervention, particularly when restoration of perturbed ecosystems is necessary (Hunter 1996).

Step 2: Models of Natural Landscapes

A necessary first step to forest restoration is to conceptualize models of what we believe a natural landscape should look like and how it should function (fig. 2). This is the step that is most dependent on input from scientific research. In the case of Sierra Nevada ecosystems, we have a substantial body of information to draw upon (SNEP 1996). The results from numerous studies show that mid-elevation Sierra Nevada forests are currently experiencing fire-free periods many times longer than at anytime in the past 2000 years (Swetnam 1993).

It appears that fire exclusion has altered the structure and composition of mid-elevation forests (Stephenson 1999), and knowledge of these changes will be valuable in creating a conceptual model of natural Sierran forests. Ideally, a model of such forests would be derived from empirical studies of forest structure under natural fire regimes. Isolated examples of forests that have been allowed over the past three decades to return to some semblance of a natural fire regime exist in the Sierra Nevada (such as Sugarloaf Valley in Kings Canyon National Park or Illilouette Basin in Yosemite National Park [NPS fire records]). Study of these forests could provide a valuable model of natural forest structure and function. One limitation to this approach is the possibility that decades of fire exclusion have so altered forest structure that when the natural process of fire is allowed to return, it will not restore the natural forest structure and composition (Bonnicksen and Stone 1985). In other words, fire regimes are a deterministic process, solely controlled by fuel load distribution—once this has been altered, the system cannot return to its natural state unless the natural fuel structure is first recreated. Alternatively, the fire regime is driven by a combination of factors, including fuels, weather and topography that vary spatially and temporally, producing multiple possible stable points (Christensen 1991a), and making it more likely that returning the process of fire is sufficient to recreate a semblance of natural forest conditions (Stephenson 1999). If this latter view is more or less correct, studies of areas subjected to quasi-natural fire regimes will ultimately provide far more information on the multitude of ecosystem components needed for true ecosystem restoration than will any alternative method of reconstructing past forests.

Other approaches have focused on reconstructions of forest dominants by comparative studies of historical photographs and written descriptions, as well as inferences drawn from contemporary forest demographics (Skinner 1997; Stephenson 1999; Swetnam and others 1999). These reconstructions provide a view of late 19th century forests that are termed the “pre-Euroamerican” condition and are commonly used as targets for restoration. One rationale for embracing this typological approach to forest restoration is that such conditions “portray to the extent feasible, either the same scene that was observed by the first Euroamerican visitor to the area or the scene that would have existed today, or at some time in the future, if Euroamerican settlers had not interfered with natural processes” (Bonnicksen and Stone 1985). This of course is debatable.

A variety of observations suggest that past forests had lower tree density, and very different demographic distribution of age classes, with limited accumulation of forest floor

fuels and greater landscape diversity of forest patches than 20th century forests (Vankat and Major 1978; Parsons and DeBenedetti 1979; Bonnicksen and Stone 1982; Vale 1987; Roy and Vankat 1999; Ansley and Battles 1998; Stephenson 1999). In order to be empirically useful, pre-Euroamerican models need to be made explicit for specific landscapes, and specifying, at least in a probabilistic sense, the proportion of landscape dominated by different forest types and forest structures (Christensen 1991a; Taylor and Skinner 1998). For much of the Sierra Nevada we lack sufficient knowledge for anything other than rather general projections. Lastly, it is a reasonable inference that, concomitant with structural changes in forests, there have been changes in important ecosystem functions but we have little direct information on processes other than fire.

In summary, fire regimes are the best understood component of the pre-Euroamerican landscape (for example, fig. 1) (Swetnam 1993; Caprio and Swetnam 1995; Swetnam and others 1998), although it is unknown to what extent Native Americans contributed to this fire regime and the debate still continues as to whether we should consider their fires as natural. Far less is known about the forest structure and landscape patterns present at the time of Euroamerican settlement, and the reconstructions that have been made deal only with a few dominant tree species. While such reconstructions are the closest we have to a forest model of natural conditions, most are based on late 19th century landscapes and the influence of Euroamerican settlers present in significant numbers since the mid 1800s has not been adequately considered (Barrett 1935; Cermak and Lague 1993). In the Sierra Nevada, fire frequencies generally declined during the settlement period (for example, fig. 1), prior to the era of organized fire suppression. This decline has been attributed to either diminished ignition sources following the demise of Native American populations (Keeley 1981) or to the reduction in fuels attributable to the rise in livestock grazing (Swetnam and others 1998). Further declines in fire frequency have occurred in the 20th century (for example, fig. 1) and this, as well as apparent changes in forest structure and function are thought to be primarily due to fire suppression, however, it remains to be seen how much of this change might be attributable to warmer, moister conditions of the 20th century (Graumlich 1993; Scuderi 1993).

Of course limitations such as these should not prevent us from applying this model, but they do caution against unequivocal acceptance of pre-Euroamerican models as definitive statements on the natural range of conditions.

Step 3: Evaluating Contemporary Landscapes

Considering the ecosystem process of fire, the contemporary landscape clearly exhibits substantial deviation from that expected of natural landscapes (for example, fig. 1). Also, there is widespread agreement that contemporary forest structure (for example, Table 1) deviates from natural conditions. In evaluating contemporary landscapes it is necessary to evaluate the situation from the perspective of whether these landscapes are “sufficiently natural” for resource management purposes. In many people’s minds this

Table 1—Aspects of forest structure and fire regime considered in evaluating contemporary landscapes restoration needs.

Structure	Fire regime
Composition	Return interval
Density	Season
Age distribution	Size
Patch size	Intensity/severity
Patch frequency	Gap size
Potential fuels	Gap distribution

means within the range of historical variability (for example, Morgan and others 1994; Millar 1997; Stephenson 1999). However, constraints such as our ability to restore natural processes, need to be considered. In addition, the range of “natural variability” may not include all ecosystem components considered important by stakeholders.

Christensen (1991b) cautions that “successful policies will have three common characteristics: (1) clearly stated operational goals, (2) identification of potential constraints, and (3) recognition of the variability and complexity of the successional process.” While resource managers may have clearly stated operational goals, scientists are some way from fully understanding the complexity of forest structure and function and how past management activities may constrain future successional responses.

Steps 4, 5, and 6: Targets and Objectives in Restoration

For much of the Sierra Nevada, forests do not meet our criteria of pre-Euroamerican conditions in terms of both structure and process (Table 1), and thus are candidates for restoration (SNEP 1996). There is widespread agreement that restoring fire to Sierran forests should focus on the “pre-Euroamerican condition” as the appropriate restoration target, a perspective consistent with the 1963 “Leopold Report” (Leopold and others 1963) guideline for reducing contemporary human impacts and restoring pre-Euroamerican conditions. The pre-Euroamerican condition model is not without criticism, as is often the case with such typological restoration targets (Noss 1985; Pickett and Parker 1994). While selecting the pre-Euroamerican time period as the appropriate target can be debated, it at least provides conditions for which we have some hope of emulating. In general, there is much more agreement on the use of this target condition than on techniques of restoring this target condition. Disagreement centers largely over whether restoring the process of fire is sufficient when forest structure may have been altered by decades of fire exclusion (Stephenson 1999). Currently these matters are being addressed in the USDA/USDI Joint Fire Science Program (<http://ffs.psw.fs.fed.us/>), which will study the ecological impacts of forest fuel reduction alone and in combination with structural manipulation.

In addition to a clear articulation of target conditions, successful restoration requires a careful evaluation of constraints, and development of a proposal with obtainable objectives.

Steps 7 and 8: Monitoring and Evaluating Ecosystem Function

Monitoring is a critically important part of the restoration process and provides the input necessary to evaluate ecosystem functioning (Keifer and Stanzler 1995; Keifer 1998; Keifer and others 2000a; Mutch and Parsons 1998; Haase and Sackett 1998). Many ecological, sociological, and political considerations will influence the decision regarding the acceptability of ecosystem function. If ecosystem functioning is unacceptable there are several potential reasons. The restoration process may have been in error, either in the planning or execution. Correcting such problems often requires more technical expertise in restoration techniques. Another reason may be that the selection of target conditions was flawed, or constraints not adequately evaluated, such as the need to retain or restore certain target species. Even if programs are successful in restoring naturally functioning ecosystems, the results may not meet goals of some stakeholders. Solving these problems might require a reevaluation of goals, perhaps even placing naturalness at a lower level of priority (for example, Graber 1995). Lastly, new research may provide information that alters the model of natural landscapes (Step 2).

Step 9: Maintaining Natural Ecosystems

Restoration is a corrective step that, if successful, should be replaced by a maintenance process (fig. 2). Maintenance also requires constant monitoring and evaluation, but potentially involves different approaches than restoration.

Fire Management and Future Global Change

Vitousek and others (2000) have reviewed the evidence for anticipated changes in climate. Rapid increases in greenhouse gases are projected to alter both temperature and precipitation patterns. Coupled with anticipated increases in lightning (Price and Rind 1994) there is reason to expect future fire regimes will differ significantly from past fire regimes (Parsons 1991; Ryan 1991; Torn and Fried 1992) and changes will occur faster than ever observed in the past (Vitousek and others 2000). In light of anthropogenically induced climate changes, focusing upon model conditions of the 19th century may be like trying to hit a moving target. Or, as Peter Vitousek noted, “in a changing world we need to distrust baselines.”

One could argue that anticipated climate changes are anthropogenic and therefore if the objective is to minimize human influence, then resource management goals should be directed at circumventing these climate changes. Not only would such an approach present intractable problems but it ignores the reality that even without human influences, there is no reason to assume environments of the 19th and 21st centuries would remain the same (for example, Anderson and Smith 1990; Swetnam 1993; Scuderi 1993; Graumlich 1993; Millar and Woelfenden 1999).

When anticipated climate changes are viewed in the context of other global changes, such as increasing population pressure and ecosystem fragmentation, 19th century

typological models of the structural conditions expected from natural fire regimes will need to be replaced by models focused more on fire as a natural ecosystem process. This requires mathematical models that capture the dynamic interaction between ignition patterns, weather and fuels. Presently we lack models sufficient to make precise predictions of future fire regimes. Perhaps more important, however, is the observation that such models are commonly limited by the validity of their underlying assumptions (Loehle and LeBlanc 1996). In other words, before we can develop such models, we need a clearer mechanistic understanding of how these parameters have and will affect fire regimes, past, present, and future.

Role of Ignition Patterns in Determining Fire Regimes

Lightning is the sole natural source of ignition in these and in most other ecosystems (Show and Kotok 1924; van Wagtenonk 1986). There is debate in the literature as to whether or not Native American ignitions should be part of our model of a natural landscape and the debate illustrates the multitude of different considerations resource managers and scientists must consider (Table 2). Resolution of this argument is needed for more than merely satisfying academic curiosity as it affects both policy and science. If we agree that natural ecosystems have minimal human impacts, then there would be little reason for including Native American burning in our model of a natural landscape, but it may justifiably be included in models of cultural landscapes. Also, our perception of pre-Euroamerican forest structure and function is heavily influenced by fire scar records and these records are used in the setting of restoration targets. If Native American burning is not adequately ascertained, then we may be targeting cultural rather than natural landscapes. In cases where cultural landscapes are

the desired condition, then teasing out the contribution of Native American burning from the fire scar record will provide information on the extent to which past burning patterns might be recreated by lightning alone, and thus the extent to which prescription burning subsidies will be needed.

Role of Weather and Fuels in Determining Fire Regimes

There is debate in the literature as to the relative importance of fuels and weather in driving fire regimes. On the one hand, fuels are considered to be of overriding importance in determining fire regimes. On the other hand prehistoric changes in fire regimes have been tied to climate (Edlund and Byrne 1991; Swetnam 1993) and contemporary fire regimes show a strong climatic signal in the southern Sierra Nevada (Chang 1999) as well as in other regions (Schroeder 1964). Future changes in weather are not known with any certainty but projections for Sierra Nevada forests suggest an increase in the annual window of opportunity for fire and potential for altered fire intensities (Parsons 1991). It is questionable to what extent resource managers will allow such changes to be expressed in future fire regimes. Indeed, presently fire managers allow fires to burn only under a subset of potential weather conditions, which probably do not capture the full range of natural variability.

Further complicating matters is the level of landscape development (such as roads and buildings) within otherwise largely natural landscapes. Such habitat fragmentation greatly affects fuel continuity and the capacity for lightning ignitions to burn landscapes in patterns that would be observed in the absence of such human interference. In addition, policies of total fire suppression on lands adjacent to natural areas will further limit the ability of lightning alone to recreate natural fire regimes in wilderness areas. These factors argue that "natural" fire regimes

Table 2—There is substantial evidence of Native American burning in the Sierra Nevada (Wickstrom 1987), but that information alone can not answer the question of whether or not Native American burning patterns should be included in restoration and maintenance of natural fire regimes.

Arguments for inclusion	Arguments for exclusion
(1) These ignitions were part of the pre-Euroamerican environment and therefore they fit the Leopold Report goals.	(1) Sustainable forest management can not focus indefinitely on Pre-Euroamerican forest conditions and the 1963 Leopold Report should be viewed only as an historically important stage in the evolution of park policy.
(2) Native Americans were "in tune" with their environment and managed landscapes in a responsible manner, unlike contemporary humans (Kilgore 1985), i.e., "open and parklike forests" are aesthetically more pleasing than "dog-hair thickets of white fir" (Graber 1995).	(2) Early Americans exploited their environment in a manner that was not qualitatively different from contemporary humans and given sufficient time they were capable of causing unwanted changes in their environments (e.g., Betancourt and van Devender 1981; Diamond 1986, 1996).
(3) Native Americans were a "natural" part of the landscape (Kilgore 1985).	(3) This Euro-centric perspective presumes the existence of unknown qualities that separate Native Americans from the rest of humanity (e.g., Callicott 2000). Restoring Native American burning is not ecological restoration but rather cultural restoration .
(4) These ignitions were not sufficient to alter burning caused by lightning alone and therefore inclusion is largely irrelevant (Swetnam et al. 1998; Stephenson 1999).	(4) Lightning ignitions alone were insufficient to account for fire scar records (Kilgore and Taylor 1979) or natural landscape patterns (Reynolds 1959) and therefore inclusion is highly relevant to how we interpret the past and manage the future.

will increasingly require human subsidy in the form of prescription burning (this justification for burning subsidy falls outside the criticism posed by Parsons and others (1985) against trying to emulate Native American burning). It is not logically inconsistent to use fire as a manipulative tool (for example, Johnson and Miyanishi 1995) for the purpose of restoring natural conditions, when the intent is to counterbalance other human impacts.

Recreating and Maintaining Natural Landscape Patterns

The natural range of variation in Sierran landscapes is a product of temporal and spatial changes in fire regime. Describing differences in fire regimes is often difficult because regimes are sometimes classified by the characteristics of the fire and sometimes by the effects produced by the fire (Brown 1995). Natural fire regimes in Sierran forests are often described as consisting of *understory* or *low intensity surface* fires, which contrasts with fires in other ecosystems, such as boreal forests or chaparral, that are typically *high intensity* or *stand-replacing* fires (fig. 3). Strictly speaking, low intensity surface fire regimes are more typical of savannas or open forests where fuels are largely herbaceous and such a regime does not adequately describe fire in mid-elevation Sierran forests (Keeley and Zedler 1998). Woody fuels, and their heterogeneous distribution in these forests, generate a mixture of low and high intensity burning. Commonly high intensity burning is restricted to individual trees or small clusters, but as Show and Kotok (1924) noted, "local crown fires may extend over a few hundred acres." Such high intensity fires in the past are suggested by dramatic growth releases in annual rings (Stephenson and others 1991; Mutch and Swetnam 1995). In addition to mortality from high intensity hot spots, surface fires also create gaps by causing mortality in younger age classes and vulnerable species such as *Abies concolor* (Kilgore 1973).

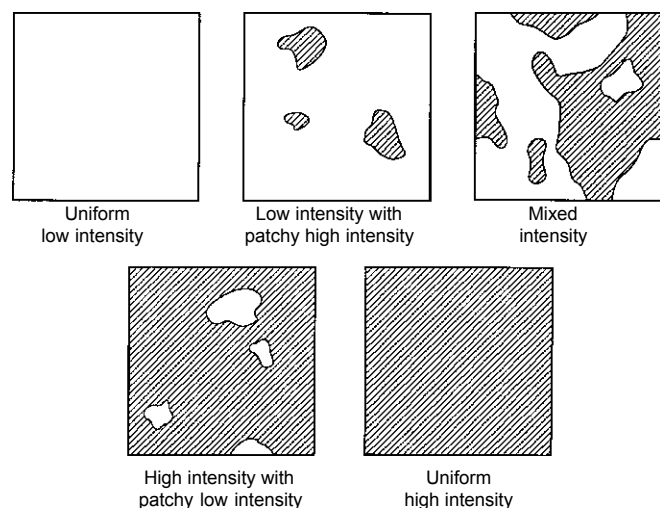


Figure 3—Range of variation of fire intensity patterns (from Stephenson and others 1991).

This mixture of surface burning and localized high intensity fires leads to a landscape mosaic of canopy gaps (fig. 3). Oftentimes this process is described as a "moderate" intensity burn, but that terminology fails to capture the action as much as describing a person who has fallen off a roof as having been, on average, midway between the roof and the ground. Agee (1995) describes such a fire regime as one "ranging from underburns, to significantly thinned stands, to stand-replacement [gaps]." The term *stand-thinning* fire regime perhaps best captures the pattern, and places appropriate emphasis on the importance of gap generation rather than fire intensity. This landscape gap pattern is critical to long term forest maintenance as many dominant trees depend upon such gaps for regeneration, which leads to quasi-even age forest patches (Show and Kotok 1924; Bonnicksen and Stone 1982, Stephenson and others 1991). The landscape mosaic of gap generated patches also likely has profound impacts on the distribution of wildlife.

Gap size varies spatially and temporally. Under a natural stand-thinning fire regime an individual fire may generate a significant number of small (single tree) gaps and a much smaller percentage of larger gaps. In order to scale up our models of natural conditions from forest stands to landscapes we need to make predictions about the expected distribution of gaps. For a natural Sierran landscape we hypothesize, with very limited data, a distribution of gap sizes distributed as depicted in figure 4. This may adequately describe past landscape patterns but following nearly a century of fire exclusion, we have altered the landscape by reducing the frequency and size of gaps (Skinner 1995). However, in the future gaps are likely to be larger due to unnatural fuel accumulation that is predicted to produce more high intensity stand-replacing fires (fig. 4). In short, heavy fuel accumulation and high intensity fires are not unnatural in Sierra Nevada forests but rather the spatial extent of high intensity fires was limited in the past, but now the potential size has increased. In more general terms, fire exclusion is moving the system from a fine scale to a coarse scale landscape.

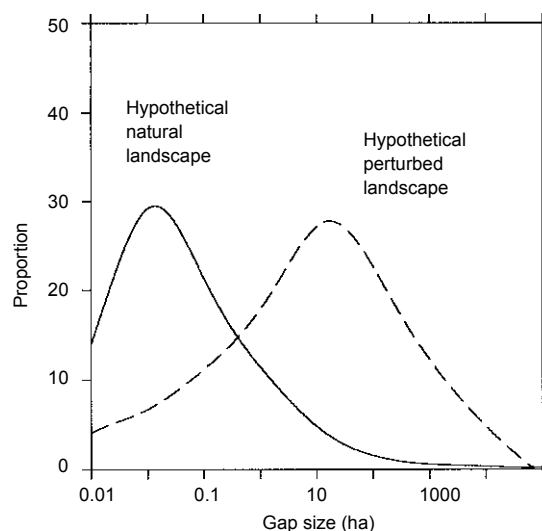


Figure 4—Hypothetical distribution of fire generated gaps (and subsequent forest patches) expected for natural Sierran landscapes and those perturbed by fire suppression.

Future Research Needs

In an era of many types of global change, sustainable wilderness management requires a clearer understanding of the natural range of variation in fire regimes and subsequent landscape mosaics (Morgan and others 1994; Millar 1997), plus an understanding of the resilience of these ecosystems to deviation from that range. In the southern Sierra Nevada, and elsewhere, fire scar dendrochronology has been extraordinarily valuable in recreating past landscapes (Swetnam 1993; Caprio and Swetnam 1995; Skinner and Chang 1996; Swetnam and others 1998). Much remains to be gleaned from this work, particularly in the determination of bounds on the natural range of variation in fire regime at both the landscape and community scales.

Making statistically valid inferences about landscape patterns of burning with fire scar dendrochronology data has limitations that need further exploration. These fire histories are not based upon random samples of the landscape, rather they, by necessity, focus on sites with fire scarred trees and possibly in densities higher than the landscape as a whole. The southern Sierra Nevada is an extraordinarily rugged mountain range and accessibility is certainly a factor in selection of sites, both for dendrochronologists as well as Native Americans. Barrett and Arno (1982) have shown (in the Rocky Mountains) that study sites proximal to Native American settlements had a much higher incidence of burning than more distal sites. In the Sierra Nevada, one approach to validating inferences beyond local study sites might be a simple comparison of fire scarred tree density at sample sites with the density from random landscape samples.

In addition to the question of Native American burning, are questions related to the extrapolation of point data (individual fire scarred trees) to the spatial pattern of burning generated by composite samples (all fire scarred trees in a stand). Fire return intervals estimated from composite samples are usually much shorter than intervals recorded by individual trees. It is important to recognize that estimates drawn from composite samples carry with them certain assumptions about fire behavior. These need to be closely examined because composite estimates play a significant role in determining burning prescriptions in forest restoration plans (Keifer and others 2000b).

Some have suggested that point data should not be used to infer a spatial pattern to a fire because of the localized nature of many lightning ignited fires (Minnich and others, in press). However, dendrochronologists often restrict inferences about spatial patterns of burning to instances where widely scattered trees reveal scars from both the same year and season, thus strengthening the assumption that they constitute different points of a single widespread fire (Caprio and Swetnam 1995; Swetnam and others 1998). The failure of an individual fire scarred tree to record a fire, when it occurs within a circumscribed burned area, is generally attributed to the vagaries of scar formation—such trees are considered uninformative about that particular fire. It would be prudent, however, to consider the possibility that such trees may reflect intra-stand variation in burning. That is, fires may not burn uniformly through a stand and individuals may not scar because the fire skipped their particular patch (Dieterich 1980; Brown and others 1995). If so, this

may alter the fire manager's perspective on the acceptable standards for evaluating prescription-burning patterns.

Knowledge of intra-stand variation in natural fire regimes will add to our ability to manage forests with the appropriate level of gap structure. Gaps are critical to the regeneration of certain species in Sierran forests, for example, *Pinus ponderosa* and *Sequoiadendron giganteum* (Kilgore and Biswell 1971; Mutch and Swetnam 1995; Keifer 1998; Stephens and others 1999). Gaps play two critical roles in the regeneration of these species – they provide a suitable site for seedling recruitment and, because of the absence of mature trees, fuels accumulate more slowly (fig. 5A). This increases the likelihood that fires burning in adjacent forests will skip—or burn incompletely—these regeneration sites for some period of time following patch initiation, thus promoting sapling survivorship (fig. 5B). Such a scenario is required for successful recruitment, since fires at a young age are commonly lethal to coniferous seedlings and young saplings (Swezy and Agee 1991; Regelbrugge and Conard 1993), and is predicted from simulation models of natural fire regimes (van Wagtenonk 1986).

Fire scar dendrochronology may provide some evidence of such intra-site variation in burning. It is a widespread custom in fire scar dendrochronology studies to ignore the first interval from the pith (~germination) to first scar

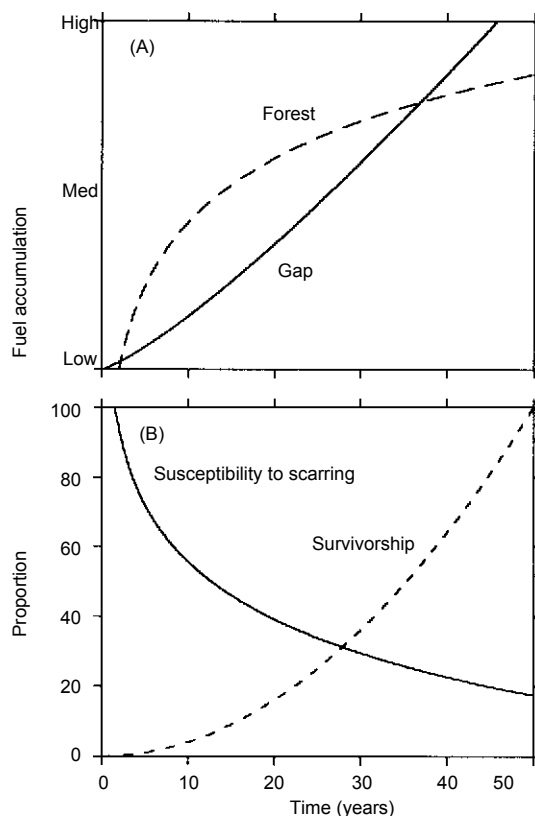


Figure 5—Expected postfire changes in (A) fuel accumulation in forests burned by low intensity underburns vs gaps generated by high intensity fire, and (B) susceptibility of saplings to formation of first scar and the expected seedling/sapling survivorship of a repeat fire.

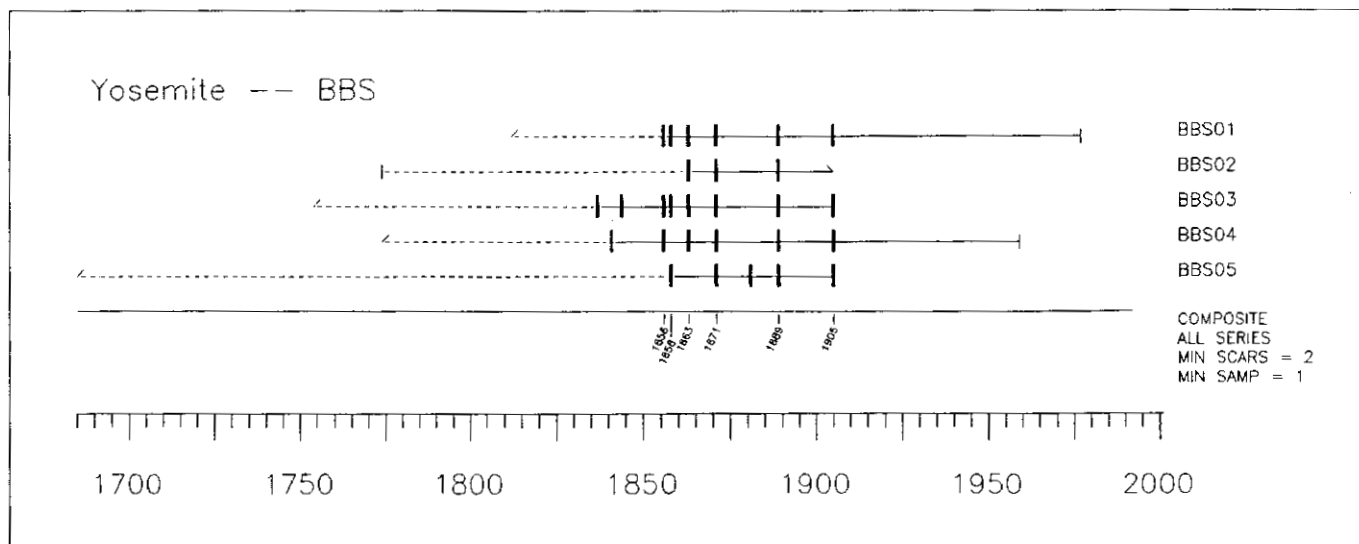


Figure 6—Fire scar dendrochronology record for individual trees at a mid-elevation site in Yosemite National Park (from Swetnam and others 1998). Bold vertical marks indicate fires and dashed horizontal lines reflect the period from the pith to the first scar.

formation (fig. 6). Omitting this fire interval has been justified because (1) it is not known if it is a fire interval and (2) the interval would not include that period of time from the last fire to germination and would thus give shorter fire intervals than was really the case (Baker 1989). At present there is insufficient information available to make either of these arguments very compelling. Justification #1 applies to all fire intervals and in fact is the basis for using composite fire histories. The second is logically justifiable, however, in the vast majority of cases the time from the pith to first scar is longer than the average fire interval for that tree and including it usually increases the estimated fire return interval. Others have suggested that prior to the first scar, saplings are less susceptible to scarring, however there is no empirical evidence of such a phenomenon (Tom Swetnam, personal communication, September 1999).

However, rather than including this interval from germination to the first fire in a composite fire history it might be worth considering the extent to which this reflects events occurring in gaps. Because bark thickness increases with age, it is reasonable to expect that the propensity for initial scar formation should be high in young saplings and decrease with time (fig. 5B). Thus, the failure to find scars in young trees is due either to fire-caused mortality eliminating young trees (Gutsell and Johnson 1996) or failure of fire to burn the patch or microsite where the seedling has established. This initial interval between establishment and first fire scar could provide a means of getting at estimates of intra-stand variation in burning and the period of time patches need to be released from fire in order to achieve successful recruitment. This is reflected in a comparison of fire return intervals calculated for the first intervals compared to the average calculated by all other intervals (Table 3). This example suggests that patches may require a significant fire-free period for successful recruitment, a conclusion that has relevance to the evaluation of post-fire monitoring of prescribed burns and future prescription plans.

Table 3—Comparison of reported fire-return interval (excluding first interval) with calculated fire return-intervals for period from pith to first scar—period from germination to first scar would be longer due to the sampling of fire scars at various heights above ground level (data from Swetnam and others 1998).

Site	Reported fire return interval		Fire interval from germination to first scar	
	X	S.E.	X	S.E.
Mariposa grove (Yosemite NP)	5.0	0.8	38.3	5.2
Giant forest (Sequoia NP)	10.2	2.0	45.1	7.9

Conclusions

After nearly a century of highly successful fire suppression there is an urgent need for restoring fire to many Sierran forests, both because the current situation jeopardizes ecosystem stability and because it represents a dangerous fire hazard (GAO 1999). Pre-Euroamerican models of forest structure may be an appropriate target for contemporary restoration efforts, but future forest maintenance will need to shift emphasis from structure to process. The ideal of allowing just natural lightning ignited fires to eventually return fire to its natural role (Parsons and others 1985) is appropriate. However, the reality of the situation is that lightning ignited fires alone are incapable of recreating natural landscapes. There are several reasons for this. Habitat fragmentation by roads creates barriers to natural fire spread. Additionally, lightning fires that threaten developments, commercial timber or watershed processes will always be suppressed, both within natural areas, such as national parks, as well as on adjacent private and public lands. It is our belief that the goal of restoring and maintaining

ecosystems with minimal human impact is not incompatible with the reality that this will require fire subsidies in the form of prescription burning.

Future management requires a better understanding of the natural range of variation in fire regimes. Due to a century of wildfire exclusion, most of our direct knowledge of fire in the Sierra Nevada is based on observations of prescribed fires—either intentional prescribed burns or unintentional natural fires, both of which are allowed to burn only under “acceptable” weather/fuel/geographic conditions. In the absence of human interference there is reason to believe that the landscape has historically burned under a greater mixture of fire intensities and severities. Future progress in our understanding of natural fire regimes is most likely to progress through modeling of both fire and forest processes, for example by coupling weather/fuel-driven fire spread models (Weise and Biging 1997) with climate-driven forest dynamics models (Urban and Miller 1996; Miller and Urban 1999). The extent to which this approach alters management of Sierran forests will depend upon other ecological and political constraints.

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References

- Agee, J. K. 1995. Alternatives for implementing fire policy. In: Brown, J. K.; Mutch, R. W.; Spoon, C. W.; Wakimoto, R. H., eds. *Proceedings: symposium on fire in wilderness and park management*. Ogden, UT: USDA Forest Service, General Technical Report INT-GTR-320: 107-112.
- Anderson, R. S.; Smith, S. J. 1990. Paleoeecology within California's Sierra Nevada national parks: an overview of the past and prospectus for the future. In: *Natural areas and Yosemite: prospects for the future*. Concord, CA: Yosemite centennial symposium proceedings: 329-337.
- Ansley, J. A. S.; Battles, J. J. 1998. Forest composition, structure, and change in an old-growth mixed conifer forest in the northern Sierra Nevada. *Journal of the Torrey Botanical Society* 125:297-308.
- Baker, W. L. 1989. Effect of scale and spatial heterogeneity on fire-interval distributions. *Canadian Journal of Forest Research* 19:700-706.
- Bancroft, L.; Nichols, T.; Parsons, D.; Graber, D.; Evison, B.; van Wagendonk, J. 1985. Evolution of the natural fire management program at Sequoia and Kings Canyon national parks. In: Lotan, J. E.; Kilgore, B. M.; Fischer, W. C.; Mutch, R. W., eds. *Proceedings—symposium and workshop on wilderness fire*. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station, General Technical Report INT-182: 174-179.
- Barrett, L. A. 1935. A record of forest and field fires in California from the days of the early explorers to the creation of the forest reserves. San Francisco, CA: USDA Forest Service, unpublished ms. on file.
- Barrett, S. W.; Arno, S. 1982. Indian fires as an ecological influence in the northern Rockies. *Journal of Forestry* 80:647-651.
- Betancourt, J. L.; van Devender, T. R. 1981. Holocene vegetation in Chaco Canyon, New Mexico. *Science* 214:656-658.
- Bonnicksen, T. M.; Stone, E. C. 1982. Reconstruction of a presettlement giant sequoia-mixed conifer forest community using the aggregation approach. *Ecology* 63:1134-1148.
- Bonnicksen, T. M.; Stone, E. C. 1985. Restoring naturalness to national parks. *Environmental Management* 9:479-486.
- Botti, S. J.; Nichols, T. 1979. The Yosemite and Sequoia-Kings Canyon prescribed natural fire programs 1968-1978. In: *Proceedings of the second conference on scientific research in the national parks*. Washington, D.C.: USDI National Park Service, Volume 10. Fire ecology: 46-63.
- Brown, J. K. 1995. Fire regimes and their relevance to ecosystem management. In: *Managing forests to meet people's needs*. Bethesda, MD: Society of American Foresters: 171-178.
- Brown, J. K.; Arno, S. F.; Bradshaw, L. S.; Menakis, J. P. 1995. Comparing the Selway-Bitterroot fire program with presettlement fires. In: Brown, J. K.; Mutch, R. W.; Spoon, C. W.; Wakimoto, R. H., eds. *Proceedings: symposium on fire in wilderness and park management*. Ogden, UT: USDA Forest Service, General Technical Report INT-GTR-320: 48-54.
- Callicott, J. B. 2000. Contemporary criticisms of the received wilderness idea. In: Cole, David N.; McCool, Stephen F.; Freimund, Wayne A.; O'Loughlin, Jennifer, comps. 2000. *Wilderness science in a time of change conference—Volume 1: Changing perspectives and future directions*; 1999 May 23-27; Missoula, MT. *Proceedings RMRS-P-15-VOL-1*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Caprio, A. C.; Swetnam, T. W. 1995. Historic fire regimes along an elevational gradient on the west slope of the Sierra Nevada, California. In: Brown, J. K.; Mutch, R. W.; Spoon, C. W.; Wakimoto, R. H., eds. *Proceedings: symposium on fire in wilderness and park management*. Ogden, UT: USDA Forest Service, General Technical Report INT-GTR-320: 173-179.
- Cermak, R. W.; Lague, J. H. 1993. Range of light - range of darkness. The Sierra Nevada - 1841-1905. San Francisco, CA: USDA Forest Service, Pacific Southwest Regional Office, unpublished report on file.
- Chang, C.-R. 1999. Understanding fire regimes. Durham, NC: Duke University, Ph.D. dissertation. 184 p.
- Christensen, N. L., Jr. 1991a. Wilderness and high intensity fire: how much is enough. *Proceedings Tall Timbers Fire Ecology Conference* 17:9-24.
- Christensen, N. L., Jr. 1991b. Variable fire regimes on complex landscapes: ecological consequences, policy implications, and management strategies. In: Nodvin, S. C.; Waldrop, T. A., eds. *Fire and the environment: ecological and cultural perspectives*. Asheville, NC: USDA Forest Service, Southeastern Forest Experiment Station General Technical Report SE-69: ix-xiii.
- Diamond, J. 1986. The environmentalist myth. *Nature* 324:19-20.
- Diamond, J.M. 1996. *Guns, germs, and steel: the fates of human societies*. New York, NY: W. W. Norton & Co.
- Dieterich, J. H. 1980. The composite fire interval—a tool for more accurate interpretation of fire history. In: Stokes, M. A.; Dieterich, J. H., eds. *Proceedings of the fire history workshop*. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-81: 8-14.
- Edlund, E. G.; Byrne, R. 1991. Climate, fire, and late Quaternary vegetation change in the central Sierra Nevada. In: Nodvin, S. C.; Waldrop, T. A., eds. *Fire and the environment: ecological and cultural perspectives, proceedings of an international symposium*. Asheville, NC: USDA Forest Service, Southeastern Forest Experiment Station, General Technical Report SE-69: 390-396.
- GAO. 1999. Western National Forests. A cohesive strategy is needed to address catastrophic wildfire threats. Washington, D.C.: Report to the Subcommittee on Forests and Forest Health, Committee on Resources, House of Representatives. (No. GAO/RCED-99-65 (<http://www.access.gpo.gov/su%5Fdocs/aces/aces160.shtml>)).
- Graber, D. M. 1985. Coevolution of national park service fire policy and the role of national parks. In: Lotan, J. E.; Kilgore, B. M.; Fischer, W. C.; Mutch, R. W., eds. *Proceedings—symposium and workshop on wilderness fire*. USDA Forest Service, Ogden, UT: Intermountain Forest and Range Experiment Station, General Technical Report INT-182: 345-349.
- Graber, D. M. 1995. Resolute biocentrism: the dilemma of wilderness in national parks. In: Soule, M. E.; Lease, G., eds. *Reinventing nature?* Washington, D.C.: Island Press: 123-135.
- Graumlich, L. J. 1993. A 1000-year record of temperature and precipitation in the Sierra Nevada. *Quaternary Research* 39:249-255.

- Gutsell, S. L.; Johnson, E. A. 1996. How fire scars are formed: coupling a disturbance process to its ecological effect. *Canadian Journal of Forest Research* 26:166-174.
- Haase, S. M.; Sackett, S. S. 1998. Effects of prescribed fire in giant sequoia-mixed conifer stands in Sequoia and Kings Canyon national parks. *Proceedings Tall Timbers Fire Ecology Conference* 20:236-243.
- Hunter Jr., M. 1996. Benchmarks for managing ecosystems: are human activities natural? *Conservation Biology* 10:695-697.
- Johnson, E. A.; Miyanishi, K. 1995. The need for consideration of fire behavior and effects in prescribed burning. *Restoration Ecology* 3:271-278.
- Keeley, J. E. 1981. Reproductive cycles and fire regimes. In: Mooney, H. A.; Bonnicksen, T. M.; Christensen, N. L.; Lotan, J. E.; Reiners, W. A., eds. *Proceedings of the conference fire regimes and ecosystem properties*. Washington, D.C.: USDA Forest Service, General Technical Report WO-26: 231-277.
- Keeley, J. E.; Fotheringham, C. J.; Morais, M. 1999. Reexamining fire suppression impacts on brushland fire regimes. *Science* 284:1829-1832.
- Keeley, J. E.; Zedler, P. H. 1998. Evolution of life histories in *Pinus*. In: Richardson, D. M., eds. *Ecology and biogeography of Pinus*. Cambridge, U.K.: Cambridge University Press: 219-250.
- Keifer, M. 1998. Fuel load and tree density changes following prescribed fire in the giant sequoia-mixed conifer forest: the first 14 years of fire effects monitoring. *Proceedings Tall Timbers Fire Ecology Conference* 20:306-309.
- Keifer, M.; Lineback, P.; Folger, K. 2000b. Incorporating a GIS model of ecological need into fire management planning. In: *Proceedings of the Fire Ecology and Management Conference*, San Diego, CA: California Association of Fire Ecologists. In press.
- Keifer, M.; Stanzler, P. M. 1995. Fire effects monitoring in Sequoia and Kings Canyon National Parks. In: Brown, J. K.; Mutch, R. W.; Spoon, C. W.; Wakimoto, R. H., eds. *Proceedings: symposium on fire in wilderness and park management*. Ogden, UT: USDA Forest Service, General Technical Report INT-GTR-320: 215-218.
- Keifer, M.; Stephenson, N. L.; Manley, J. 2000a. Prescribed fire as the minimum tool for wilderness forest and fire regime restoration: a case study from the Sierra Nevada, California. In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. *Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management*; 1999 May 23-27; Missoula, MT. *Proceedings RMRS-P-15-VOL-5*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Kilgore, B. M. 1973. The ecological role of fire in Sierran conifer forests: its application to national park management. *Quaternary Research* 3:496-513.
- Kilgore, B. M. 1985. What is "natural" in wilderness fire management? In: Lotan, J. E.; Kilgore, B. M.; Fischer, W. C.; Mutch, R. W., eds. *Proceedings—symposium and workshop on wilderness fire*. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station, General Technical Report INT-182: 57-66.
- Kilgore, B. M.; Biswell, H. H. 1971. Seedling germination following fire in a giant sequoia forest. *California Agriculture* 25:8-10.
- Kilgore, B. M.; Sando, R. W. 1975. Crown-fire potential in a sequoia forest after prescribed burning. *Forest Science* 21:83-87.
- Kilgore, B. M.; Taylor, D. 1979. Fire history of a sequoia-mixed conifer forest. *Ecology* 60:129-142.
- Leopold, A. S.; Cain, S. A.; Cottam, C. M.; Gabrielson, I. N.; Kimbal, T. L. 1963. Wildlife management in the national parks. *American Forests* 69(4):32-35, 61-63.
- Loehle, C.; LeBlanc, D. 1996. Model-based assessments of climate change effects on forests: a critical review. *Ecological Modelling* 90:1-31.
- Millar, C. I. 1997. Comments on historical variation & desired condition as tools for terrestrial landscape analysis. In: Sommarstrom, S., eds. *Proceedings of the sixth biennial watershed management conference*. Davis, CA: University of California, Water Resources Center Report No. 92: 105-131.
- Millar, C. I.; Woolfenden, W. B. 1999. The role of climate change in interpreting historical variability. *Ecological Applications* 9:1207-1216.
- Miller, C.; Urban, D. L. 1999. Forest pattern, fire, and climatic change in the Sierra Nevada. *Ecosystems* 2:76-87.
- Minnich, R. A.; Barbour, M. G.; Burk, J. H.; Sosa-Ramirez, J. In press. Californian conifer forests under unmanaged fire regimes in the Sierra San Pedro Martir, Baja California, Mexico. *Journal of Biogeography*.
- Morgan, P.; Aplet, G. H.; Haufler, J. B.; Humphries, H. C.; Moore, M. M.; Wilson, W. D. 1994. Historical range of variability: a useful tool for evaluating ecosystem change. *Journal of Sustainable Forestry* 2:87-111.
- Mutch, L. S.; Swetnam, T. W. 1995. Effects of fire severity and climate on ring-width growth of giant sequoia after burning. In: Brown, J. K.; Mutch, R. W.; Spoon, C. W.; Wakimoto, R. H., eds. *Proceedings: symposium on fire in wilderness and park management*. Ogden, UT: USDA Forest Service, General Technical Report INT-GTR-320: 241-280.
- Noss, R. F. 1985. On characterizing presettlement vegetation: how and why. *Natural Areas Journal* 5:5-19.
- NPS. 1988. *Management policies*: U.S. Department of the Interior, National Park Service. Washington, D.C.: U.S. Government Printing Office.
- Parsons, D.; Bancroft, L.; Nichols, T.; Stohlgren, T. 1985. Information needs for natural fire management planning. In: Lotan, J. E.; Kilgore, B. M.; Fischer, W. C.; Mutch, R. W., eds. *Proceedings—symposium and workshop on wilderness fire*. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station General Technical Report INT-182: 356-358.
- Parsons, D. J. 1981. The historical role of fire in the foothill communities of Sequoia National Park. *Madroño* 28:111-120.
- Parsons, D. J. 1990. Restoring fire to the Sierra Nevada mixed conifer forest: reconciling science, policy and practicality. In: Hughes, H. G.; Bonnicksen, T. M., eds. *Restoration '89: the new management challenge*. Madison, WI: University of Wisconsin Arboretum, Society for Ecological Restoration: 271-279.
- Parsons, D. J. 1991. Planning for climate change in national parks and other natural areas. *Northwest Environmental Journal* 7:255-269.
- Parsons, D. J.; DeBenedetti, S. H. 1979. Impact of fire suppression on a mixed-conifer forest. *Forest Ecology and Management* 2:21-33.
- Parsons, D. J.; van Wagtenonk, J. W. 1996. Fire research and management in the Sierra Nevada National Parks. In: Halvorson, W. L.; Davis, G. E., eds. *Science and ecosystem management*. Tucson, AZ: University of Arizona Press: 25-48.
- Pickett, S. T. A.; Parker, V. T. 1994. Avoiding the old pitfalls: opportunities in a new discipline. *Restoration Ecology* 2:75-79.
- Price, C.; Rind, D. 1994. Lightning fires in a 2xCO₂ world. In: 12th Conference on fire and forest meteorology, October 26-28, Jekyll Island, Georgia. Washington, D.C.: Society of American Foresters: 77-84.
- Regelbrugge, J. C.; Conard, S. G. 1993. Modeling tree mortality following wildfire in *Pinus ponderosa* forests in the central Sierra Nevada of California. *International Journal of Wildland Fire* 3:139-148.
- Reynolds, R. D. 1959. Effect of natural fires and aboriginal burning upon the forest of the central Sierra Nevada. Berkeley, CA: University of California, . M.A. thesis. 262 p.
- Roy, D. G.; Vankat, J. L. 1999. Reversal of human-induced vegetation changes in Sequoia National Park, California. *Canadian Journal of Forest Research* 29:399-412.
- Ryan, K. C. 1991. Vegetation and wildland fire: implications of global climate change. *Environment International* 17:169-178.
- Schroeder, M. J. 1964. Synoptic weather types associated with critical fire weather. Washington, D.C.: U.S. Department of Commerce, National Bureau of Standards, Institute for Applied Technology, (Report Number AD 448-630).
- Scuderi, L. A. 1993. A 2000-year tree ring record of annual temperatures in the Sierra Nevada Mountains. *Science* 259:1433-1436.
- Show, S. B.; Kotok, E. I. 1924. The role of fire in California pine forests. Washington, D.C.: USDA, Bulletin 1294. 80 p.
- Shrader-Frechette, K. S.; McCoy, E. D. 1995. Natural landscapes, natural communities, and natural ecosystems. *Forest & Conservation History* 39:138-142.
- Skinner, C. D. 1997. Toward and understanding of fire history information. In: Sommarstrom, S., eds. *Proceedings of the sixth*

- biennial watershed management conference. Davis, CA: University of California: Water Resources Center Report No. 92: 15-22.
- Skinner, C. N. 1995. Change in spatial characteristics of forest openings in the Klamath Mountains of northwestern California, USA. *Landscape Ecology* 10:219-228.
- Skinner, C. N.; Chang, C.-R. 1996. Fire regimes, past and present. In: *Sierra Nevada Ecosystem Project: Final report to congress. Volume II*. Davis, CA: University of California, Centers for Water and Wildland Resources: 1041-1069.
- SNEP. 1996. *Sierra Nevada Ecosystem Project: Final report to congress. Volumes II and III. Assessments, commissioned reports, and background information*. Davis, CA: University of California, Centers for Water and Wildland Resources.
- Stephens, S.; Dulitz, D. J.; Martin, R. E. 1999. Giant sequoia regeneration in group selection openings in the southern Sierra Nevada. *Forest Ecology and Management* 120:89-95.
- Stephenson, N. L. 1999. Reference conditions for giant sequoia forest restoration: structure, process, and precision. *Ecological Applications* 9:1253-1265.
- Stephenson, N. L.; Parsons, D. J.; Swetnam, T. W. 1991. Restoring natural fire to the sequoia-mixed conifer forest: should intense fire play a role? *Proceedings Tall Timbers Fire Ecology Conference* 17:321-338.
- Swetnam, T. W. 1993. Fire history and climate change in giant sequoia groves. *Science* 262:885-889.
- Swetnam, T. W.; Allen, C. D.; Betancourt, J. L. 1999. Applied historical ecology: using the past to manage for the future. *Ecological Applications* 9:1189-1206.
- Swetnam, T. W.; Baisan, C. H.; Morino, K.; Caprio, A. C. 1998. Fire history along elevational transects in the Sierra Nevada, California. Three Rivers, CA: U.S. Geological Survey, Sequoia-Kings Canyon and Yosemite National Parks, final report on file.
- Swezy, D. M.; Agee, J. K. 1991. Prescribed-fire effects on fire-root and tree mortality in old-growth ponderosa pine. *Canadian Journal of Forest Research* 21:626-634.
- Taylor, A. H.; Skinner, C. N. 1998. Fire history and landscape dynamics in a late-successional reserve, Klamath Mountains, California, USA. *Forest Ecology and Management* 111:285-301.
- Torn, M. S.; Fried, J. S. 1992. Predicting the impacts of global warming on wildland fire. *Climatic Change* 21:257-274.
- Urban, D.; Miller, C. 1996. Modeling Sierran forests: capabilities and prospectus for gap models. In: eds. *Sierra Nevada Ecosystem Project: Final report to congress. Volume III. Assessments, commissioned reports, and background information*. Davis, CA: University of California, Centers for Water and Wildland Resources: 733-744.
- Vale, T. R. 1987. Vegetation change and park purposes in the high elevations of Yosemite National Park, California. *Annals of the Association of American Geographers* 77:1-18.
- van Wagtenonk, J. W. 1985. Fire suppression effects on fuels and succession in short-fire-interval wilderness ecosystems. In: Lotan, J. E.; Kilgore, B. M.; Fischer, W. C.; Mutch, R. W., eds. *Proceedings of the symposium and workshop on wilderness fire*. Ogden, UT: USDA Forest Service, Intermountain Forest Range and Experiment Station, General Technical Report INT-182: 119-126.
- van Wagtenonk, J. W. 1986. The role of fire in the Yosemite wilderness. In: Lucas, R. C., eds. *Proceedings—national wilderness research conference: current research*. Ogden, UT: USDA Forest Service, Intermountain Research Station, General Technical Report INT-212: 2-9.
- Vankat, J. L.; Major, J. 1978. Vegetation changes in Sequoia National Park, California. *Journal of Biogeography* 5:377-402.
- Vitousek, P. M.; Aber, J. D.; Goodale, C. L.; Aplet, G. H. 2000. Global change and wilderness science. In: Cole, David N.; McCool, Stephen F.; Freimund, Wayne A.; O'Loughlin, Jennifer, comps. 2000. *Wilderness science in a time of change conference—Volume 1: Changing perspectives and future directions*; 1999 May 23–27; Missoula, MT. *Proceedings RMRS-P-15-VOL-1*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Wagner, F. H.; Kay, C. E. 1993. "Natural" or "healthy" ecosystems: are U.S. National Parks providing them? In: McDonnell, M. J.; Pickett, S. T. A., eds. *Humans as components of ecosystems*. New York, NY: Springer-Verlag: 257-270.
- Weise, D. R.; Biging, G. S. 1997. A qualitative comparison of fire spread models incorporating wind and slope effects. *Forest Science* 43:170-179.
- Wickstrom, C. K. R. 1987. Issues concerning Native American use of fire: a literature review. Yosemite, CA: USDI, National Park Service, Yosemite National Park, Yosemite Research Center Publications in Anthropology, No. 6.

Prescribed Fire as the Minimum Tool for Wilderness Forest and Fire Regime Restoration: A Case Study From the Sierra Nevada, California

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Abstract—Changes in forest structure were monitored in areas treated with prescribed fire in Sequoia and Kings Canyon National Parks. Five years after the initial prescribed fires, tree density was reduced by 61% in the giant sequoia-mixed conifer forest, with the greatest reduction in the smaller trees. This post-burn forest structure falls within the range that may have been present prior to Euroamerican settlement, based on forest structural targets developed with input from research, historic photos and written accounts. The results from this monitoring program provide an example of prescribed fire being used successfully both to reduce fuel hazard and to restore forest structure. This example may be particularly interesting to managers of other parks or wilderness areas where fire is considered the most appropriate means for restoring and managing ecosystems.

Fire has been a pervasive and important process in many ecosystems for thousands of years, but humans have disrupted fire regimes over the past century. Proponents of allowing natural processes to function in wilderness areas maintain that natural fire regimes should be restored to many of these areas. After decades of fire regime disruption, however, altered fuel and vegetation conditions throughout many Western forests have increased the risk of severe wildland fires, which could result in undesirable fire effects (Hardy and Arno 1996; Vankat and Major 1978; van Wagtenonk 1985).

Heavy amounts of surface fuels have accumulated after nearly a century of fire regime disruption. In addition, small trees have increased in the understory, many of which would have been thinned by fire in the past. Some believe that forest structure in these areas should be restored before reintroducing natural fire regimes, thus minimizing the potential for unnaturally severe ecosystem effects from fire (Bonnicksen and Stone 1985; Fule and others 1997).

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In the giant sequoia-mixed conifer forest of the Sierra Nevada, pre-Euroamerican fires burned at intervals ranging from 2–30 years, as evidenced by fire scars in the giant sequoia annual ring record dating back nearly 2,000 years (Kilgore and Taylor 1979; Swetnam 1993). This record does not distinguish fires by ignition source, and therefore, includes fires ignited by lightning, as well as those that may have been set by Native Americans (Lewis 1973). The increase in surface fuels and stand density after the century-long disruption of the fire regime are well-documented in this forest type (Kilgore 1972; Parsons 1978; Stephenson 1996, 1999; Vankat and Major 1978).

One way to restore forest structure quickly is to mechanically remove trees, thereby increasing the space between tree crowns and reducing the risk that future fires will spread through the crowns. Using mechanical means to thin forests may have unacceptable consequences in wilderness areas, including new road construction and impacts of heavy equipment (such as high noise levels, soil compaction, heavy erosion and possible increases in certain pathogens). Even with mechanical thinning, surface fuels would need to be burned before natural fire regimes are restored. One method for restoring forest structure that is more compatible with wilderness values and legislation is prescribed fire; however, whether prescribed fire alone can accomplish the needed restoration has been questioned (Bonnicksen and Stone 1985).

To determine whether prescribed fire alone can restore forest structure, changes in fuel load and forest structure are monitored in areas treated with prescribed fire in Sequoia and Kings Canyon National Parks. The Parks have had an active program of prescribed fire since 1969. The long-term monitoring program began in 1982 to assess objective achievement and document changes in fuel and vegetation in burned areas.

Until recently, the Parks' prescribed fire management program focused first on reducing heavy surface fuel loads and then on restoring and maintaining the natural fire regime where possible. The initial fire planned in an area, called a 'restoration burn', has the primary objective of reducing the heavy accumulation of surface fuels that expose park developments and cultural and natural resources to damage from severe wildland fire. For the past three decades, the Park staff has concentrated on restoration burns, in part because of the extent of the recognized fuel hazard. Initial objectives of 60–80% total fuel reduction are consistently met with the current burn prescriptions in the

giant sequoia-mixed conifer forest (Keifer 1998; Keifer and Manley, in press). In addition, large changes in stand structure occurred following prescribed fire (Keifer 1998) but there were no specific targets for stand structure, making it difficult to assess goals related to forest structural restoration.

Recently, the Park staff has developed preliminary targets for structural conditions in all vegetation types where stand structure is likely to have been greatly altered over the past century. These target conditions have been determined using the best available information, including research data, historic photographs, written accounts and expert opinion. To determine whether the prescribed fire program is making progress toward achieving forest structural goals, stand density results from the Parks' fire effects monitoring program is compared with the newly developed targets. If these and other structural target conditions are attained, the program can progress more readily toward restoring and maintaining the natural fire regime, where appropriate. If the target conditions are not achieved, changes to the prescribed fire treatment or reanalysis of the target conditions may be needed before natural fire regimes can be readily restored.

Methods

Study Area

Sequoia and Kings Canyon National Parks are located in the southern Sierra Nevada, California. The giant sequoia-mixed conifer forest is located at elevations from 1,650-2,200 meters (5,400-7,200 feet), on all aspects, in drainage bottoms, broad upland basins and, occasionally, on steep slopes and ridgetops. Soils are coarse-textured and acidic, and soil depth ranges from shallow to very deep. The giant sequoia-mixed conifer forest is dominated by mature white fir (*Abies concolor* [Gordon & Glend.] Lindley), red fir (*A. magnifica* Andr. Murray) and giant sequoia (*Sequoiadendron giganteum* [Lindley] Buchholz), but also includes sugar pine (*Pinus lambertiana* Douglas), ponderosa pine (*P. ponderosa* Laws.), Jeffrey pine (*P. jeffreyi* Grev. & Balf.) and incense cedar (*Calocedrus decurrens* [Torrey] Florin) in small, varying amounts. Understory trees are primarily composed of white fir and incense cedar. The understory vegetation is typically sparse, with few herbs and <20 percent shrub cover.

Forest Structural Targets

In the giant sequoia-mixed conifer forest, research results (Bonnicksen and Stone 1982a; Stephenson 1994), examination of written accounts (Bonnicksen 1978), qualitative analysis of historic photos and expert opinion (both National Park Service and US Geological Survey scientists) led to development of target conditions for stand density. Recognizing that climate in the Sierra Nevada has changed over time (Graumlich 1993), the targets refer to the approximately 1,000-year time period prior to Euroamerican settlement, based on issues addressed by Stephenson (1996, 1999). From age/diameter relationships, trees 80 cm or larger in diameter at breast height (DBH) were established almost exclusively prior to Euroamerican settlement (Finney and Stephenson, unpublished data).

The stand-level structural target for the giant sequoia-mixed conifer forest is to maintain the density of trees ≥ 80 cm DBH between 10-75 trees/ha and trees <80 cm DBH between 50-250 trees/ha, for a total tree density of 60-325 trees/ha. While species composition targets are also needed, the park staff focused on total tree density as a starting point for developing the stand density target conditions. Landscape-level target conditions include size and number of forest gaps and surface fuel load, which are not discussed in this paper.

Development of target conditions is ongoing, and any new knowledge gained about past conditions will be used to further refine target conditions. Additional information that will be useful in this process includes additional search for and quantitative analysis of historic photography and examination of existing park databases that contain information on the density of large trees by species (those present prior to Euroamerican settlement).

Burning Conditions

All areas in this study were burned between 1982 and 1997 within the range of burning conditions specified for the giant sequoia-mixed conifer forest (USDI National Park Service 1992a). Fuels are best described by Northern Forest Fire Laboratory (NFFL) Fuel Model 8 (Albini 1976). Time since the last fire in all plots was longer than 40 years and usually exceeded 100 years. Air temperature during prescribed fires ranged from 4-24 °C (40-75 °F), relative humidity from 25-50 percent and mid-flame wind speeds from 0-10 kilometers per hour (0-6 miles per hour). Fuel moisture ranges included: 1-hour time lag fuel moisture (TLFM), 3-13 percent; 10-hour TLFM, 4-14 percent; 100-hour TLFM, 5-15 percent; 1,000-hour TLFM, 10-20 percent. The range of backing fire rates of spread was 0-20 meters/hour (0-66 feet/hour), with flame lengths from 0-0.6 meters (0-2 feet). Head fire rates of spread ranged from 40-180 meters/hour (132 to 594 feet/hour) with flame lengths from 0-1.5 meters (0-5 feet).

Field Data Collection

Monitoring data were collected from a network of permanently marked 20 x 50 meter plots, established using a stratified-random sampling design within the Park areas designated for prescribed fire. Within each forest plot, fuel load and tree density were recorded pre-burn, immediately post-burn and 1, 2, 5, and 10 years post-burn. To obtain overstory tree density, all trees >1.37 meters (4.5 feet) in height were tagged, mapped, identified to species, measured for diameter and recorded as live or dead (USDI National Park Service 1992b).

We analyzed data from 27 plots that burned in 17 different prescribed fires between 1982 and 1991. Although the plots did not all burn during the same year, all analyses are direct comparisons of the same 27 plots at each post-burn stage. Some of the older plots were not monitored two years post-burn, so those data were not used in the analyses. To examine post-burn changes in stand structure, one-year and five-year post-burn mortality data were used because tree mortality resulting from the direct effects of fire is often not apparent immediately post-burn (Mutch and Parsons 1998).

Results and Discussion

Pre-burn mean density for trees <80 cm DBH was 625 trees/ha, which is two and a half times the maximum target value (fig. 1). The pre-burn mean density of trees ≥80 cm DBH was 46 trees/ha, well within the target range of 10-75 trees/ha. Tree density was reduced one-year post-burn, with 53% mortality of trees <80 cm DBH but only 4% mortality of trees ≥80 cm DBH. While large tree post-burn density remained within the target range, the density of trees <80 cm DBH (292 trees/ha) was still higher than the target maximum of 250 trees/ha (fig. 1).

By five years post-burn, the mean density of trees <80 cm DBH was further reduced to 222 trees/ha, which falls within the target range (fig. 1). The larger trees are only slightly reduced to 42 trees/ha by five years post-burn. Most of the density reduction occurred in the smaller trees, indicating that prescribed fire may reduce the potential for spread of crown fire in these forests by thinning smaller trees and ladder fuels, while minimizing effects on larger trees (8% reduction in density from pre-burn to 5-years post-burn). No mortality of large giant sequoia trees occurred within the monitoring plots following prescribed burning.

Although the Parks' preliminary target conditions do not yet include species composition, one of the indicators of successful fire regime restoration is the regeneration of fire-adapted species. Giant sequoia establishment and recruitment rely heavily on exposed mineral soil and canopy openings resulting from fire. Results from 12 giant sequoia-mixed conifer forest monitoring plots indicate that the relative density of giant sequoia has tripled 10 years after prescribed

fire (Keifer 1998). This increase is primarily attributed to the successful recruitment of giant sequoia post-burn regeneration into the smallest tree diameter class, along with the fire-induced mortality of many of the small white fir (fig. 2). This regeneration of giant sequoia is in stark contrast to areas that have not burned, where giant sequoia regeneration is almost entirely absent (Stephenson 1994). Demographic models suggest that the amount of sequoia regeneration following prescribed fire may be roughly comparable to that prior to Euroamerican settlement (Stephenson, unpublished data).

The results from this monitoring program provide an example of prescribed fire being successfully used both to reduce fuel load and to restore forest structure in the giant sequoia-mixed conifer forest of Sequoia and Kings Canyon National Parks (see also Stephenson 1996, 1999). Whether forest structure can be restored in other vegetation communities using prescribed fire depends on many site-specific factors, including the number of fire-return intervals missed and history of other disturbance. Some vegetation communities may be so greatly altered that using prescribed fire without first mitigating the altered structural conditions may result in unacceptable effects (Fule and others 1997). In some areas, several prescribed fires may be needed to completely restore fuel load and forest structure before natural fire regimes can be returned. This case study presents evidence that forest structure restoration using prescribed fire is possible in at least some forests where structure has been altered. Use of fire in forest restoration should be investigated in other areas where mechanical forest restoration is inappropriate or impractical, such as parks and wilderness areas.

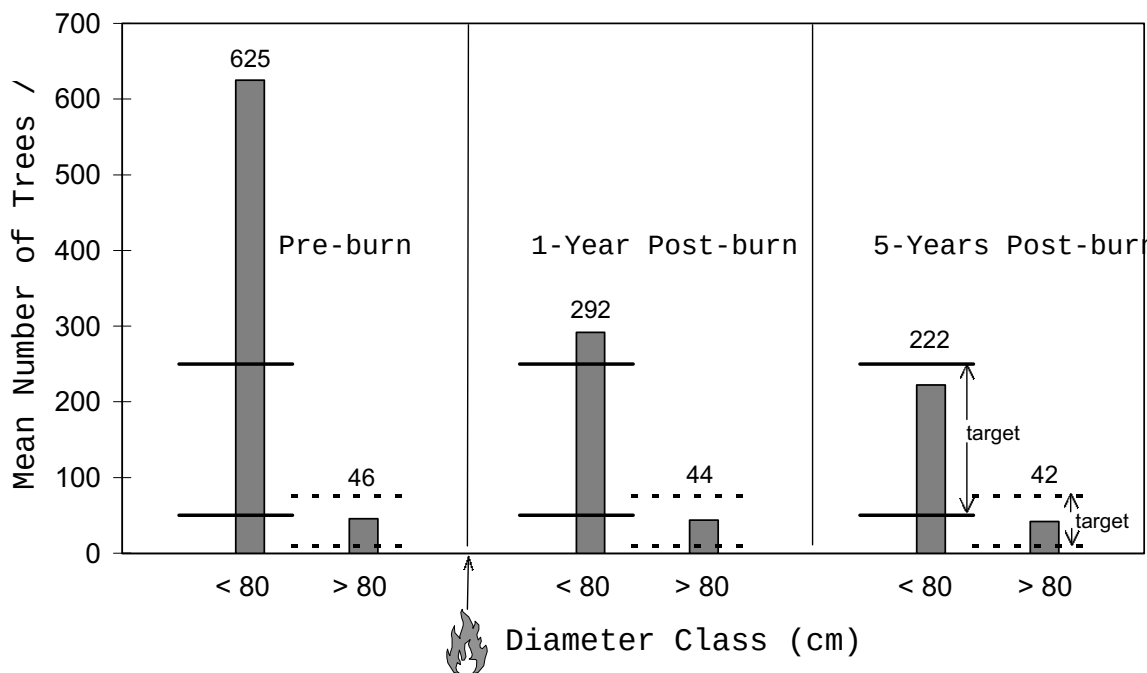


Figure 1—Stand density (all species combined) by diameter class in the giant sequoia-mixed conifer forest (n = 27 plots) pre-burn, and 1- and 5-years after prescribed fire. The target range for trees <80 cm is indicated by solid lines, and the target range for trees >80 cm is indicated by dashed lines.

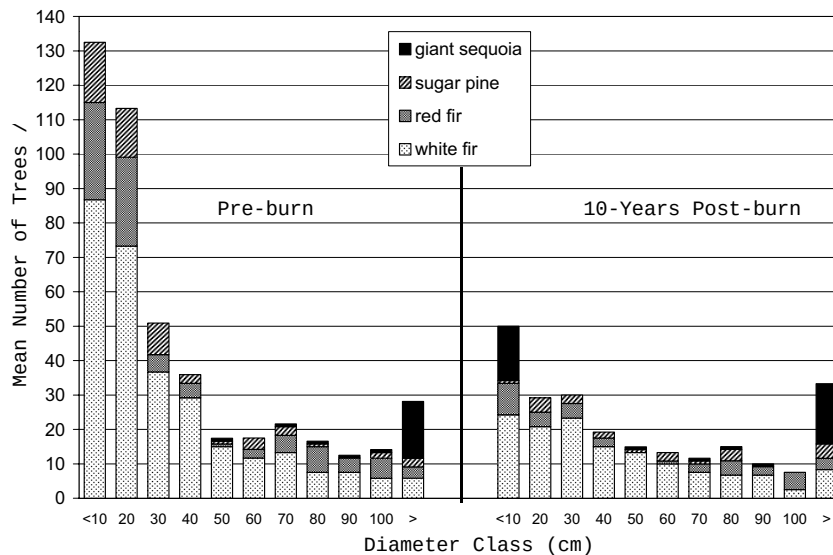


Figure 2—Pre-burn and 10-year post-burn stand density, by species and diameter class, in the giant sequoia-mixed conifer forest (n = 12 plots). Note the increase in giant sequoia in the smallest diameter class 10-years post-burn.

References

- Albini, F. A. 1976. Estimating wildfire behavior and effects. Gen. Tech. Rep. INT-30. Ogden, Utah: Intermountain Forest and Range Experiment Station, U.S. Department of Agriculture, Forest Service.
- Bonnicksen, T. M.; Stone, E. C. 1978. An analysis of vegetation management to restore the structure and function of presettlement giant sequoia – mixed conifer forest mosaics. Contract report to the U. S. National Park Service, Sequoia and Kings Canyon National Parks, California.
- Bonnicksen, T. M.; Stone, E. C. 1982. Reconstruction of a presettlement giant sequoia—mixed conifer forest community using the aggregation approach. *Ecology* 63:1134-1148.
- Bonnicksen, T. M.; Stone, E. C. 1985. Restoring naturalness to national parks. *Environmental Management* 9:479-486.
- Finney, M.; Stephenson, N.L. 1999. Unpublished data on giant sequoia-mixed conifer forest age structure. On file at U.S. Geological Survey, Western Ecological Research Center, Sequoia and Kings Canyon Field Station. Three Rivers, CA.
- Fule, P. Z.; Covington, W. W.; Moore, M. M. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. *Ecological Applications* 7:895-908.
- Graumlich, L. J. 1993. A 1,000-year record of temperature and precipitation in the Sierra Nevada. *Quaternary Research* 39:249-255.
- Hardy, C. C.; Arno, S. F., eds. 1996. The use of fire in forest restoration. Gen. Tech. Rep. INT-GTR-341. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 86 p.
- Keifer, M. 1998. Fuel load and tree density changes following prescribed fire: the first 14 years of fire effects monitoring. In: Brennan, Leonard A.; Pruden, Teresa L., editors. *Proceedings of the Tall Timbers fire ecology conference, No. 20. Fire in ecosystem management: shifting the paradigm from suppression to prescription*. Tallahassee, Florida: Tall Timbers Research Station.
- Keifer, M.; Manley, J. in press. Beyond initial fuel reduction in the giant sequoia-mixed conifer forest: Where do we go from here? In: *proceedings, Fire in California Ecosystems: Integrating Ecology, Prevention, and Management*, November 17-20, 1997, Bahia Hotel, San Diego, CA.
- Kilgore, B. M. 1972. Fire's role in a sequoia forest. *Naturalist* 23(1):26-37.
- Kilgore, B. M.; Taylor, D. 1979. Fire history of a sequoia-mixed conifer forest. *Ecology* 60(1):129-142.
- Lewis, H.T. 1973. Patterns of Indian burning in California. *Ecology and Ethnohistory*. Ballena Press. Anthropological Monographs, Pomona #1. 130 p.
- Mutch, L. S.; Parsons, D. J. 1998. Mixed conifer forest mortality and establishment before and after fire in Sequoia National Park, California. *Forest Science* 44:341-355.
- Parsons, D. J. 1978. Fire and fuel accumulation in a giant sequoia forest. *Journal of Forestry* 76:104-105.
- Stephenson, N. L. 1994. Long-term dynamics of giant sequoia populations: Implications for managing a pioneer species. Pages 56-63 in P. S. Aune, technical coordinator. *Proceedings of the Symposium on giant sequoias: their place in the ecosystem and society*, 23-25 June 1992, Visalia, California. USDA Forest Service General Technical Report PSW-151.
- Stephenson, N. L. 1996. Ecology and management of giant sequoia groves. Pages 1431-1467 in *Sierra Nevada Ecosystem Project, Final Report to Congress, vol. II, Assessments and scientific basis for management options*. Centers for Water and Wildland Resources, University of California, Davis, CA.
- Stephenson, N. L. 1999. Reference conditions for giant sequoia forest restoration: structure, process, and precision. *Ecological Applications* 9: 1253-1265.
- Stephenson, N. L. 1999. Unpublished data on giant sequoia demographic models. On file at U.S. Geological Survey, Western Ecological Research Center, Sequoia and Kings Canyon Field Station. Three Rivers, CA.
- Swetnam, T. W. 1993. Fire history and climate change in giant sequoia groves. *Science* 262: 885-889.
- U.S. Department of Interior, National Park Service. 1992a. Sequoia and Kings Canyon National Parks fire management plan. Available from Sequoia and Kings Canyon National Parks. Three Rivers, CA.
- U.S. Department of Interior, National Park Service. 1992b. Western Region fire monitoring handbook. Available from Pacific West Regional Office, San Francisco, CA.
- Vankat, J. L.; Major, J. 1978. Vegetation changes in Sequoia National Park, California. *Journal of Biogeography* 5:377-402.
- van Wagtenonk, J. W. 1985. Fire suppression effects on fuels and succession in short-fire-interval wilderness ecosystems. Pages 119-126 in J. E. Lotan, B. M. Kilgore, W. C. Fischer, and R. W. Mutch, editors. *Proceedings – symposium and workshop on wilderness fire*, 15-18 November 1983, Missoula, MT. USDA Forest Service General Technical Report INT-182.

Fire-Climate Interactions in the Selway-Bitterroot Wilderness Area

Kurt F. Kipfmueller
Thomas W. Swetnam

Abstract—Tree-ring reconstructed summer drought was examined in relation to the occurrence of 15 fires in the Selway-Bitterroot Wilderness Area (SBW). The ten largest fire years between 1880 and 1995 were selected from historical fire atlas data; five additional fire years were selected from a fire history completed in a subalpine forest within the SBW. Results of the analysis indicate summers during the fire year were significantly ($p < 0.001$) drier than average conditions. The summer preceding the fire year tended to be drier than average, but results were not statistically significant ($p > 0.05$). A significant ($p < 0.05$) wet year occurred four years prior to fire occurrence in the SBW. Further research which examines fire-climate interactions differentiated by forest type may provide an improved understanding of the dynamics between fire and climate.

In remote mountain areas where pre-20th century human occupation was limited and transient, climate variability was the dominant influence on interannual to centennial-scale fire and forest dynamics. Although the short-term weather (i.e., daily to monthly) patterns associated with large fire events in western forests have been well studied, longer-term climate patterns (i.e., seasonal to centennial) are less well understood. Interest in understanding the longer-term associations between climate and fire has increased in recent years due to concern over current and future changes in climate and fire regimes. An understanding of the interannual relationships between fire and climate could provide resource managers advance information to plan and implement mitigation efforts such as prescribed fire or could be used to guide decisions on the allowance of fires to burn under certain climate and weather conditions.

In some areas of the western U.S., the interannual relationships relating fire to interannual climate characteristics are well established. For example, it is well known that years in which large areas burn are often drier than normal. However, wetter than average winter-spring conditions are often present several years in advance of large fire years in ponderosa pine (*Pinus ponderosa* Law.) forests of the Southwestern United States (Baisan and Swetnam 1990; Swetnam and Betancourt 1998), and in the grasslands of the Great

Basin (Knapp 1995). Wetter than average antecedent conditions result in an increase in fine fuels which readily burn during subsequent dry years (Baisan and Swetnam 1990; Swetnam and Betancourt 1998). In the coniferous forests of the Northern Rocky Mountains these interannual relationships have not been intensively explored. Intuitively, large fire occurrence (often examined through the use of historical data and modern records) in the Northern Rockies is related to short-term (seasonal or monthly) intervals of drier than average conditions (Barrett and others 1997). Low elevation forests in the Northern Rocky Mountains may have similar antecedent climate relationships to forests in the southwestern United States. In addition, upper elevation forests in the Northern Rocky Mountains may require longer periods of dry weather to dry accumulated fuels sufficiently to support spreading fires. For example, upper elevation forests might be expected to burn primarily during relatively drier years than lower elevation forests, or during a sequence of two or more drier than average seasons and years.

This research explores the relationship of interannual climate variability on fire occurrence in the Selway-Bitterroot Wilderness Area (SBW) on the border of Idaho and Montana (fig. 1). The effect of summer drought, using a

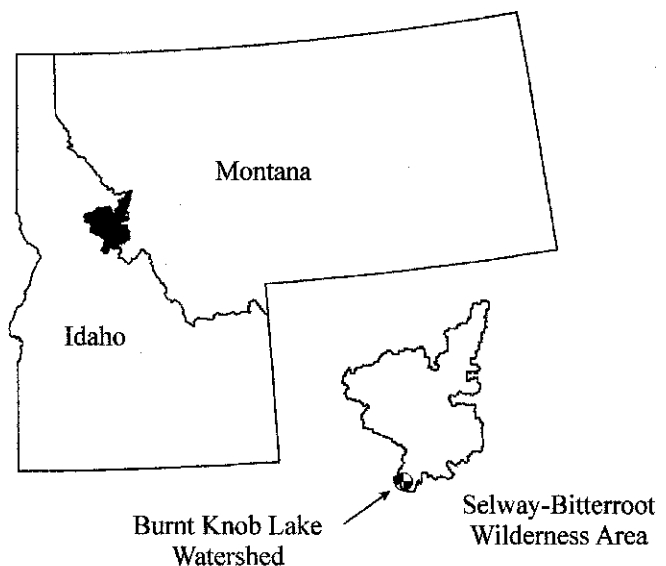


Figure 1—The location of the Selway-Bitterroot Wilderness Area on the border of Idaho and Montana in the northern Rocky Mountains. Burnt Knob Lake watershed is located in the southern portion of the Selway-Bitterroot Wilderness Area and is representative of subalpine forests of the region.

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tree-ring reconstructed summer Palmer Drought Severity Index (Cook and others 1999) is examined using superposed epoch analysis to identify important relationships between climate during the year of fire occurrence and interactions between fire and antecedent climate conditions. Fire-climate relationships are examined using fire years selected from the modern record as well as fire years identified in a crossdated fire history from a subalpine watershed in the SBW.

Study Area

The SBW, located on the border of Montana and Idaho (fig. 1), is composed of complex topography and a diversity of forest habitat types. At 547,370 ha (1,352,000 acres), the SBW is the third largest wilderness in the coterminous United States. Ponderosa pine-Douglas-fir (*Pinus Ponderosa* Laws.-*Pseudotsuga menziesii* Mirb.) habitats characterize xeric lower forest zones, with relatively mesic sites occupied by western redcedar (*Thuja plicata* Donn.) communities. Middle elevations of the SBW are composed of mixed-conifer forests with varying compositions of lodgepole pine (*Pinus contorta* var. *latifolia* Dougl.), western larch (*Larix occidentalis* Nutt.), subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.), grand fir (*Abies grandis* (Dougl.) Lindl.), and Engelmann spruce (*Picea engelmannii* Parry). Long-lived whitebark pine (*Pinus albicaulis* Engelm.) and subalpine larch (*Larix lyallii* Parl.) persist at the highest elevations and on more extreme sites (Arno and Habeck 1972). Wildfires were consistently and effectively suppressed for most of the 20th century in and around the SBW. Since 1979, however, lightning-ignited fires have been allowed to burn within prescribed conditions in the SBW (Barrett and Arno 1991; Brown and others 1994).

The climate of the SBW varies from an inland-maritime climate in its northwestern areas to a somewhat drier continental climate in the southeast (Finklin 1983). The warmest and driest months are July and August. The fire

season lasts from mid-June through late September, with peak lightning activity occurring in July and August (Finklin 1983). Fires ignite in the SBW throughout the summer, but historical records indicate most land area is burned later in the summer months when fuel conditions are driest.

Methods

A total of fifteen fire years were selected in the SBW area to examine fire-climate relationships (table 1). Ten fire years representing the largest areas burned in the 20th century were selected using historical fire atlas data (1880-1995). Five additional fire years (1709, 1719, 1729, 1741 & 1883) were selected from a crossdated fire history of a subalpine forest in the Burnt Knob Lake watershed (fig. 1) to obtain a longer time series of fire occurrence. These fire years were selected based on sample replication and the existence of associated forest stands based on age-class analysis.

A 279-year timeseries (1700-1978) of Palmer Drought Severity Indices (PDSI) reconstructed from tree-ring records was obtained from the NOAA Paleoclimatology Program (Cook and others 1999; data available on the world wide web at <http://ngdc.noaa.gov/paleo/drought.html>) to investigate relationships between large fire events and drought. PDSI is an estimate of the departure of soil moisture relative to average conditions (Palmer 1965). PDSI incorporates temperature and precipitation parameters as well as evapotranspiration and soil characteristics. Cook and others (1999) developed a systematic grid of tree-ring reconstructed PDSI to investigate the spatial characteristics of summer drought in the United States. Their grid consists of 155 grid points at a resolution of 2° x 3°. Station data used to reconstruct PDSI was selected using a 150 km search radius around each grid point. Tree-ring chronologies (consisting primarily of moisture sensitive ponderosa pine and Douglas-fir) were selected using a rule set developed to collect a minimum number of suitable chronologies at a minimal distance (see Cook and others, 1999 for more detail). Using regression techniques,

Table 1—Fire years selected for fire-climate analysis and reconstructed summer PDSI values during the fire year, and at one and four year lags. Reconstructed summer PDSI from Cook et al. 1999. Fire years with (*) are years selected from the crossdated fire history in Burnt Knob Lake watershed.

Fire event year	PDSI during fire year	PDSI Lag 1	PDSI Lag 4
1709*	0.84	-2.60	0.00
1719*	-0.58	-3.51	2.97
1729*	-2.31	-0.53	0.64
1741*	-1.99	-0.31	1.60
1883*	-0.44	-0.21	2.67
1889	-3.29	0.53	2.83
1890	-2.20	-3.29	-0.28
1895	-2.06	2.41	0.84
1910	-0.37	0.71	0.63
1919	-1.97	-2.38	1.80
1929	-1.28	-0.07	0.55
1934	-3.08	-2.30	-1.15
1979	-1.43	3.14	0.91
1987	-2.2	-0.86	3.11
1988	-3.54	-2.2	3.98

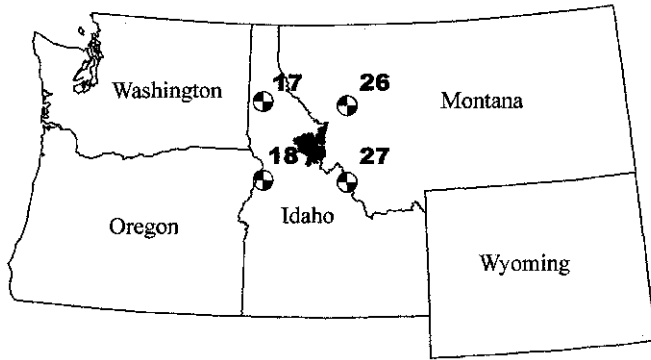


Figure 2—The location of the four grid points used to develop a regional index of tree-ring reconstructed summer PDSI (reconstructions from Cook and others, 1999, available online at <http://www.ngdc.noaa.gov/paleo/drought.html>).

Cook and others (1999) reconstructed summer (June-July-August) PDSI values back to at least A.D. 1700 in all cases, and in some instances to A.D. 1650. PDSI was not reconstructed beyond 1977, so PDSI values derived using instrumental data were used to extend the record to the present. For our analysis, we selected four grid points bracketing the SBW (fig. 2). PDSI values from these grid points were averaged to create a regional time series of summer PDSI from 1700-1995.

Superposed epoch analysis (SEA; Grissino-Mayer 1995; Lough and Fritts 1987; Swetnam 1993) was used to determine the relationship between fire occurrence and antecedent climate conditions. SEA computes the mean PDSI from the tree-ring reconstructed PDSI for the fifteen fire years selected. Mean PDSI was also calculated for each of the five years (years -1 to -5) prior to the selected fire years.

Superposed epoch analysis is a technique that characterizes one parameter (i.e., the fire year) in relation to another (i.e., reconstructed summer PDSI) (Grissino-Mayer 1995). SEA uses Monte Carlo simulations to estimate confidence intervals around the observed mean values.

Results

Reconstructed summer PDSI values indicate 14 of 15 fire years occurred during moderate drought (PDSI ≤ -1.0), though three years were very close to average conditions (fig. 3, table 1). Of the 15 fire events examined, 9 occurred during years when reconstructed summer PDSI was characterized as being an extreme drought (≤ -1.5). The frequency of reconstructed PDSI values less than -1.5 examined over 100 year intervals (e.g., 1700-1799, 1800-1899, and 1900-1995) exhibited some minor long-term differences between 1700-1995. Eighteen years between 1700 and 1799 exhibit reconstructed summer PDSI less than -1.5; the 1800-1899 period contained 13 years of reconstructed summer PDSI values less than -1.5 (table 2). Although covering the shortest time period, 19 years have PDSI values less than -1.5 between 1900 and 1995. Moderate drought (PDSI ≤ -1.0) indicated a similar pattern, with a higher frequency of drought during 1900 and 1995 (table 2).

The time series of regional summer PDSI in the SBW graphically illustrates the variability in drought from 1700-1995 (fig. 3). The minimum reconstructed value of summer PDSI was -4.52 which occurred in 1721. The wettest summer recorded by this data set occurred in 1750 (4.34). Palmer (1965) arbitrarily defines moderate drought conditions to occur when PDSI values fall below -1.0. Extreme drought conditions exist when PDSI values are ≤ -1.5 .

The results of superposed epoch analysis suggests that summer drought conditions, as measured by reconstructed summer PDSI are significantly ($p < 0.001$) related to the

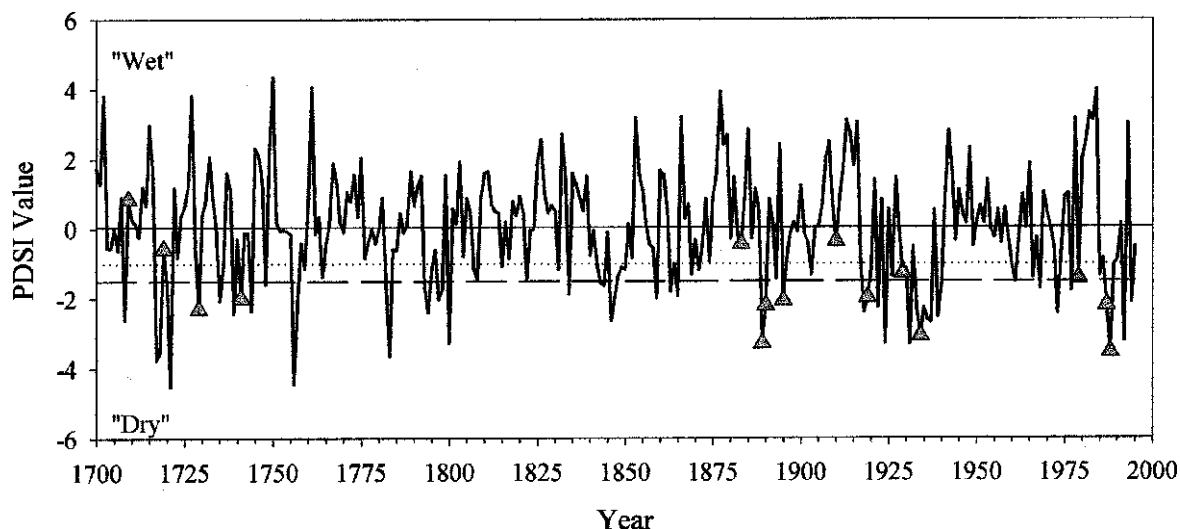


Figure 3—Time series of regional summer PDSI reconstructed from tree-rings. Fire years used in this analysis are plotted as triangles. The mean (near 0) is plotted as a solid line. Moderate drought (PDSI value ≤ -1.0) is plotted as a dotted line, and severe drought (≤ -1.5) is plotted as a dashed line. Drought definitions are as in Palmer (1965).

Table 2—Frequency of drought conditions by century in the SBW based on reconstructed July PDSI (Cook et al. 1999). Moderate and extreme drought values are based on Palmer (1965) drought classification.

	Time period		
	1700-1799	1800-1899	1900-1995
No. Moderate Drought Events (<-1.0)	23	25	30
No. Extreme Drought Events (<-1.5)	18	13	19

occurrence of large fires in the SBW (fig. 4). The year preceding fire occurrence also appears to be drier than long-term average conditions, but this result is not statistically significant ($p>0.05$). Statistically significant ($p<0.05$) wet conditions occur four years prior to the year of a fire event in the SBW (fig. 4).

Discussion

There are important relationships between inter-annual climate variability and large fire years in the SBW. Although emphasis has been placed on the importance of climate and short term weather conditions during the year of fire occurrence (e.g., Barrett and others 1997; Johnson and Larsen 1991; Johnson 1992), antecedent conditions up to four years preceding fire events may also be important. Statistically significant dry years during the fire years selected in the SBW (fig. 3) support the influence of seasonal (i.e., summer) drought leading to large areas burned. Drier-than-average conditions prior to the fire year, though not statistically significant, suggests extended drought might also play a role in the development of large fires in the SBW.

Although our results indicate important fire-climate relationships, other research in the Northern Rocky Mountains have not been conclusive. Barrett and others (1997) did not

find a strong link between drought and fire occurrence in their analysis of fire-climate relations in the Interior Columbia River Basin. However, their comparisons with drought and fire events are hampered by a lack of annual precision, precluding the analysis of interannual fire-climate interactions. Moreover, the tree-ring reconstructed climate parameters used by Barrett and others (1997) are geographically much further from the majority of their fire history sites than the fire history and summer drought data used here.

Dry summers preceding the fire event, though not statistically significant, were present before almost all fire event years selected from the Burnt Knob Lake watershed (table 1). However, fire years during the modern period (years 1895-1988) sometimes had wetter than average conditions one year prior to fire occurrence (table 1). The fire years selected from the modern record probably include areas of lower elevation forests containing ponderosa pine with grass as an understory component similar to southwestern ponderosa pine forests. Ponderosa pine forests in the SBW may respond similarly to wetter than average conditions, with buildups of fine fuels acting as an important mechanism to large fire occurrence during succeeding dry years. In contrast, upper elevation forests, such as those represented by Burnt Knob Lake (fire years 1709-1883), may require a longer period of dry weather for fuels to dry out appreciably to support large fires.

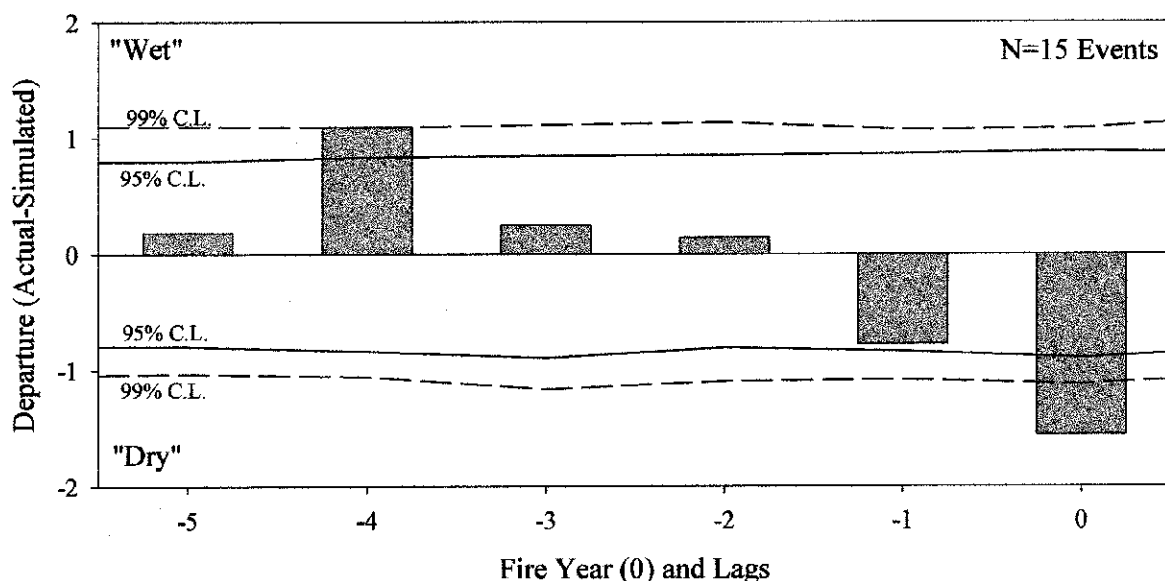


Figure 4—Results of superposed epoch analysis. Bars represent deviation from normal conditions based on 1000 simulations. The 99% and 95% confidence limits are also indicated.

The statistically significant wet year four years prior to fire occurrence may also be partially explained by the production of fuel (fig. 4). A subtle relationship exists between needle persistence and production with wet years. A wetter than average year may produce abundant needles, which are then shed during drought years, providing an important fuel source (Reich and others 1994; Swetnam and Baisan 1996). Thus, a wet year four years prior to fire occurrence may be related to the production of fine fuels, in a manner similar to the earlier discussion concerning the one year lag. The incorporation of additional fire events in the analyses may help to highlight these important lag effects. In addition, analyses which focus on fires occurring in specific vegetation types may highlight different fire-climate relationships between forest types.

The frequency of drought may have important impacts on fire regimes at long time scales. Several studies in the boreal forests of northern Canada suggest the frequency of drought has diminished following the end of the Little Ice Age, resulting in a decrease in area burned over the 20th century (Bergeron and Archambault 1993; Johnson 1992). However, Balling and others (1992) have found a 20th century trend of increasing drought conditions related to increasing temperatures during the fire season, coupled with a reduction of precipitation prior to the fire year have occurred in Yellowstone National Park. The frequency of drought, as measured by reconstructed summer PDSI, appears to have increased during the 20th century as compared to earlier time periods (table 2). The number of summers experiencing moderate drought (PDSI values ≤ -1.0) has remained relatively constant from 1700 to 1899, but increases somewhat between 1900 and 1995. Increases in the frequency of drought occurrence, coupled with unnatural fuel buildups due to fire suppression, may lead to more severe stand replacement fire events in the future.

There are several important limitations of this analysis. First, there is a mismatch in scales of our fire history data collected in a small area from the Burnt Knob lake watershed, and the broad scale data from the 20th century. The large fire events observed in the Burnt Knob Lake area may be more influenced by local scale factors such as localized fuel characteristics, topography, human set fires, or short-term, local weather patterns. These fires may have occurred during a short period (one month) of dry weather which is not accurately portrayed by PDSI reconstructed over an entire summer season.

Although this analysis suggests there are important fire-climate relationships for the SBW, these results could be further strengthened if forest types were delimited for study. For example, there is a moderate amount of ponderosa pine-Douglas-fir forest in the SBW which has a very different fire regime than that of higher elevation forests of the area (Arno 1980). It is plausible that the relationship between PDSI and large fire years would be different in these two habitat types. We suspect that ponderosa pine-Douglas-fir forests in the SBW may have similar PDSI-fire relationships to ponderosa pine forests in the Southwest. That is, prior wet years may be important to large fire occurrence due to the production of fine fuels which contribute to fire spread during subsequent dry years. However, consecutive dry years may prove to be more important in subalpine forest environments

where fuel conditions rather than abundance may be important mechanisms of large fire events (Agee 1993). To more effectively evaluate the relationship between drought conditions and fire, long high resolution data sets of fire history need to be developed over a broad region.

Monthly values of PDSI often contain high levels of autocorrelation (0.6-0.8). Because the calculation of PDSI uses prior values of PDSI to determine a current PDSI value, low values are often related to low values from a prior month (Alley 1984). Therefore, even if a summer is very dry, a wet spring prior to the fire season may have the effect of masking the dry summer PDSI somewhat. This limitation may be overcome using the Palmer Hydrologic Drought Index (PHDI) which removes the effect of prior values and is often thought of as a "real time" measure of drought (Alley 1984).

One problem identified by Cook and others (1999), which affects the strength of the fire-climate relationships identified here, is the quality of the PDSI reconstruction in mountainous landscapes. They concede that a lack of meteorological station data in many of these areas, coupled with a limited number of crossdated tree-ring chronologies in the northern Rockies, results in weaker reconstructions of PDSI for these areas. In fact, the weakest PDSI reconstructions, as identified by Cook and others (1999), occur in the mountains of Montana and along the front range of the Rockies from about central Colorado northward. In addition, a large search area was often necessary to locate a suitable number of tree-ring chronologies fitting the criteria and rule set for the reconstruction. This search radius sometimes exceeds 350 km. For this reason regional correlations among grid point reconstructions will be enhanced because the same tree-ring sites are being used to reconstruct PDSI for several grid points. Though the relationship between PDSI and large fire occurrence may not be perfect, it does identify important interannual and antecedent climate relationships with fire that can only be resolved using data with annual resolution.

Conclusions

This research suggests important relationships exist between fire and the onset of drought conditions during large fire years. Antecedent conditions may also play an important role in large fire occurrence in the forests of the SBW. Antecedent wet conditions may increase the production of fine fuels necessary to support large fires in low elevation forests. Although not statistically significant, drier than average conditions one year prior to fire occurrence may be important to the development of large fires in subalpine forests of the SBW. Further research which examines the relationship fire and climate needs to focus on individual forest types. In addition, annually resolved regional scale fire histories spanning multiple centuries will help better understand the role of climate on the fire regime of forests in the SBW.

Acknowledgment

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References

- Agee, J. K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Washington, D.C. 493 pp.
- Alley, W. M. 1984. The Palmer Drought Severity Index: Limitations and assumptions. *Journal of Climate & Applied Meteorology*, 23: 1100-1109.
- Arno, S. F. 1980. Forest fire history in the Northern Rockies. *Journal of Forestry*, 78: 450-465.
- Arno, S. F. and Habeck, J.R. 1972. Ecology of alpine larch (*Larix lyallii* Parl.) in the Pacific Northwest. *Ecological Monographs*, 42: 417-450.
- Baisan, C. H., and Swetnam, T. W. 1990. Fire history on a desert mountain range: Rincon Mountain Wilderness, Arizona, U.S.A. *Canadian Journal of Forest Research*, 20: 1559-1569.
- Balling, R. C., J. R. Meyer, G. A., and Wells, S. G. 1992. Climate change in Yellowstone National Park: Is the drought-related risk of wildfires increasing? *Climatic Change*, 22: 35-45.
- Barrett, S. W. and Arno, S. F. 1991. Classifying fire regimes and defining their topographic controls in the Selway-Bitterroot Wilderness. A paper presented at the 11th conference on Fire and Forest Meteorology, Missoula, MT.
- Barrett, S. W., Arno, S. F., Menakis, J. P. 1997. Fire episodes in the inland Northwest (1540-1940) based on fire history data. INT-GTR-370, 17 pp.
- Bergeron, Y. and Archambault, S. 1993. Decreasing frequency of forest fires in the southern boreal zone of Quebec and its relation to global warming since the end of the "Little Ice Age". *The Holocene*, 3: 255-259.
- Brown, J. K., Arno, S. F., Barrett, S. W., and Menakis, J. P. 1994. Comparing the prescribed natural fire program with presettlement fires in the Selway-Bitterroot Wilderness. *International Journal of Wildland Fire*, 4: 157-168.
- Cook, E. R., Meko, D. M., Stahle, D. W. and Cleaveland, M. K. 1999. Drought reconstructions for the continental United States. *Journal of Climate*, 12: 1145-1162.
- Finklin, A. I. 1983. Weather and climate of the Selway-Bitterroot Wilderness. University of Idaho Press, Moscow, ID. 144 pp.
- Grissino-Mayer, H. D. 1995. Tree-ring reconstructions of climate and fire history at El Malpais National Monument, New Mexico. Ph.D. Dissertation, University of Arizona, Tucson.
- Johnson, E. A. 1992. Fire and Vegetation Dynamics: Studies from the North American boreal forest. Cambridge University Press, 129 pp.
- Johnson, E. A. and Larsen, C. P. S. 1991. Climatically induced change in fire frequency in the southern Canadian Rockies. *Ecology*, 72: 194-201.
- Knapp, P. A. 1995. Intermountain west lightning-caused fires: Climatic predictors of area burned. *Journal of Range Management*, 48: 85-91.
- Lough, J. M. and Fritts, H. C. 1987. An assessment of the possible effects of volcanic eruptions on North American climate using tree-ring data, 1602-1900 A.D. *Climate Change*, 10: 219-239.
- Palmer, W. C. 1965. Meteorologic drought. U.S. Dept. of Commerce Weather Bureau, Washington, D.C., Res. Pap. No. 45.
- Reich, P. B., Koike, T., Gower, S. T., and Schoettle, A. W. 1994. Causes and consequences of variation in conifer leaf life-span. In: Smith, W. K. and Hinckley, T. M., eds. *Ecophysiology of Coniferous Forests*; Boston, MA: Academic Press: 225-254.
- Swetnam, T. W. 1993. Fire history and climatic change in giant sequoia groves. *Science*, 262: 885-889.
- Swetnam, T. W., and Baisan, C. H. 1996. Historical fire regime patterns in the Southwestern United States since A.D. 1700. In: Allen, C.D., ed. *Fire Effects in Southwestern Forests: Proceedings of the second La Mesa fire symposium*, RM-GTR-286, 11-32.
- Swetnam, T. W., and Betancourt J. L. 1998. Mesoscale disturbance and ecological response to decadal climate variability in the American Southwest. *Journal of Climate*, 11: 3128-3147.
- Touchan, R., Allen, C. D., and Swetnam, T. W. 1996. Fire history and climatic patterns in ponderosa pine and mixed- conifer forests of the Jemez Mountains, northern New Mexico. *In Fire Effects in Southwestern Forests: Proceedings of the second La Mesa fire symposium* (C.D. Allen, ed.). RM-GTR-286, 216 pp.

The Challenge of Restoring Natural Fire to Wilderness

David J. Parsons

Abstract—Despite clear legislative and policy direction to preserve natural conditions in wilderness, the maintenance of fire as a natural process has proven to be a significant challenge to federal land managers. As of 1998, only 88 of the 596 designated wilderness areas in the United States, excluding Alaska, had approved fire plans that allow some natural ignitions to burn; and even those areas with active natural fire programs continue to suppress many natural ignitions. As a result, none of the four federal wilderness management agencies have been able to restore fire to a level that even approaches pre-settlement fire regimes. Although prescribed fire has been utilized in some areas as a means to compensate for the lack of natural fire, it has been questioned as an appropriate wilderness management tool and is prohibited for most uses in Forest Service wilderness. The questions must be asked whether it is practical to expect restoration of natural fire regimes in wilderness and if they cannot be restored, what are the options and implications for wilderness resources and values?

The restoration of natural fire to wilderness poses significant challenges for federal wilderness management agencies. Following nearly a century of attempting to exclude fire from wilderness, land managers are now struggling with how best to restore fire as a natural ecological process (Christensen 1995; Czech 1996; Kilgore 1987). Despite abundant evidence of the importance of fire as a natural process, and legislative and policy direction to preserve natural conditions (including the process of fire) in wilderness, fire suppression remains the dominant wilderness fire management strategy (Parsons and Landres 1998). Administrative, political and practical constraints (Botti and Nichols 1995) result in suppression of most natural ignitions. If wilderness is to truly be preserved in its “natural condition,” ways must be found to overcome these constraints and to allow natural ignitions to burn significantly larger areas. The most practical alternatives to increased application of natural fire in wilderness include substitution of management-ignited prescribed fire as a surrogate for natural fire, or acceptance of the inevitable divergence of fire-adapted ecosystems from their historic character because of fire suppression. Prescribed fire is considered by some to be inappropriate manipulation of wilderness, yet, continued suppression can be expected, in many cases, to increase levels and homogeneity of hazardous fuels, change successional patterns and increase the threat of wildfire to surrounding areas (Arno and Brown 1991; Christensen 1995).

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This paper briefly reviews understanding of the natural role of fire in wilderness ecosystems and the evolution of wilderness fire management policy and programs. I review the status of efforts to restore fire to wilderness by the four federal wilderness management agencies, including accomplishments to date, and then revisit the significant issues and options that face wilderness fire management as it moves into the 21st Century. Specific attention is given to assessing impediments to the expanded application of fire in wilderness and to the consequences of future wilderness fire management choices.

Wilderness Fire and the Evolution of Wilderness Fire Management

When the first wilderness preserves were designated over a century ago, management emphasized protection of what was thought to be pristine, or natural, ecosystems. Disturbances such as fire and insects were viewed as undesirable, preventing forests from attaining or maintaining their natural climax state. Fire, in particular, was considered a destroyer of wilderness resources and values, including animals, vegetation and scenery (Christensen 1995). The elimination of fire became a primary goal of wilderness and park management. This approach was consistent with the understanding of ecosystems as static entities which characterized ecological thinking of the times (Botkin 1990).

Yet, even during the height of fire suppression, informed individuals warned of the dire consequences of continuing such practices (Chapman 1912; Weaver 1943). It is particularly interesting to note the early recognition of the importance of fire in natural ecosystems by members of the Leopold family, probably the pre-eminent conservation family in American history. Aldo Leopold, in laying out plans in the early 1920s for the Gila Wilderness in New Mexico, recognized the long history and ecological role of fire in the area (Meine 1988). In 1957, at the Fifth Biennial Wilderness Conference, Starker Leopold commented that fire exclusion was the “one striking exception to the trend toward naturalness in park preservation” and that he was “convinced that ground fires some day will be reinstated in the regimen of natural factors permitted to maintain the parks in something resembling a virgin state.” This discussion was not well received by National Park Service staff in attendance (Rydell 1998). Two years later, in a discussion at the Sixth Biennial Wilderness Conference, Luna Leopold asked penetrating questions about the need for controlled burning “to maintain the environment” in management of the proposed wilderness lands (Brower 1960). And then, in 1963, Starker Leopold and others (1963) recommended to the Secretary of Interior that the restoration of fire must be an important part of national park management.

Despite such early awareness of the importance of fire as a natural process, when the United States Congress passed the 1964 Wilderness Act, replete with abundant references to the preservation of “natural” conditions and the importance of maintaining “the forces of nature,” the only reference to fire was in relation to measures necessary for the “control of fire.”

Gradually, we have come to understand that fire, whether ignited by lightning or humans, has long played a critical role in the evolution and functioning of many natural ecosystems (Agee 1993; Kilgore 1987; Pyne 1982). The challenge has come in developing management strategies to restore and maintain fire as a natural part of these ecosystems. The incorporation of relatively recent understanding of the important role of pre-Europeans in using fire throughout the Americas has proven particularly challenging in efforts to define naturalness and develop management goals (Denevan 1992; McCann 1999). But, most importantly, we now realize that when we attempt to eliminate fire from fire-adapted ecosystems, we cause changes in succession and nutrient cycling, as well as buildups in flammable fuels which, in turn, threaten surrounding lands. By creating conditions that are outside the range of historic variability, we threaten our very goal of preserving natural systems.

Based on a growing recognition of the importance of fire in natural ecosystems, the National Park Service (NPS) and Forest Service (FS) began to rethink their wilderness fire policy in the 1960s. In 1968, the NPS revised its management policies to formally recognize fire as a natural process. Later that year, two lightning fires were allowed to burn in the high elevations of Kings Canyon National Park in California (Parsons and van Wagtendonk 1996). Soon, a number of national parks had operational prescribed natural fire programs, under which lightning ignitions were permitted to burn under predetermined conditions (Parsons and Botti 1996; van Wagtendonk 1991). In 1971, the Forest Service (FS) revised its policy of total suppression to permit natural fires in wilderness. The first lightning fires permitted to burn in FS wilderness were in the Selway-Bitterroot Wilderness of Montana in 1972 (Williams 1995). By 1988, 26 national parks and approximately 50 FS wilderness areas had operational prescribed natural fire programs (Parsons and Landres 1998; Williams 1995). In 1988, extensive fires, both lightning- and human-ignited, burned more than 3.7 million acres throughout the western United States. These fires, largely focused around Yellowstone National Park and the northern Rocky Mountains, had a significant and immediate effect on the wilderness fire programs of the federal agencies. There was an immediate suspension of all wilderness fire programs while a national review re-examined federal fire policy. Although it endorsed the major policy objectives, the review recommended significant changes in implementation strategies. Increased operational constraints resulting from the review limited the reestablishment of natural fire programs by limiting conditions under which lightning fires could be permitted to burn (Botti and Nichols 1995). Ten years later, the area of federal wilderness burned by natural ignitions had yet to reach pre-1988 levels (Parsons 1999).

Following a difficult fire season in 1994, another review of federal wildland fire policy and programs (USDI/USDA 1995) resulted in major revisions to wildland fire terminology, as

well as a renewed effort to standardize implementation procedures between the agencies. The most significant change influencing wilderness fire was the new emphasis on managing fire for resource benefits based on analysis of appropriate management responses (see Zimmerman and Bunnell in this proceedings for further discussion). As a result, what had been known as prescribed natural fires, became “wildland fires managed for resource benefits.”

Despite the increased emphasis on managing wilderness fire for resource benefits (including restoration of natural processes), none of the federal wilderness agencies have a fully successful wilderness fire management program. Even the more progressive FS and NPS programs have not permitted natural fire in many wilderness units, and even where they have, many fires continue to be suppressed (Parsons 1999). Outside of Alaska, where all of the agencies permit some lightning fires to burn in a limited suppression or confine/contain strategy, neither the Bureau of Land Management (BLM) nor the Fish and Wildlife Service (FWS) have yet to implement an operational natural fire program in wilderness. These situations have raised concerns about whether current wilderness fire programs are accomplishing enough to be worth the effort (Parsons and Landres 1998).

Current Status and Accomplishments

Despite recognition of the importance of fire as a natural process in the wilderness policy statements of all four federal wilderness management agencies, and the existence of approved fire management plans that permit lightning ignitions to burn in at least some units managed by all but the FWS, 508 of the 596 designated wilderness areas outside of Alaska were still in total suppression mode as of the 1998 fire season. In fact, as of 1998, only the NPS and the FS had permitted any lightning fires to burn in wilderness. Table 1 summarizes the status and accomplishments of the wilderness natural fire programs (excluding Alaska) of the four wilderness management agencies for 1995-1998. The average of 65,037 acres burned by natural fire in national forest wilderness from 1995-1997 (recent terminology changes and the lack of a centralized database for FS wilderness fire records made it impossible to report 1998 data in this paper) represented approximately 0.2% of all FS wilderness (outside of Alaska). The average of 11,439 acres burned by natural fires in national parks from 1995-1998 was equivalent to about 0.1% of all NPS wilderness; however, only 17 of the NPS areas and 4,858 of the acres reported were for parks designated as wilderness, making the percentage of NPS wilderness actually burned per year even lower.

The acreage burned by natural fire on FS wilderness in recent years exceeds that burned in most years prior to 1988 (Parsons 1999), but most of that acreage can be accounted for by a few exceptionally large fires (in Arizona and New Mexico in 1995 and 1997 and Oregon and Montana in 1996), some of which had to be suppressed when they escaped their prescribed boundaries and threatened nearby communities. Thus, it is not clear if recent accomplishments are truly indicative of a long-term trend of increasing acreage burned by natural fires. Most FS wildernesses, including many with

Table 1—Natural fire programs for the four federal wilderness management agencies, excluding Alaska, 1995–1998.

Agency	Total wilderness acres (x10 ⁶)	No. wilderness areas	No. approved natural fire plans	Ave. acres burned/yr
BLM	5.2	131	6	0
FWS	2.0	50	0	0
FS ^a	29.0	380	65	65,037
NPS	10.3	36	27 ^b	11,439 ^c

^aFS data for 1995–1997.

^bIncludes 10 nonwilderness parks.

^cOnly 4,858 acres of this is designated wilderness.

natural fire programs, have permitted few, if any, lightning fires to burn in recent years. Even in the Bob Marshall Wilderness in northwest Montana, with one of the most progressive natural fire programs, the average number of natural ignitions permitted to burn has dropped by over 50 percent, the average size of natural fires has dropped by 75 percent, and only 19 percent of all eligible lightning fire starts have been permitted to burn since 1988 (Eckert, personal communication). Some lightning fires have been permitted to burn on FS lands under a confine/contain strategy; however, the fact that such fires are classified as suppressed wildfires makes it impossible to incorporate them into an analysis of natural fire accomplishments (Parsons 1999).

Beginning in 1998 with implementation of a new Federal Wildland Fire Policy (Zimmerman and Bunnell, this proceedings), several significant changes influenced the Forest Service's wildland fire management program. These include the ability to allow natural ignitions to be managed for resource benefits, as well as a change in FS policy to allow use of suppression funds to manage wildland fire for resource benefits (only the Department of Interior had previously allowed such use). As a result, in 1998, significant more lightning fires on FS land in the northern Rocky Mountains were permitted to burn (managed for resource benefits) following analysis of the appropriate management response, as called for in the new Federal Wildland Fire Policy. This has caused Zimmerman and Bunnell to express considerable optimism regarding the future of the FS wilderness fire program under the new policy.

As of 1998, 27 national parks had approved fire management plans that permitted natural fires to burn. This included 17 of the 36 parks outside of Alaska with designated wilderness, as well as 10 nonwilderness parks (although parks like Glacier, Grand Canyon, and Voyageurs have never been congressionally designated as wilderness, they are managed as if they were). Acreage burned by natural fire in national parks in recent years has yet to approach pre-1988 levels (figure 1). Perhaps more significantly, a comparison of the NPS natural fire program for five-year periods before (1983–1987) and after (1994–1998) the 1988 Yellowstone fires shows a marked decrease in the mean number of natural fires per year, the number of parks with natural fires allowed to burn per year, the mean annual acres burned per year and the mean fire size (table 2). Specifically, the mean number of acres burned per year by natural fire has dropped from over 32,000 prior to 1988 to less than 11,000 in the most recent five-year period. Mean fire size has decreased from

209 acres to 137 acres. Since even the pre-1988 accomplishments were considered well below that required to approximate presettlement fire frequencies, these decreases are of considerable concern to the National Park Service. The lack of progress in expanding the NPS natural fire program has been largely attributed to the increased planning and management constraints following the 1988 Yellowstone fires and subsequent policy review (Botti and Nichols 1995; Parsons and Landres 1998).

Although the BLM approved its first Wilderness Management Plan that permitted natural fires to burn in 1990 (the Mount Trumbull and Mount Logan Wildernesses in northwestern Arizona), they have yet to permit a natural fire to burn. The FWS has yet to approve a program that permits the use of natural fire outside of Alaska (see Parsons and Landres 1998 for further discussion).

Ultimately, the most important question related to the restoration of natural fire in wilderness is how close do our management programs and accomplishments come to reestablishing “natural” fire regimes? To date, such comparisons are extremely limited. One reason for this is the lack of a consistent reporting process for wilderness fire accomplishments. For example, only the Forest Service distinguishes wilderness from nonwilderness fires. Such comparisons also require an understanding of fire history that is not available for many areas; where it is available, arbitrary decisions must be made about what time period is to be used as the baseline for comparison. The choice of any given time interval to represent the natural or target fire regime is unlikely to reflect the full range of historic variability (Swetnam

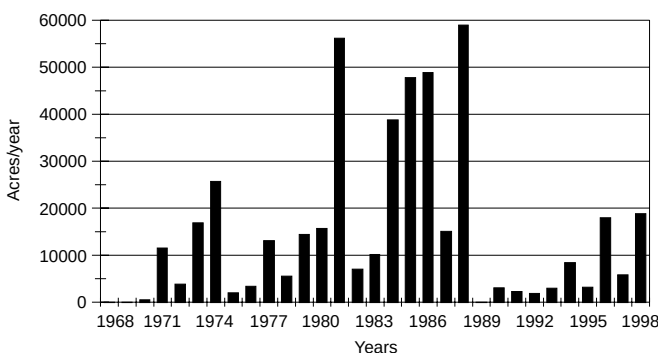


Figure 1—Natural fire acres burned per year for 1967–1998 on National Park Service lands. This data includes nonwilderness as well as wilderness parks.

Table 2—Comparison of National Park Service natural fire programs for 5-year periods before and after 1988.

	1983-1987	1994-1998
No. natural fire plans	26	27
Mean no. parks with fires	22	13
Mean no. fires per year	154	79
Mean annual acres burned	32,135	10,833
Mean fire size (acres)	209	137

1993). Despite these limitations, Brown and others (1994) have determined that the average annual area burned prior to the advent of fire suppression in the Selway-Bitterroot Wilderness in Idaho and Montana was 1.5 to 1.9 times greater than has burned during recent years. Analyses for several forest types in the Sierra Nevada of California also document that area burned under current management strategies is well below that required to approach pre-suppression fire frequencies (Parsons 1995; vanWagtendonk 1995). Recent work by Caprio and Graber (this proceedings) in Sequoia and Kings Canyon National Parks provides the most comprehensive comparison to date between accomplishments of current fire management programs (including natural fires, prescribed fires, and wildfires) and pre-settlement fire regimes for a number of vegetation types. They also found that accomplishments fall well below that required to restore presuppression fire frequencies. Although such data are limited, it is clear that recent accomplishments fall well below targets in all areas for which data have been evaluated.

In addition to the policy and administrative constraints that have limited the use of natural fire, even in the larger units, a number of other reasons explain why natural fire can never be expected to be allowed to burn in some wildernesses. These include the small size of many wilderness areas, resulting in many natural ignitions outside of wilderness being suppressed before they can burn into the area, the risk of fire escaping onto adjacent lands managed for other purposes, the threat of unnaturally intense fires causing unacceptable resource damage and the threat of smoke causing unacceptable impacts to surrounding areas. Together, such concerns raise serious questions about the potential for natural fire to ever be able to effectively restore natural fire regimes in many wilderness areas.

A Role for Prescribed Fire?

If natural ignitions are not going to be sufficient to restore desirable fire regimes in most wilderness areas, it will be necessary to look at potential consequences, as well as other options. Management-ignited prescribed fire, either as a supplement to or substitute for natural fire, has been used as a wilderness fire management tool by the Department of Interior agencies (BLM, FWS, NPS) for some time. In fact, the FWS relies almost entirely on prescribed fire to accomplish wilderness management objectives. Specific objectives of BLM and FWS prescribed fire programs include reduction of hazardous fuels, range improvement, wildlife habitat enhancement and restoration of natural fire regimes (Parsons 1999). National Park

Service policy approves the use of prescribed fire for the “protection and preservation of wilderness character and resources.” The use of management ignitions is widespread within the agency, including within designated wilderness (Botti and Nichols 1995). In tracing the history of prescribed fire in national parks, Botti and Nichols (1995) showed that the use of prescribed fire, in contrast to the natural fire program, continued to expand after the 1988 Yellowstone fires. In figure 2, I present the natural fire and prescribed fire data (mean acres burned per year) for the same 25 natural fire parks analyzed by Botti and Nichols (1995) for the most recent four-year period (1995-1998). Although the use of natural fire has yet to recover to pre-1988 levels (figure 1), the use of prescribed fire has continued to grow, well surpassing the acreage burned pre-1988, as well as that burned by natural fire. In addition, many parks that do not have natural fire programs now rely entirely on prescribed fire to accomplish wilderness fire objectives.

In contrast to wilderness managed by the Department of Interior agencies, Forest Service wilderness is subject to extremely limited prescribed fire. With the exception of the national forests of Florida, where the Chief of the Forest Service granted a 1995 blanket approval for use of prescribed fire for resource objectives, FS policy does not permit the use of prescribed fire in wilderness for purposes other than the reduction of unnatural buildups of fuel (Parsons 1999). Despite numerous calls for increased use of prescribed fire in FS wilderness (Brown 1992; Mutch 1995), and optimism that the new Federal Wildland Fire Policy will facilitate increased use of prescribed fire (Zimmerman and Bunnell, this proceedings), there continues to be considerable opposition both within and outside the agency to such a change. Perhaps of greatest concern is that the use of prescribed fire could become an accepted alternative to natural ignitions and, as such would soon become the dominant wilderness fire management strategy. Since prescribed

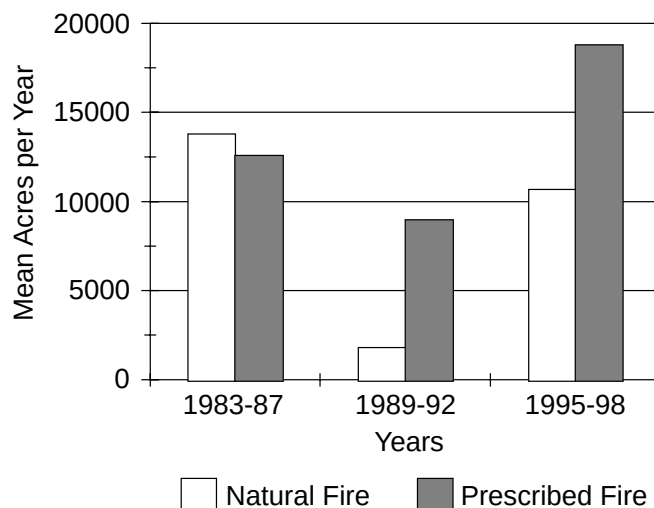


Figure 2—Mean acres burned per year, for three 4-year periods, by natural and prescribed fires in the 25 natural fire national parks analyzed by Botti and Nichols (1995). The time periods represent the period immediately before the 1998 Yellowstone fires (1983-1987), the period immediately following 1988 (1989-1992) and the most recent four years for which data are available (1995-1998).

fire is viewed by many as inappropriate intervention that detracts from the wild or untrammelled nature of wilderness, some feel that its use conflicts with the primary purposes of wilderness. Nickas (1998) exemplifies this attitude in stating "more troubling in many ways than fire suppression is the growing tendency toward utilizing management-ignited ("prescribed") fire" and "when managers light the match, fire ceases to be a natural force and instead becomes a manipulative tool." It seems clear that the use of prescribed fire in wilderness presents a fundamental dilemma that must be addressed before the future of wilderness fire management can be fully resolved.

If neither natural or prescribed fire proves sufficient to restore more natural fire regimes to wilderness, we must be prepared to consider the alternatives. There is little question that continued emphasis on fire suppression will ultimately lead to increasing numbers of unnaturally severe wildfires that threaten both wilderness and adjacent nonwilderness resources. Yet the only other option appears to be the use of mechanical manipulation to reduce unnatural vegetation, a practice sure to raise the ire of wilderness advocates as inappropriate and unnecessary. Fire, whether natural or prescribed, appears to be a preferred alternative to either continued suppression or mechanical manipulation.

Issues and Options

It is clear that fire is an important natural process in many wilderness areas and that if fire is not permitted to burn, those areas cannot be considered as truly natural. Yet, despite clear policy direction recognizing the importance of natural fire, suppression continues to be the dominant fire management strategy in most wilderness areas. Moreover, it is becoming clear that wilderness managers will probably never be able to allow enough natural fire to burn to restore even a semblance of natural fire regimes in most wilderness areas. This raises the dilemma of what to do if we are unable to significantly increase the use of natural fire. The principal options appear to be to either live with the consequences of continued suppression (shifts in vegetation and increasing hazardous fuel accumulations) or to consider prescribed fire or some other surrogate for natural fire (most likely, mechanical manipulation of vegetation). These options will result either in increasingly unnatural, although admittedly wild conditions or the use of manipulative management to restore and maintain some vision of what natural conditions should be.

Since the Wilderness Act calls for both wild (untrammelled or unmanipulated) and natural conditions this presents a significant dilemma for wilderness management (Aplet 1999; Cole 1996). For example, the use of prescribed fire may make the system more natural, but it will be at the cost of being less wild, or self-willed (Nickas 1998). Of course, we must remember that the current situation, where suppression dominates, is also a highly unnatural condition perpetuated by a different type of "management." The question of wilderness versus naturalness raises important philosophical and policy questions that have yet to be fully addressed (Aplet 1999). One option that has been raised by Cole (1996) is that different wilderness areas, or portions of wildernesses, could be managed for different purposes. In the case of fire, some areas might be managed to maintain natural fire regimes

through whatever means are necessary, while others could be managed to maximize wildness, recognizing that the system may become increasingly unnatural.

In addition to the fundamental policy issue raised above, there are a number of other issues that must be addressed in the struggle to restore natural fire to wilderness. These include the challenge of implementing the new Federal Wildland Fire Management Policy and the associated terminology changes. The concept of calling natural ignitions allowed to burn (previously called prescribed natural fires) wildland fire use for resource benefits is already presenting challenges for communication among and between fire and wilderness managers and the public (Dietrich 1999). Other changes mandated by the new fire policy are being addressed in new interagency implementation guides. Zimmerman and Bunnell (this proceedings) have shown reason to be optimistic that many of the changes will improve agency abilities to use natural fire in wilderness.

Another issue in need of immediate attention is the lack of standardized record keeping and reporting procedures for wilderness fire. At this time, only the NPS has a centralized database where records on both natural and prescribed fire can be obtained. Only the FS designates whether a fire burns in wilderness or not, but the lack of standardized record keeping and reporting procedures makes it particularly difficult to obtain FS fire records without contacting individual units. Consistent reporting is essential to evaluating the accomplishments and effectiveness of wilderness fire programs (Parsons and Landres 1998).

In many wilderness areas, natural fires simply cannot be permitted to burn. Reasons include being too small to contain a fire, location in areas where the primary ignition points are outside the boundaries, location within matrices of high value private lands where the risk of escape is too great, or location adjacent to areas particularly sensitive to air quality concerns. Although such reasons convinced the FS to permit expanded use of prescribed fire in the national forests of Florida, there has yet to be a systematic analysis of where natural fire programs have the most and least potential to be successful and, thus, where other options need to be considered. We do understand that the natural fire success stories in both the FS and NPS have been largely in a limited number of large, remote areas, characterized largely by low and mixed severity fire regimes. The challenge will be considerably more difficult when these approaches are applied to smaller areas, areas in proximity to private and other high value lands or areas characterized by infrequent, high severity fire regimes.

A need with which few disagree is that science must inform management choices. Although the ultimate choices as to what outcomes are most desirable must be made by those responsible for managing the areas, their decisions should be informed by the best available science. In the case of wilderness fire, specific roles for science include the need to compare accomplishments with our best understanding of presettlement fire regimes, improve understanding of the effects of varying fire intensities, frequencies and seasonality, improve predictive models and the ecological knowledge to drive those models and improve understanding of public acceptability of options. Most importantly, scientists need to assess the consequences of alternative future scenarios, including the effects on wilderness ecosystems and on

societal values. The most effective management decisions can be made only when they are informed by the best possible science.

Conclusions

Although the current policies of all four wilderness management agencies clearly recognize the importance of fire as a natural part of wilderness ecosystems, implementation of wilderness fire programs varies greatly between agencies and is far from what would be required to restore natural fire regimes. It can no longer be assumed that natural fire programs will be adequate to restore fire to wilderness. The various constraints under which wilderness and fire managers must work make it highly unlikely that natural fire can ever be fully restored to most wildernesses. And since continuing on the current course is only making matters worse by perpetuating the changes caused by fire suppression, it is clearly time to address management options and the consequences of those options. The fact that the alternatives identified to date all present problems of their own—philosophical, policy, and practical problems—presents a real challenge, but one that we cannot afford to postpone. We are falling farther behind each year. It is time to address the challenges and choose between the available options. The optimism evinced by Zimmerman and Bunnell (this proceedings) regarding the potential for the new Federal Wildland Fire and Management Policy to provide a mechanism to more fully achieve wilderness fire objectives is promising but must be more fully evaluated.

References

- Agee, J. K. 1993. *Fire ecology of Pacific Northwest Forests*. Washington, D.C.: Island Press. 493 p.
- Aplet, Gregory H. 1999. On the nature of wildness. *Denver University Law Review* 76(2):347-367.
- Arno, Stephen F.; Brown, James K. 1991. Overcoming the paradox in managing wildland fire. *Western Wildlands*. Spring 1991:40-46.
- Botkin, Daniel B. 1990. Discordant harmonies: a new ecology for the 21st Century. New York, NY: Oxford University Press. 241 p.
- Botti, Stephen J.; Nichols, Howard T. 1995. Availability of fire resources and funding for prescribed natural fire programs in the National Park Service. In: Brown, J. K., Mutch, R. W., Spoon, C. W., and Wakimoto, R. H., eds. *Proceedings: Symposium on fire in wilderness and park management*. 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 94-104.
- Brown, James K. 1992. A case for management ignitions in wilderness. *Fire Management Notes* 53-54:3-8.
- Brown, James K.; Arno, Stephen F.; Barrett, Stephen W.; Menakis, James P. 1994. Comparing the prescribed natural fire program with presettlement fires in the Selway-Bitterroot Wilderness. *International Journal of Wildland Fire* 4(3): 157-168.
- Brower, David. 1960. *The meaning of wilderness to science*. San Francisco, CA: Sierra Club. 129 p.
- Caprio, Anthony C.; Graber, David M. 2000. Returning fire to the mountains: can we successfully restore the ecological role of pre-European fire regimes to the Sierra Nevada? In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. *Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management*; 1999 May 23-27; Missoula, MT. *Proceedings RMRS-P-15-VOL-5*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Chapman, H. H. 1912. Forest fires and forestry in the southern states. *American Forests* 18:510-517.
- Christensen, Norman L. 1995. Fire and wilderness. *International Journal of Wilderness* 1(1):30-34.
- Czech, Brian. 1996. Challenges to establishing and implementing sound natural fire policy. *Renewable Resources Journal*. Summer 1996:14-19.
- Cole, David N. 1996. Ecological manipulation in wilderness: an emerging management dilemma. *International Journal of Wilderness* 2(1):15-19.
- Denevan, William M. 1992. The pristine myth: the landscape of the Americas in 1492. *Annals of the Association of American Geographers* 82(3):369-385.
- Dietrich, Michael J. 1999. Understanding fire policy. *Journal of Forestry* 97(2): Insert, p. 1.
- Eckert, C. 1998. [Personal communication]. July 14, Spotted Bear Ranger Station, MT: U.S. Department of Agriculture, Forest Service, Flathead National Forest, Spotted Bear Ranger District.
- Kilgore, Bruce M. 1987. The role of fire in wilderness: a state-of-knowledge review. In: Lucas, R.C., compiler. *Proceedings—national wilderness research conference: issues, state-of-knowledge, future directions*. 1985 July 23-26; Fort Collins, CO. Gen. Tech. Rep. INT-220. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 70-103.
- Leopold, A. S.; Cain, S.A.; Cottam, C.M.; Gabrielson, I.N.; Kimball, T.L. 1963. *Wildlife management in the national parks*. *Transactions of the North American wildlife and natural resources conference* 28:28-45.
- McCann, Joseph M. 1999. Before 1492: the making of the pre-Columbian landscape, Part I: the environment. *Ecological Restoration* 17(1&2):15-30.
- Meine, Curt. 1988. *Aldo Leopold, his life and work*. Madison, WI: The University of Wisconsin Press. 638 p.
- Mutch, Robert W. 1995. Prescribed fire in wilderness: how successful? In: Brown, J. K.; Mutch, R. W.; Spoon, C. W.; Wakimoto, R. H., eds. *Proceedings: Symposium on fire in wilderness and park management*. 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 38-41.
- Nickas, George. 1998. Wilderness fire. *Wilderness Watcher* 10(1):14-5.
- Parsons, David J. 1995. Restoring fire to giant sequoia groves: what have we learned in 25 years? In: Brown, J. K.; Mutch, R. W.; Spoon, C. W.; Wakimoto, R. H., eds. *Proceedings: Symposium on fire in wilderness and park management*; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 256-258.
- Parsons, David J. 2000. Restoration of natural fire to United States wilderness areas. In: Watson, Alan E.; Aplet, Greg H.; Hendee, John C., comps. *Personal, societal, and ecological values of wilderness: Sixth World Wilderness Congress proceedings on research, management, and allocation, volume II*; 1998 October 24-29; Bangalore, India. *Proc. RMRS-P-14*. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Parsons, David J.; Botti, Stephen J. 1996. Restoration of fire in national parks. In: Hardy, Colin; Arno, Stephen F., eds. *The use of fire in forest restoration*. Gen. Tech. Rep. INT-GTR-341. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 29-31.
- Parsons, David J.; van Wagendonk, Jan W. 1996. Fire research and management in the Sierra Nevada national parks. In: Halvorson, W. L.; Davis, G. E. eds. *Ecosystem management in the national parks*. Tucson, AZ: University of Arizona Press: 25-48.
- Parsons, David J.; Landres, Peter B. 1998. Restoring natural fire to wilderness: how are we doing? Pruden, Teresa L.; Brennan, Leonard A., eds. *Fire in ecosystem management: shifting the paradigm from suppression to prescription*. Tall Timbers Fire Ecology Conference Proceedings, No. 20. Tall Timbers Research Station, Tallahassee, FL: 366-373.
- Pyne, S. J. 1982. *Fire in America: a cultural history of wildland and rural fire*. Princeton, NJ: Princeton University Press. 654 p.
- Rydell, Kiki Leigh. 1998. A public face for science: A. Starker Leopold and the Leopold Report. *The George Wright Forum* 15(4):50-63.

- Swetnam, Thomas W. 1993. Fire history and climate change in giant sequoia groves. *Science* 262:885-889.
- U.S. Department of Interior/U.S. Department of Agriculture. 1995. Federal wildland fire management policy and program review. Final Report. Boise ID: National Interagency Fire Center. 45 p.
- vanWagtendonk, Jan W. 1991. The evolution of National Park Service fire policy. In: Nodvin, S.C.; Waldrop, T.A. eds. *Fire and the environment: ecological and cultural perspectives*; 1990 March 20-24; Knoxville, TN. USDA Forest Service Gen. Tech. Rep. SE-69:328-332.
- vanWagtendonk, Jan W. 1995. Large fires in wilderness areas. In: Brown, J. K.; Mutch, R. W.; Spoon, C. W.; Wakimoto, R. H., eds. *Proceedings: Symposium on fire in wilderness and park management*; 1993 March 30-April 1; Missoula, MT. Gen Tech. Rep. INT-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 113-116.
- Weaver, Harold. 1943. Fire as an ecological and silvicultural factor in the ponderosa pine region of the pacific slope. *Journal of Forestry* 41: 7-14.
- Williams, Jerry T. 1995. Managing risk in wilderness fire management. In: Brown, J. K.; Mutch, R. W.; Spoon, C. W.; Wakimoto, R. H., eds. *Proceedings: Symposium on fire in wilderness and park management*; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 22-23.
- Zimmerman, G. Thomas; Bunnell, David L. 2000. The Federal wildland fire policy: opportunities for wilderness fire management. In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. *Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management*; 1999 May 23-27; Missoula, MT. Proc. RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Twentieth-Century Fire Patterns in the Selway-Bitterroot Wilderness Area, Idaho/Montana, and the Gila/Aldo Leopold Wilderness Complex, New Mexico

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Tom Swetnam
Penelope Morgan

Abstract—Twentieth century fire patterns were analyzed for two large, disparate wilderness areas in the Rocky Mountains. Spatial and temporal patterns of fires were represented as GIS-based digital fire atlases compiled from archival Forest Service data. We find that spatial and temporal fire patterns are related to landscape features and changes in land use. The rate and extent of burning are interpreted in the context of changing fire management strategies in each wilderness area. This research provides contextual information to guide fire management in these (and similar) areas in the future and forms the basis for future research involving the empirical definition of fire regimes based on spatially explicit time-series of fire occurrence.

Large wilderness areas may be viewed as natural laboratories for studying patterns and processes in ecosystems virtually uninfluenced by human activity. They provide the opportunity to understand basic ecological principles (Parsons and Graber 1991) and to define benchmark or control areas for gauging how nonwilderness systems have been affected by human land use (Christensen and others 1996). A library of empirical studies of pattern-process interactions is crucial for understanding the relative effects of landscape patterns on ecosystem processes (Turner and others 1995), and wilderness research is a crucial component. The need for spatial and temporal research into fire regimes is particularly critical in light of predicted climate change (Price and Rind 1994; Running and Nemani 1991; Ryan 1991). This research is one of the few (if not the only) specific examples of using broad spatial databases and 20th century fire perimeter data in two disparate regions to determine relationships between landscape attributes and ecosystem processes. Comparison between two regions yields a measure of the generality of interpretations and allows observation of fire-landscape-climate relationships that may occur at regional scales.

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The objectives of our research were to characterize the interrelationships among fire, topography, and vegetation across two major Rocky Mountain Wilderness ecosystems: the Selway-Bitterroot Wilderness Area in Idaho and Montana and the Gila/Aldo Leopold Wilderness Complex in New Mexico. Comparing results between two distinct regions, the northern and southern Rocky Mountains, provides a regional perspective. Fire-climate relationships may be studied at these scales. Differences and similarities in our results enable us to determine whether fire-landscape relationships are determined by constraints at local or regional scales. This paper describes the acquisition and compilation of GIS databases for each wilderness area, a graphical analysis of spatial and temporal fire patterns, and a comparison of patterns found in each wilderness.

Study Areas

The Gila/Aldo Leopold Wilderness Complex (GALWC) is a 486,673-ha area in west-central New Mexico. The complex is composed of the Gila Wilderness Area, the Aldo/Leopold Wilderness Area, the Gila Cliff Dwellings National Monument and some nonwilderness portions of the Gila National Forest. The GALWC encompasses the headwaters of the Gila River, the Mogollon Mountains and the Black Range, 70 km north of Silver City, New Mexico. The Gila Wilderness portion of the study area is topographically diverse, with deep, narrow river canyons, flat mesa tops and steep mountains. Elevations range from 1,300 m near the main stem of the Gila River to 3,300 m on top of the Mogollon Mountains. The Aldo Leopold Wilderness Area is rugged, with elevations ranging from 1,500 m near the Mimbres River to 2,900 m on McKnight Mountain in the Black Range.

Broad valleys of desert scrub (*Ceanothus*, *Artemisia*, and *Yucca* sp.) are found at the lowest elevations. As elevation increases, piñon/juniper woodlands (*Pinus edulis*, *Juniperus deppeana*, and *J. monosperma*, and *Quercus* sp.) gain dominance. Extensive stands of ponderosa pine (*Pinus ponderosa*) mixed with Douglas-fir (*Pseudotsuga menziesii*) are found at middle elevations. At upper elevations, forests are comprised of mixed Englemann's spruce, subalpine fir (*Abies lasiocarpa*) (var. *arizonica*), Southwestern white pine (*Pinus strobiformis*), white fir (*Abies concolor*) and aspen (*Populus tremuloides*).

Fire season in the GALWC begins as early as April and may extend through September. Spring conditions are

usually dry, with thunderstorm activity increasing in June and July. The GALWC is dominated by low-severity surface fire regimes. Higher mortality, mixed-severity fire regimes are found at upper elevations (Abolt 1996, Swetnam and Dieterich 1983). During dry seasons, fire behavior can be extreme, with lethal fire common across all elevations. The Gila/Aldo Leopold Wilderness Complex has the highest number of lightning-ignited fires in the nation, with an average of 252 fires per million acres (404,687 ha) protected per year (Barrows 1978).

The Selway-Bitterroot Wilderness Area (SBWA) in Idaho and Montana is a 547,370-ha wilderness area; second in size (in the conterminous United States) only to the adjacent Frank Church-River of No Return Wilderness in Idaho. The area is characterized by extremely rugged terrain with broad topographic variation. Portions of the SBWA are found in the Bitterroot, Clearwater and Nez Perce National Forests. The northwest portion of the wilderness is characterized by Pacific maritime forests with assemblages of western red cedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*), western white pine (*Pinus monticola*) and Douglas-fir ranging from 500 m to 1,500 m. As elevation increases, mesic and mixed conifer forests dominate, with assemblages of Douglas-fir/Englemann's spruce (*Picea engelmannii*)/grand fir (*Abies grandis*) found on moist sites and ponderosa pine, while Douglas-fir and western larch (*Larix occidentalis*) forests occupy dryer sites. Subalpine forests make up the largest portion of the Wilderness, characterized by assemblages of Englemann's spruce/subalpine fir (*Abies lasiocarpa*) with lodgepole pine (*Pinus contorta*) dominant on drier sites and stands with more recent disturbance. Many of these lodgepole pine stands are extensive and have homogenous stand structure and ages. The highest subalpine sites are characterized by mixed whitebark pine (*Pinus albicaulis*)/alpine larch (*Larix lyallii*) forests.

Large thunderstorms are frequent in the SBWA, with a peak in activity during the early summer. Fire season in the SBWA begins in the early summer and extends through September. Fire regimes are mixed, with patchy stand-replacement fire dominant in upper elevation forests (70% of the SBWA) and lower severity, understory fire at lower elevations. Stand-replacement fires are dominant across all elevations during seasons with extreme fire weather (Barrett and Arno 1991, Brown and others 1994).

Methods

Twentieth-century fire perimeters were obtained in digital form (or digitized) from archival fire data at the Gila, Bitterroot, Clearwater and Nez Perce National Forests. Archives were compiled from old fire reports or operational fire perimeter maps. Using Arc/Info GIS software, digitized fire perimeters were organized by year, then compiled (using the Arc/Info 'Regions' data architecture) as separate fire atlases for each wilderness. These atlases were subset to a 5 k buffer around each wilderness boundary. The individual fire years for each area were converted to 30 m binomial raster grids using Arc/Grid. These layers were added together to create continuous surface models of fire frequency for each wilderness area.

Data for topography and vegetation were available from the USDA Forest Service Intermountain Fire Sciences Laboratory in Missoula, Montana (see Keane and others 1998; Keane and others 1999). Elevation was represented by two compiled sets of 7.5 minute USGS digital elevation models for the study areas. Slope and aspect surfaces for each study area were derived using the Arc/Grid commands SLOPE and ASPECT. Potential vegetation types (PVTs) were used to characterize the forests of each study area. Potential vegetation is a means of classifying biophysical characteristics of a site using the vegetation that would be present in the absence of disturbance (Cooper and others 1991). Classifications of potential vegetation were based on qualitative and quantitative analysis of geographic location, existing vegetation, field data, topography, local productivity and soil characteristics. GIS layers for vegetation and topography were subset to the extent of the fire atlases.

Different time periods for each wilderness area were delimited by different fire management strategies (see dashed lines in fig. 2). Area burned over time for both wilderness areas was plotted, then reported as proportions of each study area for each time period. Natural fire rotations for pre-suppression, suppression and prescribed fire periods were calculated. Distributions of 20th century fire frequency were summarized by topography and potential vegetation for each wilderness using Arc/Info GIS software.

Results

Fire perimeter data extended from 1909 to 1993 in the GALWC and from 1880 to 1996 in the SBWA. Mapped data indicated that 147,356 ha had burned in 232 fires in the GALWC and 474,237 ha in 437 fires in the SBWA. In the GALWC, 1909, 1946, 1951, 1985, 1992 and 1993 were the largest years, with 71% of the total area burned during these years. In the SBWA 1889, 1910, 1919, 1929, 1934 and 1988 were the largest years, accounting for 72% of the total area burned. Data from the SBWA show an almost total lack of mapped fire from 1935 through 1979. Mean mapped fire size in the GALWC was 637 ha, with a minimum of 2 ha and a maximum of 19,446 ha (the 1951 McKnight Fire). Median fire size was 88 ha. Mean fire size in the SBWA was 1,153 ha, with a minimum of 2 ha and a maximum of 52,223 ha (the 1910 Moose Creek fire). Median fire size was 135 ha. Perimeters from the two largest fire years are shown in figure 1.

Time periods related to different fire management strategies are indicated by vertical dotted lines in figure 2. Fire rotations for each of these time periods are summarized in table 1. Graphical analyses of re-burn patterns over the landscape indicate that 20th century fire frequency was associated with potential vegetation, elevation, slope and aspect.

Areas with the highest fire frequencies in the GALWC were found between 2,250 m and 2,525 m. In the SBWA, areas with the highest fire frequencies were found between 600 m and 1,760 m (fig. 3). Areas of highest fire frequency were skewed toward higher slopes in both wilderness areas. In the GALWC, northeast aspects burned most frequently, while southeast aspects burned less than expected based on the distribution of slope across the study area. Southeast and southwest aspects burned more frequently in the SBWA.

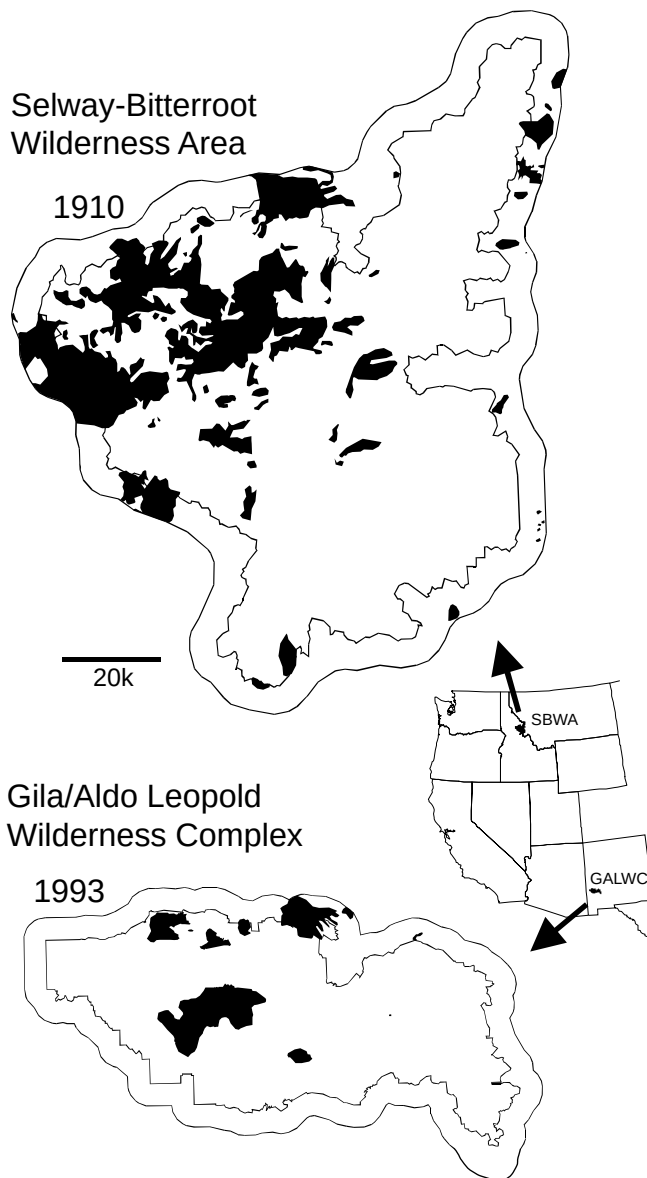


Figure 1—Perimeters from the two largest fire years in each wilderness area.

Quantitative descriptions of these relationships will be reported elsewhere.

In the GALWC, areas that had burned multiple times were more likely to be in Douglas-fir or ponderosa pine potential vegetation types. This relationship grew stronger as fire frequency increased. In the SBWA, areas that had burned two or more times were more often found in shrubfields, western red cedar or Douglas-fir potential vegetation types (fig. 4). It is important to note that ponderosa pine forests are included in the Douglas-fir PVT in the SBWA. Chi-square analyses indicated that these patterns were statistically significant.

Discussion

Each wilderness experienced large amounts of fire in the 20th century. The time-series of area burned may be divided

Table 1—Fire rotations for time periods delineated by different fire suppression levels in each study area.

Time period	Area burned (ha)	Percent of study area burned	Natural fire rotation (years)
Selway-Bitterroot Wilderness			
1880-1935	482,030	60.9	92
1936-1974	9,622	1.2	3,206
1975-1994	58,427	7.4	271
Gila/Aldo Leopold Wilderness Complex			
1880-1946	32,601	6.7	552
1947-1974	34,742	7.1	406
1975-1993	80,424	16.5	114
Selway-Bitterroot Wilderness			
1880-1935			
Gila/Aldo Leopold Wilderness Complex			
1880-1946			

into distinct periods in each wilderness, indicated by dashed lines in figure 2. Changes in fire frequency and area burned over time are attributable to land-use and fire suppression in each area. Fire suppression in each wilderness has taken place since the early 20th century. Large numbers of forest rangers and conscripted miners in the SBWA fought the great fires of 1910 (Habeck 1972; Moore 1996; Pyne 1982). Fires in the early 20th century were also fought by miners, loggers and forest rangers in the GALWC. In the mid-20th century, fire suppression technologies improved with the implementation of smokejumper and aerial retardant operations, a direct result of technologies improved during World War II (Pyne 1982). Aerial fire depots began operation in the late 1940s in Grangeville, Idaho, Missoula, Montana and Silver City, New Mexico. From the late 1940s through the mid-1970s, fires in both wilderness areas were suppressed rapidly and aggressively. Each wilderness area implemented prescribed natural fire management in the mid-1970s and both areas are currently considered to have the most advanced, broad-scale wildfire use programs in the United States.

In the GALWC, grazing was the main land use at the turn of the century. High levels of grazing are thought to have reduced the area burned, by reducing fine fuels, across the Gila National Forest from the mid-1800s through 1929 (Savage and Swetnam 1990, table 1), when grazing was reduced sharply in the forest reserve. Large fires during the mid-1940s and '50s correspond with a period of severe drought in the southwestern United States (Swetnam and Betancourt 1998). The absence of fire from the mid-1950s through the mid-1970s is probably due to aggressive fire suppression by the Gila National Forest. Area burned in the SBWA was quite high in the late 19th century and early 20th century (table 1). Grazing was never a prevalent land-use in the SBWA, unlike the GALWC. The main portions of the SBWA were so remote that little if any grazing occurred. Analyses of the relationships between area burned in each wilderness and time series reconstructions of drought may be found in Rollins and others 1999. Natural fire rotations in recent years are much shorter than rotations during the modern-suppression period (table 1). While this indicates

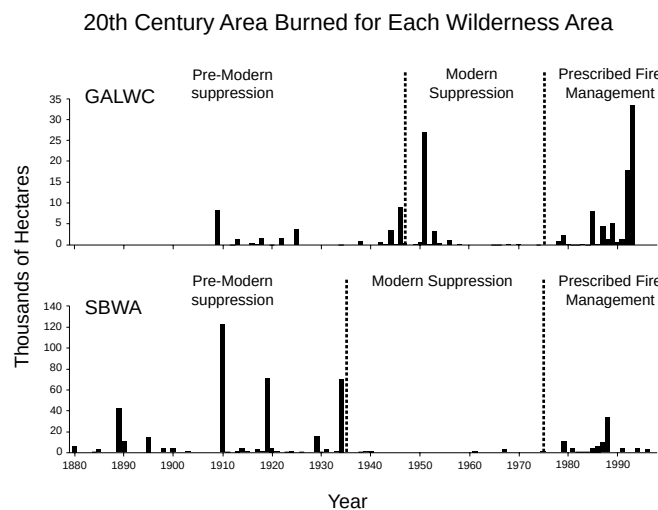


Figure 2—Area burned over time in each wilderness. Dashed lines indicate time periods delineated based on different fire management strategies.

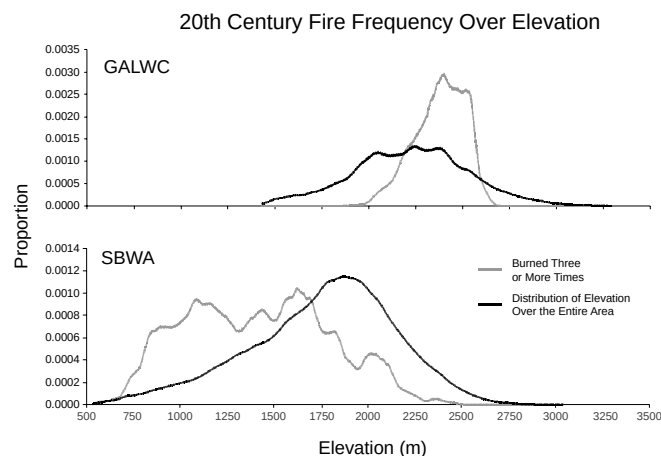


Figure 3—Distributions of areas that burned three or more times (7,020 ha in the Gila/Aldo Leopold Wilderness Complex and 7,472 in the Selway-Bitterroot Wilderness Area) over elevation. Areas are plotted as proportions of totals for direct comparison.

that wildfire use programs are successful to some degree, there are forest types where fire rotations remain much longer than pre-settlement estimates (Rollins and others, in preparation).

Topography and vegetation apparently had a strong effect on 20th century fire frequencies. In both wilderness areas, ponderosa pine and Douglas-fir PVTs had the highest 20th century fire frequencies (fig. 3). This supports evidence from Swetnam and Dieterich (1983) and Swetnam and Baisan (1996) that indicates these forest types had the highest fire frequencies, based on dendroecological evidence of pre-20th century fire. Open, ‘parklike’ ponderosa pine stands are common in both wilderness areas. These stands are maintained by frequent, low-severity fires that prune lower branches and suppress the growth of woody understory vegetation.

20th Century Fire Frequency Over Potential Vegetation

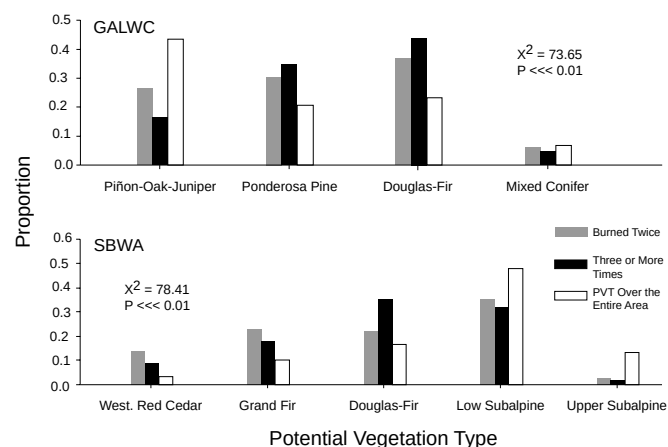


Figure 4—Distributions of areas that burned three or more times (7,020 ha in the Gila/Aldo Leopold Wilderness Complex and 7,472 in the Selway-Bitterroot Wilderness Area) by potential vegetation type. Areas are plotted as proportions of totals for direct comparison.

Elevation, slope and aspect are all important landscape-scale factors that determine fire patterns over landscapes. Our analyses show that elevation, slope and aspect affected 20th century fire frequency in both wilderness areas. In the GALWC, the highest fire frequencies were found between elevations of 2,250 m and 2,525 m (fig. 3). At these elevations, forests were dominated by ponderosa pine stands that historically showed high frequencies of large, low-severity fire events (Abolt 1996; Swetnam and Dieterich 1983). These forests have been the focus of much of the Gila National Forest’s prescribed natural fire management. In the SBWA, 20th century fire frequencies are highest between 600 m and 1,760 m. These elevations correspond with forests dominated by combinations of ponderosa pine and Douglas-fir and have been shown to exhibit high fire frequencies historically (Barrett and Arno 1991; Brown and others 1994). Both wilderness areas showed higher fire frequencies on steeper slopes. Steeper slopes cause fire to spread faster (Agee 1993). The uphill movement of flaming fronts is aided by more efficient radiant heat transfer and convective air currents generated from downslope fire. In the GALWC, northern and northeastern aspects showed higher fire frequencies than would be expected based on the distribution of aspect across the landscape. South and southeastern aspects had higher fire frequencies in the SBWA.

These results support working hypotheses that during ‘climatically average’ fire years, fire regimes in the southern Rocky Mountains are dominated by spatial distribution of fuel structure and composition, while in the northern Rocky Mountains climate (at least levels of solar insolation) constrains fire patterns during years with average weather. These hypotheses will be described in an upcoming journal article. Future research will concentrate on incorporating data for existing vegetation and climate variables, along with the variables described in this paper, into regression-based models of potential fire frequency. These surfaces of potential fire frequency will be valuable for evaluating fire management strategies based on the restoration of fire as an

ecosystem process in Rocky Mountain forests. The results of this research are a preliminary step toward quantitatively determining the complex relationships between landscape characteristics and fire patterns.

Fire regimes over the last millennia have shaped the forests of the Rocky Mountains. Large wilderness areas, like the Gila/Aldo Leopold Wilderness Complex and the Selway-Bitterroot Wilderness Area, remain the only areas where evidence of this long-term series of disturbance and recovery may be found. Our research is a first step in developing statistical models of fire regimes in these wilderness areas. Understanding of ecosystem dynamics and response to change is needed to understand what determines the distribution, extent and location of fires and to guide decisions about forest management. Globally, large wilderness ecosystems are rare, and getting rarer. In the face of rapid, broad-scale landscape change, knowledge and predictability of ecosystem function are imperative.

References

- Abolt, R. A. 1996. Surface and crown fire histories of upper elevation forests via fire scar and stand age structure analyses. M.S. Thesis. On file at the School of Renewable Resources, Department of Watershed Management, University of Arizona, Tucson, Arizona-USA. 118 p.
- Agee, J. K. 1993. Fire ecology of Pacific Northwest forests. Island press, Covelo, California-USA. 493 p.
- Barrett, S. W.; Arno, S. F. 1991. Classifying fire regimes and their topographic controls in the Selway-Bitterroot Wilderness. In Andrews, P. L.; Potts, D. F. eds. Proceedings of the 11th conference on fire and forest meteorology, Missoula, Montana-USA: 299-307.
- Barrows, J. S. 1978. Lighting fires in southwestern forests. Final Report prepared by Colorado State University for U.S. Department of Agriculture, Forest Service Cooperative agreement 16-568-CA. Intermountain Forest and Range Experimental Station, Ogden, Utah-USA. 154 p.
- Brown, J. K.; Arno, S. F.; Barrett, S. W.; Menakis, J. P. 1994. Comparing the prescribed natural fire program with presettlement fires in the Selway-Bitterroot Wilderness. *International Journal of Wildland Fire* 4(3):157-168.
- Christenson, N. L.; Bartuska, A. M.; Brown, J. H.; Carpenter, S.; D'Antonio, C.; Francis, R.; Franklin, J. F.; MacMahon, J.; Noss, R. F.; Parsons, D. J.; Peterson, C. H.; Turner, M. G.; Woodmansee R. R. 1996. The report of the Ecological Society of America committee on the scientific basis for ecosystem management. *Ecological Applications* 6(3):665-691.
- Cooper, S. V.; Neiman, K. E.; Roberts, D. W. 1991. Forest habitat types of northern Idaho: A second approximation. U.S. Department of Agriculture Forest Service GTR-INT-236. Intermountain Research Station, Ogden, Utah-USA. 143 p.
- Habeck, J. R. 1972. Fire ecology investigations in the Selway-Bitterroot Wilderness: Historical considerations and current observations. U.S. Department of Agriculture Forest Service INT-R1-72-001. Intermountain Research Station, Ogden, Utah-USA. 118 p.
- Keane, R. E.; J. L. Garner; K. M. Schmidt; D. G. Long; J. P. Menakis; M. A. Finney. 1998. Development of input data layers for the FARSITE fire growth model for the Selway-Bitterroot Wilderness complex, USA. U.S. Department of Agriculture Forest Service RMRS GTR-3. Rocky Mountain Research Station, Fort Collins, Colorado-USA. 66 p.
- Keane, R. E.; Mincey, S. A.; Schmidt, K. M.; Menakis, J. P.; Garner, J. L. 1999. Fuels and vegetation mapping for fire management on the Gila National Forest, New Mexico. In: Proceedings of the ASPRS conference, May 16-20, 1999, Portland, Oregon-USA. American Society for Photogrammetry and Remote Sensing, Bethesda, Maryland-USA: 272-282.
- Moore, B. 1996. The Lochsa story, Land ethics in the Bitterroot Mountains. Mountain Press Publishing Company, Missoula, Montana-USA. 458 p.
- Parsons, D. J.; Graber, D. M. 1991. Horses, helicopters and hi-tech: managing science in wilderness. In Reed, P.C., Preparing to manage wilderness in the 21st century. U.S. Department of Agriculture Forest Service GTR-INT-SE-66. Intermountain Research Station, Ogden, Utah-USA: 90-94.
- Price, C.; Rind, D. 1994. The impact of a 2XCO₂ climate on lightning-caused fires. *Journal of Climate* 7:1484-94.
- Pyne, S. J. 1982. Fire in America-a cultural history of wildland and rural fire. Princeton Univ. Press, Princeton, New Jersey-USA. 654 p.
- Rollins, M. G.; Swetnam, T. W.; Morgan, P. 1999. Twentieth-century fire Patterns in the Selway-Bitterroot Wilderness Area in Idaho/Montana and the Gila/Aldo Leopold Wilderness Areas in New Mexico. In J. D. Greer, ed. Proceedings of the Joint Fire Science Conference, Crossing the Millennium, Integrating spatial technologies and ecological principles for a new age in fire management. In press.
- Running, S. W.; Nemani, R. R. 1991. Regional hydrologic and carbon balance response of forests resulting from potential climate change. *Climatic Change* 19:349-368.
- Ryan, K. C. 1991. Vegetation and wildland fire: implications for global change. *Environmental Management* 17:169-178.
- Savage, M.; Swetnam, T. W. 1990. Early 19th-century fire decline following sheep pasturing in a Navajo ponderosa pine forest. *Ecology* 71(16):2374-2378.
- Swetnam T. W.; Dietrich, J. H. 1983. Fire history of ponderosa pine forests in the Gila Wilderness, New Mexico. In Proceedings-Symposium and workshop on wilderness fire; November 15-18, 1983. Missoula, Montana. General Technical Report INT-182 U.S.D.A. Forest Service Intermountain Forest and Range Experiment Station, Ogden, Utah-USA: 390-397.
- Swetnam, T. W.; Baisan, C. H. 1996. Historical fire regime patterns in the southwestern United States since A.D. 1700. In Allen, C.D., Proceedings of the 2nd La Mesa Fire Symposium, March 29-31, 1994, Los Alamos, New Mexico. U.S. Department of Agriculture Forest Service RM-GTR-286. Rocky Mountain Forest and Range Research Station, Fort Collins, Colorado-USA: 11-32.
- Swetnam, T. W.; Betancourt, J. L. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American southwest. *Journal of Climate* 11:3128-3147.
- Turner, M. G.; Gardener, R. H.; O'Neill, R. V. 1995. Ecological dynamics at broad scales: ecosystems and landscapes. *BioScience*, supplement 1: Science and Biodiversity Policy:29-35.

The Federal Wildland Fire Policy: Opportunities for Wilderness Fire Management

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Abstract—The Federal Wildland Fire Management Policy and Program Review represents the latest stage in the evolution of wildland fire management. This policy directs changes that consolidate past fire management practices into a single direction to achieve multidimensional objectives and creates increased opportunities for wilderness fire management. Objectives previously accomplished through prescribed natural fire are now achieved through application of an appropriate management response to wildland fires. The 1998 fire season provided both a test of the policy and a clear indication of future wildland fire management and benefits to wilderness management.

Throughout the 20th century, fire management capability has continued to develop in response to land and resource management needs, growing knowledge of the natural role of fire, and increased effectiveness of fire suppression. Threats from wildland fires escalate annually as long-term effects from past land use and fire management actions become manifest in natural vegetation communities. Expanding values to be protected in combination with current land use practices also intensify protection concerns. Federal land management agencies' ability to respond to these challenges is rapidly becoming overextended. However, increasing knowledge, understanding and experience have shown that complete fire exclusion does not support a balanced resource management program. In fact, in many situations, this management direction is detrimental to ecosystem health and functioning. Wildland fire management policy and procedures must change to reflect new considerations, capabilities and direction, while being responsive to the increasing complexity of wildland fire management and resource management objectives.

Since 1988, the federal fire program has experienced two policy and program reviews (U.S. Department of Agriculture and U.S. Department of the Interior 1989, U.S. Department of the Interior/U.S. Department of Agriculture 1995) and one General Accounting Office program audit (U.S. General Accounting Office 1990). These reviews have

strengthened long-term accountability of the fire program and promoted more informed decision-making.

Most recently, events of the 1994 fire season, including 34 firefighter fatalities, \$925 million dollars in suppression expenses and significant damage to natural resources and private property, created a renewed awareness and concern among federal land management agencies and constituents about safety, wildland fire impacts and the integration of fire and resource management. As a result of these concerns, the Federal Wildland Fire Management Policy and Program Review was chartered (U.S. Department of the Interior/U.S. Department of Agriculture 1995). Federal agencies are currently involved in implementing the results of this review as the Federal Wildland Fire Management Policy.

Events during the 1998 fire season in the northern Rocky Mountains are indicative of the future of wildland fire management and benefits to wilderness management. During August and September of 1998, lightning ignited numerous wildland fires that were managed for resource benefits consistent with policy implementation procedures and funding authorities. Dozens of wildland fires were successfully managed in national parks and wildernesses. Compared with past policy, constraints and capability, this reflects a significant increase in the number of successfully managed fires. In fact, in previous years, the greatest proportion of these ignitions would probably have been quickly suppressed. This period of fire activity provided an immeasurable opportunity to put the current policy into practice and evaluate its effectiveness.

Since the early 1900s, fire management policy has adapted to meet emerging land and resource management issues, fire suppression needs and expanded understanding of the natural role of fire. This policy provides management direction and procedures that markedly increase opportunities to manage fire in wilderness to accomplish multiple objectives. The success of these recommendations and policy implementation depends on actions and expectations both internal and external to federal agencies. Agencies must ensure that wildland fire management is fully integrated into land management planning. It can no longer be assumed that all wildland fires can and should be controlled and suppressed. Absolute protection is an expectation that is difficult, if not impossible to achieve, and based on federal workforce limitations, fiscal constraints, resource management needs and environmental and fire behavior variables, is unrealistic.

This paper describes the Federal Wildland Fire Management Policy and Program Review recommendations, defines implications of the policy and management opportunities for wilderness fire management, and provides an encapsulation of

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future wilderness fire management activities through a review of the 1998 fire season in the northern Rocky Mountains.

Review of Federal Wildland Fire Management Policy

The Federal Wildland Fire Management Policy currently being implemented represents the latest stage in the evolution of wildland fire management. This policy directs federal agencies to achieve a balance between suppression to protect life, property, and resources, and fire use to regulate fuels and maintain healthy ecosystems. This policy eliminates many of the previous limitations to expanded fire use.

Differences between the previous (prior to 1995) and current (post-1995) federal wildland fire management policy are typified by previous classification of all fires as either wildfires or prescribed fires. This arbitrary classification precluded maximum management effectiveness and strategic implementation. Under the current policy, all fires not ignited by managers for predetermined objectives are considered wildland fires. All wildland fires, then, have the same classification and receive management appropriate to conditions of the fire, fuels, weather and topography to accomplish specific objectives for the area where the fire is burning. These management actions are termed the “appropriate management response” and will vary among individual fires. This type of management permits a dynamic range of tactical options. The federal fire policy now advocates greater application and use of fire for accomplishing resource benefits while maintaining and implementing an effective suppression program.

The 1995 report (U.S. Department of the Interior/U.S. Department of Agriculture 1995) presents nine guiding principles fundamental to the success of the wildland fire management program and implementation of review recommendations. It also recommends a set of 13 wildland fire policies in the areas of: safety, planning, wildland fire, prescribed fire, preparedness, suppression, prevention, protection priorities, interagency cooperation, standardization, economic efficiency, wildland/urban interface, and administration and employee roles (table 1).

The following guiding principles (U.S. Department of the Interior/U.S. Department of Agriculture 1995) represent the foundation of the Federal Wildland Fire Management Program:

- *Firefighter and public safety is the first priority in every fire management activity.* Every firefighter, fireline supervisor, fire manager and agency administrator will take positive action to ensure compliance with established safe firefighting practices.
- *The role of wildland fire as an essential ecological process and natural change agent will be incorporated into the planning process.* Federal land and resource management plans will recognize and define the natural role of fire and set objectives for the use and desired future conditions of public lands.
- *Fire management plans, programs and activities support land and resource management plans and their importance.* All agencies will develop Fire Management Plans that: use information about fire regimes, current conditions and land management objectives to develop fire management goals and objectives; address

all potential wildland fire occurrences and provide for a full range of actions; use new knowledge and monitoring results to revise goals, objectives and actions; and build and maintain a close link between fire and land and resource management. Wildland and prescribed fire are not ends in themselves, but rather are means to an end. They represent planning and implementation actions done to facilitate protection and the resource management objectives described in the plans.

- *Sound risk management is a foundation for all fire management activities.* Risks and uncertainties associated with fire management activities must be understood, analyzed, communicated and managed as they affect the cost of either doing or not doing an activity. Net public benefits will be an important component of decisions.
- *Fire management programs and activities are economically viable, based on values to be protected, costs and land and resource management objectives.* Federal agency administrators are adjusting and reorganizing programs to reduce costs and increase efficiency. Investments in fire management activities must be evaluated against other agency programs in order to accomplish the overall mission, set short- and long-term priorities and clarify management accountability.
- *Fire management plans must be based on the best available science.* All wildland fire management agencies develop knowledge and experience. An active fire research program combined with interagency collaboration can make this available to all fire managers.
- *Fire management plans and activities incorporate public health and environmental quality considerations.* Fire management plans will address desired objectives but will be balanced with other societal needs, including public health and safety, air quality and other specific concerns.
- *Federal, tribal, state and local interagency coordination and cooperation are essential.* Increasing costs and smaller workforces require public agencies to pool their human resources to deal with the ever-increasing and more complex fire management tasks. Full collaboration among federal agencies and between federal agencies and tribal, state, local and private entities results in a mobile fire management workforce that can respond to the full range of public needs.
- *Standardization of policies and procedures among federal agencies is an ongoing objective.* Consistency of plans and operations provide the fundamental platform upon which federal agencies can cooperate and integrate fire activities across agency boundaries and provide leadership for cooperation with tribal, state and local fire management organizations.

To reduce misinformation and provide correct and consistent direction, the National Wildfire Coordinating Group (NWCG) developed and approved an “umbrella” flow chart which illustrates the broad framework behind policy implementation (fig. 1). This flow chart has become the cornerstone for policy description, illustration and development of implementation procedures. All fires are shown as either wildland or prescribed fires. Wildland fire management can follow one of three pathways, depending on the level of land management planning completed, resource values

Table 1—Federal wildland fire policies.

Policy area	Policy direction
Safety	Firefighter and public safety is the first priority. All Fire Management Plans and activities must reflect this commitment.
Planning	Every area with burnable vegetation must have an approved Fire Management Plan. Fire Management Plans must be consistent with firefighter and public safety, values to be protected and land and resource management plans and must address public health issues. Fire Management Plans must also address all potential wildland fire occurrences and include the full range of fire management actions.
Wildland fire	Fire as a critical natural process will be integrated into land and resource management plans and activities on a landscape scale, across agency boundaries, and will be based upon best available science. All use of fire for resource management requires a formal prescription. Management actions taken on wildland fires will be consistent with approved Fire Management Plans.
Use of fire	Wildland fire will be used to protect, maintain, and enhance resources and as nearly as possible, be allowed to function in its natural ecological role.
Preparedness	Agencies will ensure their capability to provide safe, cost-effective fire management programs in support of land and resource management plans through appropriate planning, staffing, training, and equipment.
Suppression	Fires are suppressed at minimum cost, considering firefighter and public safety, benefits, and values to be protected, consistent with resource objectives.
Prevention	Agencies will work together and with other affected groups and individuals to prevent unauthorized ignition of wildland fires.
Protection priorities	Protection priorities are (1) human life and (2) property and natural/cultural resources. If it becomes necessary to prioritize between property and natural/cultural resources, this is done based on relative values to be protected, commensurate with fire management costs. Once people have been committed to an incident these resources become the highest value to be protected.
Interagency cooperation	Fire management planning, preparedness, suppression, fire use, monitoring, and research will be conducted on an interagency basis with the involvement of all partners.
Standardization	Agencies will use compatible planning processes, funding mechanisms, training and qualification requirements, operational procedures, values-to-be-protected methodologies, and public education programs for all fire management activities
Economic efficiency	Fire management programs and activities will be based on economic analyses that incorporate commodity, non-commodity, and social values
Wildland/urban interface	The operational role of Federal agencies as a partner in the wildland/urban interface is wildland firefighting, hazard fuels reduction, cooperative prevention and education, and technical assistance. Structural fire protection is the responsibility of Tribal, State, and local governments. Federal agencies may assist with exterior structural suppression activities under formal Fire Protection Agreements that specify the mutual responsibilities of the partners, including funding. (Some Federal agencies have full structural protection authority for their facilities on lands they administer and may also enter into formal agreements to assist State and local governments with full structural protection.)
Administrator and employee roles	Employees who are trained and certified will participate in the wildland fire program as the situation demands; employees with operational, administrative, or other skills will support the wildland fire program as needed. Administrators are responsible and will be accountable for making employees available.

affected or fire cause. Fire Management Plans (FMP), prepared by each administrative unit or jointly by multiple units, are prerequisite to operational implementation. Management options are substantially reduced when a Fire Management Plan is lacking, incomplete, or not approved. Without a plan, units may only implement an appropriate management response of initial attack suppression (top pathway, fig. 1). When a Fire Management Plan has been completed and approved, and wildland fires are from natural ignition sources, the full range of appropriate management response options is available (middle pathway, fig. 1).

The concept of appropriate management response is integral to this policy. Management responses are programmed to accept resource management needs and constraints, reflect a commitment to safety, be cost-effective, and accomplish desired objectives while maintaining the versatility to change intensity as conditions change. Every wildland fire will receive an appropriate management response. The appropriate management response is defined as the specific action taken in response to a wildland fire to implement protection and/or fire use objectives. It allows managers to utilize a full range of responses. It does not lock tactical options to fire type designations. As conditions change, the

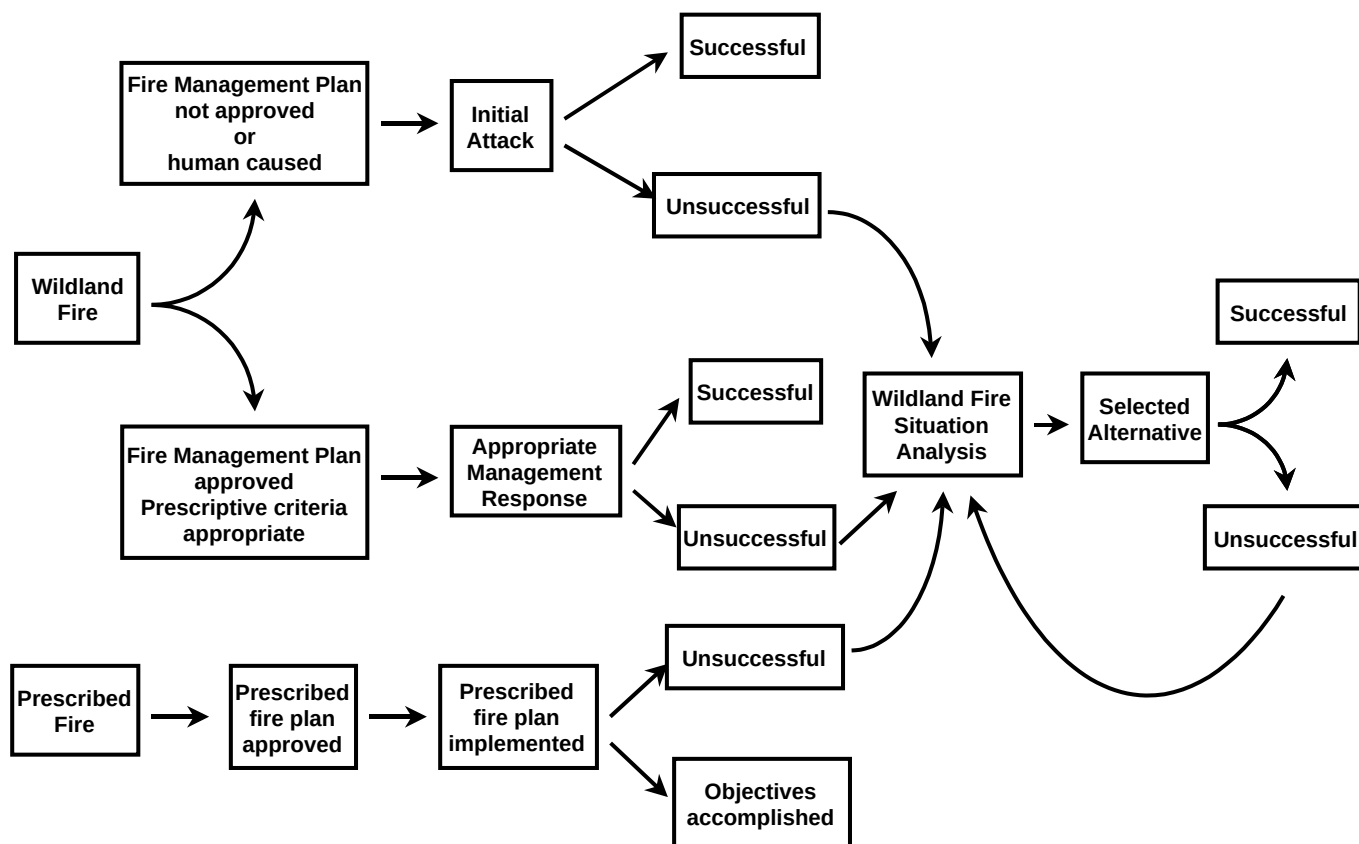


Figure 1—National Wildfire Coordinating Group (NWCG) Wildland Fire Management Policy flow chart (disseminated throughout the five Federal fire management agencies via internal agency communication directives).

particular response can change to accomplish the same objectives.

It is important to note that the appropriate management response *is not* a replacement term for prescribed natural fire, or the suppression strategies of control, contain, confine, limited or modified; but it is a concept that offers managers a full spectrum of responses (Zimmerman and Bunnell 1998). It is based on objectives, environmental and fuel conditions, constraints, safety and ability to accomplish objectives. It includes wildland fire suppression at all levels, including aggressive initial attack. Use of this concept dispels the interpretation that there is only one way to respond to each set of circumstances. Appropriate management responses can be developed along a continuum from monitoring to aggressive suppression. Under this policy, opportunities to combine strategies on individual fires are unlimited, as is implementing a variety of options concurrently, and there is no distinction between fire types or strategic responses. Through its application, managers have the ability to maximize the opportunities presented by every wildland fire situation.

Prescribed fire, as shown in the bottom pathway of the flowchart (fig. 1), differs very little from its management under previous policy. A Fire Management Plan must be completed and approved, and clearly specify the need for prescribed fire. Specific implementation plans (Prescribed Fire Plans) must be developed before a fire can be ignited.

When conditions described in the Prescribed Fire Plan occur and necessary resources are available to implement the prescribed actions, the fire is ignited and the plan implemented.

If the desired objectives cannot be met for either wildland or prescribed fire, a new strategy must be selected through the Wildland Fire Situation Analysis (WFS) process.

Misconceptions Surrounding the Wildland Fire Management Policy

It can be difficult to interpret and understand this policy and its implications to management. Comparison to previous fire management policies does not necessarily offer similarities, direct replacement terms, or defined actions. Recognizing the flexibility and range of opportunities presented by the new policy facilitates its interpretation. Understanding these opportunities and implementation mechanisms is prerequisite to efficient implementation.

Common misconceptions have developed about the policy. The Wildland and Prescribed Fire Management Policy, Implementation Procedures Reference Guide (Zimmerman and Bunnell 1998) was prepared to present a set of implementation procedures and to define what the policy is and isn't. To understand what can be accomplished, it must be realized that this policy:

- *Is not a less safe way of managing wildland fires.* The new policy is formulated on a solid basis incorporating safety; this commitment is continually reinforced. Federal agencies will develop, thorough planning processes, and implement management procedures that accomplish objectives while always maintaining a firm commitment to safety. The guiding principles, fundamental to the success of the policy implementation, describe the commitment to safety in the very first principle. One of the key points stated in the Federal Wildland Fire Management Policy and Program Review recommendation report is, "Protection of human life is reaffirmed as the first priority in wildland fire management. Property and natural/cultural resources jointly become the second priority, with protection decisions based on values to be protected and other considerations." The report further affirms the commitment to safety by stating, "Once people are committed to an incident, those resources become the highest value to be protected and receive the highest management considerations."
- *Is not a significant change in what we do.* The wildland fire management program strives to accomplish objectives designed to maintain, enhance, protect, and preserve natural and cultural resources. Fire management programs will continue to provide safe, ecologically sound and economically efficient actions in support of land and resource management plans through planning, staffing, training and equipment readiness.
- *Is not a wholesale shift to "let burn" actions.* Federal wildland fire management programs have never included "let burn" activities. The implication of this term—that fires do not receive appropriate levels of management scrutiny and attention—is not correct. In fact, wildland and prescribed fires have received and will continue to receive significant attention during management planning, implementation and evaluation. A wholesale shift to one management strategy over another is undesirable, unrealistic and inconsistent with policy goals, and it will not occur. The aggregate strategies available to implement the fire management program will achieve a better balance of protection and land and resource management objectives.

Agencies will utilize the full spectrum of fire management actions—from prompt suppression of unwanted fires to managing naturally ignited fires to accomplish specific resource management objectives. The majority of wildland fires will continue to receive a suppression-oriented response. Suppression capabilities will continue to expand and grow in sophistication and capacity to meet increasing demands such as the rapid expansion of wildland/urban interfaces.

- *Is not a less efficient way of doing business.* The policy promotes application of fire management actions along a "sliding scale," ranging from minimal on-the-ground actions to prompt, aggressive actions to fully extinguish the fire. Use of this spectrum allows agencies more flexibility to design responses closely allied with objectives and fuel, weather and topographic conditions. In the past, responses were driven by fire type as well as other considerations. Responses will be appropriate for individual conditions and the objectives associated with

that ignition; they will not be related to a fire type or classification. This will permit federal agencies to achieve effectiveness and efficiency in operations.

What the policy actually represents is:

- *A more cohesive way of approaching wildland fire management.* Management actions on wildland fires will no longer be driven by fire type designation. Fires will no longer be extinguished under a default response but will be suppressed for specific reasons. Fires managed for resource benefits will have specific rationale for such management identified in the Fire Management Plan.
- *A foundation to facilitate more efficient operations.* Classification of all fires into a single category of wildland fires will allow managers to respond to each and every fire in a manner appropriate for the objectives, constraints and conditions associated with that fire. Managers will not be forced to adopt a strategy due to fire classification. There will be more attention to ecological concerns, and each fire will have a greater probability of accomplishing desired objectives.
- *A program of action that promotes concurrent use of available management strategies.* Through the appropriate management response, managers can respond to different fires in different ways, using different strategies to accomplish different objectives. Nothing precludes this from happening concurrently. In fact, the most efficient management will make simultaneous use of fire management strategies. Different strategies may also be employed on various portions of individual fires, thus reducing costs and utilization of scarce resources. Fire Policy Review Recommendation goals support the concurrent utilization of available management strategies by stating for protection capabilities, "Federal Agencies will maintain sufficient fire suppression and support capability." They further state, for reintroduction of fire, "Based upon sound scientific information and land, resource and fire management objectives, wildland fire is used to restore and maintain healthy ecosystems and to minimize undesirable fire effects. Fire management practices are consistent for areas with similar management objectives, regardless of jurisdiction."
- *A program of action that does not automatically place priority on one strategy over another without analysis of specific information.* No wildland fire will automatically be categorized as having a lower priority than others. All wildland fires will compete for resources on the basis of objectives, values-to-be-protected, safety, risk, complexity and other specific considerations. During periods of resource shortages, fires determined to be in greater need will receive priority for resource allocation. Policy Review action items for values to be protected and preparedness planning state, "Federal agencies will define values to be protected, working in cooperation with Tribal, State, and local governments; permittees; and public users. Criteria will include environmental, commodity, social, economic, political, public-health, and other values." As part of the standardization goals, the report states that agencies will use compatible methodologies to determine values-to-be-protected. Common priority-setting standards to facilitate allocation of scarce resources will be developed.

The National Wildfire Coordinating Group has completed a report on allocation of resources for this purpose (Williams and others 1998).

- *A common planning process for all agencies, resulting in one plan.* The Fire Policy Review Recommendation for planning states, “Fire management goals and objectives, including the reintroduction of fire, are incorporated into land management planning to restore and maintain sustainable ecosystems. Planning is a collaborative effort, with all interested partners working together to develop and implement management objectives that cross jurisdictional boundaries.” Recommendations stated in the Policy Review include, “the use, by Federal Agencies, of a compatible fire management planning system that recognizes both fire use and fire protection as inherent parts of natural resource management; this system will ensure adequate fire suppression capabilities and support fire reintroduction efforts.” The Policy Review further states that federal agencies will “continue ongoing efforts to jointly develop compatible, ecosystem-based, multiple-scale, interagency land management plans that involve all interested parties and facilitate adaptive management.”
- *A process based on uniform budget and fiscal procedures.* Agency standardization and development of common procedures will reduce administrative barriers. Action items to achieve this include: develop consistent language to be included in budget appropriations, enabling the full spectrum of fire management actions on wildland fires; seek authority to eliminate internal barriers to the transfer and use of funds for prescribed fire on non-federal lands and among federal agencies; seek authority or provide administrative direction to eliminate barriers to carrying over, from one year to the next, all funds designated for prescribed fire; work with the Office of Personnel Management to acquire authority for hazard pay to compensate employees exposed to hazards while engaged in prescribed burning activities; jointly develop simple, consistent hiring and contracting procedures for prescribed fire activities; jointly develop programs to plan, fund and implement an expanded program of prescribed fire in fire-dependent ecosystems.

Implications of the Fire Policy to Wilderness Fire Management

Wilderness heritage in the United States has a long and storied history. In the late 1800s, John Muir, America’s most famous and influential naturalist and conservationist, explored California’s wilderness and was instrumental in the formation of numerous national parks (Yosemite—1890, Mt. Rainier—1899, Grand Canyon—1908). In 1919, Arthur Carhart, a young Forest Service landscape architect, recommended that Trappers Lake in Colorado’s White River National Forest be removed from development, even for recreational purposes. In 1924, Aldo Leopold, deputy regional forester in Region 3, had the satisfaction of seeing his efforts achieved when the Forest Service designated 574,000 acres of the Gila National Forest, New Mexico, as a wilderness reserve. In 1939, Bob Marshall, Chief of Division of

Recreation and Lands in the Forest Service, led establishment of the U Regulations, creating and tightening protection for wilderness, wild and roadless areas, immediate forerunners of today’s National Wilderness Preservation System.

The National Wilderness Preservation System has grown from 9 to 104 million acres since passage of the Wilderness Act in 1964. Today’s wilderness (104 million acres) collectively comprises a little more than the area of the state of Montana (94 million acres). Wilderness is important to the environment and society. It provides clean water and air, naturalness, critical habitats for endangered and non-endangered plants and animals, solitude, scenic beauty and economic benefits to communities through tourism and recreation. Wilderness condition is a barometer for measuring ecologic integrity.

This year, 1999, marks the 75th anniversary of the establishment of the Gila Wilderness and the 50th anniversary of *A Sand County Almanac*, written by Leopold in 1949, arguably America’s most read and influential book on ecologic principles and social values. These noteworthy anniversaries, combined with the 1995 Interagency Wilderness Strategic Plan, emerging Natural Resource Program management efforts and the implementation of the Federal Fire Policy prepares us to look at the future of fire management in wilderness with an eye on our past and debts to be paid to Muir, Carhart, Marshall and Leopold.

The Federal Wildland Fire Management Policy has much to offer to wilderness management objectives. The dissolution of funding mechanisms that influenced the Prescribed Natural Fire (PNF) program, primarily in Forest Service wilderness areas, from 1972-1998 is a significantly positive step toward increased use of fire in wilderness. Limited funding bases for the previous PNF program severely constrained full implementation. Consequently, many fires were suppressed due to a lack of appropriated funds for management. Other fire actions were financed by “bootleg” operations that attached funding to other fires or program elements. These aggressive and sometimes heroic financial actions clearly placed managing fire in wilderness in a “second class” position. These actions were largely viewed as problematic and a threat to traditional management efforts.

Funding authority for appropriate management response to wildland fire occurrence in wilderness has dramatically increased flexibility. This will promote both the use of fire in wilderness and support from wilderness management for critical fire implementation. Particularly critical is proper financing of under-financed wilderness field staff combined with full funding for fire management resources required to successfully manage fires. It will increase implementation action safety and internal/external coordination, as well as provide better long-range fire planning while reducing overall risk.

Increased management application of wildland fires in wilderness will build the confidence of wilderness and fire management staff. Past programmatic success (1970-1998), has produced growing advocacy at both the public and interagency management levels. Two important cultural elements have been influenced by this change. First, fire suppression as the primary fire management response to fire occurrence in wilderness has been softened in some areas. Subsequently, where adequate planning has been

completed, new fire starts may be equally considered for use or suppression. Second, wilderness management has recognized that substantial program increases will require the full integration of wilderness and fire management personnel in both the decision-making process and implementation on the wildland fires selected to meet resource benefits in wilderness.

The application of prescribed fire in wilderness areas has had consistent and substantial success in the National Park System. Combinations of Wilderness Act interpretation, administrative restrictions, and complex NEPA requirements in the planning process have severely limited the use of prescribed fire in Forest Service wilderness areas. The current policy offers no change in requirements or application from the previous policy. Subsequently, the Park Service should continue to increase accomplishments. Two major administrative changes may also allow the Forest Service to increase wilderness acres treated by prescribed fire. Forest Service Manual 2320 section is under revision and will promote increased use of prescribed fire in wilderness, both to reduce risk of escape through boundary treatments and to promote the use of wildland fire for resource benefits. This addresses an important issue identified by Parsons and Landres (1998). It is now accepted by management that prescribed fire application is a viable treatment for maintaining or restoring historic vegetative components in wilderness areas physically smaller than the historic size of fires that shaped and textured vegetation components in those ecosystems. This applies specifically to relatively small areas with fire-adapted ecosystems identified as high fire frequency nonlethal fire regimes. The net effect of this approach is a potential for increased application of wildland and prescribed fire in all areas. But, more important, it will be possible to manage more small wilderness areas with the best available management practice of prescribed fire application. The key to success of this effort will be integrated decisions, with wilderness management assuming a leadership role in promulgating direct management actions.

Putting the Federal Wildland Fire Policy Into Practice

During early August 1998, thunderstorm activity was responsible for igniting more than 200 wildland fires in the northern Rocky Mountains (in two geographic areas: Great Basin and Northern Rockies). These fires were located throughout northern Idaho and western Montana on national forests and national parks. Appropriate management responses, consistent with the federal fire policy, were developed for all fires. Evaluations of each fire and its specific set of circumstances, including land management objectives, values-to-be-protected, primary land use, external influences and other information pertinent to the fire location and situation were completed. Results indicated that many of the fires needed an immediate management response of suppression to accomplish protection objectives (46% of the fires in the Northern Rockies Area for 3696 acres). Other fires, actually a greater number than were suppressed, did not need immediate suppression responses and were, in fact, candidates for accomplishing resource benefits. These fires were evaluated with processes identified in the

implementation procedures reference guide (Zimmerman and Bunnell 1998) and received appropriate management responses to accomplish resource benefits; firefighter safety was also minimized because of reduced exposure, and the response was also cost effective (54% of the fires in the Northern Rockies Area for 26,385 acres).

Although federal agencies are in the process of actively implementing this policy and have been since its inception, not all agencies have enough direction to completely implement new procedures. Newly updated agency manuals had not been officially approved for the USDA Forest Service in 1998. As a result, it was not possible to implement the policy using all new terminology, although fiscal allowances, management coding and management responses were in place, permitting consistency with policy direction. Wildland fires on National Forest lands managed for resource benefits were described as prescribed natural fires. This situation had little influence on the eventual outcome, but it did cause some limited confusion in regard to terminology.

The 1998 fire season accomplishments can be differentiated from previous years by the numbers of fires managed for resource benefits. In past years, fixed budgets for prescribed natural fire implementation severely curtailed the scale of accomplishments. Natural fire management budgets for both the Forest Service and Park Service limited the numbers of and, often, the duration of prescribed natural fires. Once these budgets were exhausted or fully committed to potentially long-duration fires, all other new ignitions were forced into a wildfire designation and received an initial attack suppression response. Confinement responses were implemented only if large resource commitments were not warranted. Budget limitations often restricted prescribed natural fires to large, undeveloped areas that presented little risk of fire leaving the area or threatening boundaries, developments, etc. This situation did not promote efficient use of natural fire management or a balanced program. Since 1970, nearly 4,000 prescribed natural fires have occurred; since 1980, almost an additional 2,000 fires have been managed through confine, contain, limited, or modified strategies in all 50 states (table 2). Fire data from lands managed by the Bureau of Land Management in Alaska are not available and not included in table 2. However, the best available information suggests that several million acres were treated under modified suppression, producing similar landscape scale effects during the same time period.

The numbers of wildland fires managed for resource benefits over the last five years does not show well-defined trends. These data indicate a gradual increase, then slight drop-off, reflecting seasonal severity and total numbers of ignitions (table 3). The total number of fires managed for resource benefits in 1998 was not the highest on record (table 3). However, this total is comprised of fires almost exclusively concentrated in the northern Rocky Mountains rather than throughout the western United States. More than 60 wildland fires were managed for resource benefits in the Bob Marshall, Selway-Bitterroot, Sawtooth and Frank Church-River of No Return Wilderness Areas and in Glacier National Park (fig. 2). Managing this number of fires for this purpose is clearly significant when during previous years, as many as 90 percent of these fires would probably have been suppressed through aggressive initial

Table 2—Wildland fire program summary (includes all 50 states).

strategies ^b		Prescribed	natural	fires ^a	Confine-contain-modified	
Area	Agency burned (acres)	Number of fires	Area burned	(acres)	Number of fires	
National Park Service	2,596	329	860	760	704	857
U.S. Forest Service	1,360	533	215	590	951	557
U.S. Fish and Wildlife Service	— ^c	— ^c	— ^c	68	5,004	
Bureau of Indian Affairs	— ^c	— ^c	— ^c	372	3,765	
Bureau of Land Management	— ^c	— ^c	— ^c	NA ^d	NA ^d	
Other	21	3	100	—	—	
Total	3,977	866	175	1,956	1,685	183

^aData for 1970 to 1998, source U.S. Forest Service and National Park Service files at National Interagency Fire Center.

^bData for 1980 to 1998, source U.S. Forest Service and National Park Service files at National Interagency Fire Center.

^cU.S. Fish and Wildlife Service, Bureau of Indian Affairs and Bureau of Land Management did not manage fires under prescribed natural fire strategies during this time period.

^dBureau of Land Management data not available for Alaska.

Table 3—Wildland fires managed for resource benefits by the USDA Forest Service and USDI National Park Service, 1994-1998.

Agency 1998	Number of wildland fire use actions ^a				
	1994	1995	1996	1997	1998
Forest Service USFS	26	91	164	70	113
National Park Service (NPS)	68	51	83	96	101
Total number of wildland fire use actions	94	142	247	166	214

^aSource:USFS and NPS file data, National Interagency Fire Center.

attack or extended attack and would have never contributed to resource benefits. The shift in management response to wilderness fires prompted by the current policy is resulting in more fires being managed for resource benefits. This is clear when reviewing the proportion of fires managed for resource benefits, suppressed through control strategies, and managed through a confinement strategy during 1998 (fig. 3). The largest proportion of fires during

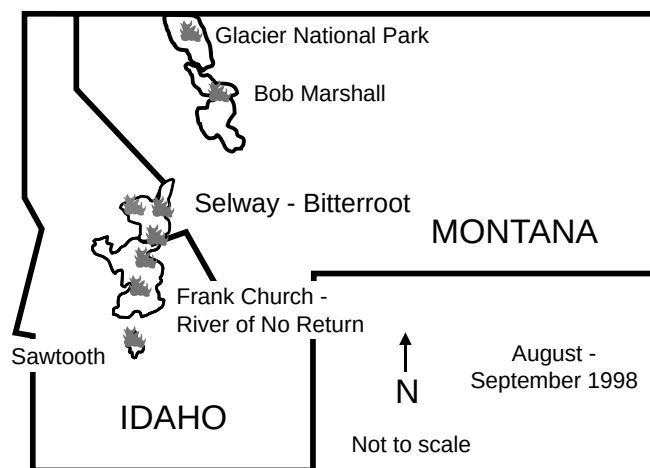


Figure 2—Wilderness and National Park wildland fire activity in Idaho and Montana from August to September 1998 (names are wilderness areas only, not specific fires).

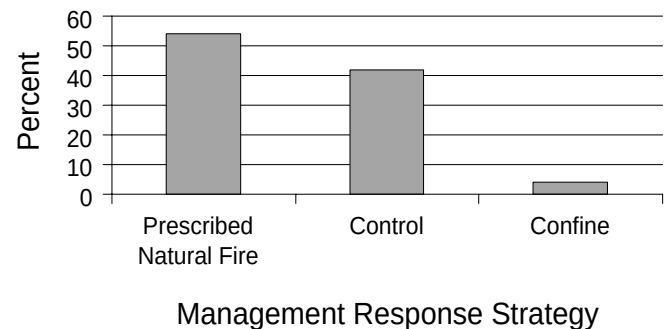


Figure 3—Proportion of wilderness fires in management response strategies of prescribed natural fire, control, and confinement during 1998 in the Northern Rocky Mountains.

the August to September period in the northern Rocky Mountains was managed through a prescribed natural fire strategy to accomplish resource benefits, while the control and confinement strategies were used considerably less. This raises speculation concerning how many fires shown in table 2 as confine-contain-modified strategies would have been wholly managed for resource benefits under the federal fire policy.

During 1998, Wildland Fire Implementation Plans (WFIP) were used to define appropriate management responses for each fire or for groups of fires when resource benefits were the primary objective. This includes all fires managed under the old terminology of prescribed natural fire. When protection objectives and/or external influences indicated a dominant need for a suppression-oriented response, either an initial attack response was originated or a Wildland Fire Situation Analysis (WFSA) was used to formulate the preferred alternative. This included all fires managed under the strategies of control and confine. It is important to remember that all fires are considered wildland fires under the current policy and receive a management response appropriate for the specific set of circumstances.

After reviewing the various appropriate management responses applied to fires in the northern Rockies in 1998, these responses can be categorized in tactical groups, as described by Zimmerman (in press). These include monitoring from a distance, monitoring on-site, confinement, monitoring plus

contingency actions, monitoring plus mitigation actions, initial attack, large fire suppression with multiple strategies, and control and extinguishment. These appropriate management response groups are defined as:

- Monitoring from a distance—fire situations where inactive behavior and low threats required only periodic monitoring from a nearby high point, lookout or aircraft.
- Monitoring on-site—fires where circumstances required the physical placement of monitors on the fire site to track movement and growth.
- Confinement—actions taken when wildland fires were not viable candidates for resource benefits, and an analysis of strategic alternatives indicated threats from the fire did not require costly deployment of large numbers of resources for mitigation or suppression. These fires were managed with little or no on-the-ground activity, and fire movement remained confined within a predetermined area bounded by natural barriers or fuel changes.
- Monitoring plus contingency actions—monitoring was carried out on fires managed for resource benefits, but circumstances necessitated preparation of contingency actions to satisfy external influences and ensure adequate preparation for possible undesirable developments.
- Monitoring plus mitigation actions—actions on fires managed for resource benefits that either posed real, but not necessarily immediate, threats or did not have a totally naturally defensible boundary. These fires were monitored, but operational actions were developed and implemented to delay, direct or check fire spread, to contain the fire to a defined area, and/or to ensure public safety (through signing, information and trail and area closures).

- Initial attack—situations where an initial response was taken to suppress wildland fires, consistent with firefighter and public safety and values to be protected.
- Large fire suppression with multiple strategies—fires where a combination of tactics such as direct attack, indirect attack and confinement by natural barriers were used to accomplish protection objectives as directed in a WFSA.
- Control and extinguishment—actions taken on fires when a WFSA alternative indicated that a control strategy using direct attack was preferred. Sufficient resources were assigned to achieve control of the fire with minimum burned area.

The purpose of aggregating fires into these groups is not to create discrete types of appropriate management responses or a new classification. It is strictly an effort to further exemplify the dynamic, full range of appropriate management responses presented by the current policy. These groups do not necessarily represent all possibilities and may not be applicable to all wildland fires. They do, however, provide a useful description of the range of appropriate management responses implemented in the wilderness areas and national parks during the wildland fire activity from August to September 1998 in the northern Rocky Mountains.

Describing groups of like responses is useful because it provides more concise, understandable information such as summaries of fire information, objectives and management actions for each appropriate management response group, reduces redundancy and offers a clear image of the fire situations and subsequent management activities (fig. 4). As land use changes from wilderness to nonwilderness and multiple use, objectives for fire management also generally

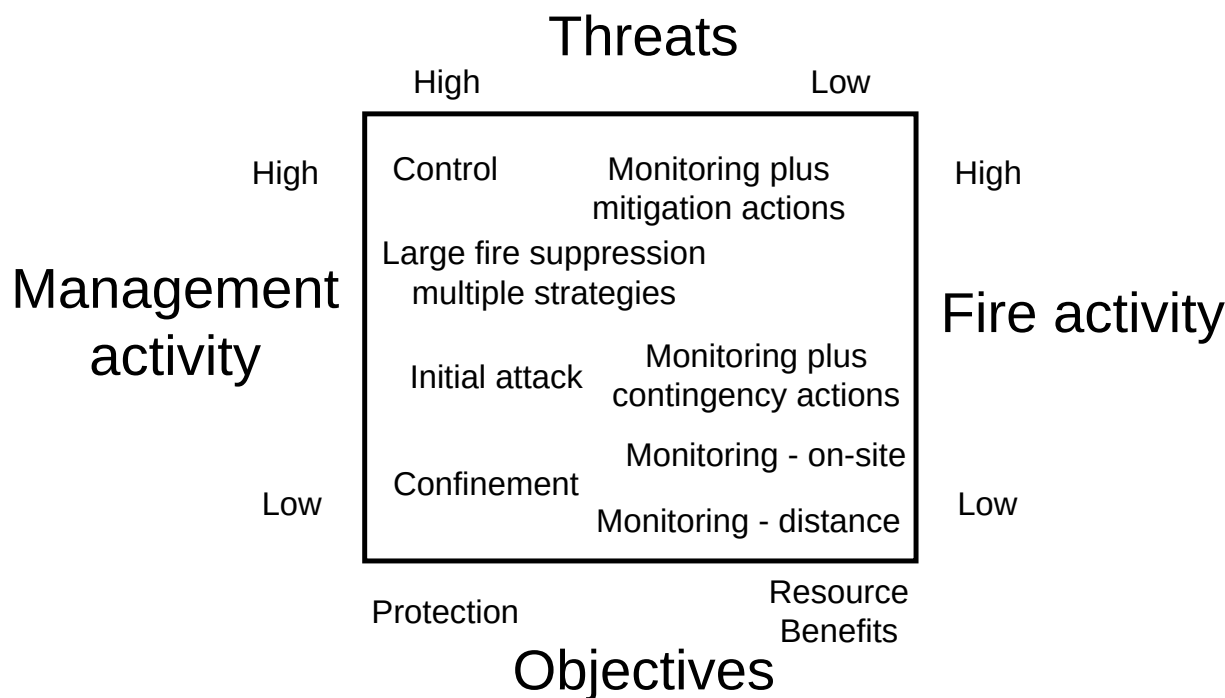


Figure 4—Appropriate management response groups applied in 1998 shown along a spectrum based on criteria of threats, fire activity, management activity, and objectives.

change from managing for resource benefits to more protection. This strongly influences appropriate management response dynamics. However, responses are not limited to one particular kind because of land use. For example, wildland fires in wilderness are not only subject to monitoring for resource benefits; they can also receive suppression responses to achieve control when necessary. In addition, within specific primary land uses, increasing threats drive appropriate management responses to include greater on-the-ground activity, both in the form of overhead and line fire management resources (fig. 4). Fire size and activity also have a major influence on the appropriate management response. Using the seven tactical management response groups identified above, it is possible to see how the appropriate management response concept presents a range of possible actions and how this was applied during August to September 1998 (fig. 4). This range indicates the flexibility available to managers under the current policy.

Summary

Wilderness and fire policies continue to be dynamic. Program management changes only after lengthy negotiation and careful deliberation following the occurrence of some significant event. The multiple deaths among people fighting fires in 1994 and lack of significant maintenance of forest and range health over large landscapes of the West have recently been noteworthy examples. This has placed management in a reactive posture. A proactive position that responds to projected needs by incorporating analysis of scientific data and social/political vagaries will place wilderness fire management on a more steady and effective course.

This course must have adequate flexibility to accommodate future uncertainty. Much of what needs to be done in wilderness fire management still lies ahead of us. Our knowledge of wilderness management needs, actions to fulfill these needs, and fire accomplishment data are lacking or seriously inadequate at best (Parsons and Landres 1998). There continues to be no interagency reporting process or database for wilderness fire occurrence or prescribed fire treatments. The nearly 2.5 million acres of wilderness that has experienced fire since 1970 seems an impressive figure, but the reality is that the bulk of the acreage comes from far too few centers of excellence such as Sequoia-Kings Canyon and Yosemite National Parks and the Selway-Bitterroot, Gila, Bob Marshall and Frank Church-River of No Return Wilderness Complexes.

Wilderness fire implementation opportunities and accomplishments will grow as federal agencies implement the 1995 Federal Wildland Fire Management. Applying an appropriate management response to all fires, rather than regulating responses by fire types, will enhance efficiency. Along with this efficiency will be more attention to ecological concerns, better responsiveness to resource management objectives, ability to better accommodate evolving objectives, more effective assignment and use of limited resources, and the most efficient expenditure of funds.

The Federal Fire Policy provides increased emphasis on consistent implementation of program elements across agency boundaries. This reduces barriers to accomplishment when joint planning efforts take full advantage of this direction. Noteworthy examples include the current effort underway consolidating a management plan for the Flathead National

Forest and Glacier National Park and the potential for a joint Yellowstone National Park and Gallatin National Forest plan.

This policy also directs changes in funding that clearly will enhance wilderness fire management and promote increased allocations in prescribed fire programs. It will require a significant increase in the combination of wildland fire use and prescribed fire application to restore many fire-dependent ecosystem components to maintenance levels.

Perhaps the most significant long-term effect of implementing the policy can be found in increased interagency cooperation, acceptance and trust. The final approval by federal agencies of the implementation procedures reference guide (Zimmerman and Bunnell 1998) heralds a major step forward and potentially ensures that increased use of fire will become a reality when the next revision/amendment of land and resource management plans is completed.

Wilderness managers have a unique opportunity to capitalize on a fire management policy and program change that provides far greater flexibility than ever before. This policy allows for better balance in management responses to fires and can meet many wilderness goals and objectives. There are no meaningful elements more pervasive in wilderness than natural processes, including fire. Complete implementation of the fire policy will require wilderness managers to redeem their management responsibility to both plan for and implement full use of fire in wilderness and facilitate growth and advances in program management.

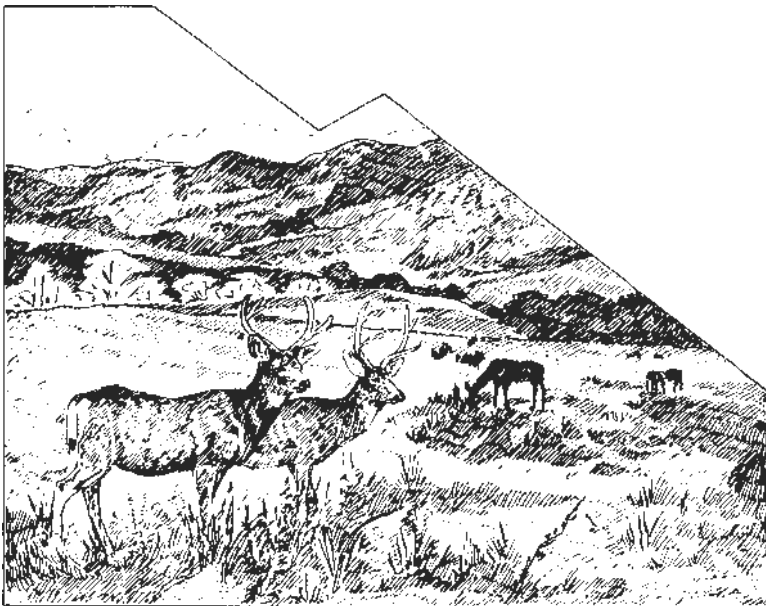
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References

- Parsons, David, J., and Peter B. Landres. 1998. Restoring natural fire to wilderness: how are we doing? Pages 366-373 in Teresa L. Pruden and Leonard A. Brennan (eds.). *Fire in ecosystem management: shifting the paradigm from suppression to prescription*. Tall Timbers Fire Ecology Conference Proceedings, No. 20. Tall Timbers Research Station, Tallahassee, FL.
- U.S. Department of Agriculture and U.S. Department of the Interior. 1989. *Final Report on Fire Management Policy*. Washington, D.C. 20 p.
- U. S. Department of the Interior/U. S. Department of Agriculture. 1995. *Federal wildland fire management policy and program review*. Final Report. Boise, ID: National Interagency Fire Center; 45 p.
- U.S. General Accounting Office. 1990. *Report to the Chairman, Environment, Energy, and Natural Resources Subcommittee on Government Operations, House of Representatives. Federal Fire Management, Limited Progress in Restarting the Prescribed Fire Program*. GAO/RCED-91-42. Washington, D.C. 15 p.
- Williams, J.T., L. Bradshaw, G. Day, C. Denton, R. Dunton, R. Gale, S. Haglund, N. Hitchcock, P. Keller, M. Long, G. Schmidt, C. Seely, and L. Wright. 1998. *Implementation of the Federal Wildland Fire Management Policy, Allocation of Resources*. Final Report to National Wildfire Coordinating Group. 29 p.
- Zimmerman, G. Thomas, and David L. Bunnell. 1998. *Wildland and Prescribed Fire Management Policy, Implementation Procedures Reference Guide*. National Park Service, USDA Forest Service, Bureau of Indian Affairs, U.S. Fish and Wildlife Service, and Bureau of Land Management, Boise, ID. 154 p.
- Zimmerman, G. Thomas. In press. *Appropriate Management Responses to Wildland Fire: Options and Costs*. Symposium on Fire Economics, Planning, and Policy, Bottom Lines, Proc. Riverside, CA. U.S. Department of Agriculture, Forest Service, Pacific Southwest Experiment Station.

5. Air, Water, and Exotic Species



Fish Stocking in Protected Areas: Summary of a Workshop

Paul Stephen Corn
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Abstract—Native and nonnative sport fish have been introduced into the majority of historically fishless lakes in wilderness, generating conflicts between managing wilderness as natural ecosystems and providing opportunities for recreation. Managers faced with controversial and difficult decisions about how to manage wilderness lakes may not always have ready access to research relevant to these decisions. To address this problem, and to expose scientists to the concerns and constraints of managers and wilderness users, a workshop was held in October 1998 at the Flathead Lake Biological Station in Polson, Montana. Participants included 43 scientists, state and federal managers, wilderness users and advocates and students. Four subject areas were addressed: federal, state, tribal and user perspectives, community and ecosystem effects, species effects and management recommendations. Papers from the workshop are being developed for an issue of the journal *Ecosystems*.

The conflicts between managing wilderness as “natural” ecosystems and providing opportunities for recreation are especially acute in fisheries management. Native and nonnative sport fish have been introduced into the majority of historically fishless lakes in wilderness (Bahls 1992), usually to the detriment of the native biota (Bradford and others 1993; Chess and others 1993; Tyler and others 1998). Alpine lakes are the primary target for recreation in wilderness (Hendee and Schoenfeld 1990), and fishing opportunities may further concentrate use in these areas, resulting in resource damage and compromising solitude in the wilderness experience. Fish stocking, especially using aircraft, is also considered to conflict with wilderness values (Duff 1995).

However, fish stocking in mountain lakes long predates the Wilderness Act of 1964, and fishing is the objective of a sizable proportion of wilderness visitors (Fraley 1996; Hendee and Schoenfeld 1990). Language in the Wilderness Act, reserving the rights of the States with respect to management of fish and wildlife, is often cited as justification for continued active management of fisheries in wilderness (Duff 1995; Fraley 1996). Conversely, other language in the Wilderness Act promoting the preservation of natural systems, and increasing emphasis on wilderness as a reference point for the study and management of ecosystems

(Hendee and others 1990; Kaufmann and others 1994) are difficult to reconcile with many of the current practices of fisheries management.

Consequently, managers are faced with controversial and difficult decisions about how to manage wilderness lakes, and they do not always have ready access to research relevant to these decisions. Considerable research has been conducted recently on the biological effects of fish stocking on resident biota. Many managers tend to minimize these effects, however, instead promoting untested alternative hypotheses (Fraley 1996). Thus, we organized a workshop, held for three days in October 1998 at The University of Montana Flathead Lake Biological Station.

The objectives were to present wilderness managers with the latest research results and management recommendations on the effects of fish introductions on wilderness lakes; to encourage discussion of issues, areas of agreement, conflicts and recommendations for future management and research among managers, scientists and wilderness and recreation users; and to publish a compilation of research results and management recommendations that will be useful for scientists and managers, alike.

The workshop was organized into four sessions, which included formal presentations and a block of time for group discussion. The workshop began with an overview of fish stocking in wilderness from federal, state, tribal and user perspectives, including summaries of key legislation, policy and description of current management practices. A session on community and ecosystem effects included effects of fish stocking on lake nutrient cycling, algal dynamics and invertebrates and interactions between predators, hydroperiod and amphibians. The third session focused on effects on vertebrate species and included discussions on effects of stocking on native fish and amphibians. The final session described restoration and management. This paper briefly describes the presentations and summarizes the findings and comments from the discussions. The complete agenda and abstracts can be found at the Aldo Leopold Wilderness Research Institute's web site (www.wilderness.net/leopold/bulletin.htm).

Participants

Participation in the workshop was by invitation to try to achieve representation by scientists, managers and interested wilderness users and advocates and to keep the size of the meeting small enough for productive discussions. Organizations represented by the 43 participants included the National Park Service (2 participants), U. S. Fish and Wildlife Service (2), U. S. Forest Service (9), U. S. Geological Survey (4), California Department of Fish and Game (1),

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Montana Fish Wildlife and Parks (3), Oregon Department of Fish and Wildlife (1), Washington Department of Fish and Wildlife (1), Confederated Salish and Kootenai Tribes (1), University of California (1), Colorado State University (1), Idaho State University (2), The University of Montana (4), Salish Kootenai College (2), Savannah River Ecology Laboratory (1), Utah State University (2), University of Washington (1), Alliance for the Wild Rockies (1), Trail Blazers (1), Trout Unlimited (1), Wilderness Watch (1) and a private consultant (1).

Overview of Fish Stocking Policies and Attitudes

The workshop emphasized the biological effects of fish stocking, mostly, but not exclusively, in western North America. First we reviewed the history of the issue, current policies of the various management agencies and the views of wilderness users and advocates. We began with an introduction to the issue by Bruce Bury (U. S. Geological Survey) and continued with an overview of federal viewpoints. Sue Matthews (U. S. Fish and Wildlife Service and Arthur Carhart National Wilderness Training Center) reviewed the Wilderness Act and the issue of federal versus state control of fisheries management in wilderness. Linda Ulmer (U. S. Forest Service) summarized Forest Service policy guidance on fish stocking in wilderness and Bruce Freet (National Park Service) described the history and controversy of fish stocking in the creation and management of North Cascades National Park.

Next, there were talks on the policies of states and tribes, including Montana (James Satterfield, Jr., Montana Fish, Wildlife and Parks), Washington (James Johnston, Washington Department of Fish and Wildlife), Oregon (Terry Farrell, Oregon Department of Fish and Wildlife), California (Betsy Bolster, California Department of Fish and Game) and the Flathead Indian Reservation (Joe Dos Santos, Confederated Salish and Kootenai Tribes).

Lastly, we heard from several conservation organizations. Michael Swayne (Trail Blazers, Seattle) described his group's efforts during the past 65 years to assist the State of Washington with stocking wilderness lakes, conducting fish surveys and maintaining a database of high-elevation lakes. Bruce Farling (Trout Unlimited) described his organization's desire to emphasize science and wild fish management in wilderness. George Nickas (Wilderness Watch) stated that nonnative species should not be introduced into wilderness and that fish stocking is generally at odds with wilderness values.

Considerable information was presented in this session and lively discussion followed. One major point was that there is no single definition of what constitutes an indigenous species of fish, with differences between state and federal policies and even internally among Forest Service documents. This is clearly contributes to the greater problem that there is no clear or consistent set of policies for how federal and state agencies cooperate, an issue also discussed by Duff (1995) and Fraley (1996). However, participants generally agreed that cooperation and objective research, rather than conflict and litigation, was necessary to effectively manage fisheries in wilderness. The point was well

made that the public doesn't care about the squabbles among agencies.

Community and Ecosystem Effects

The second session began with Daniel Schindler (University of Washington) describing changes to nutrient cycling and algal dynamics resulting from fish introductions. Brook trout introduced into a fishless lake in Banff, Alberta, altered grazing on phytoplankton by eliminating large zooplankton, resulting in an increase in primary productivity. Food webs were altered, with nutrients, particularly phosphorus, transported from storage in the benthos into the pelagic zone. Charles Hawkins (Utah State University) reported results from a study of zooplankton and macroinvertebrates in 48 lakes in the Uintah Mountains in Utah (Carlisle and Hawkins 1998). The study included three predator regimes—no fish, brook trout and cutthroat trout; and three habitat types—sand, cobble and macrophyte-dominated substrates. Differences among lakes were not due to differences in structural complexity. Lakes with fish had smaller zooplankton and few macroinvertebrates compared with fishless lakes. Joel Snodgrass (Savannah River Ecology Laboratory) described interactions between fish and amphibians in Carolina bays, which are small depression ponds on the Atlantic Coastal Plain. These ponds are typically temporary, but ditches now connect many to creeks and rivers, and fish have colonized some of them. A diverse amphibian fauna occurs in this area, but presence at a pond depends on amphibian body size, presence of fish and hydroperiod. For example, small-bodied salamanders are restricted to temporary ponds without fish, while large-bodied species may occur in more permanent ponds containing fish.

Effects on Vertebrates

Ted Koch (U.S. Fish and Wildlife Service) began this session with an overview of the Endangered Species Act, including factors leading to listing and procedures to be followed in interactions with other federal and state agencies. He pointed out that the states have the primary legal responsibility for managing fish and wildlife, but the federal role has been growing since the Lacey Act of 1906 and the Migratory Bird Act of 1918. Effective use of the Endangered Species Act for conservation is often hampered by poor understanding of taxonomic relationships (including ability to define distinct population segments), poor understanding of species' status and difficulty in monitoring trends of most species.

Christopher Frissell and Susan Adams (The University of Montana) described the effects of stocking on native fish. Several widely-distributed species, including bull trout, west slope cutthroat trout and Yellowstone cutthroat trout are threatened by habitat destruction and the stocking of nonnative trout. Interactions between native and nonnative trout include predation, competition, disease, hybridization and effects on food webs. Native trout have been largely extirpated from lower elevation waters and secure habitats are predominately in nonwilderness roadless areas. Stocking nonnative trout into headwater lakes can have severe

consequences, because there are few barriers to downstream migration.

Michael Adams (U. S. Geological Survey) observed that fish are often overlooked as a cause of amphibian declines in low-elevation, nonwilderness habitats. Largemouth bass, nonnative to the western U. S., have a negative effect on native frogs, particularly if bullfrogs (another nonnative species) are present. At a landscape scale, habitat gradients like those studied by J. Snodgrass in South Carolina may allow native amphibians to persist. Kathleen Matthews (U. S. Forest Service) described comparisons of amphibian and fish distributions between the John Muir Wilderness and Kings Canyon National Park in the Sierra Nevada in California. Surveys of 2162 lakes from 1995 to 1997 found fewer lakes with fish in the Park (where stocking was terminated in 1977). Mountain yellow-legged frogs were more common in the Park and rare in the adjacent John Muir Wilderness, where stocking continues. David Pilliod (Idaho State University) has studied Columbia spotted frogs in 73 lakes in the Frank Church-River of No Return Wilderness in Idaho. Adult frogs were found in equal numbers at lakes with and without fish, but reproduction was successful only in a small number of small ponds without fish. He suggested that the removal of fish populations to restore frogs should be done only at sites that would derive the greatest benefit.

Restoration and Management

The session on restoration and management included descriptions of current management practices and a proposal for a watershed-based reserve for native species in the Sierra Nevada. Amy Harig (Colorado State University) described attempts to restore lakes in the Adirondack Mountains in New York, where acid rain and introduced fish (perch and planktivorous cyprinids) have negatively affected native fish communities. Several measures of zooplankton, phytoplankton and native fish were combined for an index of biological integrity to judge success of restoration efforts. James Johnston (Washington Department of Fish and Wildlife) and James Darling (Montana Fish, Wildlife and Parks) described current fish management in the northern Cascades in Washington and the Beartooth Plateau in Montana, respectively. Finally, Roland Knapp (University of California) presented a proposal to restore populations of mountain yellow-legged frogs and macroinvertebrates in four watersheds in the John Muir Wilderness. Of 130 lakes, 117 currently have trout and small populations of frogs occur at the other 13. Selective removal of trout from 16 lakes would result in improved breeding habitat for frogs and greater connectivity among frog populations. Implementation of this restoration project is currently underway.

Products

Several papers, based on presentations at the workshop, are currently being developed. These will be submitted as a group to the journal *Ecosystems*, intended as a special feature. Our goal is for papers to be submitted by the end of 1999, with publication by mid-2000. Participants in the workshop were enthusiastic about the information presented and the discussions that followed. We hope that the published papers will bring this information to the larger scientific and management communities.

References

- Bahls, Peter. 1992. The status of fish populations and management of high mountain lakes in the western United States. *Northwest Science* 66: 183–193.
- Bradford, David F.; Tabatabai, Farinaz; Graber, David M. 1993. Isolation of remaining populations of the native frog, *Rana muscosa*, by introduced fishes in Sequoia and Kings Canyon National Parks, California. *Conservation Biology* 7: 882–888.
- Carlisle, Daren M.; Hawkins, Charles P. 1998. Relationships between invertebrate assemblage structure, 2 trout species, and habitat structure in Utah mountain lakes. *Journal of the North American Benthological Society* 17:286–300.
- Chess, D. F.; Gibson, F.; White, R. J. 1993. The introduction of Lahontan cutthroat trout into a previously fishless lake: feeding habits and effects upon the zooplankton and benthic community. *Journal of Freshwater Ecology* 8: 215–225.
- Duff, Donald A. 1995. Fish stocking in U. S. Federal wilderness areas—challenges and opportunities. *International Journal of Wilderness* 1: 17–19.
- Fraley, John. 1996. Cooperation and controversy in wilderness fisheries management. *Fisheries* 21(5): 16–21.
- Harig, Amy L.; Bain, Mark R. 1998. Defining and restoring biological integrity in wilderness lakes. *Ecological Applications* 8:71–87.
- Hendee, John C.; Schoenfeld, Clay. 1990. Wildlife in wilderness. In: Hendee, John C.; Stankey, George H.; Lucas, Robert C. *Wilderness management*. 2nd edition. Golden, CO: North American Press: 263–294.
- Hendee, John C.; Stankey, George H.; Lucas, Robert C. *Wilderness management*. 2nd edition. Golden, CO: North American Press. 546 p.
- Kaufmann, Merrill R.; Graham, Russell T.; Boyce, Douglas A., Jr.; Moir, William H.; Perry, Lee; Reynolds, Richard T.; Bassett, Richard L.; Mehlhop, Patricia; Edminster, Carlton B.; Block, William M.; Corn, Paul Stephen. 1994. An ecological basis for ecosystem management. Ft. Collins, CO: U. S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. General Technical Report RM-246. 22 p.
- Knapp, Roland A.; Matthews Kathleen R. In Press. Nonnative fish introductions and the decline of the mountain yellow-legged frog from within protected areas. *Conservation Biology*.
- Tyler, Torrey; Liss, William J.; Ganio, Lisa M.; Larson, Gary L.; Hoffman, Robert; Demling, Elisabeth; Lomnický, Gregg. 1998. Interaction between introduced trout and larval salamanders (*Ambystoma macrodactylum*) in high-elevation lakes. *Conservation Biology* 12:94–105.

Visitor Perceptions and Valuation of Visibility in the Great Gulf Wilderness, New Hampshire

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John M. Halstead
Thomas H. Stevens
Ina Porras
Kenneth D. Kimball

Abstract—New Hampshire's White Mountain National Forest is well known for its mountain scenery and its diverse outdoor recreational opportunities. Within The Forest are two federally protected Class I wilderness areas, the Great Gulf Wilderness, and the Presidential Dry-River Wilderness. The expansive scenic vistas from these two wilderness areas are commonly impaired by regional haze, largely a byproduct of fossil fuel electric energy production upwind of the region. Consumer choice of electric suppliers, the U.S. Environmental Protection Agency's 1999 regional haze regulations, and other regional emissions reductions programs may work to change visibility in the White Mountain National Forest. This paper characterizes existing visibility conditions in the Great Gulf Wilderness, and outlines the design and preliminary results of an ongoing study of visitor perceptions of visibility. The objective of the study is to understand: a) visibility conditions in the Great Gulf Wilderness, b) the sensitivity of visitors to haze, and c) the economic value of potential visibility changes to visitors.

The Appalachian Mountain Club (AMC) and the White Mountain National Forest (WMNF) have jointly examined regional haze-related visibility impairment and its causes in New Hampshire's Great Gulf Wilderness for over a decade (Hill and others 1996). The Great Gulf Wilderness, located in northern New Hampshire (fig. 1), is one of two federally protected Class I airsheds in the White Mountain National Forest, and one of only seven in the Northeast. The Wilderness lies just north of the summit of Mount Washington, the highest peak in New England (6,288 feet). The approximate quarter-million visitors to the summit of Mount Washington travel by car, mountain train or by foot to see the breathtaking views. Approximately seven million visitor days are logged in the White Mountains annually. On a perfectly

clear day, one can see West across the state of Vermont to the Adirondack Mountains in New York State—130 miles distant or east to the Atlantic Ocean. However, on very hazy days, nearby peaks become indistinct, and the scenic vistas from the summit lose clarity, color and contrast. Under these conditions, the closest towns, approximately 7-17 miles distant, may disappear into the haze altogether, seriously degrading the quality of the wilderness experience for some visitors.

In the eastern United States, the annual mean visibility is estimated at 18-40 miles (EPA 1997). Visual range in New England's Class I airsheds (measured in Acadia National Park, ME, Lye Brook Wilderness, VT, Great Gulf Wilderness, NH) is generally about 35 miles compared to about 20 miles in the mid-Atlantic and southern United States (EPA 1998). The poorest mean annual visibility for Class I areas in the United States is 18 miles, recorded in Shenandoah National Park (VA), Mammoth Cave National Park (KY) and Great Smoky Mountains National Park (NC, TN).

As compared to estimated natural conditions, the visibility in the entire Eastern United States is significantly



Figure 1—Visibility image, Great Gulf Wilderness as if viewed from visibility camera location at Camp Dodge, Pinkham Notch, NH. Image produced using WinHaze 2.7.0 (Air Quality Specialists Inc.) Image represents natural visibility juxtaposed with the 80th percentile summer time visibility value.

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impaired. One estimate of median natural visibility is given by Trijonis (1982), 60 miles plus or minus 30 miles. EPA (1998) estimates mean natural visibility to be about 80-90 miles, which takes into account natural organic haze in the Southeast. In New England, due to less stagnant atmospheric conditions, average natural visibility may be higher, in the range of 90-120 miles. Thus, comparing current visibility with estimated natural average visibility, current visual range is about one quarter to third of estimated natural visual range in the eastern United States. In addition, current trends in visibility conditions on the haziest days at many eastern Class I airsheds suggest little or no improvement in visibility (Sisler and Damburg 1997) despite national reductions in sulfur dioxide emissions, from 23.2 million tons in 1988 to 20.4 million tons in 1997, as reported by EPA (1998).

Typically, visitors come to the White Mountains of New Hampshire from the Boston, New York and Montreal metropolitan areas for respite from the urban life. Visitors to these areas reasonably expect fresh clean air and crystal clear vistas. In reality, however, some days are as smoggy as the urban areas from which they came. In a three year study of hikers to the summit of Mount Washington, AMC, Harvard School of Public Health, and Brigham and Women's Hospital demonstrated measurable reductions in short-term lung function at levels well below the national ambient air quality standards (Korrick and others 1998). From an early unpublished study of hiker's response to photographs of visibility conditions (Kimball, unpublished data, 1989) it was apparent that the same smog that affects hiker health could further diminish the quality of the wilderness experience, resulting from haze-impaired vistas. This paper focuses on our investigations into regional haze in the Great Gulf Wilderness, its potential effects on visitors and how visitors value visibility in the wilderness area.

To understand how people perceive visual air quality in a wilderness area, the AMC piloted its study in the Great Gulf Wilderness in 1996 based on the Denver visibility study (Ely 1991). The objective of the study was to determine: a) if people could distinguish between a continuum of hazy and clear vistas, b) the acceptability of haze to visitors to a wilderness area, and c) whether people may be willing to pay for cleaner air in these areas. The study was joined and broadened in 1997 by University of New Hampshire and University of Massachusetts economists interested in how visitors and people off-site value visibility in the Great Gulf Wilderness (Porras 1999). The following briefly describes: a) research on the causes of visibility impairment in the Great Gulf Wilderness and b) the design and preliminary results of a wilderness visibility perception/valuation study.

Characterizing Visibility Impairment and Its Causes in the Great Gulf Wilderness, 1988-1998

Section 169 of the Clean Air Act requires federal land managers to protect federal wilderness areas, national parks and national wildlife refuges, designated as Class I, from visibility impairment. As a part of this obligation, a visibility monitoring program was established in the Great Gulf

Wilderness in 1985 by the White Mountain National Forest. A camera designed to assess visibility conditions was installed near the Great Gulf Wilderness in 1985 which was subsequently supplemented with AMC air quality monitoring in 1988 under a partnership with the White Mountain National Forest to characterize visual air quality conditions. The AMC's monitoring program and results are briefly outlined below as a context for understanding the preliminary results and significance of the visibility survey.

Visibility Monitoring Methods

The Great Gulf Wilderness monitoring site is located at Camp Dodge in Pinkham Notch, New Hampshire. The site is located in an active AMC volunteer trails management facility adjacent to the Great Gulf Wilderness, a glaciated valley surrounded by the steep headwalls and ravines of the Northern Presidential Range (fig. 2).

From its installation by WMNF in 1985 until its elimination in 1997, the visibility camera was automated to take three daily photographs (at 9:00 AM, noon and 3:00 PM) of the visibility scene "target," Mount Jefferson (5,712 feet). Mount Jefferson is situated along the western border of the Great Gulf wilderness approximately 4.4 miles from the camera. The photographic monitoring was undertaken typically from mid-May until late September/early October. For each of the visibility photographs, systematic estimates of standard visual range (SVR), an empirical measure of visibility generally expressed in kilometers, were

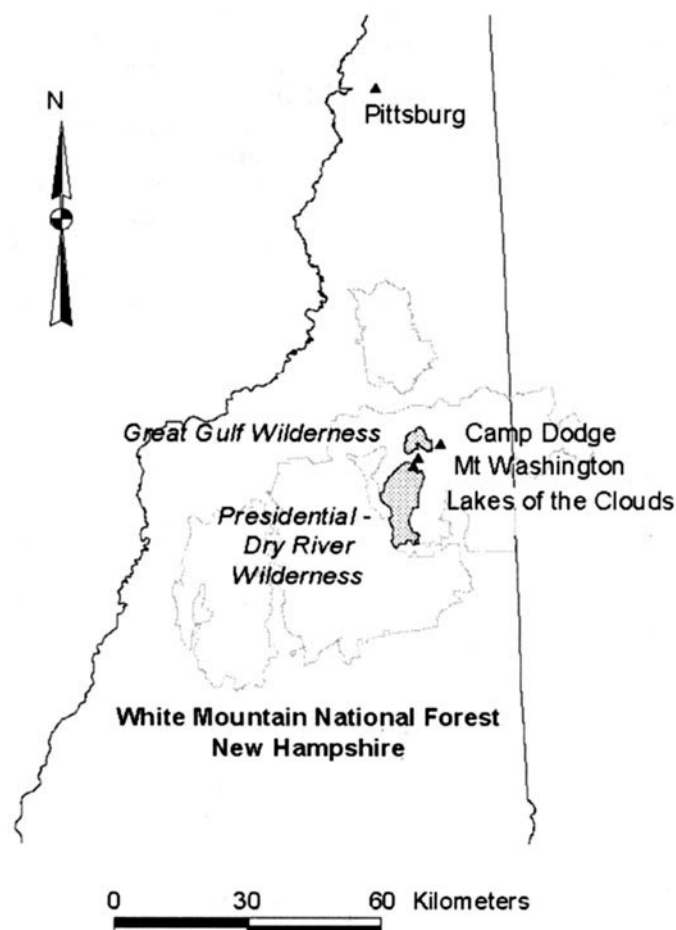


Figure 2—Location map showing AMC and cooperators' four air quality monitoring sites in northern New Hampshire (Pittsburg site: ozone; Camp Dodge site: PM_{2.5}, ozone and full IMPROVE; Mount Washington site: ozone; Lakes of the Clouds site: PM_{2.5}, acid precipitation).

determined using a scanning densitometer (NRC 1993). In 1997 the camera measurements were superseded by a nephelometer (a continuous electronic visibility measurement device based on light scattering).

To determine the causes of visibility impairment, particularly the relationship between visual range and particulate matter aerosols in the Great Gulf, a fine particle monitor (PM_{2.5}) was colocated with the visibility camera at Camp Dodge in 1988. Further, in order to explore high elevation air quality conditions a high-elevation fine particle monitor was sited in at the AMC's Lakes of the Clouds Hut in 1990 at 5,050 feet elevation on Mount Washington. From 1988 until 1997, AMC acquired fine particle (PM_{2.5}) samples from mid-June through August. Samples were typically collected from 7 AM to 5 PM daily (fewer samples in the first few years of operation)—during the daytime hours when visitors are typically hiking or sightseeing. In 1998, daily, consecutive 24-hour sampling replaced 10-hour daytime sampling in the program following the fine particulate matter monitoring protocol introduced as part of the 1997 PM_{2.5} air quality standards. Fine particle measurements at Camp Dodge use a the Harvard/Turner Impactor (HI) to fractionate the sample, effectively removing all particles larger than 2.5 microns in diameter but collecting the fine particles, less than 2.5 microns, on a pre-weighed Teflo filter inside the HI monitor. Mass per unit volume of air for each filter (in micrograms per cubic meter of air—ug/m³) were subsequently determined from: 1) the measurement of fine particle mass on a filter, and 2) the measured volume of air pumped across the filter. Following these gravimetric analyses, sulfate mass, aerosol acidity and ammonium concentration (degree of neutralization) were measured from the same filters.

In 1996 WMNF added a full IMPROVE (Interagency Monitoring for Impaired Visual Environments) monitoring site at Camp Dodge. IMPROVE protocol includes coarse and fine particle monitors (PM₁₀ and PM_{2.5}), and an Optek nephelometer (a visibility monitor that continuously measures light caused by scattering and absorption of particles and gases). In addition to the fine particle and visibility monitoring, AMC and WMNF conduct tropospheric ozone monitoring at the same site.

Visibility Monitoring Results

Figure 3 shows the distribution of daily summer visibility measurements from 1988-1996. These data represent visibility measurements from both photographic and electronic (nephelometer) methods when PM_{2.5} was monitored in that 9 year period (369 days). Note that this does not represent all of the days visibility was monitored—only the days when fine particles measurements are available—and therefore these data only represent the approximate conditions in the wilderness area during the summer months. For these days, the median summertime daily visibility was 15 deciviews (87 km/54 mi.), with a maximum (poorest visibility) of 39 deciviews (8 km/5 mi.).

1988-1998 PM_{2.5} data is summarized in figure 4. For simplicity this box plot combines 10-hour daytime sample data from 1988-1997 with the 24 hour sample data acquired beginning in 1998. Based on these data, PM_{2.5} concentrations in the Wilderness have been measured as high as 86 micrograms per cubic meter of air over 10 hours (86 ug/m³) in comparison to the 24-hour national standard of 65 ug/m³. The approximate summer mean for continuous PM_{2.5} monitoring ranges from 9.5-15.0 ug/m³ (Hill and others 1996) as compared to mean 1996 summer conditions in Boston of 14.4 ug/m³ (Unpublished data, courtesy Harvard School of Public

Visibility Distribution 1988-1996 Great Gulf Wilderness

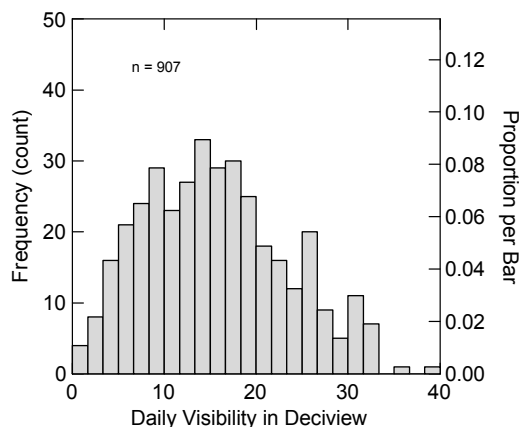


Figure 3—Distribution of average daily visibility measurements 1988-1996.

Health, Boston Edison). Chemical analyses suggest that the dominant particle phase in the fine mass is the partially neutralized form of sulfate, ammonium bisulfate (Hill and others 1996).

Figure 5 is a plot of PM_{2.5} versus visibility, where visibility is measured in deciview (for explanation of deciview scale see figure 5 caption). The graph shows a clear cause and effect relationship between the dependant variable, visibility, and the independent variable PM_{2.5}. Moreover, the relationship has a positive slope which demonstrates that an increase in fine particulate matter is accompanied by a systematic, although somewhat non-linear, decrease in perceived visibility in the Great Gulf Wilderness (note that changes in deciview of 1-2 increments are approximately “just noticeable” and that the deciview scale is linear to human perception). Correlations between visibility and sulfate also show an even stronger predictable relationship between sulfate compounds in the Great Gulf Wilderness (Hill and others 1996) To summarize, average visibility in the Great Gulf Wilderness is approximately one third of estimated natural conditions, impaired by anthropogenic aerosol particles, which, in turn, are dominated by hygroscopic (moisture-absorbing) sulfate compounds.

Visibility Perception Study: Acceptability Survey

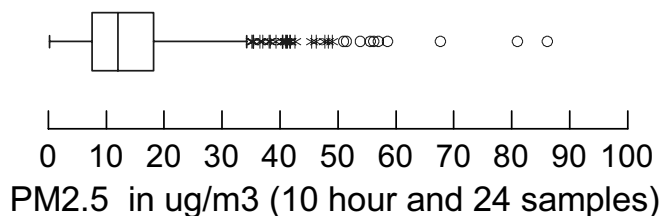


Figure 4—Distribution of PM_{2.5} Concentrations, Great Gulf Wilderness 1988-1998, combining daytime 10 hour and 24 hour samples.

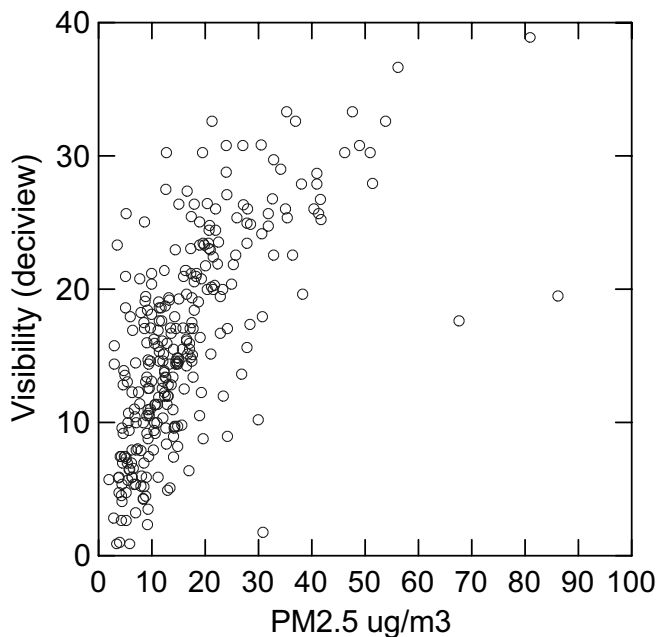


Figure 5—Relationship of visibility in deciview to fine particle mass, Great Gulf Wilderness. For reference, deciview (dv)= $10 \ln(\text{bext}/0.01 \text{ km}^{-1})$ where bext is the coefficient of light scattering expressed in inverse kilometers (Pitchford and Malm, 1994). (For example, 0 dv = 391 km, 10 dv = 144 km = 53 km, 30 dv = 19 km, 40 dv = 7 km.)

In the 1977 amendments to the Clean Air Act, Congress established a goal of remedying visibility impairment in wilderness areas, national parks and national wildlife refuges federally designated as “Class I.” Yet little action was taken in meeting this goal in the first two decades after the goal was established. However, in April 1999, the U.S. Environmental Protection Agency promulgated the “regional haze rule” establishing a flexible timeline for states to implement programs to bring visual air quality in Class I areas back to natural conditions in 60 years. Anticipating the development of these recently published rules in 1995, the AMC designed a pilot visibility perception survey to investigate visitor awareness of haze using photographs of a range of visibility conditions in the Great Gulf Wilderness. If meaningful, the White Mountain National Forest, land manager for the Great Gulf and Presidential Dry River Wilderness areas, could use the results of the survey as guidance in establishing visibility as an “air quality related value” (AQRV). AQRVs are resources sensitive or in some way related to air quality conditions in a Class I airshed. As an AQRV, a threshold of “unacceptable” visibility could be established to screen, for recommended approval or denial, permit applications for new and modified smokestacks in the vicinity of the wilderness areas. As an analogy, ozone and acid deposition are current AQRVs in the Great Gulf Wilderness with established limits called “red line values” (which effectively operate as ozone and acid deposition standards). Exceeding an established AQRV “red line” value, permits for new and modified plants emitting pollutants that cause haze could be recommended for denial by the federal land manager. This study could help determine at what point the red line value might be set.

Visibility Survey Methods

To investigate the sensitivity of visitors to these protected areas, the AMC embarked on a pilot study in 1996 to see if visitors could consistently rate and rank changes in visual air quality. The study was

continued with largely the same protocol in 1997 and then modified into a digital format in 1998. The 1996 pilot study was designed to determine by survey:

- 1) if forest visitors could consistently distinguish, rate and rank photographs of a spectrum of visibility conditions, and
- 2) if respondents perceived visual range as “unacceptable” at some consistent value when viewing clear to haze-obscurd vistas of Mount Jefferson in the Great Gulf Wilderness.

The survey was initially designed after the Denver Visibility Study (Ely and others 1991). In our pilot field study (Harper and others 1997), visitors viewed a suite of 23 photographs of the wilderness scene. They were told that they were participating in a study of “how people perceive visibility conditions in wilderness areas” and that “the photographs in the binder represent a range of conditions in the Great Gulf Wilderness.” In addition, participants were advised that “your responses will be used to develop visibility standards in wilderness areas and to assess the economic impact of visibility changes in the area.” Participants were asked to “decide if the amount of haze depicted in the photograph would be acceptable or unacceptable under your standard.”

The 5 x 7 images of the Great Gulf Wilderness scene were printed, with careful control of contrast, from visibility slides obtained from the White Mountain National Forest. Images were viewed individually by flipping through individual photos mounted on a white background over so that side-by-side was not possible. They were, however, allowed to flip back through. First, as a warm up, participants in the survey rated 5 photos, representing the range of visibility conditions in the following section of the survey, on a scale of 1-5 (where 1 is clear and 5 is most hazy). In the second section of the survey, participants rated a series of 23 photographs on the same scale. Finally, participants were asked to go back through the same suite of 23 photos, and rate each as either “acceptable” or “unacceptable.”

The survey was conducted at three sites. The primary site, using a trained interviewer, was located at the Tuckerman Ravine trailhead to the summit of Mount Washington at the AMC’s Pinkham Notch Visitor Center, one of the busiest trails in the White Mountain National Forest which logs over 7 million visitor days per year. Mount Washington provided an ideal location for the study because of its near proximity of the wilderness area; the summit is located less than 1 mile north of the Presidential-Dry River Class I Wilderness and about 0.25 mile south of the Great Gulf Wilderness area. A second self-service survey location was established at the summit of Mount Washington in the Mount Washington Observatory. Surveys collected at the summit self-service site presumably represented a broader demographic group, from sightseers that rode up to the summit in cars, trains and on foot. The third site was located at AMC’s Cardigan Lodge, the trailhead of a popular hike to the bald summit of Mt. Cardigan in central New Hampshire. These surveys were collected both by staff and as self-service surveys when staff were unavailable at this fairly remote location. In total, approximately 300 useable, valid surveys were collected in the 1996 pilot from the three survey sites. A parallel study was undertaken in 1999 by Porras (1999, unpublished Master’s thesis, University of Massachusetts) to examine off-site responses in Amherst Massachusetts, using virtually the same survey design and using the Great Gulf images, as described below.

The design of the survey included pairs of cloudy and cloud-free photos, at the same visibility/visual range levels to estimate the effect of clouds in confounding the perceptions of views. We concluded from the survey results that cloudy images were consistently rated as less acceptable. For example, one pair of photos with the same visibility, one cloudy and one clear at the 44 km visibility level, were tested. On the 1-5 scale, the clear image garnered a mean rating of 2.9 (rated clearer) while the cloudy photo (but with same visibility/visual range) received a mean rating of 4.0 (rated hazier), significant at the 95 percent confidence level. Moreover, the cloudy photograph received a rating of “acceptable” from 15 percent of the respondents, and the cloud-free (clear) photo was acceptable to 71 percent of the respondents. Therefore, subsequently, in the 1997 and 1998 surveys, cloudy images were eliminated to remove the observed bias. Interestingly, this result suggests that natural sky conditions (clouds) may have a negative impact of a viewer’s overall rating of a scene as well as uniform haze does. By analogy, this result also raises the question of whether respondents make decisions based on health impacts unconsciously when viewing hazy scenes although we have clearly informed them that the goal of the study was to understand “how people perceive visibility conditions in wilderness areas.”

One of the first questions to address was whether respondents could distinguish between photos representing a variety of visibility conditions and then secondly, whether they could accurately rate them, placing them indirectly in order of visual range. Using cloud-free images, 34 percent of participants ranked the images in the correct order, 63 percent ranked all but one image accurately and 88 percent correctly ranked all but two images. In addition, duplicate photographs were also used to assess the precision of the method in the first survey. Three sets of duplicate photos in the series of 23 photographs garnered similar ratings, leading us to conclude that the precision or repeatability of the method was good. Therefore, we conclude that viewers could consistently distinguish between, and accurately rank, photographic images of visibility based on the Great Gulf Wilderness/Mount Jefferson scene. As the Great Gulf scene is characterized by a short viewer to scene distance, this result suggests that, given a longer sight path to the horizon, viewers might be sensitive to smaller decrements in visibility.

In 1998, the survey was redesigned with computer modeled images using the WinHaze Visual Air Quality Modeler (Air Resource Specialists, Fort Collins, Colorado). This allowed us to generate a clear to hazy continuum of cloud-free visibility images of the Great Gulf Wilderness. Also in 1998, automated data collection by embedding scene images in a Microsoft Access database program and by subsequently collecting data using a laptop computer in the field, eliminating paper surveys and photographs.

Results

In general, the study in progress confirms the expected relationship between visibility and perception: As visibility decreases acceptability decreases. 1998 results indicate that half of all respondents (the median) found a visibility of about 20 deciviews (53 km or 33 mi.) or greater, unacceptable for the Great Gulf/Mount Jefferson vista (fig. 6). As noted above, the scene depth of the Mount Jefferson vista, approximately 5 miles is a comparatively short range with respect to many other visibility monitoring sites in Class I areas. This may introduce a bias into the acceptability results, since the image may not represent the distant features which become obscured sooner. In other words, in a scene with greater distance to the scene target being viewed, distant ridgelines would disappear into the haze before haze

may even be noticeable in a scene with a shorter distance to the viewing target. This would have the effect of shifting the unacceptability threshold to a greater visual range. From the Great Gulf Wilderness results we have learned that given a short range from viewer to scene, visitors can clearly distinguish between and rank images of a variety of visual air quality conditions. However, to further test the sensitivity of the acceptability question to the scene depth, location and visitor demographics, we are considering a control study at one or more eastern national parks, such as Acadia National Park in Maine, for summer 2000. The objective of the Acadia study would be to see how the longer distance/greater depth in the scene from Cadillac Mountain’s summit (12 miles to Blue Hill, the target, and further depth beyond) may affect the acceptability relationship derived from perceptions of the Great Gulf image. Moreover, the visitor demographics in a National Park may be quite different.

Valuation of Visibility in the Great Gulf Wilderness

Methods

Economists have long been interested in placing a value on goods that are not traded in a market setting (see for example, Mitchell and Carson 1989; Cummings Brookshire and Schulze 1986). Examples of such goods include environmental amenities such as clean air and water. In order to make informed policy decisions it is often important to understand the economic value that individuals place on environmental goods. There are two methods used by economists to value these goods, revealed preference and stated preference. Stated preference methods are survey-based and involve asking individuals directly how they value an environmental good (Boxall and others 1996). The most commonly used stated preference method is the contingent valuation method. This method asks respondents directly about their willingness to pay or willingness to accept compensation for a given change in an environmental amenity. Revealed preference methods use observations of mar-

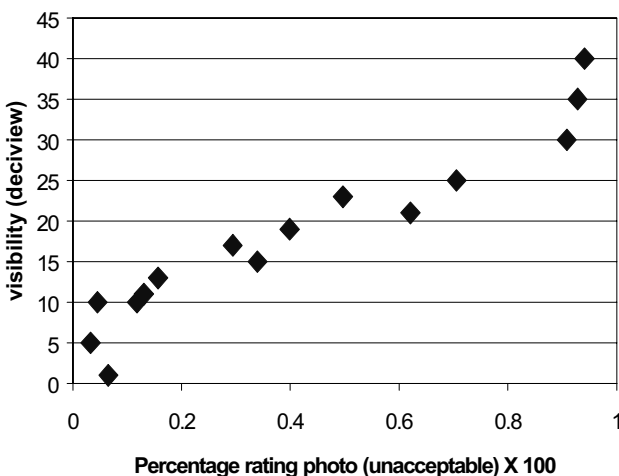


Figure 6—Percentage of respondents to visibility survey rating the Great Gulf scene unacceptable at a given visibility level in deciview.

ket behavior to infer the value that individuals place on environmental goods. For example, we might look at how much individuals will pay to travel to an environmental amenity, or we might examine differences in housing prices to infer the value of proximity to an environmental amenity. The study discussed here uses stated preference methods. While there are advantages to both methods, in this case the stated preference is the most appropriate.

The idea of a stated preference methodology was first proposed by an economist in the 1940s, however it was not put into practice until the mid 1960s with a study of hunters in the Maine woods. (Hunters were asked about the value of their experience, and their answers were then compared with values obtained from a revealed preference methodology.) Improvements on this technique have continued to be made since the mid-1970s. Many studies have focused in on clean air (for example, Brookshire and others 1985). A majority of these studies focused on vistas in the southwestern United States, primarily the Grand Canyon. In these studies, visitors and non-visitors alike were asked to state their willingness to pay to either avoid further visibility degradation or willingness to accept compensation if visibility worsened. In the willingness to pay scenarios, respondents would state their willingness to pay an increased admission fee, contribute to a special fund or pay a higher monthly electric utility bill.

In the current study, we attempt not only to value a change in the visual range, but also to compare two types of stated preference methods. These methods are the contingent valuation method (CVM), which is described above and conjoint analysis (CA). Conjoint analysis has been used widely in marketing research to determine how individuals value different attributes of a good (e.g. Green and Srinivasan, 1990). It has only recently been introduced in the environmental economics literature and this is the first study which has compared conjoint analysis and CVM to examine air quality. The conjoint method asks respondents to rate rather than directly price changes in an environmental amenity; However, the method also allows for the calculation of consumer surplus estimates comparable to the CVM (e.g. Stevens, Barrett and Willis 1997; Mackenzie 1993; Roe and Teisl 1996). In comparing these two methods we hope to gain insight into the individuals decision making process and continue to make progress in improving and refining stated preference methods.

The survey was administered during the summer of 1998 at the Pinkham Notch Visitor center in New Hampshire. Respondents were approached and asked to complete a visual image-based survey. In addition to soliciting information on how visitors value changes in the visual range, this survey collected information on respondents' perception of visual conditions as well as travel and demographic data. Both valuation questions asked respondents to make a trade off between a reduction in their monthly electric bill and degraded visibility. In the CVM question, respondents were asked if they would accept degraded visibility in exchange for a reduction in their monthly electric bill. In the CA question, they were presented two scenarios and asked to rate the two individually. Scenario A was a "status quo" scenario and Scenario B had worsened visibility and a lowered monthly electric bill. In thinking about the format of the question, it was decided that the individual should be assigned the property rights to the clean air, thus giving them the right to exchange that clean air for some monetary compensation; thus the choice of the willingness to accept wording. Also of note is the use of the payment vehicle of an electric bill. A change in electric bill has been employed in earlier visibility studies and had several advantages over other commonly used payment vehicles. Further, with electric utility deregulation upcoming in the

New England states, the use of this payment vehicle seemed to be the most realistic.

Results

Under both methods, the preliminary analysis of the data suggest that only 20% of the respondents were willing to accept a lower electric bill if it would result in hazier air. This indicates that respondents' value changes in the visual range more highly than our compensating offer. Econometric analysis was unable to explain the behavior of the respondents in any satisfactory way. There are several possible reasons for our inability to successfully capture respondents' value of a change in the visual range. One possible reason could be sample bias. As stated earlier, the survey was conducted at a major trailhead/ visitor center in the White Mountain National Forest. Simply by their presence at this location we can infer that the respondents will have a high valuation for visibility. It is possible that this particular group is not willing to make a trade-off regarding a change in visibility. A second (and related) possible explanation is limitations within the payment vehicle. The electric bill makes up at the maximum 5.8% of a respondent's income, and on average 3.3%. By limiting the realistic amount we can offer in means of compensation, we may simply not be able to offer a sufficient reduction to induce respondents to make this trade off.

A parallel off-site control study supporting these results was completed by Porras (1999, unpublished Master's thesis, University of Massachusetts) employing both personal and mail-in survey methods, similar questions and the same Great Gulf image created using the WinHaze Visual Air Quality Modeler (Air Resource Specialists, Fort Collins, Colorado). Results show that respondents in the parallel study were also able to rate and rank the images consistently. Visual ranges of 36 miles or less were unacceptable to half or more of the respondents. Based on a total of 60 personally acquired survey observations, using the CVM (contingent method), the off-site study indicated that the average electric bill reduction offer of 20 percent (average \$11.16) was insufficient to compensate most (80 percent) of the participants for the reduction in visibility in the White Mountains from 90 miles (cleaner end of annual median summer visibility) to 20 miles (approx. 90th percentile haziest condition, and deemed by most viewers as "unacceptable"). Results were similar using the CA (conjoint) method indicating that respondents could rank the images accurately, that acceptability decreases with increasing haziness and that a threshold level can be determined using this method. Based on the CA method, visual ranges of 50 miles or less are unacceptable to more than half of the viewers of the Great Gulf Wilderness image.

The mail-in survey results, (1,000 sent, 106 CVM, 106 CA= 212 responses) indicated that for an average reduction of visibility to 12 miles (the "viewed" average of 4 images used at 30, 20, 7.3, 4.4 miles), 23 percent of the respondents would accept a 35 percent decrease in their electric bill (average \$45.40 reduction) using the CVM method. Using the CA method, the average rating of 3 (where 1 is unacceptable and 10 is acceptable) suggests that respondents would not be willing to accept similarly degraded visibility regardless of the 35 percent reduction in their electric bill (average \$25.50 reduction). Eighty-seven percent of the respondents were planning to visit the White Mountains in the future, but if visibility conditions worsened, 64 percent of the CVM

respondents stated that they would be less likely to visit, while 36 percent would not change their plans. Similarly for the CA respondents, 68 percent of the 72 percent that said they were planning a visit to the White Mountains would be less likely to visit if conditions worsened.

Future Work

Visibility monitoring and human perception and econometric research was continued in summer 1999 utilizing the computer-aided survey method for the Great Gulf Wilderness. Daily PM_{2.5} and visibility monitoring was continued at both high and lower elevation sites on Mount Washington by AMC and WMNF (IMPROVE). The Great Gulf IMPROVE site is slated to be upgraded to an enhanced site and full annual operation in the near future. Sensitivity to survey question wording may be tested in future surveys (for example, testing willingness to accept versus willingness to pay). Additional data is necessary to make the results more robust. To examine potential differences in response with a different scene image and visitor demographics, a pilot project in Acadia National Park and other locations is under consideration for summer 2000.

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References

- Boxall, P.C.; Adamowicz, W.; Swait, J.; Williams, M.; Louviere, J. 1996. A comparison of stated preference methods for environmental valuation. *Ecological Economics*. 18: 243-253.
- Brookshire, D.; Schulze, W.D.; Ben-David, S.; Walther, E.; Thayer, M.A.; Whitworth, R. 1985. The benefits of preserving visibility in the National Parklands of the Southwest, Volume VIII of methods development for environmental assessment. EPA # 230-12-85-026. Washington, D.C. Office of Policy Analysis, Office of Policy Planning and Evaluation, U.S. Environmental Protection Agency.
- Cummings, R.G.; Brookshire, D.S.; Schulze, W.D. 1986. Valuing environmental goods. An assessment of the contingent valuation method. Totowa, N.J. Rowman and Allenheld.
- Duce, R.A. (Chair) 1993. Protecting Visibility in National Parks and Wilderness Areas. Washington. D.C. National Research Council. National Academy Press. 446 p.
- Ely, A.W.; Leary, J.T.; Stewart, T.R.; Ross, D.M. 1991. The establishment of the Denver visibility standard. In: proceedings of the Air and Waste Management Association, 84th Annual Meeting, June 1991, 91-48.4.
- EPA. 1998. National Air Quality and Emissions Trends Report, 1997. U.S. Environmental Protection Agency Report 454/R-98-016. Research Triangle Park, N.C. Office of Air Quality Planning and Standards.
- Green E.; Srinivasan, V. 1990. "Conjoint analysis in marketing: new developments with implications for research and practice. *Journal of Marketing*, October 3-19.
- Harper, Wendy; Hill, L. Bruce; Carlson, Joan. 1997. Perceptions and valuation of visibility: a case study in New Hampshire's White Mountain National Forest. In: Air and Waste Management Association, Pittsburgh PA. VIP-76: 589-600.
- Hill, L. Bruce; Allen, George A.; Carlson, Joan. 1996. Characterization of acid aerosols and regional Haze -related visibility impairment, Great Gulf and Presidential-Dry River Class I Wilderness Airsheds, New Hampshire. In: Air and Waste Management Association proceedings, 89th annual meeting, 1996, paper 96-MP1A.06.
- Korrick, Susan a.; Neas, Lucas M.; Dockery, Douglas W.; Gold, Diane R.; Allen, George A.; Hill, L. Bruce; Kimball, Kenneth D.; Rosner, Bernard A.; Speiser, Frank E. 1998. Effects of ozone and other pollutants on the pulmonary function of adult hikers. *Environmental Health Perspectives*. 106(2): 93-100.
- Mackenzie, J. 1993. A Comparison of Contingent Preference Models. *American Journal of Agricultural Economics*. 75(3): 593-603.
- Mitchell, R.C.; Carson, R.T. 1996. Using surveys to value public goods: the contingent valuation method. Washington, D.C. Resources for the Future.
- Pitchford, Marc L.; Malm, William C. 1994. Development of a standard visual index. *Atmospheric Environment*. 28(5): 1049-1054.
- Porras, Ina T. 1999. Off-site value of visibility: New Hampshire's White Mountains case study. Masters Thesis, University of Massachusetts, Department of Resource Economics. July 1999. 108 p.
- Roe, B.; Teisl, M. 1996. "Using conjoint analysis to derive estimates of compensating variation. *Journal of Environmental Economics and Management*. 31: 145-159.
- Sisler, James F.; Damburg, Richard. 1997. Interpretation of trends of PM_{2.5} and reconstructed visibility from the IMPROVE network. In: Visual Air Quality: Aerosols and Global Radiation Balance. Pittsburgh, PA. Air and Waste Management VIP-76: 35-44.
- Stevens, T.H.; Barrett, C.; Willis, C.E. Conjoint analysis of ground-water protection programs. *Agricultural and Resource Economics Review*. 27(2): 229-238.
- Trijonis, John 1982. Existing and natural background levels of visibility and fine particles in the rural East. *Atmospheric Environment*. 16(10): 2431-2445.

Effects of Nonnative Fishes on Wilderness Lake Ecosystems in the Sierra Nevada and Recommendations for Reducing Impacts

R. A. Knapp
K. R. Matthews

Abstract—Wilderness areas of the Sierra Nevada, California contain thousands of lakes and ponds, nearly all of which were historically fishless. After more than a century of fish stocking, introduced trout are now present in up to 80% of larger lakes. These nonnative fishes have had profound impacts on native fishes, amphibians, and invertebrates. Several of these native species are either already listed under the Endangered Species Act, or are likely to be petitioned for listing in the near future. Reducing impacts to aquatic ecosystems within wilderness areas should be a high priority, and will require that some lakes be restored to their historic fishless condition.

One of the primary purposes of the National Wilderness Preservation System is to protect natural ecosystems. As human-caused modification of lands outside of wilderness intensifies, the protection of wilderness ecosystems will be increasingly important and challenging. These protected areas already are being affected by anthropogenic impacts both internal and external to wilderness areas (Cole and Landres 1996). Attempts to minimize these impacts have typically focused on protection of terrestrial ecosystems, using tactics such as regulation of visitor use and allowing the return of natural fire regimes. In contrast, little attention has been focused on impacts to aquatic ecosystems in wilderness, habitats that have also been substantially altered.

In the western United States, where wilderness areas typically encompass high-elevation montane ecosystems, the most ubiquitous impact to aquatic ecosystems is the introduction of nonnative fish species (Bahls 1992). Many of the lakes in these areas were historically fishless, but have been stocked with several different trout species to create a recreational fishery. When the National Wilderness Preservation System was created in 1964, language was included in the governing legislation to ensure that fish stocking could continue in these areas (Kloepfer and others 1994). Today, fish stocking in the western U.S. continues in many national forest wilderness areas, as well as at least one national park.

In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management; 1999 May 23–27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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Despite the potential impacts caused by fish introductions into wilderness lakes, Bahls (1992) concluded, after interviews with state fishery managers, that the practice of stocking trout into mountain lakes was generally conducted with “little concern for protection of native fish species in lakes or downstream systems, no evident concern for maintaining representative pristine lakes, and no consideration of the effects of trout stocking on indigenous fauna, aquatic ecosystems, and lakeshore recreational impacts”. In addition, Bahls (1992) found that although stocking effort is intensive, research is minimal. As a result, changes in the distribution of fish caused by fish stocking and the effects of fish introductions on aquatic ecosystems remain relatively poorly understood. This lack of information has generally precluded comprehensive efforts to reduce these impacts.

The effects of nonnative fish introductions on aquatic ecosystems in wilderness areas of the Sierra Nevada, California are relatively well-studied, compared with aquatic ecosystems in wilderness areas in other parts of the western U.S. Although the results of these studies have important implications for the management of wilderness fisheries throughout the western U.S., this body of research has only rarely been reviewed. The goals of this paper are to 1) review the changes in fish distribution resulting from over a century of fish stocking in wilderness lakes of the Sierra Nevada, 2) review the impacts of these fish introductions on lake ecosystems, and 3) provide recommendations aimed at reducing these impacts. By making this information more accessible to scientists, federal wilderness managers, and state fisheries managers, we hope that this paper will help to focus much needed attention on how to better balance the interest of providing recreational fisheries in wilderness with the need to maintain or restore natural ecosystems.

Study Area

The Sierra Nevada of California is largely federally owned, with the majority of its five million ha (12 million acres) lying within national parks, national monuments and national forests (Palmer 1988). Eighty-four percent of the national park acreage and 24% of the national forest acreage is officially designated as wilderness (Palmer 1988). The area above 1,800 m (6,000 ft) contains thousands of lakes and ponds, and most of these habitats are located within designated wilderness. More than 99% of these lakes and ponds were historically fishless (Moyle and others 1996). Instead of fish, these water bodies were inhabited by a unique assemblage of amphibians, zooplankton and benthic invertebrates.

Starting in the mid-1800's, trout were introduced into formerly fishless lakes to provide recreational fishing (Moyle and others 1996). Although some of these introductions were interbasin transfers of trout native to the Sierra Nevada (Little Kern golden trout (*O. mykiss whitei*), California golden trout (*O. mykiss aguabonita*), Lahontan cutthroat trout (*O. clarki henshawi*), Paiute cutthroat trout (*O. clarki seleniris*) and coastal rainbow trout (*O. mykiss irideus*)), many were introductions of trout species not native to California. These included brook trout (*Salvelinus fontinalis*), lake trout (*Salvelinus namaycush*), and Atlantic salmon (*Salmo salar*) from eastern North America, kokanee salmon (*Oncorhynchus nerka*) from northwestern North America, and brown trout (*Salmo trutta*) from Europe (Moyle and others 1996). Early trout planting efforts were aimed primarily at establishing trout in formerly fishless waters, and were carried out largely by sporting groups and the U.S. military. In the early 1900's, the California Fish and Game Commission (the precursor to the current California Department of Fish and Game) began coordinating the fish planting effort, and by the 1940's fish stocking was conducted almost entirely by the California Department of Fish and Game (CDFG). Today, the CDFG is responsible for nearly all authorized trout stocking throughout the Sierra Nevada, although the emphasis has changed from introducing trout into fishless lakes and streams to stocking waters to augment existing nonnative trout populations.

Sequoia, Kings Canyon, and Yosemite National Parks began phasing out trout stocking in 1969 as a result of recommendations made in the Leopold Report (Leopold 1963). Limited stocking in these parks was continued until 1991, when all stocking was halted. Trout continue to be stocked into lakes within national forest wilderness areas, and stocking is accomplished using airplanes.

Distribution of Nonnative Fishes in Sierra Nevada Lakes

As a result of more than a century of fish stocking, the majority of historically fishless wilderness lakes in the Sierra Nevada now contain introduced trout. Bahls (1992) estimated that 63% of California's 4,000+ mountain lakes (natural lakes at elevations above 800 m, most of which are found in the Sierra Nevada) now contain nonnative fish populations and 52% are currently stocked. Of the estimated 37% of lakes that remain fishless, most are small (< 2 ha surface area), shallow (< 3 m), and generally incapable of supporting trout populations (Bahls 1992). Only 3% of larger lakes (> 2 ha surface area, > 3 m deep) remain fishless. Similar results were obtained by Jenkins et al. (1994), who projected that one or more species of nonnative trout would occur in 63% of high elevation lakes in the Sierra Nevada (lakes at elevations > 2400 m and > 1 ha surface area). Golden trout were projected to occur in 36% of lakes, rainbow trout in 33%, brook trout in 16%, brown trout in 8%, and cutthroat trout in 0.5% of lakes.

A greater proportion of lakes within national forest wilderness areas contain introduced trout populations than lakes within national parks. Based on a survey of fish populations in 2,000+ wilderness lakes in the Sierra Nevada, Matthews and Knapp (1999) reported that 80% of

lakes larger than 1 ha within the John Muir Wilderness contained introduced trout versus 40% of lakes within the adjacent Kings Canyon National Park. Similar proportions of fish-containing lakes in national forest wilderness and national parks in the Sierra Nevada were reported by Botti (1977), Bradford and others (1993), Knapp (1996) and Wallis (1952). The lower percentage of trout-containing lakes in Sierra Nevada national parks than in national forests is likely the result of both a lower historical stocking intensity and the recent termination of all stocking in the national parks. The termination of fish stocking in Sequoia-Kings Canyon National Park and Yosemite National Park was expected to cause 30% and 40% of previously stocked lakes, respectively, to revert to a fishless condition (Botti 1977; Zardus 1977).

Ecological Effects of Fish Introductions Into Sierra Nevada Lakes

Native Fishes

Although fish were absent historically from nearly all mid- to high-elevation lakes and ponds in the Sierra Nevada, native trout species were present in streams in several watersheds. These included the Little Kern golden trout, California golden trout, Lahontan cutthroat trout, Paiute cutthroat trout and coastal rainbow trout (Knapp 1996). Distributions of some of these native trout populations were altered when nonnative trout species were stocked into fishless headwater lakes, and subsequently moved downstream to hybridize with or displace the native populations. For example, the Little Kern golden trout (*O. mykiss whitei*) is native only to the Little Kern River in the southern Sierra Nevada. The introduction of nonnative brook trout into the headwater lakes of this drainage and their dispersal downstream caused the near-extinction of the Little Kern golden trout as a result of competitive displacement. Consequently, the Little Kern golden trout was listed as "threatened" under the Endangered Species Act (Stephens 1999).

Similarly, the California golden trout is native only to the South Fork Kern River and Golden Trout Creek in the southern Sierra Nevada. The stocking of hybridized golden trout into the headwater lakes of the Golden Trout Creek drainage resulted in extensive hybridization with the downstream native California golden trout population. Partly as a result of this threat, the U.S. Fish and Wildlife Service is currently considering listing the California golden trout under the Endangered Species Act (Stephens 1999). Similar impacts to native fishes resulting from the stocking of headwater lakes with nonnative trout species are apparently common in wilderness areas throughout the western U.S. (Adams 1999; Bahls 1992).

Amphibians

Populations of four of the nine anurans (frogs and toads) native to the Sierra Nevada are reported to be declining (Yosemite toad: *Bufo canorus*; California red-legged frog: *Rana aurora draytonii*; foothill yellow-legged frog: *R. boylei*;

and mountain yellow-legged frog: *R. muscosa*; Jennings 1996). Only one of these species, the mountain yellow-legged frog, was a common inhabitant of lakes historically and is therefore the species most likely to be affected by the introduction of trout into these habitats. Despite its former wide distribution throughout the Sierra Nevada (Zweifel 1955), a recent resurvey of historic localities in the central Sierra Nevada indicated that the mountain yellow-legged frog is now present at fewer than 15% of the sites where it was found in 1915 (Drost and Fellers 1996). Severe declines have also been noted elsewhere in the Sierra Nevada (Bradford and others 1994).

Increasing evidence from the Sierra Nevada indicates that introduced trout are a primary factor in the decline of the mountain yellow-legged frog. As early as 1915, Grinnell and Storer (1924) reported that predation by introduced trout on mountain yellow-legged frog larvae prevented the co-occurrence of these two taxa in lakes and ponds. This observation has now been quantified in several different areas in the Sierra Nevada (Bradford 1989; Bradford and others 1998). Although these studies have generally been done using a relatively small number of sites (< 100), recent research based on surveys at more than 1,700 sites in Kings Canyon National Park (KCNP) and John Muir Wilderness (JMW) provided similar results (Knapp and Matthews 2000). These results can be summarized as follows:

- 1) The KCNP study area had fewer trout-containing lakes than the adjacent JMW study area, and this difference in trout distribution was associated with a seven-fold higher percentage of lakes containing mountain yellow-legged frogs in the KCNP study area than the JMW study area.
- 2) Drainages with a higher percentage of total water body surface area containing trout had a lower percentage of total water body surface area containing frogs.
- 3) After accounting for habitat differences between lakes with and without trout, the probability of occurrence for mountain yellow-legged frog larvae in individual water bodies was three times higher and the abundance of larvae was six times higher in fishless than in fish-containing water bodies.

Together with the results of previous studies, there is now compelling evidence that the mountain yellow-legged frog has been extirpated from much of its historic habitat by the introduction of trout into historically fishless lakes. The results presented in Knapp and Matthews (2000) suggest that these impacts have been particularly severe in national forest wilderness areas, and that the severity of these impacts could eventually require the listing of the mountain yellow-legged frog under the Endangered Species Act.

In addition to the direct impact that nonnative trout have on mountain yellow-legged frogs via predation, Bradford and others (1993) suggested that fish could impact mountain yellow-legged frogs indirectly by isolating remaining populations. They reported that fish introductions into lakes in Sequoia and Kings Canyon National Parks have resulted in a four-fold reduction in effective mountain yellow-legged frog population size and a 10-fold reduction in connectivity between populations. Because amphibian populations often fluctuate widely under natural conditions (Hecnar and M'Closkey 1996; Pechmann and others 1991), and small populations are more likely to go extinct as a result of

stochastic population fluctuations than large populations (Hanski 1994), Bradford and others (1993) proposed that the reduction in mountain yellow-legged frog population size caused by trout introductions probably increased the rate at which individual populations go extinct. In addition, they suggested that the increased isolation of mountain yellow-legged frog populations would reduce the probability of recolonization of formerly occupied sites. This lower probability of recolonization could result from the smaller size of potential source populations, increased distance from source populations and predation by introduced trout on dispersing frogs (Bradford and others 1993).

Several attributes of the mountain yellow-legged frog make it particularly vulnerable to predation and subsequent extirpation by nonnative trout. First, adult mountain yellow-legged frogs are highly aquatic and are found primarily in lakes (most of which now contain trout). Second, in contrast to the larvae of other Sierran anurans that complete metamorphosis to the terrestrial stage in a single summer, mountain yellow-legged frog larvae generally require at least two years to complete metamorphosis. This overwintering requirement restricts successful breeding to permanent water bodies (typically those deeper than 2 m; Bradford 1983; Knapp and Matthews 2000; Mullally and Cunningham 1956). The majority of these deeper lakes, however, now contain introduced trout.

Zooplankton

One of the best studied and most consistent effects of introduced fishes on lake ecosystems is the alteration of zooplankton communities (Brooks and Dodson 1965; Zaret 1980). The introduction of zooplanktivorous fishes into fishless lakes generally shifts the zooplankton community from one dominated by large-bodied species to one dominated by smaller-bodied species, as a result of size-selective predation. Several studies have documented this effect of introduced trout on zooplankton communities in wilderness lakes of the Sierra Nevada. Stoddard (1987) found that the occurrence of introduced trout was the most important predictor of zooplankton species composition in alpine and subalpine lakes, with large-bodied species found in fishless lakes and small-bodied species found in lakes with trout. A recent study by Bradford and others (1998) reported comparable results. Similar effects of trout on zooplankton communities have also been reported for mountain lakes throughout western North America (Anderson 1980; Bahls 1990; Carlisle and Hawkins 1998; Liss and others 1995).

Benthic Macroinvertebrates

The introduction of fish into fishless lakes also causes predictable effects on benthic macroinvertebrate communities in which large conspicuous species are eliminated, while burrowing or otherwise inconspicuous species are relatively unaffected (Zaret 1980). In the Sierra Nevada, the benthic invertebrate communities of high-elevation fishless lakes are typically dominated by several conspicuous taxa of mayfly larvae (Ephemeroptera), caddisfly larvae (Trichoptera), aquatic beetles (Coleoptera) and true bugs (Corixidae). These taxa are rare or absent in lakes with introduced trout.

Instead, the benthic macroinvertebrate community of trout-containing lakes is typically dominated by midge larvae (Chironomidae), alderfly larvae (*Sialis*), aquatic mites (Acari) and fingernail clams (*Pisidium*) (Bradford and others 1998; Reimers 1958), all taxa that either burrow into lake bottom sediments or are distasteful. Similar effects of trout on benthic macroinvertebrate communities have been reported from mountain lakes throughout the western United States (Bahls 1990; Carlisle and Hawkins 1998; Walters and Vincent 1973). As noted by Liss and others (1995), however, the effects of introduced trout may be less pronounced in areas where lakes contain naturally occurring vertebrate predators such as salamanders. In these situations, the long evolutionary history between predatory salamanders and their invertebrate prey may have resulted in adaptations by the prey to reduce predation risk, and these adaptations may also reduce their vulnerability to introduced trout. This possibility merits additional study.

Food Web Effects

The effect of introduced trout on native aquatic taxa is often presented as an interaction between two trophic levels (trout preying on amphibians, trout preying on zooplankton). However, changes in one trophic level can have important indirect effects on all parts of the food web. Although multiple trophic-level consequences of fish introductions have not received much attention until recently, at least one such effect has been suggested for aquatic ecosystems in the Sierra Nevada. Jennings and others (1992) demonstrated that the garter snake, *Thamnophis elegans*, depends heavily on frog larvae as prey items, and they suggested that the decline of amphibians in the Sierra Nevada may also result in the decline of *T. elegans*. Because introduced trout are an important factor in the decline of at least one Sierran amphibian (Bradford 1989; Bradford and others 1993; Knapp and Matthews 2000), trout may also indirectly cause the decline of *T. elegans*.

Trout introductions may also cause trophic cascades, in which changes caused by the introduction of a new top predator (fish) propagate to cause substantial changes at the primary producer level (Carpenter and others 1985). Trophic cascades have now been reported from a diverse array of lake types (Carpenter and Kitchell 1993), but studies of trophic cascades from trout introductions into Sierra Nevada lakes are just beginning. In a study of alpine lakes in Canada, the introduction of nonnative trout resulted in a decrease in large-bodied herbivorous zooplankton and an increase in phytoplankton abundance (Leavitt and others 1994; McNaught and others 1999). The elimination of amphibian larvae following trout introductions may also influence lower trophic levels, since amphibian larvae can have important effects on algal biomass (Dickman 1968) and lake nutrient cycling (Seale 1980).

Conclusions and Management Recommendations

The management of nonnative fish populations in wilderness lakes of the western U.S. has been the focus of considerable controversy for at least two decades (Gottschalk 1976;

Hall and May 1977), with debate generally focusing on the question of whether nonnative fishes impact wilderness ecosystems. The preponderance of evidence collected during the past two decades leaves little doubt that the introduction of nonnative trout into historically fishless lakes causes a series of predictable changes in the recipient ecosystems; therefore, discussions over the management of nonnative fishes in wilderness lakes should be shifted from whether there are impacts to determining what level of impact is acceptable and how to reduce current impacts to this level.

In the Sierra Nevada, introduced trout have caused dramatic changes in the distributions of several native trout species, one amphibian, and several invertebrate species. This current level of impact is clearly unacceptable if wilderness areas are to serve the purpose of maintaining natural processes. We suggest that an acceptable level of impact would be one that, at a minimum, allows for the long-term persistence of all native taxa across their historic distribution within wilderness lands. In the Sierra Nevada, reducing current impacts to this level will take significant resources from the state and federal agencies with jurisdiction over management of these ecosystems.

To reduce the impacts of introduced trout on wilderness lake ecosystems in the Sierra Nevada, it will be critical to (i) eliminate the stocking of lakes harboring self-sustaining trout populations, and (ii) restore fishless habitat for the native taxa most seriously effected by nonnative trout. The current California Department of Fish and Game stocking program for wilderness lakes is based on the untested assumption that stocking is required to maintain the target fisheries. Similar assumptions are commonly made by fisheries managers throughout the western U.S., and appear to result in the frequent stocking of self-sustaining trout populations (Bahls 1992). Available evidence for wilderness lakes in the Sierra Nevada indicates that the majority of stocked lakes have sufficient natural reproduction to maintain these fisheries in the absence of stocking (Botti 1977; Matthews and Knapp 1999; Zardus 1977). This unnecessary stocking brings with it considerable risks to native aquatic species, as a result of stocked fish hybridizing with or displacing native fishes and of fish being stocked into the wrong water bodies (fishless lakes).

Because most trout populations in Sierra Nevada wilderness lakes would be self-sustaining in the absence of stocking (Matthews and Knapp 1999), restoration of mountain yellow-legged frog populations to even a fraction of their historic habitat will require the active eradication of fish populations from some lakes. Remaining mountain yellow-legged frog populations within national forest wilderness areas are typically extremely isolated (Knapp and Matthews 2000), and are therefore unlikely to persist over the long term (Bradford and others 1993). To expand the few remaining mountain yellow-legged frog populations and enhance their likelihood of persistence, one of us (R. Knapp) is currently using gill nets (Knapp and Matthews 1998) to remove fish populations from lakes in the immediate vicinity of existing frog populations. The goal of this work is to create clusters of interconnected fishless lakes and ponds that would provide high quality habitat for mountain yellow-legged frogs and that could be naturally recolonized from nearby source populations. Preliminary results indicate that frogs are rapidly recolonizing these lakes after fish

removal. The success of these pilot projects suggests that the creation of clusters of fishless habitat across the historic range of the mountain yellow-legged frog could reverse the decline of this species and reduce the need to list it under the Endangered Species Act. In addition to benefiting the mountain yellow-legged frog, these fishless habitat clusters would also benefit fish-sensitive invertebrate species.

Implementation of these recommendations would represent a significant step toward reducing impacts to Sierra Nevada wilderness lakes from nonnative fishes. However, outside of the study area surveyed by Matthews and Knapp (1999), information on self-sustainability of fish populations and locations of fish-sensitive native species in the Sierra Nevada remain rudimentary at best. Resolving the ongoing controversy over the management of nonnative fisheries in these wilderness lakes will take a considerable and sustained effort to survey aquatic habitats for nonnative fish and native aquatic taxa, evaluate the self-sustainability of fish populations, and design and implement restoration measures for these sensitive species.

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References

- Adams, S. B. 1999. Mechanisms limiting a vertebrate invasion: brook trout in mountain streams of the northwestern USA. Dissertation, University of Montana. 215 p.
- Anderson, R. S. 1980. Relationships between trout and invertebrate species as predators and the structure of the crustacean and rotiferan plankton in mountain lakes. In: *Evolution and Ecology of Zooplankton Communities*. Edited by W.C. Kerfoot. Hanover, NH: University Press of New England: 635-641.
- Bahls, P. 1990. Ecological implications of trout introductions to lakes of the Selway Bitterroot Wilderness, Idaho. Master's Thesis, Oregon State University. 85 p.
- Bahls, P. 1992. The status of fish populations and management of high mountain lakes in the western United States. *Northwest Science*. 66: 183-193.
- Botti, S. 1977. Status of fish populations in 102 planted lakes. Unpublished file report, U.S. Department of the Interior, National Park Service, Yosemite National Park, California.
- Bradford, D. F. 1983. Winterkill, oxygen relations, and energy metabolism of a submerged dormant amphibian, *Rana muscosa*. *Ecology*. 64: 1171-1183.
- Bradford, D. F. 1989. Allotopic distribution of native frogs and introduced fishes in high Sierra Nevada lakes of California: implication of the negative effect of fish introductions. *Copeia*. 1989: 775-778.
- Bradford, D. F., F. Tabatabai, and D. M. Graber. 1993. Isolation of remaining populations of the native frog, *Rana muscosa*, by introduced fishes in Sequoia and Kings Canyon National Parks, California. *Conservation Biology*. 7: 882-888.
- Bradford, D. F., D. M. Graber, and F. Tabatabai. 1994. Population declines of the native frog, *Rana muscosa*, in Sequoia and Kings Canyon National Parks, California. *Southwestern Naturalist*. 39: 323-327.
- Bradford, D. F., S. D. Cooper, T. M. Jenkins, K. Kratz, O. Sarnelle, and A. D. Brown. 1998. Influences of natural acidity and introduced fish on faunal assemblages in California alpine lakes. *Canadian Journal of Fisheries and Aquatic Sciences*. 55: 2478-2491.
- Brooks, J. L. and S. I. Dodson. 1965. Predation, body size and composition of plankton. *Science*. 150: 28-35.
- Carlisle, D. M., and C. P. Hawkins. 1998. Relationships between invertebrate assemblage structure, 2 trout species, and habitat structure in Utah mountain lakes. *Journal of the North American Benthological Society*. 17: 286-300.
- Carpenter, S. R., J. F. Kitchell, and J. R. Hodgson. 1985. Cascading trophic interactions and lake productivity. *Bioscience*. 35: 634-639.
- Carpenter, S. R., and J. F. Kitchell. 1993. *The trophic cascade in lakes*. New York, NY: Cambridge University Press. 385 p.
- Cole, D. N., and P. B. Landres. 1996. Threats to wilderness ecosystems: impacts and research needs. *Ecological Applications*. 6: 168-184.
- Dickman, M. 1968. The effect of grazing by tadpoles on the structure of a periphyton community. *Ecology*. 49: 1188-1190.
- Drost, C. A., and G. M. Fellers. 1996. Collapse of a regional frog fauna in the Yosemite area of the California Sierra Nevada, USA. *Conservation Biology*. 10: 414-425.
- Gottshalk, J. S. 1976. A symposium on the management of wilderness area waters: proceedings; 1976; Washington D.C., Washington, D.C.: American Fisheries Society: 23 p.
- Grinnell, J., and T. I. Storer. 1924. *Animal life in the Yosemite*. University of California Press. Berkeley, California. 752 p.
- Hall, A. and R. May. 1977. A symposium on the management of high mountain lakes in California's national parks: proceedings; 1976 January 29; Fresno, CA. San Francisco, CA: California Trout: 69 p.
- Hanski, I. 1994. Patch-occupancy dynamics in fragmented landscapes. *Trends in Ecology and Evolution*. 9: 131-135.
- Hecnar, S. J., and R. T. M'Closkey. 1996. Regional dynamics and the status of amphibians. *Ecology*. 77: 2091-2097.
- Jenkins, T. M. Jr., R. A. Knapp, K. W. Kratz, S. D. Cooper, J. M. Melack, A. D. Brown, and J. Stoddard. 1994. Aquatic biota in the Sierra Nevada: current status and potential effects of acid deposition on populations. Final Report, Contract A932-138. California Air Resources Board, Sacramento.
- Jennings, M. R. 1996. Status of amphibians. In: *Sierra Nevada Ecosystem Project: final report to Congress. Volume II*. Davis, CA: Centers for Water and Wildland Resources, University of California: 921-944.
- Jennings, W. B., D. F. Bradford, and D. F. Johnson. 1992. Dependence of the garter snake *Thamnophis elegans* on amphibians in the Sierra Nevada of California. *Journal of Herpetology*. 26: 503-505.
- Kloepfer, D., J. Watson, and P. Byrnes. 1994. *The Wilderness Act handbook*. Washington, D.C.: The Wilderness Society. 68 p.
- Knapp, R. A. 1996. Nonnative trout in natural lakes of the Sierra Nevada: an analysis of their distribution and impacts on native aquatic biota. In: *Sierra Nevada Ecosystem Project: final report to Congress. Volume III*. Davis, CA: Centers for Water and Wildland Resources, University of California: 363-407.
- Knapp, R. A., and K. R. Matthews. 1998. Eradication of nonnative fish by gill-netting from a small mountain lake in California. *Restoration Ecology*. 6: 207-213.
- Knapp, R. A., and K. R. Matthews. 2000. Nonnative fish introductions and the decline of the mountain yellow-legged frog (*Rana muscosa*) from within protected areas. *Conservation Biology*. 14: 1-12.
- Leavitt, P. R., D. E. Schindler, A. J. Paul, A. K. Hardie, and D. W. Schindler. 1994. Fossil pigment records of phytoplankton in trout-stocked alpine lakes. *Canadian Journal of Fisheries and Aquatic Sciences*. 51: 2411-2423.
- Leopold, A. S. 1963. *Wildlife management in the National Parks*. Report to the Secretary of the Interior (available online at www.halcyon.com/rdpayne/nps-leopold.html).
- Liss, W. J., G. L. Larson, E. Deimling, L. Ganio, R. Gresswell, R. Hoffman, M. Kiss, G. Lomnický, C. D. McIntire, R. Truitt, and T. Tyler. 1995. Ecological effects of stocked trout in naturally fishless lakes, North Cascades National Park Service Complex, WA, USA. Technical Report NPS/PNROSU/NRTR-95-03.
- Matthews, K. R., and R. A. Knapp. 1999. A study of high mountain lake fish stocking effects in the U.S. Sierra Nevada wilderness. *International Journal of Wilderness*. 5: 24-26.

- McNaught, A. S., D. W. Schindler, B. R. Parker, A. J. Paul, R. S. Anderson, D. B. Donald, and M. Agbeti. 1999. Restoration of the food web of an alpine lake following fish stocking. *Limnology and Oceanography*. 44: 127-136.
- Moyle, P. B., R. M. Yoshiyama, and R. A. Knapp. 1996. Status of fish and fisheries. In: *Sierra Nevada Ecosystem Project: final Report to Congress. Volume II*. Davis, CA: Centers for Water and Wildland Resources, University of California: 953-973.
- Mullally, D. P., and J. D. Cunningham. 1956. Ecological relations of *Rana muscosa* at high elevations in the Sierra Nevada. *Herpetologica*. 12: 189-198.
- Palmer, T. 1988. *The Sierra Nevada: a mountain journey*. Washington D.C.: Island Press. 339 p.
- Pechmann, J. H. K., D. E. Scott, R. D. Semlitsch, J. P. Caldwell, L. J. Vitt, and W. Gibbons. 1991. Declining amphibian populations: the problem of separating human impacts from natural fluctuations. *Science*. 253: 892-895.
- Reimers, N. 1958. Conditions of existence, growth, and longevity of brook trout in a small, high altitude lake of the eastern Sierra Nevada. *California Fish and Game*. 44: 319-333.
- Seale, D. B. 1980. Influence of amphibian larvae on primary production, nutrient flux, and competition in a pond ecosystem. *Ecology*. 61: 1531-1550.
- Stevens, S. 1999. Conversation between S. Stevens (California Department of Fish and Game, Region 4, Visalia, California) and R. Knapp, August 10.
- Stoddard, J. L. 1987. Microcrustacean communities of high-elevation lakes in the Sierra Nevada, California. *Journal of Plankton Research*. 9: 631-650.
- Wallis, E. O. 1952. Comprehensive review of trout fishery problems of Yosemite National Park: A report of the Yosemite Trout Investigations, 1951-1953. U.S. Department of the Interior, National Park Service, Yosemite National Park.
- Walters, C. J., and R. E. Vincent. 1973. Potential productivity of an alpine lake as indicated by removal and reintroduction of fish. *Transactions of the American Fisheries Society*. 102: 675-697.
- Zardus, M., T. Blank, and D. Schultz. 1977. Status of fishes in 137 lakes in Sequoia and Kings Canyon National Parks, California. Unpublished file report, U.S. Department of the Interior, National Park Service, Sequoia-King Canyon National Parks.
- Zaret, T. M. 1980. *Predation and freshwater communities*. New Haven, NJ: Yale University Press: 187 p.
- Zweifel, R. G. 1955. Ecology, distribution, and systematics of frogs of the *Rana boylei* group. *University of California Publications in Zoology*. 54: 207-292.

A Survey of Exotic Plants in Federal Wilderness Areas

Marilyn Marler

Abstract—I conducted a survey of wilderness areas to provide an overview of plant invasions in the National Wilderness Preservation System. Fifteen per cent of responding managers reported that exotic plants were among their top 10 management concerns, either because they are actively dealing with control of exotic pest plants or have prioritized prevention of their establishment. Seventy per cent of responding wilderness areas do not monitor or inventory for exotic plants. The majority of respondents reported that exotic plants have not impacted their areas, so it is important to emphasize prevention and early detection of exotic plant establishment. Responses varied greatly among regions, with the highest priority being given to exotic plants by agencies in the California Mediterranean region and the Rocky Mountain montane region. The National Park Service was most likely to monitor or inventory for exotic plants. The greatest needs for most areas are increased funding, education and training to prevent further establishment of exotic plants.

The 1964 Wilderness Act was passed to protect designated natural areas from human impacts in order to preserve research and recreation opportunities and to protect the intrinsic value of natural areas and wildlife. Although legislatively protected, all wilderness areas have been impacted to some degree by human disturbances (Cole and Landres 1996). Among these disturbances is the introduction of exotic species that have been transported by human activities beyond their native ranges, and variously referred to as nonnative, nonindigenous, or alien. Many of these plants have significant and measurable ecological effects on invaded ecosystems, and are considered pests or “weeds.”

Increased global travel by humans and the resulting breakdown of geographic barriers to plant dispersal has dramatically increased the rate of intentional and accidental introductions of exotic species (Vermeij 1991, D’Antonio and Vitousek 1992, Lodge 1993, Vitousek and others 1997). Changes in species’ distributions are natural phenomena that operate on various time and spatial scales (Vermeij 1991, Lodge 1993). However, human mobility allows introductions at rates that are without precedent over the past several million years. The resulting exotic plant invasions have long been recognized as serious ecological problems (Leopold 1941, Stewart and Hull 1949, Elton 1958) and are increasingly considered one of the greatest anthropogenic threats to preservation of biodiversity and the regional

distinctiveness of the planet (Soulé 1990, Vitousek 1990, 1994, D’Antonio and Vitousek 1992, Dudley and Collins 1995, Huenneke 1997).

Because exotic plant management is expensive, time consuming and complex, researchers and managers need to identify priority areas to focus weed control efforts. Many researchers agree that relatively undisturbed areas should be high priorities for weed control efforts (MacDonald and others 1989, Asher and Harmon 1995, Hobbs and Humphries 1995). Relative to other managed lands, wilderness areas usually have more limited access, more natural conditions and fewer impacts of human activity. Therefore, they are an appropriate focal point for prevention and control of exotic plant invasions.

In 1997 and 1998, I conducted a survey of managers of Bureau of Land Management (BLM), Forest Service (FS), Fish and Wildlife Service (FWS) and the National Park Service (NPS) wilderness areas. The goal of the survey was to compile information on exotic plant species in order to identify research needs, generate awareness and facilitate information exchange about exotics in wilderness areas. Three immediate objectives were to document the occurrence of exotic plant species in wilderness areas and control efforts being used, identify factors contributing to exotic plant establishment and spread and characterize the quality of available data. The resulting database is available over the Internet (www.umn.edu/biology/leopold). Here I discuss the current status of exotic plants in wilderness areas in the context of identifying research priorities and appropriate management actions for wilderness preservation.

Terminology

There is no universally accepted term to describe nonindigenous plants, and human values frequently complicate terminology (Luken 1994). Terminology for exotic plants is problematic due to biological problems in defining the status of “native” or “indigenous” plants. It is difficult to define natives because of the naturally dynamic nature of species distributions and the relatively brief time frame in which we have been documenting those distributions (Webb 1985, Lodge 1993, Tausch and others 1993, Carlton 1996, Schwartz 1997). One commonly used temporal standard for determining the indigenous status of plants in North America is pre- versus post- European settlement. This is often an appropriate reference point, since European settlement marked the point when the rate of introductions was dramatically accelerated. However, it is still problematic since it overlooks the fact that indigenous people were practicing agriculture and introducing plants beginning 10,000 years ago (Webb 1985, Schwartz 1997). Nonetheless, pre- versus post-colonization is often a helpful standard to use.

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The status of a given plant could be debated indefinitely, but the relevant task is when to determine that an exotic plant has become a problem. According to Loope (1993), the threshold is crossed when the introduced species results in a "significant decline in populations of one or more native species, significantly alters ecosystem processes, (or) causes aesthetic damage perceived to be unacceptable."

"Weed" is a commonly used, subjective term for any plant that is not wanted. This term can be confusing when used by different people; but when management goals are clearly identified, defining weeds becomes straightforward (Randall 1997). It is easy to identify management goals in situations where benefits and costs can be assessed economically. When benefits of "weed" control are aesthetic, social or scientific, it becomes more difficult to say which plants interfere with management goals, and why.

For this report, "exotic" will be used for plant species that are not considered indigenous to a given area. "Weed" will be used for any exotic plant that has proven to be a nuisance or environmental threat by causing any of the problems mentioned above. This survey asked respondents to list all exotic plant species. However, the majority reported only serious weeds, making generalizations about large-scale patterns in the number of exotics difficult.

Methods

A survey was distributed to all national parks, national forests, BLM offices and national wildlife refuges that administer wilderness areas. The survey form was intended to standardize responses on attitudes and priorities toward exotic plants and to gather specific information on exotic species, management responses and control efforts. Respondents were asked whether there was a weed management plan that applied to the wilderness area, whether exotic plants were monitored in the wilderness area, and to indicate the source of the information they were providing on the survey (best guess, systematic monitoring, etc.). They were also asked to rank the problem of exotic plants relative to other management issues.

Respondents were asked to list all exotic plants that they knew of or suspected in the wilderness area, and to give a categorical ranking for the abundance and perceived threat of each plant. Space was provided to list research projects and control efforts for each species.

Survey responses were entered into a database with tables for contact information, species information (plant names, notes, pattern of infestation, control methods, and so on), and general information for wilderness areas. Examples of possible queries include "Which wilderness areas have leafy spurge present?" "Which are using herbicides to control salt cedar?" or "What are some contacts for managers in the Northwest who have sweet clover in their wilderness areas?"

Survey Results

The response rate, quality of available information and level of priority assigned to exotic species management all varied considerably within and between regions. For this report, results are discussed regionally, with each wilderness area assigned to a biome.

Overall, 322 designated wilderness areas in 30 states responded by mail, phone, or e-mail. There are 667 wilderness areas in 44 states, so the responses represent 48%. Some wilderness areas are managed by more than one unit; for example, the Lee Metcalf Wilderness in Montana is managed by the Gallatin National Forest, Beaverhead-Deerlodge National Forest and the BLM. If each unit is considered a separate wilderness area, there are 756 wilderness areas, of which 342 responded (45%). This report treats each management unit as a separate wilderness area, since available information, attitudes and projects vary from one unit to the next.

Quality of Available Information and Ranking of Exotics as a Priority Issue

In addition to nonrandom participation in our survey effort, there were large differences between regions and between agencies in the quality of available information, and in the level of importance assigned to exotic plants in general. Overall, about 31% of wilderness areas reported some kind of monitoring or documentation of exotics (42% confirmed that they did not monitor, and 27% did not respond to the question). National Park Service wilderness areas were the most likely to monitor exotic plants (table 1). Fewer than 10% of respondents have written plans for weed management in the wilderness areas.

Overall, about 15% of respondents ranked exotic plants among their top 10 concerns, and 17% reported it as one of many small problems. About 42% said that exotic plants were not much of a problem. Sometimes this was because exotics were not known or suspected to occur, but occasionally respondents indicated that management was not interested in exotic plants in general, or that wilderness areas are lower priorities for active management. The level of importance assigned to exotic plants varied greatly among regions.

Regional Organization

Wilderness areas from the Sonoran desert to the Arctic tundra reported problems with exotic plants (table 2). Organization of results by biome is intended to identify regional differences in the perception of exotic plants as a management priority and to help interested parties identify and prioritize plants that are likely to occur locally or regionally.

Table 1—Percent of wilderness areas that monitor for exotic plants, by agency.

Agency	Percent
BLM	46
FS	29*
FWS	32
NPS	80

*Excludes wilderness areas that received the first version of the survey form, which did not include this question.

Table 2—Biomes included for regional discussion
(based on Barbour and Billings 1988).

Arctic tundra and boreal forest
Pacific coastal and Cascadian forests
California mediterranean
California forests
Intermountain basin
Northwest (Palouse) prairie
Sonoran and Mojave Deserts
Rocky Mountain montane forests
Central prairie
Eastern temperate forests
Great Lakes
Southwest states
Appalachian forest
Northeast states
Southeast coastal marsh, swamp, bog, forests

For convenience, the Arctic tundra and boreal forest are combined, as are the Pacific coastal and Cascadian forests, and Intermountain communities and the Palouse prairie. The Eastern temperate forest biome is very large, and I discuss it in four subsections that are geographically convenient and biologically relevant: Appalachian forests, Northeast states, Great Lakes, and the Southeast states. Those interested should also see Loope (1993) for an overview of problem exotic plants in national parks and biosphere preserves of the United States.

Arctic Tundra/Boreal Forest—All wilderness areas of this biome are located in Alaska. All respondents in this biome indicated that the low level of human disturbance and remoteness of the areas are not conducive to invasion by exotics. For example, Selawik NWR reported that no one on the current staff had visited the Selawik wilderness area. Access is very difficult, requiring foot travel through an adjacent wilderness areas or plane. It is unlikely that exotic plants have been introduced to such a remote area.

Although eight of nine wilderness areas reported that exotic plants are “not much of a problem,” a few exotic plants are found there, including some that cause problems in the lower 48 states. Several places reported the presence of cosmopolitan ruderal species that do not appear to pose a threat to native plant communities, including pineapple weed (*Matricaria matricoides*), dandelion (*Taraxacum officinale*) and shepard’s purse (*Capsella pursa-pastoris*). Most wilderness managers are not concerned with these species, but Denali National Park has an active exotic plant control program which includes pulling dandelion and vetch (*Vicia cracca*) (Balay, personal communication).

Serious pest plants have also been reported in this region. Bull thistle (*Cirsium vulgare*) has been found (and sprayed with herbicides) near the Izembek Wilderness. Orange hawkweed (*Heiracium scabriusculum*), ox-eye daisy (*Chrysanthemum leaucanthemum*, syn. *Leucanthemum vulgare*), timothy (*Phleum pratense*) and white sweet clover (*Melilotus alba*) have been reported in or near Denali and/or Togiak. These plants cause considerable problems for natural area management in other parts of the United States, although no large problems have yet developed in Alaska.

Pacific Coastal/Cascadian Forests—This region ranges from southeastern Alaska to northern California, and the wilderness areas range in elevation from sea level to over 8,000 feet. Only 8% of respondents ranked exotic plants among the top 10 problems. Those included Olympic National Park (WA), the Middle Santiam Wilderness (OR), and the Okanogan National Forest portion of the Lake Chelan-Sawtooth Wilderness (WA), which are actively dealing with weed problems. The Leavenworth Ranger District portion of Alpine Lakes Wilderness (WA) has taken a proactive approach to preventing weed establishment in the wilderness area (Therrell, personal communication). There is the potential for weeds to spread from heavily infested roads and trailheads leading into the wilderness area, and Forest staff have been aggressively hand pulling weeds at trailheads for the past six summers. Monitoring of weeds and control efforts are informal and unfunded.

In coastal and montane regions of northern California, Oregon, and Washington, little information was submitted for exotic plant distributions in Forest Service wilderness areas. Biologists on the Six Rivers, Deschutes and Mt. Baker-Snoqualmie National Forests confirmed that their wilderness areas have never been inventoried for exotic plants. Although respondents did not expect that weed problems existed in these wilderness areas, there was a general lack of information to assess the situation. Many biologists cited a lack of funding and staff for inventories. The Deschutes National Forest is seeking funding for weed projects and has a GIS project to identify weed locations on the entire Forest. While there is currently no information on the distribution of exotics in the wilderness, trailheads have been identified as priority spots for weed monitoring, and weed populations near wilderness areas are considered priorities for control (Grenier, personal communication).

Olympic National Park has a large amount of information on exotics and an active management program. In California, Oregon and Washington, common burdock (*Arctium minus*), ox-eye daisy (*Chrysanthemum leucanthemum*, syn. *Leucanthemum vulgare*), tansy ragwort (*Senecio jacobaea*), Scotch broom (*Cytisus scoparius*) and St. Johnswort (*Hypericum perforatum*) were the most widely reported problem species.

Biologists in southern and coastal Alaska reported that weeds were not a problem in their wilderness areas, due to their remoteness and inaccessibility. However, due to the same logistical limitations, most did not have specific information. While no exotics were reported for the Alaskan wilderness areas in this biome, biologists and managers should be aware that problem species have been found elsewhere in Alaska (see Tundra/Boreal Forest).

California Mediterranean—Wilderness areas in the California mediterranean biome were more likely to report exotic plants as a high priority than any other biome. Of those that responded, 65% considered exotic plants to be at least among the top 10 management concerns, and 11 of 19 monitor exotics in wilderness in some way. These wilderness areas are at lower elevations than in many regions, and therefore probably have more susceptible habitat. At the Phillip Burton wilderness area of Point Reyes National Seashore, exotic plants are a major management problem, especially since a large fire in 1995, which precipitated an explosion of weed populations (Cooper, personal communication).

Throughout California, native perennial grasses have largely been replaced by exotic annual grasses following European settlement. The replacement is so complete that the original grassland communities are virtually unknown (Heady and others 1992). These exotic annual grasses now also make up a significant portion of understory in chaparral/scrub communities. Biologists on the Los Padres National Forest reported that annual brome grasses have displaced native plants and are altering fire regimes, but no control efforts are underway (nor do they seem feasible). Salt cedars (*Tamarix* species) and yellow star thistle (*Centaurea solstitialis*) were the most frequently reported problem species known or suspected to occur in wilderness areas in the Mediterranean biome. On the Los Padres National Forest, salt cedar is being treated by cutting and herbicide application to stumps and by manual removal by a volunteer group which has an annual work day (Austin, personal communication).

California Montane Forests—Wilderness areas of the Sierra Nevada reported few weed problems. These wilderness areas occur mostly at high elevations and all 19 that responded expected elevation to limit exotic plant invasions. However, yellow star thistle (*Centaurea solstitialis*) has been found at Tuolumne Meadows in Yosemite National Park, at an elevation of 8,600 feet (Fritzke, personal communication). Although it is uncertain whether the plants would be able to complete their life cycle at this elevation, biologists on the Sierra National Forest and in Yosemite National Park expect that strong prevention efforts will be necessary to avoid establishment of this major pest plant, even at high elevations.

The few exotics that do occur in the high-elevation portions of the wilderness areas were probably brought in with livestock and packstock (Shevock, personal communication). Musk thistles (*Carduus* species) are suspected to occur near high-elevation lakes on the Sequoia National Forest. Kentucky bluegrass (*Poa pratense*) is well established at higher elevations in Sequoia-Kings Canyon wilderness area. Many wilderness areas in the Sierra have been or are currently grazed by livestock, a disturbance that was reported to increase susceptibility to exotic plant invasion (see Dudley and Embury (1995) for an in-depth discussion of grazing in California wilderness areas).

At lower elevations, annual bromes (*Bromus tectorum*, *B. madritensis* ssp. *rubens*, *B. diandrus*) are well established. The exotic grass *Vulpia myuros* and yellow star thistle (*C. solstitialis*) are problem invaders at low elevations.

Few exotics are known to occur in California montane wilderness areas, but 16 of 19 areas that responded have not been inventoried for weeds and indicated that their data were too rough for confident assessment of the problem. However, Sequoia and Inyo National Forests both began weed inventories in 1998, which will probably include parts of some wilderness areas, and the Modoc National Forest is preparing a noxious weed Environmental Impact Statement that includes the South Warner Wilderness. The Tahoe National Forest depends on volunteers and knowledgeable hikers for reports of weed populations in the Granite Chief Wilderness. This is probably the case for other forests, too, which do not currently have funding for weed inventories. Yosemite and Sequoia-Kings Canyon National Parks have more active exotic plant inventory programs than the Forest Service wilderness areas.

Intermountain Basin and Palouse Prairie—These biomes were combined since there are relatively few wilderness areas in each. Nearly 20% of responding wilderness area ranked exotic plants as among the top 10 priorities. Nine of the 11 wilderness areas included in this biome are on the Humboldt-Toiyabe National Forest in Nevada, where an exotic plant prevention program is currently being developed. The Forest is in the process of documenting specific occurrences of weeds in wilderness. Most of the Forest's information and focus on weeds is outside wilderness areas, since problem infestations are generally thought to be outside of wilderness boundaries. The exception is cheatgrass (*B. tectorum*), which is limited to lower elevations of wilderness areas (Jean, personal communication).

Although weeds are identified as a high priority on Forest Service lands in Nevada, monitoring and documentation of exotics only occurs when staff visit wilderness areas for other purposes. In other words, this work is getting done without specific allocated funding. The Humboldt-Toiyabe National Forest reported using biocontrols and herbicides for leafy spurge (*Euphorbia esula*), Canada thistle (*Cirsium arvense*) and toadflax (*Linaria* spp.) outside of wilderness areas.

Lake Mead National Recreation Area (which is not a designated wilderness area) is dealing with many weed problems, especially salt cedar (*Tamarix ramosissima*). Other priority plants for control are the exotic palms *Phoenix dactylifera* and *Washingtonia filifera*, which have spread from plantings as ornamentals. Eradication of these plants is controversial because the public finds them attractive and desirable (Powell, personal communication).

Sonoran and Mojave Deserts—This biome includes wilderness areas in California, Arizona, and New Mexico. The response rate was high from Arizona and New Mexico, but only a few responses were received from California. Since the BLM recently acquired the 69 wilderness areas in this biome under the California Desert Protection Act of 1994, there may be little information available on this vast amount of land (almost 4.3 million acres).

While fewer than 10% of respondents ranked exotic plants among their top 10 priorities, at least 40% of responding wilderness areas have no information available on exotics in wilderness areas. However, the Gila National Forest in New Mexico and most of the BLM field offices in Arizona have done exotic plant surveys in wilderness areas and have found very few exotics. In contrast, Organ Pipe Cactus National Monument and Saguaro National Park are dealing with many problem exotic species. In these parks, and in several of the Arizona BLM wilderness areas, problems with exotic plants are mostly restricted to lower elevation desert communities, whereas the higher elevation pine and chaparral communities have few weed problems.

Organ Pipe Cactus National Monument is targeting buffelgrass (*Pennisetum ciliare*) for control in 1998-99. Felger (1990) compiled an extensive list of exotics at Organ Pipe, with a discussion of the types of exotics (disturbance-dependent, capable of invading intact communities, etc.). This report is a good resource reference for the region. In 1990, the total proportion of actual and "potential" exotic species in the flora was 11.5%. Felger states that this low proportion is indicative of a healthy ecosystem. Although the proportion is low, several of those species are capable of dominating vast

acreage, with significant ecological impacts; thus, abundance, and not just the number of species, needs to be considered.

Red brome (*Bromus rubens*), fountain grass (*Pennisetum setaceum*), buffelgrass (*P. ciliare*), black mustard (*Brassica tournefortii*) and salt cedar (*Tamarix ramosissima*) were the most widely reported weeds of the Sonoran desert. Red brome, fountain grass and buffelgrass alter fire regimes, and *Tamarix* can alter hydrology. Fire suppression and subsequent big fires, along with intense grazing history, were widely reported causes of weed establishment and spread.

Rocky Mountain Montane Forests—About 45 wilderness areas in the Rocky Mountain region received an early version of the survey form that did not include all of the questions present in the final version, including the question to rank exotics plants relative to other management issues. Among those that did respond to all of the questions (24 wilderness areas), 68% ranked exotic plants among the top 10 management priorities, 58% have some kind of weed management plan that includes the wilderness area, and 87% reported that exotics are monitored in the wilderness area. Many of the surveys were completed by a “noxious weed specialist,” a position rarely found in other regions.

Widely reported exotics included Canada thistle (*Cirsium arvense*), spotted knapweed (*Centaurea maculosa*), hound’s tongue (*Cynoglossum officinale*), leafy spurge (*Euphorbia esula*), smooth brome (*Bromus inermis*), timothy (*Phleum pratense*), toadflax species (*Linaria dalmatica*), mullein (*Verbascum thapsus*) and sulfur cinquefoil (*Potentilla recta*). Many wilderness areas reported manual control efforts, regular herbicide treatments, and biological control of these and other plants in and immediately adjacent to wilderness areas. Control efforts with user groups are also in place. For example, the Frank Church- River of No Return has a hand-pulling campaign along the Salmon River. The river corridor is a heavily used area where many of the weed problems are focused, so the volunteer pulling program has two benefits. It results in some weed control in the heavily impacted areas, and it helps to educate wilderness users about exotic plants as a conservation problem (Anderson, personal communication). Researchers on the Frank Church-River of No Return Wilderness Area also conducted a trial using remote sensing for early detection of noxious weeds, with mixed results (Lake 1996a,b).

Central Prairie—Prairie systems are poorly represented in the National Wilderness Preservation System, with only 10 federal wilderness areas (Landres and Meyer, 1998). Of the five responding, two ranked exotics as significant problem, and three out of the five do some kind of monitoring of exotics in the wilderness. The fact that these are at lower elevations probably contributes to weed problems. Canada thistle (*Cirsium arvense*) was the most widely reported (by 4 of the wilderness areas), followed by smooth brome (*Bromus inermis*), Kentucky bluegrass (*Poa pratense*) and leafy spurge (*Euphorbia esula*). The extent of invasion seems to be small enough that these could be addressed; however, populations of spotted knapweed, Canada thistle (at Badlands) and leafy spurge (at Medicine Lake) are increasing despite biological and mechanical control efforts. Populations of salt cedar and sweet clover are expanding in places where there are no control efforts.

Eastern Temperate Forests—This is a tree-dominated biome, in which the woody taxa are mostly winter deciduous (Greller 1989). This biome covers most of eastern US, and is usually divided into many subregions (see Greller 1989). Response rate was low overall for this biome, and exotic plants are not as widely perceived as a problem in Eastern states. Many agency employees, including the Forest Service’s Eastern Regional Wilderness Coordinator, said that there are few weed problems in Eastern states; however, The Nature Conservancy and a few federal biologists reported several exotic plant concerns.

For convenience, I discuss 4 subsections (Great Lakes, Southwest, Appalachian, and Northeast). Overall, about 20% of wilderness areas ranked exotics as a significant problem, although this varied greatly among subregions.

Great Lakes (Michigan and Wisconsin Wilderness Areas)—Seven of 14 areas ranked exotics as a significant problem. Although most wilderness areas had not been inventoried, several had lists of exotic plants likely to be found there. The invasive plant list for the Forest Service Eastern Region has 57 exotic invasive species, 10 native invasive species and over 150 widespread exotics that are not considered invasive. Of these, 37 were known or suspected in wilderness areas on the Hiawatha National forest (14 of these are considered invasive).

Nordhouse Dunes Wilderness in Michigan has a noxious weed program and is manually removing exotic poplars (*Populus nigra*) and St. Johnswort (*Hypericum perforatum*), which are not thought to be spreading anymore. About 50 acres of spotted knapweed (*Centaurea maculosa*) were manually cleared in 1997. Although it is still early to determine the effectiveness, the populations seem to be spreading despite these efforts. This is also the case for spotted knapweed in the Round Island, Big Island and Horseshoe Bay wilderness areas on the Hiawatha National Forest in Michigan, where spotted knapweed has been hand pulled for the last 1-3 years. The Hiawatha National Forest does not have staff available to prepare a noxious weed plan, although it seems like this is justified (Shultz, personal communication).

In northern Wisconsin, at least 44 exotics have been verified in wilderness areas, including the major pests spotted knapweed and Japanese barberry (*Berberis thunbergii*). All five Forest Service wilderness areas in Wisconsin are small, surrounded by roads and have been logged at some point (Sheehan, personal communication). There are no control efforts for exotics, and exotics are not monitored.

Southwest States (Alabama, Arkansas, Kentucky)—No wilderness areas in these states ranked weeds among their top 10 priorities, and exotics are monitored in only one of the nine responding areas. An interesting aspect of the species list from Arkansas is that many of the pest species are native plants that have become invasive. Botanists in this area complained that many agencies are actively planting aggressive exotic species for wildlife forage and erosion control. For example, the Ouachita National Forest and other agencies in Arkansas widely plant lespedeza (*Lespedeza cuneata*) for wildlife forage. Yet the Forest has spent several thousand dollars trying to control this same plant in the last few years (Owen, personal communication). Widely reported problem species were mimosa (*Albizia*

julibrissina), rose (*Rosa multiflora*), privet (*Ligustrum* species), Russian olive (*Elaeagnus* species), kudzu (*Pueraria montana*, also reported under the synonym *P. lobata*; see Ward [1998]) and Japanese honeysuckle (*Lonicera japonica*). The latter three species were originally intentional introductions, which later became invasive. Wilderness areas in Kentucky are adjacent to wildlife management areas, where aggressive exotics are planted as wildlife forage as well (Taylor, personal communication). Although agencies have not prioritized exotic plants as a management concern, several respondents considered exotic plants pose a serious threat to natural areas in this region.

Appalachian Forest (Georgia, Pennsylvania, South Carolina, Virginia, West Virginia)—Hickory Creek Wilderness in Pennsylvania is included in this section for convenience; this is the only Forest Service wilderness area that had any information on exotic plants. The other 16 that responded confirmed that they had no information on exotics in the wilderness areas, but they were not expected to be a problem there. A botanist in West Virginia estimated that 60% of the Monongahela National Forest flora is composed of naturalized exotics, and considerably fewer than that are considered problem “weeds” (Concannon, personal communication). However, they had no information on exotic plant distribution in wilderness areas, since weed surveys are done in conjunction with rare plant surveys and at silvicultural sites, neither of which occur in wilderness.

In contrast, exotics in Shenandoah National Park in Virginia are considered a significant management concern, and are monitored in the wilderness portion. The aggressive weeds garlic mustard (*Alliaria petiolata*), kudzu (*Pueraria montana*), Japanese knotweed (*Polygonum cuspidatum*), Japanese honeysuckle (*Lonicera japonica*) and tree-of-heaven (*Ailanthus altissima*) have all been found in this in wilderness area. Some of these plants are treated or monitored, but the garlic mustard is too widespread for either.

Perhaps Shenandoah has a worse weed problem than other in wilderness areas in the Appalachians, because as a national park it may receive more visitors. Several studies have shown that the number of visitors to a natural area is correlated with the number of exotic plants present (MacDonald 1985, Lesica and others 1993). Although it seems likely that in wilderness areas throughout the Appalachians have similar exotic plants, other agencies have not attempted to document exotic species in their in wilderness areas.

Northeast States (New Jersey, New York, Vermont, New Hampshire, Maine, Massachusetts)—None of the 17 federally designated in wilderness areas in the northeast states responded. One Forest Service biologist in Vermont indicated it would be difficult to respond to the survey without a better definition of “exotic,” since a large component of their flora is composed of naturalized species. Fortunately, Adirondack State Park, a large in wilderness area managed by the state of New York, did submit information on their exotic plant concerns. Historically, there has been little attention given to exotic or invasive species in the Park except for the Lake Champlain area and the St. Lawrence Valley, which are outside the in wilderness areas, and have problems with water milfoil (*Myriophyllum spicatum*) and purple loosestrife (*Lythrum salicaria*). The wilderness areas of Adirondack Park fortunately do not currently have many problem exotic

plants, and The Nature Conservancy staff is prioritizing prevention for these in areas. While those exotics that are present appear to be restricted to roads and trails, Conservancy staff expect that some problem plants may be establishing in remote areas. They are conducting a survey of exotics this year in order to identify problems and preventative actions needed to maintain the natural plant communities (Brown, personal communication). Their “watch out” list includes crown vetch (*Coronilla varia*), garlic mustard (*Alliaria petiolata*), Eurasian water milfoil, and black locust (*Robinia pseudoacacia*).

Southeast Coastal Marsh, Swamp, Bog, Forests—Seven of 14 responding wilderness areas in this biome monitor exotics, and 29% of respondents ranked exotics as a significant problem. Biologists in this region were aware of and concerned about exotics. Managers knew where their exotic plant problems were and could confirm confidently that there were few weed problems in wilderness areas. Almost all respondents were using combinations of mechanical and herbicide treatments to control weeds in and near wilderness areas. Tom Wilmers at the National Deer Key Refuge reported that they have successfully controlled exotics by detecting problems early, acting quickly and following up on treated sites.

Wilderness areas on the Apalachicola and Ocala National Forests in Florida are checked for exotics and are not known to have any weeds. However, aggressive exotic species are close enough to be considered a serious threat. Japanese honeysuckle (*Lonicera japonicum*), Chinese tallow tree (*Sapium sebiferum*) and privet (*Ligustrum sinense*) were the most widely reported problem species throughout the region.

Mechanisms of Spread in Wilderness Areas

Both natural disturbance and disturbances associated with human activity contribute to the establishment and spread of exotics in wilderness areas. Common human disturbances listed as causal agents included livestock use, trail use, camping and existing roads adjacent to wilderness areas. Not surprisingly, land use history prior to wilderness designation had a large effect on the extent of nonnative plant distribution. Historical and active grazing allotments were often cited as a source of exotic plants (Rutman *in press*, Isle, personal communication). Dudley and Embury (1995) discuss grazing impacts in California wilderness areas in detail.

Natural disturbances, including gopher pockets, floods, storms and fire were also reported to contribute to weed establishment or spread. There is a growing understanding of the role of natural disturbance in shaping natural communities and ecosystems (Sprugel 1991, Cole and Landres 1996). Paradoxically, reintroducing natural disturbances into wilderness ecosystems may facilitate exotic plant invasions.

Many respondents reported fire as an important factor of weed spread, both in areas that are not adapted to fire (for example, desert communities) and in areas that area adapted to fire (Rocky Mountain states) (Anderson, Sanger, Rutman, Fritzke, personal communications). There are documented cases that some exotics will respond positively to a fire, then alter the fire regime to the exotic species' favor, resulting in

a positive feedback loop that maintains the exotic community (D'Antonio and Vitousek 1992). There also appears to be a plethora of anecdotal evidence on the relationship between fire and exotic plant invasion, as far back as an essay by Aldo Leopold (1941). Biologists from the Mojave and Sonoran deserts, Great Basin, and California mentioned that exotic annual grasses cause more frequent fires, which promote exotic grass expansion and suppress native species. Similar feedback loops have been documented for soil nutrients (Vitousek and others 1987) and soil salinity (Brotherson and Winkel 1986, Shafroth and others 1995).

The phenomenon of plant invasions is so complex that it is difficult to identify any single or few factors that are responsible across large scales. Sue Rutman, plant ecologist for the National Park Service, pointed out that many things contribute to the current distribution of exotic species at Organ Pipes National Monument. While areas with grazing impacts are often the most impacted by exotics, other problem areas exist where fires have destroyed native plant communities. Furthermore, species that depend on soil disturbance can colonize rodent mounds or other completely natural disturbances, and some species, include fountain grass (*Pennisetum ciliare*) will move into areas that have no apparent disturbances at all (Rutman, personal communication).

It is important to emphasize that although many invasive species depend on some level of disturbance to establish, the disturbance does not have to be large, and it does not have to be the result of human activity. We should not assume that the lack of recent disturbance precludes invasion by exotic weeds.

Appropriate Management Responses

Prevention and Early Detection

It is generally better to spend time eradicating a newly arrived exotic that might not have become a weed than to wait until a certain problem has developed (Randall 1991, Schwartz and Randall 1995, Hobbs and Humphries 1995, Reichard and Hamilton 1997, Reichard 1997). Early detection and prevention are the best ways to avoid huge sinks of financial and human resources in the long term.

Many survey forms were returned with success stories of early detection and containment. While the cost of controlling invasions may initially be more expensive than doing nothing, the long-term benefits of early action far outweigh the costs.

Education and Training

To minimize further spread of exotics in protected natural areas, awareness of the problem must increase substantially. This will require training agency personnel to recognize exotics early. In addition, increasing the general public's awareness of the problem is an important step (Krummerow 1992, Asher and Harmon 1995, Marion and others 1996, Marcus and others 1998). Several wilderness areas depend on casual observation and reports by nonstaffers for detection of weeds. Asher and Harmon (1995) outlined 5 strategies, including incorporating weed

awareness into the "Leave No Trace" mentality. In fact, with funding from state and federal agencies, Forest Service personnel at the Lolo National Forest in Montana have developed a "Leave No Weeds" campaign directed at elementary school students (Kulla, personal communication).

Many survey respondents pointed out that there is little to no funding at this time for weed monitoring or inventories and that data are collected opportunistically while other work is being conducted. The Selway-Bitterroot, Absaroka-Beartooth, Glacier National Park and other wilderness areas provide weed identification training for backcountry rangers and carefully document weed populations in the backcountry. This is a time- and cost-effective strategy for obtaining data on remote locations (Krummerow 1992, Marcus and others 1998).

Prioritization of Exotics

Only 6% of responding wilderness areas reported using some system to rank exotic plants for priority. Of the national parks wilderness areas, almost 30% use a ranking system to prioritize exotics. The National Park Service developed a generalized ranking system for exotic plants in 1993 (Hiebert and Stubbendieck 1993, Hiebert 1997). The system ranks each species in terms of (1) significance of impact, (2) feasibility of control, and (3) urgency of action. These should be considered in combination with the amount of habitat that is susceptible. The purpose of the ranking system is to separate real threats from benign species, so efforts can be directed most effectively.

Communication

Awareness and communication are key in avoiding mistakes of others and detecting problem species early. One of the most important tools of weed management is information exchange. It is important to know which plants are likely to become problematic and what to do about it once they have established. As previously mentioned, the best predictor of whether a plant will become invasive is whether it has invaded in other areas (D'Antonio and others 1994, Reichard 1997). This accounts for a significant portion of a species' ranking in the Park Service system. Communication and access to centralized information can clearly keep managers informed of which species are likely to cause problems.

The database compiled from the results of this survey is accessible over the Internet (www.umn.edu/biology/leopold) for wilderness area managers and other interested parties. The database can be queried by species (to see which wilderness areas reported it), by wilderness area name (to find out what species were reported). Ideally, access to information on what species are problems, which control methods have been used and contact information will help managers identify priorities for control.

Intentional Introduction of Exotics

Researchers have estimated that 99% of all exotic plants in North America were introduced intentionally (Reichard 1997, OTA 1993). Japanese honeysuckle, Russian olive,

purple loosestrife, kudzu and tree-of-heaven are a few examples of plants that were introduced intentionally and are now widespread problems for land managers. Surprisingly, agencies continue to introduce nonnative species for erosion control or wildlife forage. In many cases, exotics known to be aggressive are used. While most managers are not actively seeding any plants within the wilderness boundaries, exotics can and do spread across political boundaries.

In a 1997 survey of Forest Service Ranger Districts in Montana, Lesica and Miles (1998) found that over 80% of total area revegetated in Forest Service projects completed in 1994-96 used nonnative (or predominately nonnative) species. Some of the plant species used are considered aggressive, including smooth brome (*Bromus inermis*), crested wheatgrass (*Agropyron cristatum*), orchard grass (*Dactylis glomerata*) and yellow sweet clover (*Melilotus officinalis*). Collectively, these species were seeded on 1,529 acres in 1994-96 (Lesica and Miles 1998).

Many exotic plants used for revegetation projects do not appear to be invasive. However it is important to consider that (1) many species seem benign at first and subsequently "explode" after an initial lag phase, and (2) such revegetation projects are a chance to promote native species. Species that we plant now for immediate benefits may become serious problems in the future. Revegetation projects near wilderness areas should especially stay away from nonnative species.

Research Needs

The general lack of information in most wilderness areas suggests that basic surveying and monitoring should be prioritized. Many wilderness areas have successful monitoring and data management programs; perhaps general guidelines could be agreed upon for what kind of information to collect, how often to collect it, and how to store it.

The Nature Conservancy has identified early detection and action as one of the biggest needs in their weed program (Randall 1991). Part of their response is to maintain stewardship abstracts and a regularly updated database on weeds in their preserve system, as well as make information on new invaders available to their preserve managers. Government agencies should follow their example and promote awareness and communication on this issue.

Wilderness areas are high priorities for weed control, and rigorous monitoring should accompany efforts. Monitoring is an overlooked but important part of research that contributes to knowledge of effective management, which must be based on science to be effective. Few of the respondents in this survey were able to determine objectively whether their control efforts were effective, because monitoring is rarely funded. Monitoring is necessary to detect changes from a current state or following a treatment (such as plant removal, herbicide treatment, population response to prescribed burns and so on) and should be designed as rigorously as a controlled experiment (Huenneke 1995, Morrison 1997). This is an area where academic ecologists and those directly involved in management plans can collaborate.

We need to consider how to restore natural disturbance regimes if they increase the chance of surrounding aliens establishing (Cole and Landres 1996). For example, reintroduction of fire to natural areas in the Rocky Mountains, an ecosystem

adapted to fire, may have a positive effect on weed spread. Several respondents mentioned concern over this possibility. The Frank Church-River of No Return Wilderness has initiated a long-term study of weed populations in prescribed burn areas, but no other wilderness areas reported such projects.

Finally, we do not understand all of the ecological effects of plant invasions. There are well-documented examples of significant ecological changes resulting from exotic plant invasions (increased soil nitrogen input, decreased fire intervals, altered phosphorus cycling, loss of species diversity and so on) but many more remain. This is an active area of research, with endless possibilities for investigating interactions between plants and an environment in which they did not evolve. Most of the plants that have been introduced cannot be eradicated, and we need to understand the ecological impacts of these species.

Conclusions

This survey indicated that exotic weeds are increasingly invading wilderness areas. Most wilderness area managers are not aware of major weed problems, and therefore it is important to emphasize prevention and early detection. However, 70% of responding wilderness areas do not monitor for exotic plants, and several had no information at all on exotic plant distribution. Thus, to some extent, the real status is still unknown. Even though 15% of wildernesses ranked weeds as a top priority, most management units cited lack of funds and staff to deal with weed issues, and many reported that they could not confidently assess the situation.

Almost all of the information compiled in this survey is anecdotal. This illustrates the need for standardized data collection, or at least for common objectives across agencies for wilderness areas. Increased funding, awareness and training, and regular monitoring and treatment at trailheads would be helpful starting points.

Most importantly, prevention, early detection and rapid response are necessary to deal with this problem, and exotic plants should be a top priority for wilderness management. Exotic plants do not require a large disturbance to spread, and managers should not assume that exotic species are absent from wilderness areas in general. The database and findings of the survey can be used to promote awareness of the issue, help prioritize areas and species for attention, and facilitate communication and discussion of weeds in our wilderness areas.

References

- Anderson, B. 1997 [Phone conversation with M. Marler] Noxious Weed Specialist, Nez Perce National Forest, ID.
- Asher, J. A. and D. W. Harmon. 1995. Invasive exotic plants are destroying the naturalness of US WAs. *International Journal of Wilderness* 1:35-37.
- Austin, T. 1997. [Letter to M. Marler] Los Padres National Forest, CA.
- Balay, J. 1997. [Letter to M. Marler] Biological technician, Denali National Park, AK.
- Brotherson, J. D., and Winkel, V. 1986. Habitat relationships of saltcedar (*Tamarix ramosissima*) in central Utah. *Great Basin Naturalist* 46:535-541.
- Brown, B. 1997. [Letter to M. Marler] Adirondack Nature Conservancy, NY.

- Carlton, J. T. 1996. Biological invasion and cryptogenic species. *Ecology* 77:1653-1655.
- Cole, D. N. and Landres, P. B. 1996. Threats to wilderness ecosystems: impacts and research needs. *Ecological Applications* 6:168-184.
- Concannon, J. 1997. [Phone conversation with M. Marler] Forest Botanist, Monongahela National Forest, WV.
- Cooper, K. 1997. [Letter to M. Marler] National Biological Survey, Point Reyes, CA.
- D'Antonio, C. M. and Vitousek, P. M. 1992. Biological invasion by exotic grasses, the grass/fire cycle and global change. *Annual Review of Ecology and Systematics* 23:63-87.
- Dudley, T. and Collins, B. 1995. Biological Invasions in California wetlands: Impacts and control of non-indigenous species in natural areas. Pacific Institute for Studies in Development, Environment and Security, Oakland, CA, USA.
- Dudley, T. and Embury, M. 1995. Non-indigenous species in Wilderness areas: the status and impacts of livestock and game species in designated Wilderness in California. Pacific Institute for Studies in Development, Environment and Security, Oakland, CA, USA.
- Elton, C. 1958. The ecology of invasions by animals and plants. Methuen and Co. Ltd. London.
- Felger, R. S. 1990. Non-native plants of Organ Pipe Cactus National Monument, Arizona. Technical Report No. 31, Cooperative National Park Resources Studies Unit, School of Renewable Natural Resources, University of Arizona, 85721.
- Fritzke, S. 1997. [Email letter to M. Marler] Yosemite National Park, CA.
- Greller, A. M. 1988. Deciduous Forests. Pages 288-316 in Barbour, M. G. and Billings, W. D., editors. North American terrestrial vegetation. Cambridge University Press, Cambridge.
- Grenier, K. 1998. [Phone conversation with M. Marler] Forest Biologist, Deschutes National Forest, OR.
- Heady, H. F., Bartolome, J. W., Pitt, M. D., Savelle, G. D., Stroud, M. C. 1992. California prairie. Pages 313-335 in Coupland, R.T., editor, *Ecosystems of the world. v8a: Natural Grasslands: Introduction and Western Hemisphere*, Elsevier Scientific Publishing Company, Amsterdam, The Netherlands.
- Hiebert, R. D. and Stubbendieck, J. 1993. Handbook for ranking exotic plants for management and control. U.S. National Park Service Natural Resources Rep NPS/NRMWRO/NRR-93/08.
- Hobbs, R.J. and Humphries, S.E. 1995. An integrated approach to the ecology and management invasions. *Conservation Biology* 9:761-770.
- Huenneke, L. F. 1995. Involving academic scientists in conservation research: perspectives of a plant ecologist. *Ecological Applications* 5:1995.
- Huenneke, L. F. 1997. Outlook for plant invasions: interactions with other agents of global change. Pages 95-103 in J.O. Luken and J.W. Theiret, editors. *Assessment and Management of Plant Invasions*. Springer, New York.
- Isle, D. 1998. [Email to M. Marler] Botanist, Mendocino National Forest, CA.
- Jean, C. 1998. [Letter to M. Marler] Humbolt-Toiyabe National Forest, NV.
- Krummerow, M. 1992. Weeds in Wilderness: A threat to biodiversity. Western Wildlands. Summer: 12-17.
- Kulla, A. 1997. [Conversation with M. Marler] Resource specialist, Lolo National Forest, MT.
- Lachowski, H. and Varner, V. 1996. Noxious weeds and remote sensing: A literature review prepared for the Remote Sensing Steering Committee. Remote Sensing Applications Center, USDA Forest Service, Salt Lake City, UT.
- Lake, L., Anderson, B., Varner, V., and Lachowski, H. 1997a. Mapping and monitoring noxious weeds using remote sensing: Final project report. Remote Sensing Applications Center, USDA Forest Service, Salt Lake City, UT.
- Lake, L., Anderson, B., Varner, V. and Lachowski, H. 1997b. Mapping and monitoring noxious weeds using remote sensing. Remote Sensing Applications Center, USDA Forest Service, Salt Lake City, UT.
- Landres, P. and Meyer, S. 1998. A National Wilderness Preservation System database: Key attributes and trends, 1964-1997. USDA Forest Service General Technical Report. Rocky Mountain Research Station, Ogden, UT.
- Leopold, A. 1941. Cheat takes over. *The Land* 1:310-313.
- Lesica, P. and Miles, S. 1998. Revegetation on Montana's National Forests. *Kelsey* 11:1-10.
- Lodge, D. M. 1993. Biological invasions: Lessons for ecology. *Trends in Ecology and Evolution* 8:133-137.
- Loope, L. 1993. An overview of problems with introduced plant species in National Parks and biosphere reserves of the United States. In C.P. Stone, C.W. Smith, and T. Tunison, editors. *Alien Plant Invasions in Native Ecosystems of Hawai'i: Management and Research*. University of Hawai'i Cooperative National Park Resources Studies Unit. Honolulu.
- Luken, J. O. 1994. Valuing plants in natural areas. *Natural Areas Journal* 14:295-299.
- MacDonald, I. A. W., Loope, L., Usher, M. B., and Hamann, O. 1989. Wildlife conservation and the invasion of nature reserves by introduced species: a global perspective. Pages 215-255 in J. A. Drake, H. A. Mooney, F. diCastri, R. H. Groves, F. J. Kruger, M. Rejmanek, M. Williamson, editors. *Biological Invasions: a Global Perspective*. John Wiley and Sons, Ltd, Chichester.
- Marcus, W. A., Milner, G., and Maxwell, B. 1998. Spotted knapweed distribution in stock camps and trails of the Selway-Bitterroot Wilderness. *Great Basin Naturalist* 58:156-166.
- Marion, J. L., Cole, D. N., and Bratton, S. P. 1986. Exotic vegetation in Wilderness areas. Pages 114-120 in R.C. Lucas, editor. *Proceedings of the National Wilderness Research Conference*. General Technical Report INT-212. Intermountain Research Station, USDA Forest Service, Ogden, UT.
- Morrison, M. L. 1997. Experimental design for exotic plant removal and restoration. Pages 104-116 in J. O. Luken and J. W. Theiret, editors. *Assessment and Management of Plant Invasions*. Springer, New York.
- Office of Technology Assessment [OTA], U.S. Congress. 1993. Harmful Non-indigenous species in the United States, OTA-F-565. U.S. Government Printing Office, Washington, DC.
- Owen, W. 1997. [Letter to M. Marler] Forest Botanist, Ouachita National Forest, OK.
- Powell, E. 1998. [Letter to M. Marler] Lake Mead National Recreation Area, NV.
- Randall, J. M. 1991. Exotic weeds in North American and Hawaiian Natural Areas: The Nature Conservancy's plan of attack. Pages 159-172 in B.N. McKnight, editor. *Biological Pollution: the control and impact of invasive exotic species*. Indiana Academy of Science, Indianapolis.
- Randall, J. M. 1997. Defining weeds of natural areas. In J.O. Luken and J. W. Theiret, editors. *Assessment and Management of Plant Invasions*. Springer, New York.
- Reichard, S. H. 1997. Prevention of invasive plant introductions on national and local levels. Pages 215-227 in J. O. Luken and J. W. Theiret, editors. *Assessment and Management of Plant Invasions*. Springer, New York.
- Reichard, S. H. and C. W. Hamilton. 1997. Predicting invasions of woody plants introduced into North America. *Conservation Biology* 11:193-203.
- Rutman, S. 1997. Dirt is not cheap. Symposium on environmental, economic, and legal issues related to rangeland water developments. November 13-15 1997. Arizona State University College of Law, AZ. *Proceeding in press*.
- Rutman, S. 1998. [Letter to M. Marler] Ecologist, Organ Pipe Cactus National Monument, AZ.
- Sanger, A. [Letter to M. Marler] Modoc National Forest, CA.
- Schwartz, M.W. 1997. Defining indigenous species: an introduction. Pages 7-17 in J.O. Luken and J.W. Theiret, editors. *Assessment and Management of Plant Invasions*. Springer, New York.
- Schwartz, M.W. and J.M. Randall. 1995. Valuing natural areas and controlling exotic plants. *Natural Areas Journal* 15:98-100.
- Shafroth, P.B., J.M. Friedman, L.S. Ischinger. 1995. Effects of salinity on establishment of *Populus fremontii* (cottonwood) and *Tamarix ramosissima* (saltcedar) in southwestern United States. *Great Basin Naturalist* 55:58-65.
- Sheehan, Q. 1998. [Letter to M. Marler] Botanist, Chequamegon-Nicolet National Forest, WI.
- Shevock, J. 1998. [Phone conversation with M. Marler]. Regional Botanist, Region 5 Forest Service Headquarters.
- Shultz, J. 1998. [Letter to M. Marler] Botanist, Hiawatha National Forest, WI.

- Soulé, M.F. 1990. The onslaught of alien species, and other challenges of the coming decades. *Conservation Biology* 4:233-239.
- Sprugel, D.G. 1991. Disturbance, equilibrium, and environmental variability: What is "Natural" vegetation in a changing environment? *Biological Conservation* 58:1-18.
- Stewart, G. and A.C. Hull. 1949. Cheatgrass (*Bromus tectorum* L.) an ecologic (sic) intruder in southern Idaho. *Ecology* 30:58-74.
- Stone, CP, C.W. Smith, and J.T. Tunison, editors. 1993. Alien plant invasions in native ecosystems of Hawai'i: management and research. Cooperative National Park Resources Studies Unit, Hawaii. University of Hawaii, Cooperative National Park Resources Studies Unit. Honolulu, HA.
- Tanner, L. 1998. [Phone conversation with M. Marler] Botanist, Sequoia National Forest, CA.
- Tausch, R.J., P.E. Wigand and J.W. Burkhardt. 1993. Viewpoint: Plant community thresholds, multiple steady states, and multiple pathways: legacy of the Quaternary? *Journal of Range Management* 46:439-447.
- Taylor, D. 1998. [Letter to M. Marler] Botanist, Daniel Boone National Forest, KY.
- Therrell, L. 1998. [Letter to M. Marler] Wenatchee National Forest, WA.
- Vermeij, G.J. 1991. When biotas meet: Understanding biotic interchange. *Science* 253:1099-1103.
- Vitousek, P.M., L.R. Walker, L.D. Whiteaker, D. Mueller-Dombois. 1987. Biological invasion by *Myrica faya* alters ecosystem development in Hawaii. *Science* 238:802-804.
- Vitousek, P.M. 1990. Biological invasions and ecosystem processes: towards an integration of population biology and ecosystem studies. *Oikos* 57:7-13.
- Vitousek, P.M. 1994. Beyond global warming: ecology and global change. *Ecology* 1861-1876.
- Vitousek, P.M., C.M. D'Antonio, L.L. Loope, M. Rejmanek, and R. Westbrooks. 1997. Introduced species: a significant component of human caused global change. *New Zealand Journal of Ecology* 21:1-16.
- Ward, D.B. 1998. *Pueraria montana*: the correct scientific name of kudzu. *Castanea* 63:76-77.
- Webb, D.A. 1985. What are the criteria for presuming native plant status? *Watsonia* 15:231-236.

Evaluating Effects of Fish Stocking on Amphibian Populations in Wilderness Lakes

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Abstract—To balance wilderness lake use between recreational fisheries and protected habitat for native species, managers need to understand how stocking non-native predaceous fish affects amphibian populations within a landscape. The goal of this paper is to help managers design and conduct studies that will provide such information. Desirable study characteristics include multiple-visit surveys of all wetlands within a watershed to provide information on amphibian distribution, abundance, breeding, recruitment and seasonal variation in habitat use. By identifying the distribution of critical amphibian habitat and source populations, this approach should enable managers to target specific lakes for protection or restoration as fishless amphibian habitat without overly compromising wilderness fishing opportunities.

Wild areas, large or small, are likely to have values as norms for land science. Recreation is not their only, or even principal utility.

—Aldo Leopold, Sand County Almanac

In the last few years, the long-accepted practice of stocking “sport fish” in wilderness lakes has attracted both professional and public attention (Forstenzer 1998; Knapp 1994; Matthews and Knapp 1999; Murray 1994; Yuskavitch 1999). Concern that introduced trout may be threatening the persistence of native species has put pressure on managers to evaluate stocking practices and their impacts on native biota. After decades of providing recreational fisheries for backcountry anglers, state and federal agencies are now reconsidering how to manage wilderness lakes (Duff 1995; Fraley 1996; Gill and Matthews 1998; Rahel 1997).

Responding to these concerns, several agencies have initiated studies to examine the impacts of fish stocking in federally designated wilderness and other protected public lands. Many of these studies are focusing on fish-amphibian interactions to determine whether introduced trout may be contributing to the documented decline of amphibian species from the mountainous regions of the western United States (Blaustein and others 1994; Corn, in press; Fellers and Drost 1993; Fisher and Shaffer 1996; Hayes and Jennings 1986).

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Wilderness Fish Stocking

Despite the apparent novelty of concerns over wilderness fish stocking, organized dialogue among public interest groups, biologists and managers actually began more than two decades ago. In 1976, the American Fisheries Society and the International Association of Game, Fish, and Conservation Commissioners held a symposium entitled Management of Wilderness Area Waters (Gottschalk 1976). Recently, in October 1998, biologists and managers convened again to discuss the Effects of Fisheries Management on the Amphibians and Other Biota of Wilderness Lakes (Corn and Knapp, this volume). Both of these meetings emphasized that potential legal, social and biological problems exist for wilderness fish stocking, additional research is needed to evaluate the scope of the problem, and wilderness fish stocking policies require adaptive management between state fisheries agencies and federal land managers.

The legal, social and biological controversies surrounding wilderness fisheries issues can be summarized as follows. Due to the steep topography of the western United States, few fish colonized mountain watersheds since the last glaciation, so approximately 95% of roughly 16,000 high-elevation lakes were historically fishless. However, in the last century, sportsman clubs and state game agencies have stocked over 60% of these high mountain lakes, including about 95% of the larger (>2 ha surface area), deeper (>3 m maximum depth) lakes that looked like they might support fish (Bahls 1992). The widespread introduction of regionally exotic and locally non-native trout (such as eastern brook trout, *Salvelinus fontinalis*) into historically fishless lakes has dramatically altered these communities. Finally, most high-elevation lakes are located in wilderness and national parks, areas set aside to remain “untrammeled by man” and provide protected habitat for native species (Hendee and others 1990). Consequently, conflicts between state management of wilderness fisheries (section 4(d)(8) of the 1964 Wilderness Act; P.L. 88-577) and federal mandate to protect the biological integrity of wilderness (USDA 1986; USDA 1990) are inherent.

Comparative studies of high-elevation lakes with and without introduced trout have suggested that fish reduce or eliminate some amphibian species from stocked lakes (Bradford 1989; Braña and others 1996; Funk and Dunlap, in press; Knapp and Matthews, this volume; Liss and Larson 1991; Munger and others 1997; Pilliod and Peterson 1997; Tyler and others 1998a). Although the causes of this negative relationship remain uncertain, controlled experiments (Tyler and others 1998b) and field observations (Braña and others 1996; Tyler and others 1998a) indicate that fish predation on embryonic and larval life stages is responsible.

Although several studies have documented the effects of introduced fish on amphibian populations on a lake-by-lake basis, few studies have addressed these effects within a spatial context. Future studies need to look at watersheds as systems, identifying sources and sinks and prioritizing areas of critical amphibian habitat, so that information is available to target specific lakes for management actions. The goal of this paper is to provide information to help managers design and conduct such studies, and evaluate possible management actions. Our recommendations were developed from reviewing the literature and conducting a five-year, landscape-scale amphibian and trout study in 73 headwater lakes in the Frank Church-River of No Return Wilderness, Idaho. This paper is structured around six key questions that managers should try to answer when setting up and conducting these studies.

1. What Pre-Existing Information Can Help Evaluate Threats and Plan a Study?

One of the first steps in evaluating the potential threats of trout stocking in an area is to determine which amphibian and fish species may occur there. At least some of this information may be obtained from existing sources, such as state databases maintained by natural heritage programs and state fish and wildlife agencies. State databases often include hard-to-find information such as museum records, agency reports and contributed field observations. State GAP Analysis programs may also provide some of these data, including current and predicted distribution maps. State and federal agency biologists may be able to provide a list of studies that have been conducted in a geographical area as they are usually more familiar with the considerable amount of data available in the gray literature (such as government reports). Finally, state fish and wildlife agencies can provide a fish stocking history of specific waters, although these data generally do not represent fully accurate and complete fish distributions because of historic name-changes, pilot error and fish colonization.

Identifying which species require attention, such as state and/or federally listed species, is important and may be information best obtained from a local or regional herpetologist for several reasons. First, formal designations may not reflect current or local status; some species may be declining locally, and there are time lags before species are placed on Endangered, Threatened or Species of Special Concern lists. In addition, the trend in molecular systematics is to split species, in which case single species may become two or more. For example, the spotted frog (formerly *Rana pretiosa*) was shown to be made up of two species, the Oregon spotted frog (*Rana pretiosa*) and the Columbia spotted frog (*Rana luteiventris*) (Green and others 1997). This change in taxonomy influences the status, distribution and management of these frogs. *Rana pretiosa* now refers only to populations in the Pacific Northwest. These have undergone serious declines compared to *Rana luteiventris*, which is widely distributed and common in the northern Rocky Mountains.

After identifying the potential species and their status in an area, the next step is to prioritize which amphibians should be targeted for surveys by determining whether any life stages of a species may occur in fish habitat. Amphibian species with minimal interaction with trout are likely those

that breed in ephemeral wetlands (pools, wet meadows) and over-winter in terrestrial locations. Tree frogs, spadefoots and some salamanders fit these life history characteristics. Species with the greatest interaction with fish are those that breed and over-winter in permanent wetlands (lakes, ponds, creeks). Many anurans and some salamanders fall into this category, such as spotted frogs, mountain yellow-legged frogs (*Rana muscosa*), leopard frogs (*Rana pipiens*) and larval long-toed salamanders (*Ambystoma macrodactylum*).

Susceptibility to predation also should be considered when deciding on which species to focus. For example, many stream-dwelling salamanders and newts are able to coexist with trout because of behavioral and chemical defenses (Kats and others 1988; Petranks and others 1987; Sih and others 1992). Many toads also have toxic or repellent skin secretions in the egg, larval and adult life stages that enable them to coexist with predaceous fish (Jones and others 1999; Voris and Bacon 1966).

Future studies need to investigate variation in predation pressures of different trout species commonly stocked in mountain lakes. For example, in some circumstances, eastern brook trout may have stronger effects on zooplankton (Anderson 1980) and amphibian (Bahls 1990) communities than do other species of trout. However, other studies suggest that the feeding behaviors of brook and cutthroat trout (*Oncorhynchus clarki*) are fairly similar (Carlisle and Hawkins 1998).

2. What Techniques Are Appropriate for Landscape-Scale Studies?

There is a range of techniques that can be used to determine the distribution and abundance of amphibians in different habitats (see Heyer and others (1994) and Olson and others (1997)). For example, a variety of techniques can be employed to sample the different life stages of three common, lentic-breeding amphibians found in the Pacific Northwest (Table 1). Although no single technique is appropriate for sampling all species or even all life stages of one species across different habitats, one of the most common survey techniques is the visual encounter survey (VES).

Visual encounter surveys are particularly reliable for many lentic-breeding amphibians (especially ranid frogs) in habitats with relatively low structural complexity (sparse aquatic vegetation, firm substrate and delineated shoreline). In large marshes with dense vegetation, we recommend other techniques, such as trapping.

Because Thoms and others (1997) provide an excellent description of the VES technique, we will not elaborate here other than to emphasize a few points relevant to surveying lakes with fish. First, surveys should include any wetlands adjacent to lakes (such as ephemeral pools, wet meadows) because these sites are often utilized by breeding amphibians when fish are present in a lake. In addition, because amphibian larvae generally become less active and seek cover in lakes with fish (Taylor 1983; Tyler and others 1998a), dip-netting aquatic vegetation, submerged woody debris and unconsolidated bottoms may be particularly important to detect this life stage (Wassersug 1997).

Enumerating the life stages of amphibians observed during VES's can provide important abundance information, even though these data may or may not be indicators of

Table 1—A summary of collection and detection techniques for three common lentic-breeding amphibians found in the Pacific Northwest.

Species life stage	Location	Season	Techniques ^a	Difficulty ^b	Remarks
Long-toed Salamander (<i>Ambystoma macrodactylum</i>)					
Adult - breeding	lakes, ponds, creeks	late winter/early spring	aft, pit, ves, cov	easy	
Adult - active	wetlands, uplands	spring - summer	cov, pit	difficult	best during rain
Eggs	lakes, ponds, oxbows	late winter/early spring	ves	easy	eggs deposited at ice-out
Larvae	lakes, ponds, oxbows	all year, may over-winter	aft, dip, snk	easy	in shallows & open water
Metamorphs	under cover at shoreline	late summer	pit, cov	difficult	
Juveniles	uplands	spring - summer	pit, cov	difficult	
Western Toad (<i>Bufo boreas</i>)					
Adult - breeding	lakes, ponds, creeks	spring - summer	ves, cal, lit	variable	
Adult - active	wetlands, uplands	spring - fall	ves, drv	variable	crepuscular
Eggs	lakes, ponds, oxbows	spring - summer	ves	moderate	may be covered by silt
Larvae	lakes, ponds, oxbows	all year, may over-winter	ves, dip	easy	aggregate in shallows
Metamorphs	shoreline	summer - fall	ves, pit, cov	easy	may be very numerous
Juveniles	wetlands, uplands	spring - fall	ves	difficult	
Columbia Spotted Frog (<i>Rana luteiventris</i>)					
Adult - breeding	lakes, ponds, oxbows	spring	ves, dip, pit	variable	calls difficult to hear
Adult - active	riparian, wetlands	spring - fall	ves, dip, pit	easy	near water or in wet meadows
Eggs	lakes, ponds	spring	ves	easy	floating, communal oviposition sites
Larvae	lakes, ponds, oxbows	spring - summer	ves, dip, aft	easy	may hide in bottom detritus
Metamorphs	shoreline, meadows	late summer - fall	ves, pit	easy	
Juveniles	riparian, wetlands	spring - fall	ves, dip, pit	easy	

^aTechniques: aft-aquatic funnel traps, cal-calling surveys, cov-turning cover, dip-dip netting, drv-night driving, lit-spot lighting, pit-pitfall traps, sho-electroshocking, snk-snorkeling, ves-visual encounter surveys. Techniques in table are listed in the order of effectiveness.

^bDifficulty: Estimate of the difficulty of detecting individual animals under optimal conditions using appropriate techniques.

population size. Amphibian population sizes are notoriously variable, partly as a function of the number present (which is complicated by episodically high mortality or recruitment, and seasonal movements) and the weather conditions during the survey. For example, determining whether low numbers of individuals observed during a survey is due to a population crash, seasonal migration away from a lake or low observation rates associated with weather is difficult. Furthermore, the VES may be particularly unreliable for enumerating amphibians in certain habitats. In a survey of 115 marshy wetlands in Utah's west desert, fewer than 200 adult spotted frogs were observed, despite the presence of nearly 7,000 egg masses (Ross and others 1994).

Despite these caveats, in some circumstances, abundance data may be fairly reliable and provide information not represented by presence-absence analyses (see section 5). In our spotted frog surveys, we usually observed only 50% of the frogs at each site (even when we thought we had seen all the frogs), yet we were able to confidently categorize frog populations as low or high after obtaining similar results from several visits. We verified these results using mark-recapture, which certainly provides the most unambiguous information on abundance and seasonal habitat use, but may be beyond the scope of most studies.

3. How Should Sampling Effort Be Spatially and Temporally Distributed?

Ideally, all wetlands in a study area should be sampled, providing a complete survey (Fellers 1997). However, time and monetary constraints rarely permit this level of effort,

so we suggest stratified sampling at the watershed level. In other words, surveying all wetlands within randomly or systematically chosen watersheds (for example, selecting on topography or stocking history). Watershed-level sampling should provide the most unbiased, complete information about the distribution, breeding and habitat use patterns of amphibians across a landscape at a scale that can be used by managers to effectively manage for fish and amphibians (Pilliod and Peterson, unpublished data).

An advantage to subsampling at the watershed scale is a reduction in the amount of travel time between distant sites. When working in remote locations, surveying all wetlands within fewer watersheds is usually more efficient than surveying only a few wetlands in each.

A limitation of this approach is that it concentrates survey efforts at a few locations. Subsampling wetlands over a larger area would improve generality; however, this approach loses the spatial context of amphibian distribution, abundance, and habitat use patterns that is needed to make effective management decisions. In the first year (1994) of our study, we subsampled wetlands across seven watersheds, based on stocking history (Pilliod and others 1996). In each watershed, lakes to be sampled were chosen from 1:24,000 topographical maps. This site-selection strategy missed unmapped smaller ponds and wet meadows that were important breeding habitat for amphibians. As a result, in 1994, we greatly underestimated the amount of frog reproduction and completely missed one of the major source populations in a watershed. Subsampling at the wetland level erroneously indicated a worse situation for frogs than did comprehensive surveys conducted at the watershed level in subsequent years (Pilliod and Peterson 1997).

If resources are available, sites should be visited several times per year. Visiting a site only once provides potentially unreliable occurrence data (it is usually difficult to detect all species and life stages that occur at a site). In addition, some amphibian species use different habitat depending on the time of year or weather conditions, resulting in seasonal variability in occurrence and abundance. Performing two to three surveys in a year at all wetlands in a watershed should provide adequate and reliable information about occurrence and relative abundance, as well as important life history and habitat use information. If possible, conducting an additional survey in the spring of the following year will allow evaluation of between-year variability in the populations, as well as provide data on the survival of metamorphs through their first winter.

In our study, we tried to survey each site two to three times per year. The first survey was conducted in early July, one to two weeks after ice-out. This early-season survey enabled us to detect oviposition sites, count egg masses or tadpoles, count adults congregated at oviposition sites and count juveniles (an indicator of relative recruitment from the previous year's cohort). A second survey was conducted in early to mid-August to verify the reliability of VES data and to document use of summer foraging areas. Finally, we conducted a third survey in early September, when air temperatures were beginning to drop but before nighttime temperatures were cool enough to form ice. This late-season survey allowed us to document congregations of adults and juveniles at over-wintering locations, which were often sites not used by frogs in the spring and summer. For example, in the spring and summer, most frogs were isolated from trout in shallow breeding ponds or wet meadows. However, in the fall, many frogs congregated at deeper lakes, most of which contained trout, presumably to over-winter at ice-free depths. Late surveys also enabled us to document reproductive success, in terms of the number of metamorphs observed, compared with the number of egg masses or tadpoles previously counted.

Conducting landscape-scale studies involving multiple comprehensive surveys requires a significant amount of time and effort. We do not want to discourage studies with limited resources, but realizing that limited data can be misinterpreted is important. Using available resources for obtaining more complete information about a few areas is better than sparse and incomplete information across a larger region. The consequences of this approach are that information for management will be available for some areas, but not for others.

4. Can Amphibian Surveys Be Integrated Into Fisheries Studies to Evaluate Fish Stocking?

The considerable overlap of information gathered during fisheries and amphibian studies provides an opportunity for fisheries biologists to collect information about amphibians while conducting fish surveys. We wish to encourage this collaboration, but we emphasize the importance of effectively integrating herpetological sampling with existing fisheries research. For example, simply surveying for amphibians at sites visited for fisheries research may provide

useful baseline information about amphibian occurrence, but inadequate data on abundance and habitat use within a watershed. Fisheries studies rarely include ephemeral sites such as ponds and flooded meadows, which usually do not contain trout, but are often used by breeding amphibians. Furthermore, most fisheries studies visit each site once, missing information on seasonal habitat use of amphibians.

For fisheries biologists to provide data appropriate for managing for fish and amphibians, we recommend the following. Two field biologists, trained in amphibian identification, should accompany the fisheries crews, performing amphibian VES's at stocked lakes and all other wetlands within a watershed. As this team will spend more time in each watershed and return to watersheds to complete mid- and late-summer surveys, they may only be able to visit one third to one half of the watersheds that fish crews visit. Although this strategy will result in fewer areas surveyed, this approach should provide the necessary data for making effective management decisions in those areas.

5. What Information Is Needed to Evaluate Effects of Fish Stocking?

Most studies have approached this question on a lake-by-lake basis, documenting the occurrence and occasionally abundance of amphibians in lakes with and without trout. However, few studies have addressed these relationships within a spatial context (but see Bradford and others 1993). We recommend documenting the spatial and temporal distribution and abundance of the different life stages of amphibians in all wetlands to identify the spatial configuration of source populations and critical amphibian habitat within a watershed. This information can then be used to target specific lakes or groups of lakes that should be managed as amphibian reserves, instead of recreational fisheries.

Studies of this nature need to document occurrence, as well as abundance, of post-metamorphic amphibians because amphibians often colonize fringe habitat; occurring as very small sub-populations maintained by frequent immigration. For example, in our research, we found that spotted frogs were just as likely to occur in stocked and fishless lakes (78% and 84%, respectively), yet the abundance of frogs was significantly lower in the stocked lakes (fig. 1). Typically, the stocked lakes contained fewer than 10 post-metamorphic frogs. Munger and others (1997) found similar results for spotted frog and long-toed salamander populations in the Sawtooth Wilderness, Idaho. These studies suggest that documenting presence-absence of a species, without considering abundance, may be inadequate for determining the effects of introduced trout on amphibian populations.

Furthermore, many studies have assumed that the presence of amphibian reproduction at a site indicates a sustainable population, however this also may be misleading. In our study, we observed spotted frog tadpoles in 40% of the stocked lakes, yet few of those tadpoles survived to metamorphosis or through their first winter; resulting in very low recruitment of juveniles into those populations (fig. 2). This low recruitment indicates that stocked lakes may be population sinks, maintained only by colonization from source populations in surrounding fishless lakes (Hoffman and Pilliod 1999).

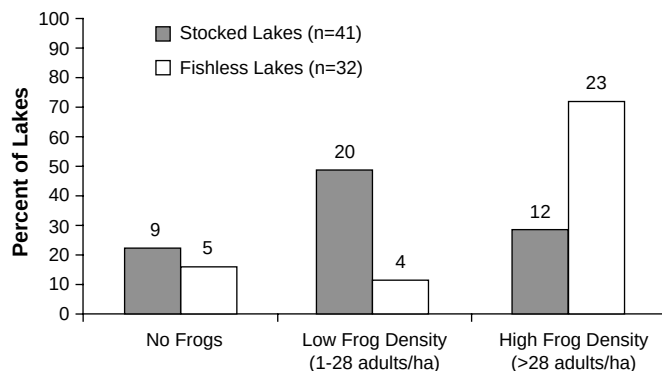


Figure 1—Differences between adult spotted frog occurrence and abundance in stocked (patterned bars) and fishless (open bars) lakes. Although frogs were just as likely to occur in stocked and fishless lakes ($\chi^2 = 0.464$, $df = 1$, $p = 0.496$), stocked lakes had significantly lower densities of adult frogs than fishless lakes ($\chi^2 = 13.799$, $df = 1$, $p < 0.001$). Densities were estimated as the average number of frogs observed per area searched (lake perimeter $\times$ 4) at each lake from 1994 to 1998. High and low frog densities represent lakes with frog densities above or below the median density for all lakes (28 adults/ha). The number of lakes in each category is given above each bar.

In addition, studies need to examine the seasonal habitat use patterns of amphibians within a watershed, to avoid missing important habitat conflicts between fish and amphibians. Despite a common misconception that amphibians hatch, live, and die in the same body of water, many amphibians require and utilize different habitat over the course of a year or lifetime (Duellman and Trueb 1986). Because many amphibians over-winter in similar habitats as fish (ice-free water) and most deep lakes now contain introduced trout (Bahls 1992), amphibians may have to over-winter in lakes with fish (Bradford 1989). Winter predation on amphibians is known to occur even under ice (Emery and others 1972; Griffith, Personal Communication), possibly contributing to low recruitment and low numbers of adults typical of lakes with fish. Furthermore, if frogs migrate from shallow, fishless wetlands to deep, stocked lakes to over-winter, winter predation of frogs from surrounding fishless wetlands could reduce recruitment in those populations as well. The loss of fishless over-wintering habitat may be one of the leading landscape-scale threats to amphibian persistence in mountain lake ecosystems and needs to be addressed in future studies.

Finally, understanding the influences of fish predation on amphibian distribution and abundance, requires an understanding of how habitat characteristics influence the presence of amphibians and fish, and mediate fish predation on amphibians. Several studies have identified certain physical, chemical, and biological lake characteristics that, if not addressed, could confound interpretations of fish effects on amphibians. For example, Bradford (1989) found that maximum lake depth influenced the occurrence of trout and mountain yellow-legged frog tadpoles, because shallow lakes (<1.5 m) did not provide over-wintering habitat for either taxa. Tyler and others (1998a) found long-toed salamander densities were associated with both water chemistry (total Kjeldahl nitrogen) and introduced trout. In lakes with low

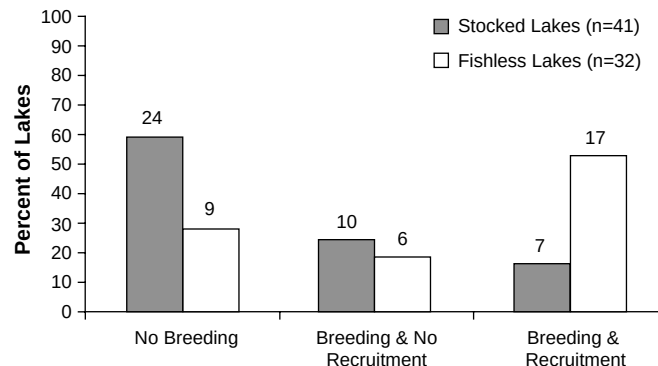


Figure 2—The percentages of stocked (patterned bars) and fishless (open bars) lakes without spotted frog breeding, with breeding but no recruitment, and with breeding and recruitment. The majority of stocked lakes had no breeding, whereas the majority of fishless lakes had both breeding and recruitment ($\chi^2 = 11.043$, $df = 2$, $p = 0.004$). Recruitment was based on the proportion of one-year-old juveniles observed in the spring relative to the number of adults and juveniles from the previous year. The number of lakes in each category is given above each bar.

nitrogen (<0.045 mg/L), salamander densities were low, even when trout were absent. Hence, evaluating the effects of introduced trout was only appropriate in lakes with high nitrogen concentrations. Bradford and others (1998) found that mountain yellow-legged frogs did not successfully breed in acidic lakes (pH <6.0) and rarely bred in lakes with trout. Consequently, they examined the effects of introduced trout only in non-acidic lakes. Finally, biological characteristics, such as shoreline emergent vegetation, may provide refugia for amphibians from fish predators, such that amphibian populations may be able to persist with trout (Hecnar and M'Closkey, 1997; Hoffman and Pilliod, 1999).

6. How Can This Information Be Used to Evaluate Potential Management Actions?

Like many ecological problems, the anthropogenic effects of trout stocking on amphibians can vary for different species and even different populations of the same species under a variety of conditions. This variability makes it difficult to make general management recommendations that will adequately protect all species and their habitats. However, research can greatly improve the evaluation and implementation of effective management actions that may balance the needs of the recreational public with conservation of native species. Ideally, any alterations in stocking practices should strive for the lowest cost-benefit ratio in terms of decreasing threats to amphibian persistence with the fewest changes to current recreational fishing opportunities.

Possible management actions include: (1) ceasing stocking in all lakes, (2) ceasing stocking and possibly removing fish from some lakes, (3) reducing stocking frequency and density, (4) reducing naturally reproducing populations of fish by restricting access to spawning areas and/or gill

netting, (5) changing species stocked (cutthroat may be less predatory than rainbow or brook trout), (6) stocking sterile fish, or (7) making no changes in stocking practices if fisheries threats to amphibian persistence are negligible.

Cessation of stocking in all wilderness lakes would most likely benefit amphibians and reduce threats to persistence (fig. 3). Undoubtedly, this action would be extremely unpopular for many anglers and could result in less support for wilderness. Economic impacts on outfitters and guides may also occur. Despite the potential socioeconomic costs of this management strategy, some wilderness proponents argue these costs will be minimal and will not overly jeopardize public support for wilderness (Murray and Boyd 1996). This view appears to be supported by resolutions from potentially opposing groups like the Society for Conservation Biology (SCB) and Trout Unlimited. The SCB recommends “phas[ing] out incongruent stocking practices and restor[ing], where appropriate and feasible, previously damaged ecosystems” (SCB 1995). Trout Unlimited states that it “oppose[s] salmonid stocking in historically documented non-salmonid waters where scientific evaluation indicates that such stocking would be likely to adversely affect native biodiversity” (Trout Unlimited 1998).

An example of the potential costs and benefits of restoring wilderness lakes through the cessation of fish stocking comes from the National Park Service, which recommended phasing out and eventually terminating all fish stocking (NPS 1975). In Sequoia, Kings Canyon and Yosemite National Parks, fish stocking was curtailed in the 1970’s and completely halted in 1991. This management decision resulted in the loss of recreational fisheries from 29% to 44% of previously stocked lakes (Knapp 1996). Due to a reduction in the proportion of lakes containing fish, as well as historic differences in stocking intensity, the mountain yellow-legged frog currently has a greater distribution in Kings Canyon National Park, compared with the neighboring John Muir Wilderness, where lakes have continued to be stocked and frog persistence is at risk (Matthews and Knapp 1999).

A similar pattern was observed in the Bitterroot Mountains, Montana where six of 18 stocked lakes (33%) no longer supported trout populations in 1996, following cessation of stocking in 1984 (Funk and Dunlap, in press). Funk and Dunlap (in press) found that long-toed salamanders recolonized five of these currently fishless, but previously stocked lakes within two decades, even in lakes over 5 km from the nearest salamander populations. These studies indicate that widespread cessation of stocking does not result in the loss all trout populations and that amphibians will recolonize lakes after fish disappear.

Cessation of fish stocking, and even removal of fish, in some but not all lakes may be more amenable to recreational anglers. If conducted properly, this management strategy could provide the necessary amphibian habitat for species recovery. The success of this management action, however, is dependent on which lakes are selected for fish elimination. Choosing lakes to be restored to a fishless condition based solely on anthropogenic variables, such as difficulty of access and amount of angler use, may have little effect on reducing threats to amphibian persistence (fig. 3). However, restoring fishless lakes based on their potential for amphibian recolonization and their importance as amphibian habitat should improve the success of this action.

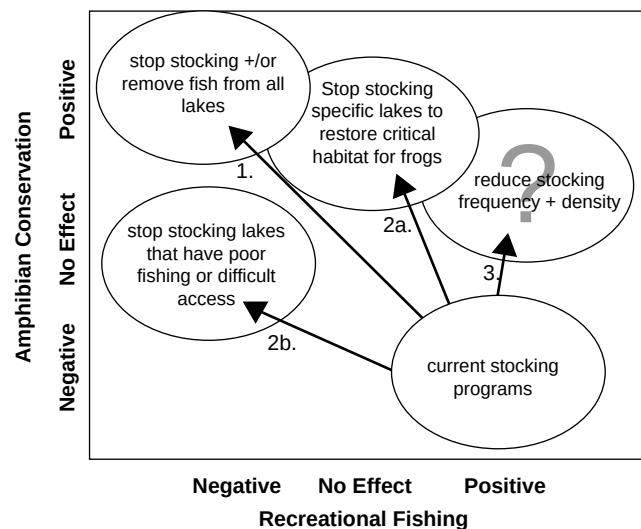


Figure 3—Diagram illustrating the effects of different management actions on recreational fishing and amphibian conservation. 1. Cessation of stocking in wilderness lakes can only help amphibians, however this will be unpopular with anglers. 2a. Restoring some lakes to their fishless state may increase amphibian persistence if lakes provide critical amphibian habitat, but have little affect if not (2b). 3. Reducing fish densities may benefit both frogs and fish, but this remains to be tested.

For fish elimination, we recommend targeting: (1) stocked lakes that already have some amphibian breeding occurring, (2) lakes that appear to provide deep-water over-wintering habitat for amphibians in surrounding shallow, fishless lakes, (3) lakes that have the potential for fish elimination (low or no natural reproduction), and (4) lakes that are the least important for recreational anglers. Of these recommendations, the first three should take priority over the last. In our study, over 40% of the stocked lakes had at least some frog reproduction, yet few of these lakes had any frog recruitment. Eliminating fish from a lake where frogs are already breeding should result in faster frog recovery than eliminating fish in a lake that has no amphibian reproduction. Furthermore, restoring lakes that provide over-wintering habitat for amphibians can benefit amphibians both locally and potentially across a watershed. Finally, when selecting a lake for fish elimination, choosing a lake that will require the least amount of invasive management (fish removal) is important. Nonreproducing fish can be eliminated from a lake by simply removing that lake from the stocking schedule. However, if fish removal is required, techniques such as gill netting (Knapp and Matthews 1998), coupled with blocking spawning habitat, are preferable to piscicides, such as rotenone and antimycin A. Both of these chemicals may harm other aquatic vertebrates, including amphibians (Fontenot and others 1994; Schnick 1974), and their use in wilderness is controversial.

The relatively easy, potentially risky, and yet untested management strategies include reducing the frequency, density, species, and/or fertility of fish stocked (fig. 3). This action has the potential to benefit both anglers and amphibians. In the best circumstance, densities of trout could be

reduced, even to the point of providing fishless or near fishless habitats for short intervals of time (several years). This strategy may be attractive to the angling public, if larger trout are caught during periods of low fish density (when lakes are designated as "trophy waters"). If amphibians could produce a successful cohort during these intervals, this action could help sustain populations of those amphibians that are long-lived. However, this strategy does not take into consideration the stochastic variables that can greatly influence amphibian recruitment, namely weather. In addition, larger fish have a greater gape and may prey on adult amphibians that were invulnerable to smaller fish (Semlitsch and Gibbons 1988; Zaret 1980). In amphibian populations, threats to older, reproductively mature individuals may be the most damaging to a population's persistence (Green 1997). In yet other circumstances, natural fish reproduction may reduce the effectiveness of this strategy at changing the density or size structure of fish populations. Clearly, further investigation of this strategy is warranted.

Finally, managers should keep in mind that most systems are not isolated, and fish stocking practices in adjacent regions can significantly affect restoration efforts. For example, fish dispersal from upstream locations may colonize wetlands that are actively managed as fishless habitats. In addition, fish predation in streams may act as barriers to migration, dispersal and hence colonization of amphibians (Bradford and others 1993).

Despite the range of possible management actions, we believe the best management strategy is to use species and watershed-specific biological information to make management decisions. This information can be obtained only through carefully designed and conducted studies that provide adequate information about the distribution, abundance and life history characteristics of amphibian species across local landscapes. Hopefully, using appropriate information at the watershed scale will enable managers to restore critical amphibian habitat and the biological integrity of wilderness lakes. Creating a few fishless lakes to provide the necessary habitat requirements of amphibians in a watershed may disproportionately reduce the threats of fish stocking on amphibian persistence. For example, having two amphibian source populations in a watershed, instead of one, may increase the probability of amphibian persistence in that watershed by an order of magnitude. With proper management, we believe amphibian populations can be recovered and protected while maintaining recreational fishing opportunities in many wilderness lakes.

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References

- Anderson, R. S. 1980. Relationships between trout and invertebrate species as predators and the structure of the crustacean and rotiferan plankton in mountain lakes. In: Kerfoot, W. C., ed. Evolution and ecology of zooplankton communities. Hanover, N.H.: University of New England Press: 635-641.
- Bahls, P. F. 1990. Ecological implications of trout introductions to lakes of the Selway Bitterroot Wilderness, Idaho. Unpublished Masters Thesis. Corvallis: Oregon State University. 85 p.
- Bahls, P. F. 1992. The status of fish populations and management of high mountain lakes in the western United States. Northwest Science. 66: 183-193.
- Blaustein, A. R.; Wake, D. B.; Sousa, W. P. 1994. Amphibian declines: judging stability, persistence, and susceptibility of populations to local and global extinctions. Conservation Biology. 8: 60-71.
- Bradford, D. F. 1989. Allotopic distribution of native frogs and introduced fishes in high Sierra Nevada lakes of California: Implication of the negative effect of fish introductions. Copeia. 1989: 775-778.
- Bradford, D. F.; Tabatabai, F.; Graber, D. M. 1993. Isolation of remaining populations of the native frog, *Rana muscosa*, by introduced fishes in Sequoia and Kings Canyon National Parks, California. Conservation Biology. 7: 882-888.
- Braña, F.; Freschilla, L.; Orizaola, G. 1996. Effect of introduced fish on amphibian assemblages in mountain lakes of northern Spain. Herpetological Journal. 6: 145-148.
- Carlisle, D. M.; Hawkins, C. P. 1998. Relationships between invertebrate assemblage structure, 2 trout species, and habitat structure in Utah mountain lakes. Journal of the North American Benthological Association. 17: 286-300.
- Corn, P. S.; Knapp, R. A. 2000. Fish stocking in protected areas: summary of a workshop. In: Cole, D. N.; McCool, S. F., eds. Proceedings: Wilderness Science in a Time of Change. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Corn, P. S. (in press). Amphibian declines: review of some current hypotheses. In: Sparling, D. W.; Bishop, C. A.; Linder, G., eds. Ecotoxicology of amphibians and reptiles. Pensacola: Society of Environmental Toxicology and Chemistry.
- Duellman, W. E.; Trueb, L. 1986. Biology of Amphibians. Baltimore: The John Hopkins University Press. 670 p.
- Duff, D. A. 1995. Fish stocking in U.S. Federal wilderness areas - challenges and opportunities. International Journal of Wilderness. 1: 17-19.
- Fellers, G. M. 1997. Design of amphibian surveys. In: Olson, D. H., W. P. Leonard, and R. B. Bury, eds. Sampling amphibians in lentic habitats. Olympia, WA: Society for Northwestern Vertebrate Biology: 23-34.
- Fellers, G. M.; Drost, C. A. 1993. Disappearance of the Cascades frog *Rana cascadae* at the southern end of its range, California, U.S.A. Biological Conservation. 65: 177-181.
- Fisher, R. N.; Shaffer, H. B. 1996. The decline of amphibians in California's Great Central Valley. Conservation Biology. 10: 1387-1397.
- Fontenot, L. W.; Noblet, G. P.; Platt, S. G. 1994. Rotenone hazards to amphibians and reptiles. Herpetological Review. 25: 150-156.
- Forstenzer, M. 1998. When trout move in, frogs move out: Fish blamed for amphibians' decline. Seattle: Seattle Times. 6 January.
- Fraley, J. 1996. Cooperation and controversy in wilderness fisheries management. Fisheries. 21: 16-21.

- Funk, W. C.; Dunlap, W. W. (in press). Colonization of high-elevation lakes by long-toed salamanders (*Ambystoma macrodactylum*) after the extinction of introduced trout populations. *Copeia*.
- Gill, C.; Matthews, K. 1998. Frogs or fish? Forestry Research West. August 1998: 1-4.
- Gottschalk, J. S. 1976. Management of wilderness area waters. Washington, D.C.: American Fisheries Society. 23 p.
- Green, D.M. 1997. Perspectives on amphibian population declines: defining the problem and searching for answers. In: Green, D.M., ed. *Amphibians in decline: Canadian studies of a global problem*. Herpetological Conservation. 1: 291-308.
- Green, D. M.; Kaiser, H.; Sharbel, T. F.; Kearsley, J.; McAllister, K. R. 1997. Cryptic species of spotted frogs, *Rana pretiosa* complex, in western North America. *Copeia*. 1997: 1-8.
- Griffith, J. Personal Communication. Conversation regarding tiger salamander larvae found in stomach of brown trout caught ice fishing at Chesterfield Reservoir, Idaho. January 1997.
- Hayes, M. P.; Jennings, M. R. 1986. Decline of ranid frog species in western North America: Are bullfrogs (*Rana catesbeiana*) responsible? *Journal of Herpetology*. 20: 490-509.
- Hecnar, S. J.; McCloskey, R. T. 1997. The effects of predatory fish on amphibian species richness and distribution. *Biological Conservation*. 79: 123-131.
- Hendee, J. C.; Stankey, G. H.; Lucas, R. C. 1990. *Wilderness management*. Golden, CO: North American Press, Fulcrum Publishing.
- Hoffman, R. L.; Pilliod, D. S. 1999. The ecological effects of fish stocking on amphibian populations in high-mountain wilderness lakes. Corvallis: USGS Biological Resources Division, Forest and Rangeland Ecosystem Science Center. 85 p.
- Jones, M. S.; Goettl, J. P.; Livo, L. J. 1999. *Bufo boreas* (Boreal Toad). Predation. *Herpetological Review*. 30: 91.
- Kats, L. B.; Petranks, J. W.; Sih, A. 1988. Antipredator defenses and the persistence of amphibian larvae with fishes. *Ecology*. 69: 1865-1870.
- Knapp, R. A. 1994. The high cost of high Sierra trout. *The Wilderness Record*. 19: 1-3.
- Knapp, R. A. 1996. Non-native trout in natural lakes of the Sierra Nevada: an analysis of their distribution and impacts on native aquatic biota. Sierra Nevada ecosystem project: final report to Congress, Volume III, assessments, commissioned reports, and background information. Davis, CA: Wildland Resource Center Report: 363-407.
- Knapp, R. A.; Matthews, K. R. 1998. Eradication of nonnative fish by gill netting from a small mountain lake in California. *Restoration Ecology*. 6: 207-213.
- Knapp, R. A.; Matthews, K. R. 2000. Non-native fishes in wilderness lakes of the Sierra Nevada: finding a balance between providing recreational fisheries and maintaining natural ecosystems. In: Cole, D. N.; McCool, S. F., eds. *Proceedings: Wilderness Science in a Time of Change*. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station.
- Liss, W. J.; Larson, G. L. 1991. Ecological effects of stocked trout on North Cascades Naturally Fishless Lakes. *Park Science*. 11: 22-23.
- Matthews, K. R.; Knapp, R. A. 1999. A study of high mountain lake fish stocking effects in the U.S. Sierra Nevada wilderness. *International Journal of Wilderness*. 5: 24-26.
- Munger, J. C.; Barnett, B. R.; Ames, A. 1997. 1996 Sawtooth Wilderness amphibian survey. A challenge cost share agreement between the Sawtooth National Forest, USDA Forest Service and Boise State University. Boise: Boise State University. 10 p.
- Murray, M. P. 1994. Reconsidering fish stocking of high wilderness lakes. *Wild Earth*. 4: 50-52.
- Murray, M. P.; Boyd, A. W. 1996. Selway-Bitterroot Wilderness lakes and streams naturalization plan. Proposal. Moscow, ID: Ecocentric Wildlands Management Institute. 59 p.
- National Park Service. 1975. Management policies. Section IV: 7-8: U.S. Department of the Interior, National Park Service.
- Petranks, J. W.; Kats, L. B.; Sih, A. 1987. Predator-prey interactions among fish and larval amphibians: use of chemical cues to detect predatory fish. *Animal Behavior*. 35: 420-425.
- Pilliod, D. S.; Duncan, D.; Peterson, C. R.; Yeo, J. J. 1996. Spatial distribution and habitat associations of amphibians in the Bighorn Crags of the Frank Church River of No Return Wilderness, 1994 Report. INT-94697-RJVA. Boise, ID: USDA Forest Service, Intermountain Research Station. 39 p.
- Pilliod, D. S.; Peterson, C. R. 1997. Alpine lake ecology: Effects of fish stocking on amphibian populations, Bighorn Crags, Frank Church River of No Return Wilderness, 1995 Progress Report. Missoula, MT: USDA Forest Service, Aldo Leopold Wilderness Research Institute. 29 p.
- Pilliod, D. S.; Peterson, C. R. (unpublished data). Unpublished data from a manuscript in prep for research conducted in the Bighorn Crags, Frank Church-River of No Return Wilderness, Idaho.
- Public Law. 88-577. Wilderness Act. 78 Stat 890: U.S. Public Law.
- Rahel, F. J. 1997. From Johnny Appleseed to Dr. Frankenstein: Changing values and the legacy of fisheries management. *Fisheries*. 22: 8-9.
- Ross, D. A.; Stanger, M. C.; McDonald, K.; Shirley, D. L.; White, P. A.; Lentsch, L. D. 1994. Distribution, habitat use, and relative abundance indices of spotted frogs in the west desert, Utah, 1993. 93-15. Salt Lake City: Native Division Section, Utah Division of Wildlife Resources. 32 p.
- Society for Conservation Biology. 1995. Species stocking in the U.S. National Wilderness Preservation System. Resolution. Malden, MA: Society for Conservation Biology. 1 p.
- Schnick, R. A. 1974. A review of the literature on the use of Antimycin in fisheries. La Crosse, WI: USDI Fish and Wildlife Service. 85 p.
- Semlitsch, R. D.; Gibbons, J. W. 1988. Fish predation in size-structured populations of treefrog tadpoles. *Oecologia Berl.* 75: 321-326.
- Sih, A.; Kats, L. B.; Moore, R. D. 1992. Effects of predatory sunfish on the density, drift, and refuge use of stream salamander larvae. *Ecology*. 73: 1418-1430.
- Taylor, J. 1983. Orientation and flight behavior of a neotenic salamander (*Ambystoma gracile*) in Oregon. *American Midland Naturalist*. 109: 40-49.
- Trout Unlimited. 1998. Threats to aquatic systems: fish stocking. Written statement from conference: Effects of Fisheries Management on the Amphibians and Other Biota of Wilderness Lakes; October 14-16, 1998; Flathead Lake Biological Station, MT. 1 p.
- Tyler, T.; Liss, W.; Ganio, L. M.; Larson, G. L.; Hoffman, R.; Deimling, E.; Lomnický, G. 1998a. Interaction between introduced trout and larval salamanders (*Ambystoma macrodactylum*) in high-elevation lakes. *Conservation Biology*. 12: 94-105.
- Tyler, T. J.; Liss, W. J.; Hoffman, R. L.; Ganio, L. M. 1998b. Experimental analysis of trout effects on survival, growth, and habitat use of two species of Ambystomatid salamanders. *Journal of Herpetology*. 32: 345-349.
- US Department of Agriculture. 1986. Policies and guidelines for fish and wildlife management in National Forest and Bureau of Land Management Wilderness. Forest Service Handbook section 2309.19. Wilderness Management Handbook 23.1. Washington, D.C.: Forest Service, Bureau of Land Management, and International Association of Fish and Wildlife Agencies.
- US Department of Agriculture. 1990. FSM 2300: Recreation, wilderness, and related resource management. WO Amendment 2300-90-1. Forest Service Manual. Washington, D.C.: USDA Forest Service.
- Voris, H. K.; Bacon, J. P. Jr. 1966. Differential predation on tadpoles. *Copeia*. 1966: 594-598.
- Wassersug, R. J. 1997. Assessing and controlling amphibian populations from the larval perspective. In: Green, D. M., ed. *Amphibians in decline: Canadian studies of a global problem*. Herpetological Conservation. 1: 271-281.
- Yuskavitch, J. 1999. High lakes drifters. *Inner Voice*. March/April: 18-21.
- Zaret, T. M. 1980. *Predation and freshwater communities*. New Haven, CT: Yale University Press. 187 p.

Air Quality Management in U.S. Fish and Wildlife Service Wilderness Areas

Ellen M. Porter

Abstract—Proper management of air resources is vital to maintaining the wilderness character of an area. Air pollution can affect natural resources and has caused injury to vegetation, bioaccumulation of mercury in fish, eutrophication of coastal ecosystems and visibility impairment in U.S. Fish and Wildlife Service (FWS) wilderness areas. Sources of air pollution include power plants, incinerators, industry, automobiles, dust and fires. Emissions from these sources can be transported long distances and affect areas otherwise considered to be pristine. The FWS uses a combination of monitoring, special studies, participation in the regulatory process and review of new sources of air pollution in its air quality management strategy.

The U.S. Fish and Wildlife Service (FWS) manages 20.7 million acres in 76 wilderness areas. These wilderness areas range in size from the eight-million acre Mollie Beattie Wilderness Area in Alaska to the two-acre Wisconsin Islands Wilderness Area in Wisconsin. Twenty-one of the wilderness areas managed by FWS in the National Wildlife Refuge System are designated Class I air quality areas (fig. 1) and receive special protection under the Clean Air Act (Public Law No. 101-549). Only a very small additional amount of air pollution (from 1977 levels) can be permitted in Class I areas.

Class I areas include the following federal lands that were in existence on August 7, 1977: national parks exceeding 6,000 acres; national wilderness areas exceeding 5,000 acres; national memorial parks exceeding 5,000 acres; and international parks. In addition, tribes have designated certain tribal lands as Class I. Congress gave the FWS and the other federal land managers for Class I areas an "affirmative responsibility to protect all those air quality related values (including visibility) of such lands..." (Senate Report No. 95-127, 95th Congress, 1st Session, 1977). Air quality-related values include vegetation, wildlife, water, soils, visibility and geological, archeological, historical and cultural resources. Despite this special protection, many of the resources in these areas are being impacted or have the potential to be impacted by air pollutants.

Common air pollutants of ecological significance include sulfur and nitrogen oxides, ammonia, ozone, particulate matter, volatile organic compounds and metals. These pollutants are either emitted directly from sources, including

power plants, incinerators, industries, automobiles and fires or, as in the case with ozone, are formed downwind of sources as emissions react and transform. Other downwind reactions produce fine aerosols and particles, including sulfates and nitrates, which may eventually be deposited into ecosystems.

Impacts to wilderness resources from air pollution include acidification of lakes, streams and soil; eutrophication of estuaries and near-shore coastal waters; direct toxicity to sensitive species; changes in species composition; changes in nutrient cycling; bioaccumulation of toxins in food chains; and visibility impairment.

Acidification may occur when sulfur and nitrogen compounds combine with moisture and transform to acids in the atmosphere, soil or water. Acids may be buffered by naturally occurring base cations, such as calcium and magnesium. However, in lakes, streams and soils with low amounts of base cations (or with high acid inputs), acid-neutralizing capacity is lost, and acidification occurs (National Acid Precipitation Assessment Program 1990a). Acid-sensitive species of fish and invertebrates, and the wildlife that depend on them, may be lost from the ecosystem. (Griffith and others 1995). In addition, increased acidity mobilizes metals, such as aluminum and mercury, that are toxic to plants and wildlife (National Acid Precipitation Assessment Program 1990b). In high-elevation spruce-fir forests increased acidity has resulted in winter foliar injury and subsequent dieback (Thornton and others 1994). Acidity may also cause changes in soil nutrient cycling (Aber and others 1995; Johnson and Lindberg 1992).

In addition to having an acidification effect nitrogen from air pollution may have a fertilizing effect on ecosystems (Vitousek and others 1997; Paerl 1993). Nitrogen can be deposited into ecosystems in the form of nitrates, ammonium ions and other compounds. In natural systems, including designated wilderness areas, nitrogen may cause an unwanted increase in primary production and a shift in species composition to nitrogen-loving species. In estuaries and coastal waters along the Atlantic and Gulf coasts, for example, excess nitrogen stimulates eutrophication characterized by algae blooms, decreased water clarity, deterioration and loss of sea grasses, and hypoxia (Ecological Society of America 1997). In some areas, this has resulted in the loss of important invertebrate, fish and wildlife species. Although much of the nitrogen entering estuaries is from terrestrial runoff, a significant portion comes from the atmosphere. In estuary studies to date, atmospheric nitrogen comprises from 10%-50% of the total nitrogen entering the system (Paerl 1995).

Certain air pollutants have a direct toxic effect on sensitive species. Ozone is the most important of the phytotoxic pollutants and enters the stomates of plants along with the normal constituents of air. Once inside the leaf, ozone (or

In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management; 1999 May 23–27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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Figure 1—U.S. Fish and Wildlife Service Class I areas.

its byproducts) reacts with cell membranes and other cell components, causing injury or death of leaf tissues. On broad-leaved plants, ozone injury may appear as dark stipples. On conifers, ozone injury may appear as chlorotic mottle (Chappelka and Chevone 1992). In addition, ozone may cause reductions in plant growth and reproduction (Manning and Krupa 1991).

Other air pollutants, including mercury and other toxic metals, bioaccumulate when deposited into ecosystems. Mercury, for example, can accumulate up the food chain by a factor of a million or more (Schroeder and Munthe 1998). Mercury tends to accumulate in aquatic food chains, reaching toxic levels in certain fish species (Facemire and others 1995). Wildlife and humans consuming such fish may be at risk of neurological and reproductive damage (U.S. Environmental Protection Agency 1997).

In addition to effects on plants, wildlife, water and soils, air pollutants reduce visibility. Fine particles of sulfates, nitrates, organics, soot and other compounds absorb or scatter light, reducing our ability to see wildland features clearly. Pollutant haze has become a common feature of the landscape (National Research Council 1993).

Air Quality Management Strategy

To better understand the effects of air pollutants on FWS lands, and to ensure protection of air quality and air quality-related values, the FWS has developed an air quality management strategy. This strategy includes monitoring, special studies, participation in the regulatory development process and review and evaluation of new sources of air pollution near FWS areas.

Air Pollutant Monitoring

The FWS conducts air quality monitoring in partnership with several national programs, including the National Atmospheric Deposition Program, the Mercury Deposition Network Program and the Interagency Monitoring of Protected Visual Environments Program. Because air pollutants tend to be well-dispersed in the atmosphere (in the absence of strong local sources), monitoring to characterize wilderness air quality is conducted in an adjacent nonwilderness area. Thus, impacts to the wilderness from monitoring activities are avoided.

National Atmospheric Deposition Program (NADP)—The NADP provides long-term spatial and temporal trend information on the concentration and deposition of major cations and anions (both natural and human-caused) in precipitation at over 200 sites nationwide. NADP, now in its third decade of collecting precipitation chemistry data, is a cooperative effort supported by national, state and local governmental agencies, State Agricultural Experiment Stations, universities and private organizations. Rain or snow is collected on a weekly basis and analyzed at a central laboratory for sulfate, nitrate, ammonium, calcium, magnesium, potassium, sodium, chloride, phosphate and hydrogen ions, as well as conductivity. Rainfall is also measured at the sampling sites, allowing deposition rates to be estimated. FWS supports NADP samplers at three Class I areas: Brigantine (part of the Edwin B. Forsythe National Wildlife Refuge-NWR in New Jersey), Okefenokee NWR (Georgia), and Chassahowitzka NWR (Florida). The U.S. Geological Survey funds NADP samplers at five other national wildlife refuges including Salt Plains NWR (Oklahoma), Santee NWR (South Carolina), Hatchie NWR (Tennessee), Muleshoe NWR (Texas), and Attwater Prairie Chicken NWR (Texas).

NADP information and data are available at the NADP website: <http://nadp.sws.uiuc.edu>

Data from NADP indicate that the monitored FWS areas are experiencing elevated levels of air pollutants in deposition, as are many wilderness areas in the contiguous United States.

Mercury Deposition Network (MDN) Program—The MDN, a network of NADP, provides long-term spatial and temporal trend information on the concentration and deposition of total mercury in precipitation at nearly 40 sites nationwide. Samples are collected weekly, using trace metal protocols, and analyzed at a central laboratory. Methylmercury can also be analyzed. FWS supports two MDN sites: Okefenokee (Georgia) and Chassahowitzka (Florida).

MDN information and data are available at the NADP website: <http://nadp.sws.uiuc.edu/mdn/>

Elevated levels of mercury have been recorded in the rainfall at the FWS sites. Fish sampled from Okefenokee and Chassahowitzka also contain elevated mercury levels (Facemire and others 1995; Brim and others 1994).

Interagency Monitoring of Protected Visual Environments (IMPROVE)—In 1977, Congress established as a goal, "...the prevention of any future, and the remedying of any existing, impairment of visibility in mandatory class I Federal areas which impairment results from manmade air pollution." (Public Law No. 101-549). In its 1993 report, "Protecting Visibility in National Parks and Wilderness Areas," the National Research Council concluded that visual range (a measure of visibility) in the western U.S. is one-half to two-thirds of the natural visual range (that is, without manmade air pollution). In the eastern U.S., the visual range is, on the average, only one-fifth of the natural visual range (National Research Council 1993). Visibility impairment occurs when fine particles and aerosols scatter or absorb light, that is, cause "light extinction." Light extinction is inversely proportional to visual range and is, therefore, much greater in the East than in the West (fig. 2).

In response to the goal set by Congress, federal land managers, together with the Environmental Protection Agency and regional and state organizations, developed the IMPROVE program. IMPROVE monitors visibility conditions at approximately 40 sites nationwide, primarily Class I areas. More sites (approximately 80) will be added in 1999-2000. An IMPROVE site includes a fine-particle sampler that measures the composition and concentration of fine particles in the air that reduce visibility. A site may also include an automatic camera to characterize haze and an optical instrument (such as a transmissometer or nephelometer) to measure light extinction or scattering.

The Fish and Wildlife Service supports IMPROVE fine-particle samplers at five Class I areas: Brigantine (New Jersey), Cape Romain (South Carolina), Chassahowitzka (Florida), Moosehorn (Maine), and Okefenokee (Georgia). Data indicate that visibility at these sites is impaired much of the time. Sulfate particles (primarily from coal-burning power plants) cause most of the light extinction at these sites, which is typical of Eastern IMPROVE sites (National Acid Precipitation Assessment Program 1990d; Colorado State University 1996).

Evaluation of Air Pollution Effects to Resources

In addition to monitoring the types and amounts of pollutants in the air and in deposition, the FWS conducts special studies to evaluate the effects of pollution on air quality-related values. These studies have focused on vegetation and water quality.

Surveys of Vegetation for Air Pollution Injury—Surveys have been conducted at a number of FWS Class I areas to date, to evaluate vegetation for symptoms of ozone injury. Ozone produces distinctive stippling and chlorosis on sensitive species that has been well characterized by controlled fumigations in open-top chambers. Observations by trained observers of similar symptoms in the field can be used to verify ozone injury. Ozone injury has been documented at most of the FWS Class I areas surveyed, including Brigantine, Cape Romain, Moosehorn and Mingo (Davis 1996; Davis 1998; Davis 1999a; Davis 1999b). Species affected include black cherry (*Prunus serotina*), wild grape (*Vitis* spp.), common milkweed (*Asclepias syriaca*), tree-of-heaven (*Ailanthus altissima*), ash (*Fraxinus* spp.), cucumbertree (*Magnolia acuminata*), flowering dogwood (*Cornus florida*), sassafras (*Sassafras albidum*), sweetgum (*Liquidambar styraciflua*), spreading dogbane (*Apocynum androsaemifolium*), trembling aspen (*Populus tremuloides*), pin cherry (*Prunus pennsylvanica*), serviceberry (*Aamelanchier laevis*), elderberry (*Sambucus canadensis*), and winged sumac (*Rhus copallina*), and salt-marsh cordgrass (*Spartina alterniflora*).

Acidification Vulnerability Study—Water chemistry of lakes in Moosehorn NWR and Wilderness was evaluated to determine the lakes' sensitivities to acidic deposition. Results indicated that although the lakes are sufficiently buffered to tolerate current loads of sulfates and nitrates, increases in loadings of these pollutants could

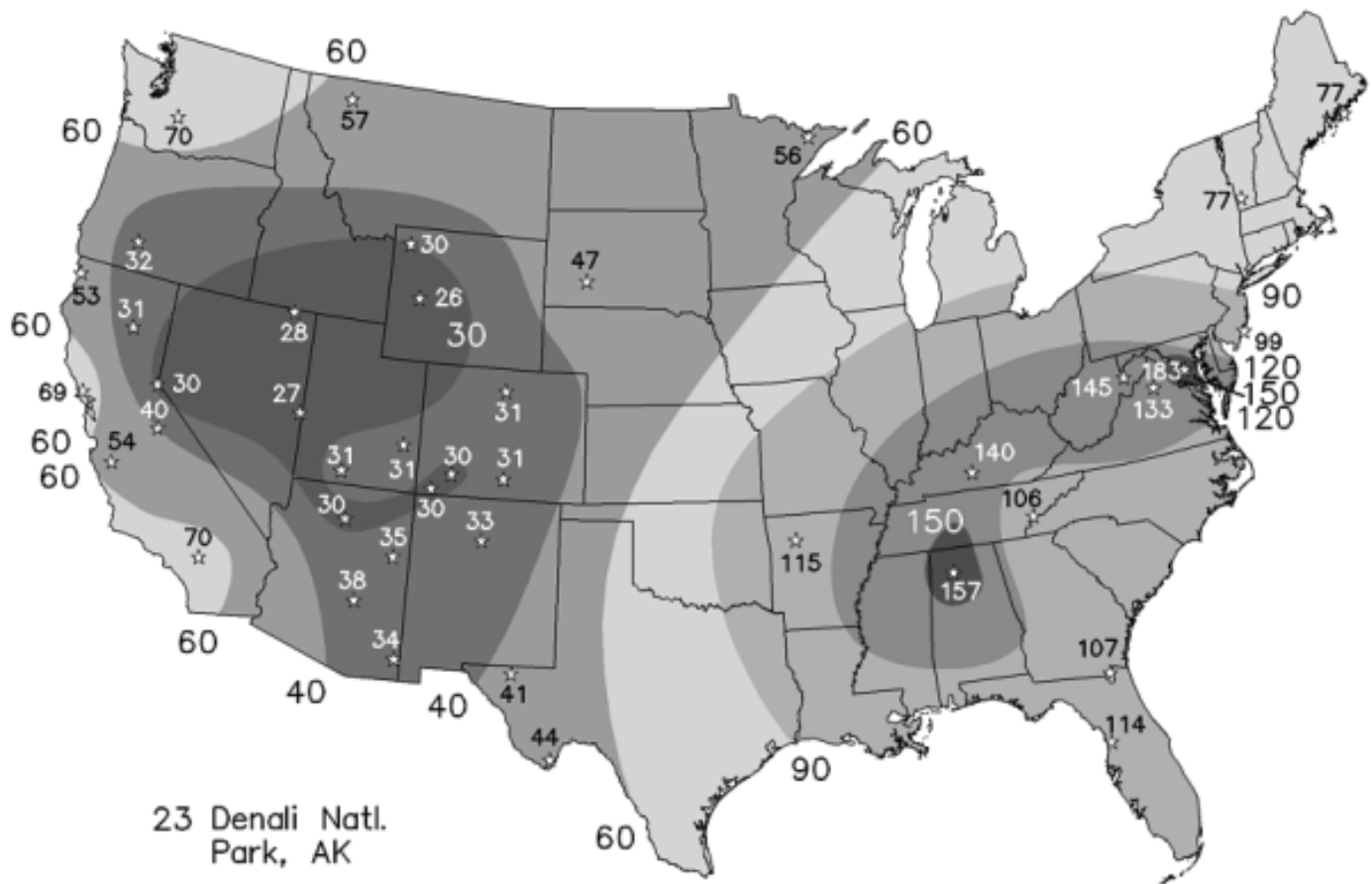


Figure 2—Light extinction coefficient, calculated from aerosol concentrations measured by IMPROVE (National Park Service Air Resources Division).

reduce buffering capacity and increase the risk of acidification (Kahl and James 1996).

Eutrophication Vulnerability Study—Water chemistry, phytoplankton and sea grasses have been examined from 1996 to the present at Chassahowitzka to evaluate nutrient and trophic status, phytoplankton species composition and density, and sea grass health and distribution. Table 1 summarizes water quality parameters for Chassahowitzka. In the first two years of the study, 1996-1997, water quality and trophic state were considered good. Water

clarity was high, and chlorophyll and nutrient concentrations were low. However, in 1998, water quality and trophic state were considered poor. Significant algae blooms were noted, with loss of water clarity and low dissolved oxygen concentrations. Nitrogen was found to be the limiting nutrient in this system (Dixon 1998). Further work will be conducted in 1999 to evaluate whether the poor water quality observed in 1998 affected sea grass health.

The studies described above were limited to a small number of FWS Class I areas. However, it is likely that other FWS areas are experiencing similar effects. For example, ozone injury to vegetation probably occurs in many FWS areas, particularly in the eastern U.S. and certain areas in the West (California), because of the high ozone concentrations that are typical of these areas. Atmospheric nitrogen is probably contributing to eutrophication at many FWS areas along the Gulf and Atlantic coasts. And, visibility impairment affects all FWS areas in the contiguous United States.

Table 1—Average water quality parameters for coastal stations in Chassahowitzka National Wildlife Refuge (Dixon 1998).

Date	Chlorophyll a micrograms/liter	IN:IP ^a	TSI ^b
May 1996	3	3	31
May 1997	2	5	30
September 1997	2	11	35
May 1998	18	4	60
July 1998	18	3	57

^aInorganic nitrogen:inorganic phosphorous (IN:IP) ratio below 10 indicates that phytoplankton growth is nitrogen-limited.

^bTrophic State Index (TSI) value below 50 indicates good water quality. TSI above 50 indicates poor water quality.

Participation in the Regulatory Development Process

The FWS routinely reviews and comments on legislation and rule-making regarding air quality, including recent regulations pertaining to ozone, particulate matter, regional haze and new source review. In addition, the agency

participates in regional air quality partnerships including the Grand Canyon Visibility Transport Commission, the Western Regional Air Partnership, and others. The FWS works with states to develop their State Implementation Plans for the Clean Air Act. Participation in these forums ensures that FWS concerns for air quality and air quality-related values under its jurisdiction are addressed.

Review and Evaluation of New Sources of Air Pollution

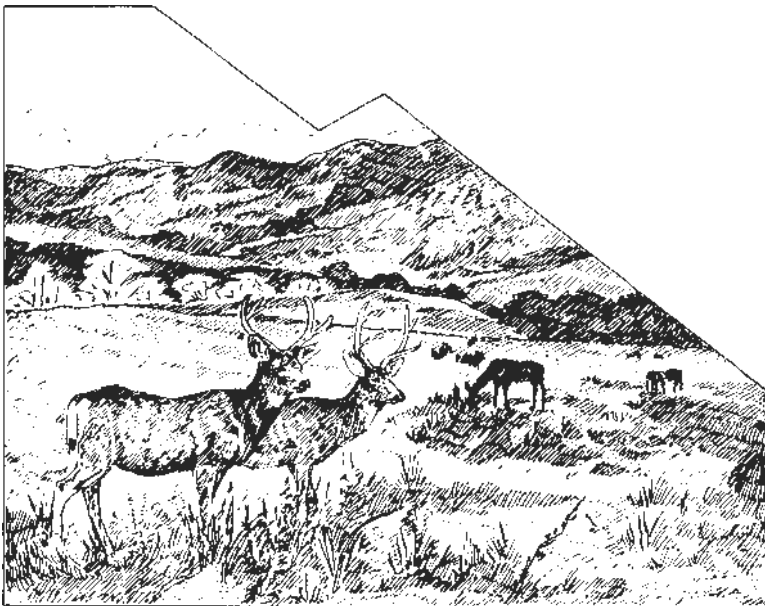
The Environmental Protection Agency or state permitting authority is required to notify the FWS (or appropriate federal land manager) of any permit application from a major source of emissions that may affect a Class I area. The facility must (1) use best available emissions control technology, (2) demonstrate that emissions will not cause or contribute to violations of the National Ambient Air Quality Standards or Class I increments (the maximum allowable increase in a pollutant, designated by the Clean Air Act), and (3) demonstrate that emissions will not cause or contribute to adverse impacts to air quality-related values in the Class I area. The FWS, together with the National Park Service, has developed guidance for Class I area analyses that is available to air pollution permit applicants (Bunyak 1993). The FWS is given the opportunity to review and comment on the proposed source's control technology, air quality impacts and air quality-related values impacts. The FWS may recommend better control technology, lower emission rates or lower production rates to mitigate potential impacts to the Class I area. In addition, the agency may ask for additional analyses to provide adequate information to evaluate potential impacts. If its concerns are not addressed, and the FWS determines that there is potential for adverse impacts to a Class I area from the proposed source's emissions, the FWS may appeal the permit.

In summary, the air quality management strategy of the FWS is designed to increase understanding and ensure protection of air quality and air quality-related values on FWS lands. Continued progress in understanding air pollution and its effects will enable the FWS Air Quality Management Program to more effectively protect its wilderness resources.

References

- Aber, J. D.; Magill, A.; McNulty, S. G.; Boone, R. D.; Nadelhoffer, K. J.; Downs, M.; Hallett, R. 1995. Forest biogeochemistry and primary production altered by nitrogen saturation. *Water, Air and Soil Pollution*. 85: 1665-1670.
- Brim, M.; Bateman, D.; Jarvis, R. 1994. Mercury concentrations in fishes of the Chassahowitzka National Wildlife Refuge. U.S. Fish and Wildlife Service Publication No. PCFO-EC 94-01. Panama City, FL.
- Bunyak, John. 1993. Permit application guidance for new air pollution sources. National Park Service Natural Resources Report 93/09. Denver, CO.
- Chappelka, Arthur H.; Chevone, Boris I. 1992. Tree responses to ozone. In: Lefohn, Allen S., ed. *Surface level ozone exposures and their effects on vegetation*. Chelsea, MI: Lewis Publishers: 271-324.
- Colorado State University. Cooperative Institute for Research in the Atmosphere. 1996. IMPROVE, Interagency Monitoring of Protected Visual Environments. Spatial and seasonal patterns and long term variability of the composition of the haze in the United States: an analysis of data from the IMPROVE network.
- Davis, Donald D. 1996. Evaluation of ambient ozone injury on the foliage of vegetation in the Edwin B. Forsythe National Wildlife Refuge, Brigantine, NJ: 1996 Observations. Pennsylvania State University.
- Davis, Donald D. 1998. Evaluation of ambient ozone injury on the foliage of vegetation in the Cape Romain National Wildlife Refuge, South Carolina: 1997 Observations. Pennsylvania State University.
- Davis, Donald D. 1999a. Evaluation of ozone injury on vegetation in the Moosehorn National Wildlife Refuge, Maine: 1998 Observations. Pennsylvania State University.
- Davis, Donald D. 1999b. Evaluation of ozone injury on vegetation in the Mingo National Wildlife Refuge, Missouri: 1998 Observations. Pennsylvania State University.
- Dixon, L. Kellie. 1998. Chassahowitzka National Wildlife Refuge Status and Trends. Mote Marine Laboratory Technical Report Number 579. Sarasota, Florida.
- Facemire, C.; Augspurger, T.; Bateman, D.; Brim, M.; Conzelmann, P.; Delchamps, S.; Douglas, E.; Immon, L.; Looney, K.; Lopez, F.; Masson, G.; Morrison, D.; Morse, N.; Robison, A. 1995. Impacts of mercury contamination in the southeastern United States. *Water, Air, and Soil Pollution*. 80: 923-926.
- Griffith, Michael B.; Perry, Sue A.; Perry, William B. 1995. Macroinvertebrate communities in headwater streams affected by acidic precipitation in the Central Appalachians. *Journal of Environmental Quality*. 24: 233-238.
- Johnson, D.; Lindberg, S. (eds). 1992. *Atmospheric Deposition and Forest Nutrient Cycling*. Springer-Verlag, New York. 707 p.
- Kahl, Steve; James, Cary. 1996. *Lake Chemistry in Moosehorn National Wildlife Refuge, Maine, 1995*. Water Research Institute, University of Maine, Orono.
- Manning, William J.; Krupa, Sagar V. 1992. Experimental methodology for studying the effects of ozone on crops and trees. In: Lefohn, Allen S., ed. *Surface level ozone exposures and their effects on vegetation*. Chelsea, MI: Lewis Publishers: 93-156.
- National Acid Precipitation Assessment Program. 1990a. *Acidic Deposition: State of Science and Technology, Report 10, Watershed and lake processes affecting surface water acid-base chemistry*. Washington, D.C.
- National Acid Precipitation Assessment Program. 1990b. *Acidic Deposition: State of Science and Technology, Report 13, Biological effects of changes in surface water acid-base chemistry*. Washington, D.C.
- National Acid Precipitation Assessment Program. 1990c. *Acidic Deposition: State of Science and Technology, Report 16, Changes in forest health and productivity in the United States and Canada*. Washington, D.C.
- National Acid Precipitation Assessment Program. 1990d. *Acidic Deposition: State of Science and Technology, Report 24, Visibility: Existing and Historical Conditions—Causes and Effects*. Washington, D.C.
- National Research Council. 1993. *Protecting visibility in national parks and wilderness areas*. Washington, D.C.: National Academy Press. 466 p.
- Paerl, Hans W. 1993. Emerging role of atmospheric nitrogen deposition in coastal eutrophication: biogeochemical and trophic perspectives. *Canadian Journal of Fisheries and Aquatic Science*. 50: 2254-2269.
- Paerl, Hans W. 1995. Coastal eutrophication in relation to atmospheric nitrogen deposition: current perspectives. *Ophelia*. 41: 237-259.
- Schroeder, William H.; Munthe, John. 1998. Atmospheric mercury—an overview. *Atmospheric Environment* 32(5): 809-822.
- Thornton, F. C.; Joslin, J. D.; Pier, P. A.; Neufeld, H.; Seiler, J. R.; Hutcherson, J. D. 1994. Cloudwater and ozone effects upon high elevation red spruce: a summary of study results from Whitetop Mountain, Virginia. *Journal of Environmental Quality*. 23(6): 1158-1167.
- U.S. Environmental Protection Agency. 1997. *Mercury Study Report to Congress, Vol. VII: Characterization of human health and wildlife risks from mercury exposure in the United States*. EPA-452/R-97-009.
- Vitousek, Peter M.; Aber, John D.; Howarth, Robert W.; Likens, Gene E.; Matson, Pamela A.; Schindler, David W.; Schlesinger, William H.; Tilman, David G. 1997. Human alteration of the global nitrogen cycle: sources and consequences. *Ecological Applications*. 7(3): 737-750.

6. Wilderness Management



Legislative Interpretation as a Guiding Tool for Wilderness Management

Shannon S. Meyer

Abstract—The Wilderness Act of 1964, which established the National Wilderness Preservation System, contains both a clear definition of wilderness and multiple “nonconforming” exceptions to this definition. Managers are given discretion to manage these nonconforming uses but must do so within the framework of wilderness the Act sought to preserve. This paper presents a process for assessing congressional intent by closely examining both the legislative language and the legislative history. This process is based on the works of legal scholars, case law and judicial practice. The paper then demonstrates the application of this process to the management of airstrips and jetboat use in the Frank Church - River of No Return Wilderness.

Human ecologists recognize that all environmental decisions involve three groups of humans: scientists, constituents and governments. While wilderness management should be based on scientific principles, it must also operate within the constraints posed by the governmental framework; this, in turn, is constantly influenced by the citizenry. In the United States, this governmental framework involves laws, regulations and management practices. When governmental directives conflict or are ambiguous, it is even more difficult for managers to make decisions based on sound science.

One frequently overlooked tool for resolving such conflicts is legislative interpretation. Legislative interpretation involves the careful examination of relevant laws and their legislative histories. Legislative interpretation can be confusing to the uninitiated and, as a result, may be misused. All legislative history is not created equal, nor is it always an appropriate recourse. Therefore, it is important to clearly understand the implications, merits and limitations of legislative history before applying it to management questions.

I will present a process for using legislative history to understand difficult management questions. I have adapted this framework from the conventions of the courts in order to make it easily accessible to managers, decision-makers, scientists and citizens. It provides a road map to lead interpreters through the confusing maze of legislative

documents that comprise legislative history. To illustrate its efficacy, I demonstrate its application to two specific management issues: managing airstrips on the Frank Church River of No Return Wilderness and jet boat use of the Salmon River.

Background

In 1964, the Wilderness Act established the National Wilderness Preservation System (NWPS) (P.L. 88-577). The system was initially composed of only 54 Forest Service areas, with a total of 9.1 million acres. Today, the NWPS contains 625 areas and more than 104 million acres managed by four federal agencies. Every wilderness area is governed not only by the Wilderness Act, but by the legislation that established it. These subsequent wilderness laws have all incorporated the management provisions of the Wilderness Act by reference. Some also add special provisions relevant only to the areas designated within. Therefore, legislative interpretation must analyze the Wilderness Act and any other legislation relevant to a particular wilderness.

The Wilderness Act defines wilderness as an “area where the earth and its community of life are untrammelled by man, where man himself is a visitor who does not remain” (sec. 2(c)). To protect this wilderness character, the Act prohibits roads, structures and installations, commercial enterprises and the use of mechanized transport and motorized equipment. But the Act also includes exceptions to these prohibitions. There are exceptions for air access, motorboat use and grazing where these uses were established prior to wilderness designation (secs. 4(d)(1) and 4(d)(4)(2)). The Act also contains a time-limited exception for mining activities, and allowances for water developments and commercial services under certain circumstances. These uses are called ‘nonconforming wilderness uses.’ While many who partake in these uses object to this term, it has been used throughout the history of wilderness legislation. During discussion of the bill that would eventually become the Wilderness Act, Representative John Saylor, one of the original sponsors, defined “nonconforming uses” as “certain existing intrusions that literally or by nature do not conform to the first two sentences of the definition [but that] can be tolerated for practical purposes, and indeed are so tolerated in establishing the system” (U.S. Congress 1962, 20268). This term is used repeatedly throughout the legislative history of the Wilderness Act. Some of the exceptions for such nonconforming uses are found in mandatory clauses, while others are subject to the managing agency’s discretion. It is when managers exercise this discretion that most controversies arise - and where this process can be most useful.

In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O’Loughlin, Jennifer, comps. 2000. Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management; 1999 May 23–27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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The Analytical Process

When a manager is faced with an ambiguous or discretionary situation, the following analytical process can be very helpful. The primary steps in the process are: 1) use statutory construction to determine whether ambiguity exists and attempt to resolve it, 2) if the ambiguity still persists, use legislative interpretation to clarify congressional intent.

Step 1—Statutory Construction

The reviewer must initially determine whether a law is truly ambiguous as it affects the situation at hand. The courts often apply the plain meaning doctrine to determine whether a law is truly ambiguous or not. This doctrine holds that when statutory language is clear and unambiguous, it represents the final meaning of the statute. A clear statement of this doctrine is found in *United States v. Missouri Pacific Railroad* (1929). In this decision the court wrote that;

where the language of an enactment is clear and construction according to its terms does not lead to absurd or impractical consequences, the words employed are to be taken as the final expression of the meaning intended.

Only when such a strict analysis of the law would yield “absurd” results, or the words are unclear, can other interpretive methods be used.

The U.S. Supreme Court retreated from this earlier interpretation in *United States v. American Trucking Association* (1940). In what has become a new standard for the plain meaning rule the court noted that “when aid to construction of the meaning of words, as used in the statute, is available, there certainly can be no ‘rule of law’ which forbids its use, however clear the words may appear on ‘superficial examination.’” Thus, if legislative materials can help, this court argued, they should not be ignored.

Another method for deciphering statutory language is the canons of linguistic construction. They are not codified in law, but these canons stem from decades of case law. They are general linguistic truisms such as: general words should be considered more broadly and specific words more narrowly; when general words follow the designation of particular things, they should be construed to include only those things specifically enumerated; associated words may be used to understand an ambiguous word or phrase; and the mention of one thing implies the exclusion of another; to name a few (Crawford 1940). While it is unlikely that managers will have a list of these canons on their office walls, they remind the interpreter to read laws systematically and logically.

Step 2—Legislative Interpretation

If statutory construction fails to eliminate the confusion, Stephen Breyer, a current Supreme Court Justice, lists five circumstances where the use of legislative history is appropriate: 1) to avoid an absurd result, 2) to discover and correct drafting errors, 3) to determine whether a special meaning exists for a word within a statute, 4) to determine the purpose of a word in the statutory scheme, and 5) to help choose between reasonable alternative interpretations of a

politically controversial statute (Breyer 1992). If any of these five circumstances apply, Breyer suggests turning to legislative interpretation.

The final step involves analyzing the applicable legislative history. Legislative history is the “explanations of the legislators themselves, or the documents officially used by them, in the course of making a specific law” (Folsom 1972). It includes committee reports, congressional debates and committee hearings. These documents provide the “authoritative explanations of the purposes or meaning of the language of the resulting law” (Folsom 1972). Legislative history cannot be used to change the general meaning of the statute, but it can be used to resolve controversies over interpretation or to determine the intended scope of statutory provisions.

All legislative history is not created equal, and the weight given to different aspects of legislative history varies. Figure 1 illustrates the relative importance of these documents. This hierarchy was created from a variety of sources, including scholarly writings and the standard legal guide to statutory construction, and reflect common usage by the courts (de Sloovere 1940, Dickerson 1975, Folsom 1972, Singer 1992). When attempting to interpret legislative history, these documents must be analyzed in order of importance.

Committee reports are generally given the most weight (McDonald 1991). On the next level are the explanations made by the committee chairperson when reporting a bill out of committee. In the process of explaining a bill to the full legislature, a committee chair must answer specific questions about it and defend it against opposition and therefore must be familiar with both the bill and the situation in need of remedy (Singer 1992).

Statements made by the legislative sponsors of a bill to the whole chamber are next in importance. They reveal “a legislative intent more significant than that revealed by those of a more casual legislative adherent” (Dickerson 1975). In contrast, the views of opponents are rarely assigned much importance, as their statements “may tend to overstate the reach of the provision opposed” (Folsom 1972).

Committee hearings are given less weight because they are generally “concerned with the more diffuse matters of ulterior legislative purpose” (Dickerson 1975). However, issues may be discussed in hearings that may not be

- I. Committee Reports
- II a. Statements of sponsors to the whole chamber
b. Explanations of the Committee Chair
- III a. Committee hearings
b. Statements in general debate
- IV a. Statements of members of the opposition
b. Amendments or language rejected in committee or on the floor

Figure 1—Significance of legislative documents in descending order of importance.

revisited in other documents. Amendments or previous bill language that were discarded also play a role. The elimination of words or phrases from a draft bill indicates that the meaning in question was not intended or was no longer acceptable to the majority. Finally, testimony given by non-congressional parties during committee hearings have little value other than to provide context (Singer 1992).

Applying the Process to Wilderness Management Issues

Airstrip Management in Idaho

The first example that I apply this process to is airstrip management on the Frank Church River of No Return Wilderness (FC-RONRW) in Idaho. This is an interesting example because it involves two separate wilderness laws. The Central Idaho Wilderness Act (CIWA) of 1980 established the 2.3 million acre FC-RONRW (P.L. 96-312). This remote area had a long tradition of access by airplane, and some users wished to ensure that this means of access would continue after wilderness designation (U.S. Senate 1979). As a result, the CIWA deviated from the standard language of the Wilderness Act's section 4(d)(1) to state that certain established uses "shall" rather than "may" be permitted to continue subject to the Secretary's regulations. It also added that:

the Secretary shall not permanently close or render unserviceable any aircraft landing strip in regular use on national forest lands on the date of enactment of the Act for reasons other than extreme danger to aircraft, and in any case not without the express written concurrence of the agency of the State of Idaho charged with evaluating the safety of backcountry airstrips (sec. 7(a)(1)).

Compared with the language of the Wilderness Act, this provision significantly limits the agency's discretion to close airstrips in the FC-RONRW.

There is currently an effort to create a comprehensive wilderness management plan. The Draft Environmental Impact Statement (DEIS) for the plan analyzes different strategies for managing these airstrips that include instituting commercial use permits, seasonal closures for wildlife and resource protection reasons, and limiting maintenance on certain airstrips (USDA Forest Service 1998). It is up to managers to decipher what management discretion remains under section 7(a)(1) and how it should be exercised.

Statutory Construction—To apply the analytical process to this question, the interpreter must first determine whether ambiguity exists. To do so, the text of both the Wilderness Act and the CIWA must be analyzed. The initial ambiguity regarding airstrip management stems from the Wilderness Act's exception in section 4(d)(1), permitting a use that is incompatible with the definition of wilderness found in section 2. The CIWA adds to this ambiguity by increasing statutory protection for airstrips, without resolving the underlying conflict between managing for air access and for wilderness protection.

With section 7(a)(1) of the CIWA, Congress clearly limits the agency's ability to close airstrips on the FC-RONRW. In doing so, Congress demonstrated that it could reduce the

agency's management discretion if it desired. Remember that one linguistic canon states that "the mention of one thing implies the exclusion of another" (Singer 1992, 334). Congress did not specifically limit the agency's discretionary ability to restrict use levels. By expressly restricting closures but not restricting other managerial discretion, Congress indicates that only the ability to close airstrips is limited.

Legislative Interpretation—Legislative interpretation is still necessary to address the question of management discretion. Breyers' fifth scenario applies in this case. Both the Wilderness Act and the Central Idaho Wilderness Act are politically controversial statutes, and varying reasonable interpretations can be made from both about how airstrips should be managed.

While aircraft landings are permitted in wilderness areas where they occurred before designation, the Wilderness Act defines wilderness in terms that do not include motorized travel. Section 2(c) of the Act defines a wilderness as an area "where the imprint of man's work [is] substantially unnoticeable" that has "outstanding opportunities for solitude or a primitive and unconfined type of recreation." These definitions are clarified by the legislative history of the Wilderness Act. Senator Hubert Humphrey, the legislation's original sponsor, defined wilderness as "the native condition of the area, undeveloped, . . . untouched by the hand of man or his mechanical products" (U.S. Congress 1957, 19). He saw wilderness as a place "for people to make their way into . . . without all of the so-called advances of modernization and technology" (U.S. Congress 1957, 20). None of the statements defining wilderness in the final law or in its legislative history include motorized uses. While airstrips did not conform with the ideal qualities of wilderness, the proponents' political strategy was to protect the status quo (Mercure and Ross 1969).

Although allowed to continue, airstrips are subject to regulation at the discretion of the managing agency. Congress abdicated its right to statutorily terminate the use of wilderness airstrips in the 1964 Act, but it also explicitly gave the Forest Service discretion to regulate aircraft access as the agency "deems desirable" (P.L. 88-577, sec. 4(d)(1)).

At the time of the bill's passage, only a few of the area's airstrips were actively maintained and some had been closed due to their dangerous conditions. Senator Church emphasized that with this provision, "the Forest Service is expressly prohibited from closing airstrips on national forests within the wilderness, which are in regular use at present, except for the reason of aircraft safety" (U.S. Congress, Senate 1980, S17780). The CIWA clearly restricts the Forest Service's ability to close airstrips on the FC-RONRW, except in the case of extreme danger to aircraft. However, it does not reduce the Forest Service's discretion to manage use levels on, and maintenance of, these strips. The legislative history of the CIWA supports the conclusion that closure, not management discretion, was the evil being remedied by this provision. The bill's sponsor, Senator Church, wanted to prevent the Forest Service from arbitrarily closing airstrips. There is no indication in the statute's legislative history that Congress intended to reduce the agency's discretionary ability to manage use-levels pursuant to agency regulations and policies.

Acceptable Levels of Jet Boat Use on the Salmon River

In addition to establishing the FC-RONRW, the CIWA designated 125 miles of the Salmon River as part of the Wild and Scenic Rivers System (P.L. 96-312). The CIWA contains a stipulation concerning continued jet boat use of the river. It states that:

The use of motorboats (including motorized jetboats) within this segment of the Salmon River shall be permitted to continue at a level not less than the level of use which occurred during calendar year 1978. (sec. 9(a)(C))

The Forest Service calculated the 1978 level from historical data and promulgated it as a standard in 1980. Private jet boat use is currently limited on the Salmon River during the control season of June 20 to September 3 to 15 Boat Use Days (BUDs) per week. Use outside of the control season is unregulated.

CIWA also required the agency to complete a comprehensive management plan for the entire wilderness. This plan was not begun until 1994. The preferred alternative in the 1998 DEIS for the FC-RONRW Wilderness Management Plan would extend the control season for private jet boaters and limit noncommercial BUDs to two per week (USDA Forest Service 1998). This proposal drew intense criticism from private jet boaters, who believe that the intention of CIWA was not to restrict motorboat use of the river. In response to these comments, and as part of a larger project aimed at understanding this class of river users, I undertook a legislative interpretation of the motorboating clause in CIWA.

Statutory Construction—The language of the statute clearly indicates that jet boat use can continue above a certain level. The ambiguity appears in regards to how much regulation was intended above that level. The issue is very politically controversial, and there is heated argument over whether or not regulation was intended. Therefore, a recourse to legislative interpretation is warranted.

Legislative Interpretation—The first legislative history documents to examine are committee reports. In November 1979, the Senate Committee on Energy and Natural Resources released a bill that included a jet boat provision similar to the final, except that the phrase “approximately equal to” was used in place of “not less than.” The report justified this provision “primarily because jetboats provide a means to reach deep into the wilderness in a relatively short time”(U.S. Congress 1979, 23). It further clarified that:

The Committee went beyond existing law . . . to assure that this traditional means of access to the river and the wilderness beyond will be allowed in the future. This section of the bill provides for the continuation of this use without preempting the prerogatives of the Secretary under the provisions of the Wild and Scenic Rivers Act to regulate motorized travel on the river in times of low water, or high fire hazard, or for other reasonable purposes. (U.S. Congress 1979, 23)

During debate in the Senate, Senator Church proposed an amendment that would change the language of section 9(a)(C) from “approximately equal to” to a “level not less than.” The reason for this change was to:

make clear that the purpose of the section is not to establish a ceiling on motorboat use on the mainstream of the Salmon but, rather, to use the year 1978 as a floor. . . the Secretary would retain the necessary flexibility to increase the use of motorboats on the basis of a management plan. . . The language would not result in overuse of motorboats in the future, but would simply prevent a decision on the part of the Secretary that would curtail their use below the level of calendar year 1978.

This amendment was accepted and reappeared in the House’s report in March of 1980.

The legislative history of this section indicates that while the level of motorboat use should not be administratively reduced below the 1978 level, Congress’ intention was never to limit the agency’s discretion to manage use levels to protect the wilderness resource. The question may still remain as to the accuracy of the 1978 level used by the Forest Service, but that will not be resolved through congressional research.

Conclusions

As these two examples illustrate, wilderness managers are asked to make a host of discretionary decisions in a very polarized atmosphere. They are constantly faced with pressures from interest groups demanding opposing interpretations of wilderness regulations. Where the Wilderness Act is clear and directive, these requests are easily dealt with; where the Act is ambiguous, the result is often controversy and confusion. These are just a few examples of the discretionary quandaries facing managers. This analytical process provides them with a clearer view of both their congressionally mandated responsibilities and the ideals that underlie them.

References

- Breyer, Stephen. 1992. On the uses of legislative history in interpreting statutes. *Southern California Law Review* 65: 845-874.
- Crawford, Earl T. 1940. *The construction of statutes*. Saint Louis: Thomas Law Book Company.
- de Sloovere, Frederick J. 1940. Extrinsic aids in the interpretation of statutes. *University of Pennsylvania Law Review* 88: 527-555.
- Dickerson, Reed. 1975. *The interpretation and application of statutes*. Boston, MA: Little, Brown, and Co.
- Folsom, Gwendolyn B. 1972. *Legislative history: research for the interpretation of laws*. Charlottesville, VA: University Press of Virginia.
- McDonald, Leigh Ann. 1991. “The role of legislative history in statutory interpretation: a new era after the resignation of Justice William Brennan?” *Missouri Law Review*. 56: 121-141.
- Mercure, Delbert V. and William M. Ross. 1969. The Wilderness Act: a product of congressional compromise. In *Congress and the Environment*, ed. R.A. Cooley and G. Wandesford-Smith, 47-64. Seattle, WA: Univ. of WA Press.
- Singer, N.J. editor, *Sutherland Statutes and Statutory Construction*. 1992. 5th ed. Deerfield, IL: Clark Boardman Callaghan.
- USDA Forest Service. 1998. Frank Church-River of No Return Wilderness programmatic and operational management plans, Vol. I and II, Draft Environmental Impact Statement (DEIS). Bitterroot, Boise, Nez Perce, Payette, and Salmon-Challis National Forests. Intermountain and Northern Regions.
- U.S. Congress. Senate. 1957. Committee on Interior and Insular Affairs, National Wilderness Preservation Act, Hearings on S. 1176, June 19-20.

- U.S. Congress. Senate. 1961. Debate on the National Wilderness Preservation System. 87 Cong. 1st sess. Congressional Record 107, September 6, 1961.
- U.S. Congress. House. 1962. Debate on the National Wilderness Preservation System. 87 Cong. 2nd sess. Congressional Record 108, Part 7. September 21, 1962.
- U.S. Congress. Senate. "River of No Return Wilderness." S. Rept. 96-414 of the Senate Committee on Energy and Natural Resources to accompany S. 2009. 96th Cong. 1st sess. November 14, 1979.
- U.S. Congress. House. 1980. Designating Certain Public Lands in Central Idaho as the River of No Return Wilderness, Designating a Segment of the Salmon River as a Component of the National Wild and Scenic Rivers System and for other Purposes. S. Rept. 96-838 Part 1 of the Committee on Interior and Insular Affairs to accompany S. 2009. 96th Cong. 2nd Sess.
- U.S. Congress. Senate. 1980. Debate on the Central Idaho Wilderness Act. 96th Cong. 2nd sess. Congressional Record 126, June 26.
- U.S. Public Law 88-577. 1964. The Wilderness Act. 78 Stat. 890.
- U.S. Public Law 96-312. 1980. Central Idaho Wilderness Act. 94 Stat. 948.
- United States v. Missouri Pacific Railroad*. 278 U.S. 269 (1929).
- United States v. American Trucking Association*. 310 U.S. 534 (1940).

Seeking a Scientific Approach to Backcountry Management in Yellowstone National Park

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Abstract—Three criteria are used to assess how Yellowstone's wilderness managers incorporate science into management: preciousness, vulnerability and responsiveness to management. Four observations are proposed. First, where scientists lead, managers will follow. Scientists that leave the best trail will be followed most closely. Second, managers need to refocus efforts on landscape-scale impacts, and they need scientists to give them the techniques to do this. Third, managers need to refocus efforts on impacts to visitors. Finally, managers need to refocus efforts to assess cumulative effects; however, scientist must first develop usable, accurate models.

Yellowstone Park staff began developing the park's first Backcountry Management Plan in 1991 using a Limits of Acceptable Change model (Stankey and others 1985). During the planning process, we relied on existing wilderness science to identify indicators of resource and social conditions, inventory and set standards for those conditions, identify management actions, and outline monitoring programs. While working on the plan (which was never signed and remains a draft), we focused science-based management efforts on site-specific impacts. Since the draft plan was completed in 1994, we have worked on several plans that influence wilderness conditions, including the draft Winter Use and Commercial Services Plans. We have also made or are making decisions on several proposals for new types of use, ranging from goat packing to whitewater access on Yellowstone's rivers. In each case, concerns about landscape-scale impacts have superseded concerns about site-specific impacts, needed science is missing, or managers are unaware of science that is available. In this paper, we discuss what we learned during and since the time we developed the backcountry plan about the science available to managers, how we tried to incorporate that science into our wilderness management, and where we think more work is necessary.

The National Park Service (NPS) does not have a strong tradition of incorporating science into management decisions (Sellers 1997). The agency has, however, recognized the need for better science-based management. In *National*

Parks for the 21st Century: THE VAIL AGENDA, the NPS (1991) recommended that the agency "must engage in a sustained and integrated program of natural, cultural, and social science resource management and research aimed at acquiring and using the information needed to manage and protect park resources." In 1998, Congress authorized and directed the NPS to "assure that management of units of the National Park System is enhanced by the availability and utilization of a broad program of the highest quality science and information" (National Parks Omnibus Management Act of 1998).

There is a clear delineation between science and science-based management. Managers are not, for the most part, conducting original research; we are not testing hypotheses. That is simply not our role. We depend on scientists to conduct original research that we can apply to management decisions. We use their information to develop policy, create regulations, implement inventory and monitoring schemes for assessing local conditions, choose indicators of resource and social conditions, set standards and assess management techniques. Stoltenberg and others (1970) state "the value of the resource scientists must ultimately be determined by how much their efforts increase the efficiency of the resource manager." They go on to outline the major purposes of natural resources research: first, to "develop new alternatives for the resource manager;" second, to "answer questions of fact that arise during management;" and third, to "answer questions of fact that arise during resource research, since it is only after some of these basic questions have been satisfactorily answered that the first two objectives can be achieved most efficiently." As wilderness managers, we have found that the available science is strong in some areas, but weak or nonexistent in others of considerable concern to us or our user groups.

Background

More than 95 percent of Yellowstone National Park's 2.2 million acres is considered backcountry. Seven designated wilderness areas administered by the U.S. Forest Service adjoin the park. In accordance with the 1964 Wilderness Act, a wilderness study was completed for Yellowstone in 1972. It recommended that more than two million acres of Yellowstone National Park be designated as wilderness. This recommendation was recently updated under President Clinton's Lands Legacy Initiative. Although Congress has not acted on these recommendations, the land is managed so as not to preclude wilderness designation, in accordance with NPS *Management Policies* (1988) and Yellowstone's

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Master Plan (1973). Yellowstone's backcountry has not been developed, with the exception of a relatively sparse trail system, a network of designated campsites and 43 patrol cabins and lookouts, most of which are historic.

Presently, the park has 90 trailheads, more than 900 miles of trails, and 302 designated backcountry campsites (with a total capacity for 316 individual parties). More than 45,000 visitor-use nights were recorded in 1998. The majority of use occurs between June and the end of September.

Few records pertain to backcountry use and management prior to 1973, when Yellowstone developed its present system of managing overnight backcountry use through a designated-campsite permit system. Previously, campsites were not defined or established; however, overnight camping and fire permits were required. A central backcountry office was created to record and track campsite use. Yellowstone developed operating procedures for backcountry management in 1974. From 1973 to 1982, backcountry use generally increased, peaking in 1977 and again in 1981, with 55,331 and 55,030 visitor-use nights per year, respectively. Backcountry use then declined by approximately one-third between 1982 and 1986. Since 1987 (except for 1988 when most of the backcountry was closed due to fires), human use has increased steadily and currently exceeds 45,000 visitor-use nights each year. No permits are as yet required for day use.

Stock use began to increase as early as 1986 and, with the exception of 1988, climbed through 1993, when it leveled out. Stock use is currently approximately 8,000 use nights per year, the highest level since records began to be kept in 1973.

Day use was monitored in 1992. Use levels varied, depending on trail location and distance from the trailhead, and ranged from zero to 109 people per day per trail. Overall, we estimate that the level of day use is approximately four times the level of overnight use.

A reservation system for commercial outfitters was implemented in 1985. In 1999, the park had 49 stock outfitters, 27 backpacking outfitters and 18 canoeing/kayaking outfitters. Outfitters and their clients share the same system of campsites that private parties use; they are not assigned areas or sites in which they can operate, and there is currently no limit on the number of trips an individual outfitter can take. Only 30% and 75% of campsites may be reserved in advance by commercial nonstock and stock outfitters, respectively. About 12%, 27% and 89% of overnight use is currently comprised of commercially led backpacking, boating and stock parties, respectively.

Overnight use is managed through a system of backcountry permits. Backcountry staff at 12 locations throughout the park work in conjunction with the Central Backcountry Office to dispense up-to-date information about trail and campsite conditions and special restrictions designed to minimize public safety hazards and resource conflicts. The park has implemented a computerized network for backcountry permitting, including the opportunity (since 1996) for the general public to make advanced reservations for backcountry campsites. Park rangers, in cooperation with trail crew staff and others, are responsible for the supervision of trails, campsite maintenance and evaluations, law enforcement and resource protection patrols, outfitter evaluations, monitoring visitor use, mitigating resource impacts,

recommending any additional needed corrective action and other resource management activities.

Assessing Yellowstone's Science-Based Management Efforts

The goal of this paper is to assess how Yellowstone's wilderness managers incorporate science into management and how we allocate our science resources. We based this assessment on three criteria for allocating wilderness management resources proposed by Cole (1997): 1) Preciousness: More resources ought to be allocated to areas that are more precious, defined as areas that are the most undisturbed or undeveloped; 2) Vulnerability: More resources ought to be allocated to areas that are likely to degrade further; and 3) Responsiveness to Management: More resources ought to be allocated to areas that are likely to respond positively to good management. Using these criteria, we assessed our current and proposed science-based program in five subject areas: 1) campsite inventories; 2) a program of monitoring grazing at stock sites; 3) the spread of exotic organisms by recreational users; 4) impacts of recreational users on wildlife; and 5) impacts on visitor experience. We focused on these areas because we are either allocating significant resources toward them or because significant concerns have been raised about them by visitors and park staff.

Campsite Inventories

Like many park and wilderness managers, we have invested much of our inventory and monitoring capital in campsite inventories. We completed two separate inventories. A Code-A-Site inventory was completed in 1979, but the key was lost and the data rendered unusable. A much more intensive inventory was completed from 1989-92. We inventoried 226 (75%) of our designated campsites by locating eight transects, radiating from the campsite center to measure the edge of bare ground and trampled vegetation; establishing a photopoint and taking a series of photos; and measuring the amount of and distance to firewood, the number of damaged trees and social trails and the distance to water and the main trail. The draft Backcountry Management Plan called for monitoring these sites every five years.

Campsite inventories have become a standard way for managers throughout the National Park System to monitor visitor impacts on wilderness resources. In a 1993 survey, Marion and others (1993) found that nearly 40% of parks that participated in the survey used campsite monitoring to evaluate visitor impacts. This compared to fewer than 10% of parks that used trail impacts, wildlife impacts, water quality or visitor experience. It was interesting to note that Marion and others (1993) found that although more National Park Service managers were concerned about trail impacts than campsite impacts (50% to 36% respectively), more managers monitored campsites than trails (nearly 39% to 9%).

In Yellowstone, we chose to put effort into campsite inventories for valid reasons. Managers are concerned about impacts at campsites because, as Cole (1982) pointed out, "In many areas, the most severe impacts occur on campsites

where use is highly concentrated, both spatially and temporally.” Campsite impacts are very evident, and wilderness rangers spend a lot of time working at campsites. Perhaps most important, though, is that a campsite inventory is easy to do because the scientists have left a well-worn trail. Techniques have been developed through research and have been widely distributed to managers (Marion 1991, Cole 1983, Cole 1989). Where scientists lead, managers will follow, or try to follow. Scientists that leave the best trail will be followed most closely.

But is this a productive use of scarce resources? These campsites do not meet the criterion of preciousness; by definition, they are some of the most developed, thus the least precious, backcountry areas. These areas are also some of the least vulnerable. Research indicates that most impact occurs rapidly at low levels of use (Marion and Cole 1996). When Yellowstone moved to a designated campsite management regime in 1973, sites were selected primarily from existing campsites. Most of Yellowstone’s designated campsites have been in use for over 30 years, and site use varies from six to more than 400 visitor use nights each year. These sites are not changing rapidly, if at all. Finally, these campsites respond slowly to management action. We have had some limited success with restricting wood fires, limiting the number of people allowed per night at sites and revegetating site margins. But these management actions tend to reduce sites from being highly impacted to being only moderately impacted. Based on this assessment, we ought to shift emphasis away from campsite monitoring toward more productive activities.

Stocksite Grazing Monitoring

The use of packstock presents special challenges to managers. McClaran and Cole (1993) state:

Even low levels of packstock use can cause substantial impacts. Compared to impacts caused by backpackers, packstock impacts to trails and campsites are more severe, and packstock impacts to grazing areas have no corollary to backpackers’ impacts.

Yellowstone managers became concerned enough about backcountry stock grazing impacts to begin developing a system for monitoring such impacts in 1984, when stock use was beginning to increase in Yellowstone; stock grazing practices led to overuse in close proximity to campsites and little or no use in the far reaches of grazing meadows. Managers began to search for a grazing monitoring system that was simple to use, easy to explain and accurate. We searched through the range science literature and, after a couple of false starts, borrowed the grazed loop method from the Forest Service for estimating range utilization (USDA Forest Service 1977). By 1994, we had established stock use night limits, based on monitoring results, at all of our (approximately 50) popular stock sites.

Here, too, we must ask ourselves: Is this monitoring system a productive use of scarce resources? These areas do not meet the criterion of preciousness; again, they are some of the most developed, thus the least precious, backcountry areas. Stock sites are vulnerable to grazing impact. Heavy grazing can lead to changes in species composition and reduce biomass production, plant size and seed output

(Briske 1991, McClaran and Cole 1993). Preliminary inventories (Sauer 1989, Whipple pers. commun.) indicate that Yellowstone stock-site meadows are, with a few exceptions, still comprised of native vegetation; thus, they are still important to maintain. Finally, these sites have responded to management action. Managers have used the results of stock-site monitoring to limit use and educate stock-site users about grazing management. Advanced reservations are limited based on the results of several years’ of monitoring at each stock site.

During the field season, wilderness rangers can adjust use levels based on monitoring results (results are influenced by weather and stock handling). As stock users have come to understand the monitoring system, they have changed the way they handle horses, grazing the farther reaches of stock meadows to conserve user days. In one example, utilization in the meadow adjacent to Soldier’s Corral, a campsite on the Gardner River, was reduced from almost 70% to 20%, maintaining consistent use levels but dispersing use, after the monitoring system was installed. Based on this assessment, we ought to continue conducting stock-site monitoring; however, it should probably not be expanded much beyond the present scope since impacts are not likely to spread to more precious areas.

Exotic Species

At least 180 nonnative plant species have been found in Yellowstone (Olliff and others, in press). Wilderness managers have conducted a few sporadic surveys for exotic plants and concluded that weeds listed as noxious in the tri-state area (Idaho, Montana and Wyoming) are primarily restricted to roadsides and developed areas.

Of more concern in recent years is the discovery of two exotic organisms that pose significant threats to native fish and aquatic ecosystems: *Myxobolus cerebralis*, the parasite that causes whirling disease, and the New Zealand mudsnail (*Potamopyrgus antipodarum*).

M. cerebralis is a parasite native to Eurasia that was introduced into North America in the 1950s. It penetrates the head and spinal cartilage of fingerling trout, causing fish to swim erratically and have difficulty feeding and avoiding predators. The disease can cause high rates of mortality in young-of-the-year fish. When an infected fish dies, thousands of parasite spores are released into the water (Whirling Disease Foundation 1999). So far, severe damage has been documented in wild rainbow trout populations. For example, in the Madison River in Montana, whirling disease caused a 77% decrease in the rainbow trout population in each mile of the severely infected sections (Whirling Disease Foundation 1999). *M. cerebralis* was discovered in Yellowstone Lake in 1998. Fisheries managers are concerned that Yellowstone cutthroat trout may be highly susceptible to whirling disease. Yellowstone Lake is the last refuge of the native Yellowstone cutthroat trout; 91% of the remaining range is located in Yellowstone National Park, mostly in Yellowstone Lake and the Yellowstone River (Varley and Schullery 1995). Coupled with the discovery of exotic lake trout in Yellowstone Lake in 1994, whirling disease may pose a significant threat to the Yellowstone cutthroat trout.

The New Zealand mudsnail was discovered in the Snake River south of Yellowstone in 1985 (Gangloff and others

1998). Since 1987, it has been discovered in four widespread localities in the U.S: the Middle Snake, Idaho; Lake Ontario; the Snake River from American Falls to the Thousand Springs area; and the Madison River. It has recently been discovered in Yellowstone, in the Firehole River, the Gardner River and the Snake River near South Entrance. Specimens can survive out of water for several hours. If kept in damp surroundings (such as wading boot tread or a Velcro strap), the snail's terrestrial survival time increases markedly (Gangloff and others 1998). Ecological impacts include competition with native species and changes in community biodiversity and ecosystem function. The mud snail seems to be a poor food source for secondary consumers such as fishes and terrestrial animals such as birds. In Yellowstone, one fisheries manager has observed that the occurrence of New Zealand mudsnail seems to be correlated with areas where people typically swim (Mahony, personal communication).

Should wilderness managers spend more resources monitoring exotic species? These species have high potential to spread off-site and invade the most pristine, undeveloped areas—those that fit the very definition of preciousness. The pristine waters that these species invade are definitely vulnerable. Finally, do these exotic species respond to management action? Without further research and monitoring, that is unclear. But because these species invade the most pristine, vulnerable areas, we ought to step up our inventory and monitoring efforts for exotic plant species, as well as aquatic exotics. Scientists need to help us by increasing research on how exotic organisms are spread, which will help us identify areas in which to concentrate both inventories and management efforts such as regulations on human use. Keeping up with research on exotic species may be a stretch for many wilderness managers since the results are typically reported outside the “wilderness science” literature.

Recreation Effects on Wildlife

It is well-documented that nonhunting recreation can have negative impacts on wildlife (Aune 1981, Hammitt and Cole 1987, Cassirer and others 1992, Knight and Gutzwiller 1995, Olliff and others 1999). A lot of research has been done; however, when a manager needs to make decision about a certain type of use, the information is often confusing, conflicting, counter-intuitive or unapplicable. Research conducted in one area or habitat or on one species is often hard to extrapolate. Managers have a difficult time applying the research to site-specific decisions. For example, in the draft Backcountry Management Plan, Yellowstone wilderness managers proposed an increase in dispersed camping (camping in nondesignated sites), but the U. S. Fish and Wildlife Service expressed concern that the effects on threatened grizzly bears would be unacceptable (U. S. Fish and Wildlife Service 1994). Unfortunately, most of the research related to human effects on grizzly bears has been focused on roads and developments rather than on dispersed activities such as backcountry camping.

Should wilderness managers spend more capital monitoring recreational effects on wildlife? Wildlife are among the most precious wilderness resources. In a recent survey, 82% of the respondents rated wildlife as extremely important; in fact, it was rated much higher than any other wilderness

resource (Littlejohn 1996). Like exotic species, impacts to wildlife can migrate off-site to become a landscape-scale issue. Are wildlife vulnerable? Yes, and wildlife that inhabit the most pristine areas may be most likely to be affected since they have less chance to habituate to human activity (Aune 1981, Cassirer and others 1992). Finally, do wildlife species respond to management action? It is likely that restricting human use to designated trails and campsites, or away from some areas altogether, can help protect wildlife from human influence. It is easy to conclude that more effort needs to be expended to determine the effects of visitors on wildlife and to determine appropriate management responses, but it is difficult to determine where managers should focus their effort. We seek scientists' help in understanding the effects of visitors on wildlife. But this is difficult, time-consuming work. It must be accomplished species by species, habitat by habitat and considering a broad array of recreational activities. Again, keeping up with such research may be a stretch for many wilderness managers since the results are typically reported outside the “wilderness science” literature in a wide spectrum of wildlife, bird and fish-related scientific forums.

Based on the previous four examples, we believe that on the continuum of site-specific to landscape-scale impacts, wilderness managers have focused more on site-specific impacts. We need to refocus on landscape-scale impacts, and we need scientists to give us the techniques to do this. Wilderness managers currently need to look outside of the “wilderness science” literature to obtain information on landscape-scale impacts.

Visitor Experience Surveys

The National Park Service has not, to our knowledge, conducted any visitor attitude studies directed at backcountry users in Yellowstone. We hear from visitors in letters, comments and other ways. Last year, as a group of Yellowstone managers were hiking out of Slough Creek, we came across a place where someone had thrown rocks and logs down on the wagon road that leads to the Silvertip Ranch, a private facility located outside the park but accessed through Yellowstone. They were obviously angry, and they were making a statement. The section of road was so steep that it would have been impossible for a wagon to pass.

While working on the Backcountry Management Plan, we did not commission or try to conduct any visitor experience surveys. We focused our monitoring on counting the number of visitors leaving trailheads, the number of overnight visitors in the backcountry and otherwise monitoring resource impacts. During the public comment period, however, we found that most of the controversial issues were driven by visitor attitudes: conflicts between stock users and hikers, controversy over whether to have designated campsites or dispersed camping, whether to have wooden or metal directional signs and whether to have more or fewer orange trail markers (or none at all).

Since the draft Backcountry Management Plan was completed, one survey has been conducted independent of the National Park Service on visitor perceptions of backcountry llama packing (Blahna and others 1995). Wilderness managers have written a proposal, with Dr. Alan Watson,

Aldo Leopold Wilderness Research Center, to conduct a visitor survey at Slough Creek. However, social science research in the backcountry has never been a high enough park priority to fund.

Why haven't we done a better job of conducting visitor surveys? A lot of science has been conducted in this area, although not in Yellowstone or many other NPS areas (Marion and others 1993). Some reasons likely include: 1) Many managers have a strong background in biological sciences, not in social sciences. So visitor surveys are foreign to us; we tend to focus more on resource impacts. 2) Visitor surveys can be expensive and limited by government regulations. 3) Unlike campsite inventories, surveys do not have step-by-step instructions for managers to do it themselves. 4) Data analysis is difficult—at least, it is perceived to be difficult. 5) Managers may either think they know what is best or, since they have a lot of contact with visitors, they know what visitors want (or should want).

Should wilderness managers spend more capital monitoring visitor attitudes? The concept of visitor experience does not seem to fit the area-based definition of preciousness. However, we might argue that our visitors, and the support they provide for wilderness, are our most precious resource. Are visitors' experiences vulnerable, and do they respond to management? Yes, perhaps more than any other resource. Management actions, such as requiring permits, erecting signs, designating campsites and maintaining trails, may have a disproportionate effect on visitors' perception of their visit. On the continuum of impacts-to-resource to impacts-to-visitors, wilderness managers have focused more on impacts to resources. We conclude here that more effort should be focused on surveying visitor experience and attitudes and helping managers understand options for managing visitors.

Cumulative Impacts

Assessing cumulative impacts is an issue we did not assess against Cole's criteria, but it is critical for park managers. During Section 7 consultation for the Backcountry Management Plan, the U.S. Fish and Wildlife Service (1994) commented, "Due to the fact that recreational and other demands are and can only be expected to increase, cumulative impacts to the grizzly bear remain a concern and an issue that needs to be addressed." This sentiment can be extended to all resources and to the visitor experience.

Cumulative impacts are especially critical when assessing requests for new uses or proposals to dramatically change existing uses. Traditional knowledge of recreational impacts and methods for measuring recreational impacts do little to inform the debate on whether or not to allow many of these new uses. A cumulative impacts assessment lends itself to assessing landscape-scale changes. How will new activities, or major changes to existing activities, add to the total impact from all recreational activities?

This is difficult work. First, it is very difficult to know how a new use will grow. If the park managers that allowed the first 200 snowmobiles into Yellowstone in the 1960s were told that the park is now visited by over 170,000 snowmobilers each winter, they would surely be amazed. Second, the amount of data and the complexity of the

modeling involved in a realistic Cumulative Effects Model (CEM) are staggering. The Grizzly Bear CEM team has been working on the CEM for over 18 years. While it is a good start, managers still do not incorporate it into most decision-making and it has limited application to a wider range of management. To be applied to a wider variety of species, the model coefficients would have to be developed for each individual species. Managers need to refocus efforts to assess cumulative effects; however, scientists must first develop usable, accurate models.

Conclusions

Managers are under a lot of constraints from court orders (Yellowstone managers are being guided by federal judges on bison management, wolf management, winter use management and thermophile management), pressure from politicians, pressure from special interest groups and the tyranny of history. Having access to good, applicable science helps to reduce these constraints. Wilderness managers need to do a better job of searching out relevant science and applying our efforts to the most important problems. In some areas, the science is missing or needs to be more fully developed. If managers can integrate science into decisions on wilderness management in a responsible way, people will more readily accept, and actually support, decisions.

Specifically, we propose four observations based on our experience. First, where scientists lead, managers will follow, or try to follow. Scientists that leave the best trail will be followed most closely. Second, on the continuum of site-specific to landscape-scale impacts (impacts that migrate off-site), wilderness managers have focused on site-specific impacts. We need to focus more on landscape-scale impacts, and we need scientists to give us the techniques to do this. Wilderness managers currently need to look outside of the "wilderness science" literature to obtain information on most landscape-scale impacts. Third, wilderness managers have focused more on impacts to resources than on impacts to visitors. We need to refocus more of our efforts on assessing impacts to visitors. Fourth, managers need to refocus efforts to assess cumulative effects; however, scientists must first develop usable, accurate models.

References

- Aune, Keith E. 1981. Impacts of winter recreationists on wildlife in a portion of Yellowstone National Park, Wyoming. M.S. Thesis, Montana State Univ., Bozeman. 110 p.
- Blahna, Dale J.; Smith, K.S.; Anderson, J.A. 1995. Backcountry llama packing: visitor perceptions of acceptability and conflict. *Leisure Sciences*. 17(3):185-204.
- Briske, D.D. 1991. Developmental morphology and physiology of grasses. In: *Transactions of the 51st North American Wildlife and Natural Resources Conference*: 265-271.
- Cassirer, E. Frances; Freddy, David J.; Ables, Ernest D. 1992. Elk responses to disturbance by cross-country skiers in Yellowstone National Park. *Wildlife Society Bulletin*, 20(4):375-381.
- Cole, David N. 1982. Wilderness campsite impacts: effect of amount of use. Res. Pap. INT-284. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 34 p.
- _____. 1983. Monitoring the condition of wilderness campsites. Res. Pap. INT-302. Ogden, UT: U.S. Department of Agriculture, Forest Service, Research Station. 10 p.

- _____. 1989. Wilderness campsite monitoring methods: a sourcebook. Gen. Tech. Rep. INT-259. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 57 p.
- _____. 1997. Soul of the wilderness. *International Journal of Wilderness*, 3(4):4-8.
- Gangloff, Michael M.; Dybdahl, M.F.; Kerans, Billie L. 1998. The invasion of the New Zealand Mud Snail (*Potamopyrgus antipodarum*) in North America: Distribution, Ecology and Potential Impacts. Unpublished report, Montana State University, Bozeman, MT.
- Hammit, William E.; Cole, David N. 1987. Wildland recreation: ecology and management. New York, NY: John Wiley and Sons. 341 p.
- Knight, Richard L.; Gutzwiller, Kevin J., editors. 1995. Wildlife and recreationists: coexistence through management and research. Island Press, Washington, D.C. 369 p.
- Littlejohn, Margaret. 1996. Yellowstone National Park visitor study. Visitor Services Project Report Number 75. Cooperative Park Studies Unit, University of Idaho, Moscow, ID. 79 p.
- Mahony, Dan. Personal Communication 1999. Lead Fisheries Biologist, National Park Service, Yellowstone National Park.
- Marion, Jeffrey L. 1991. Developing a natural resource inventory and monitoring program for visitor impacts on recreation sites: a procedural manual. Natural Resources Report NPS/NRVT/NRR-91/06. Denver, CO: U.S. Department of the Interior, National Park Service. 59 p.
- Marion, Jeffrey L.; Roggenbuck, Joseph W.; Manning, Robert E. 1993. Problems and practices in backcountry recreation management: a survey of National Park Service Managers. Natural Resources Report NPS/NRVT/NRR-93/12. Denver, CO: U.S. Department of the Interior, National Park Service. 48 p.
- McClaran, Mitchel P.; Cole, David N. 1993. Packstock in wilderness: use, impacts, monitoring, and management. Gen. Tech. Rep. INT-301. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 33p.
- Marion, Jeffrey L.; Cole, David N. 1996. Spatial and temporal variation in soil and vegetation impacts on campsites. *Ecological Applications*, 6:520-530.
- National Park Service. 1973. Yellowstone master plan. FES 74-31. Yellowstone National Park, WY. 59 p.
- National Park Service. 1988. NPS 88: Management Policies. U.S. Department of the Interior, National Park Service, Washington, D.C.
- National Park Service. 1991. National parks for the 21st century: the Vail agenda. Chelsea Green Publishing Company, Post Mills, VT. 142 p.
- National Parks Omnibus Management Act of 1998.
- Olliff, S. Thomas; Legg, Kristin; Kaeding, Beth, editors. 1999. The effects of winter recreation of wildlife of the Greater Yellowstone Area: a literature review and assessment, draft report. A report to the Greater Yellowstone Coordinating Committee. Yellowstone National Park, WY. 315 p.
- Olliff, S. Thomas; Renkin, Roy; McClure, Craig; Miller, Paul; Price, Dave; Reinhard, Dan; Whipple, Jennifer. In Press. Managing a complex exotic vegetation program in Yellowstone National Park. *Western North American Naturalist*, 0:000-000.
- Olson-Rutz, K.M.; Marlow, C.B.; Hansen, K.; Gagnon, L.C.; Rossi, R.J. 1996. Packhorse grazing behavior and immediate impact on a timberline meadow. *Journal of Range Management*, 49:546-550.
- Sauer, Brad. 1989. Unpublished field notes, Yellowstone backcountry campsite inventory. Yellowstone National Park Backcountry Office, WY.
- Sellers, Richard W. 1997. Preserving nature in the national parks: a history. Yale University Press, New Haven, CT. 380 p.
- Stankey, George H.; Cole, David N.; Lucas, Robert C.; Peterson, Margaret E.; Frissell, Sidney S. 1985. The Limits of Acceptable Change (LAC) system for wilderness planning. General Technical Report INT-176. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain and Range Experiment Station. 37 p.
- Stoltenberg, Carl H.; Ware, Kenneth D.; Marty, Robert J.; Wray, Robert D.; Wellons, J.D. 1970. Planning research for resource decisions. The Iowa State University Press, Ames, Iowa. 183 p.
- U.S. Fish and Wildlife Service. 1994. Section 7 consultation. Unpublished memorandum ES-61411 from the State Supervisor, Ecological Services, to the Superintendent of Yellowstone National Park, June 29, 1994. Cheyenne, WY. 6 p.
- USDA Forest Service. 1977. Range analysis handbook. U.S. Forest Service, Region 1, Missoula, MT.
- Varley, John D.; Schullery, Paul, editors. 1995. The Yellowstone Lake crisis: confronting a lake trout invasion. A Report to the Director of the National Park Service. Yellowstone Center for Resources, National Park Service, Yellowstone National Park, WY. 36 p.
- Whipple, Jennifer. Personal Communication 1999. Botanist, National Park Service, Yellowstone National Park.
- Whirling Disease Foundation. 1999. 1999 Whirling Disease Symposium synopsis. Whirling Disease Foundation website, www.whirling-disease.org.

Grizzly Bears as a Filter for Human Use Management in Canadian Rocky Mountain National Parks

Derek Petersen

Abstract—Canadian National Parks within the Rocky Mountains recognize that human use must be managed if the integrity and health of the ecosystems are to be preserved. Parks Canada is being challenged to ensure that these management actions are based on credible scientific principles and understanding. Grizzly bears provide one of only a few ecological tools that can be used to guide the management of human activities. Grizzly bear needs, as they relate to habitats, movement corridors, habituation and human risk management, were assessed from three spatial scales (regional landscape, landscape management unit, and area planning) and provide the basis for the implementation of numerous human use management actions. The relationship between the analysis of grizzly bear needs and the management actions are illustrated in the case studies.

Parks Canada's mission statement is:

To protect for all time representative natural areas of Canadian significance in a system of national parks, and to encourage public understanding, appreciation, and enjoyment of this natural heritage so as to leave it unimpaired for future generations (Canadian Heritage 1994).

This mission statement makes reference to the following requirements:

- “representative natural areas” (implies a provision of ecological services)
- “encourage public understanding, appreciation and enjoyment” (implies a provision of experiential services)
- “to protect for all time” (implies a need for sustainability)

During previous eras, when human visitation was low and ecological understanding was limited, Parks Canada had little difficulty fulfilling the above requirements. However, as ecological understanding increased and social and economic conditions changed, with pressure for development and increased use in some of the more popular Canadian National Parks (Rocky Mountain Block of Yoho/Kootenay/Banff & Jasper), managing for a continued balance of protection and use proves difficult. These challenges were acknowledged by Parks Canada (Parks) when it revised both its operating policy (Guiding Principles and Operational Policies) and its legislative framework (the National Parks

Act) to reflect the need to ensure protection of the ecological integrity of the parks while providing for a range of visitor opportunities.

This need for a balanced management is further emphasized through statements such as: “While ecological integrity is clearly the priority in the Park, it is recognized that tourism has been and will continue to be its primary form of human and economic activity” (Banff Bow Valley Study 1996). It is therefore essential to understand how tourism affects ecological integrity. “Equally important is how to manage this diverse phenomenon so Canadians may continue to enjoy the many experiences Parks offers and to obtain its substantial economic benefits without undermining ecological integrity.” To achieve this objective, there is a need for both an integrated and a systems approach to the management of protected areas.

Within the context of the social and economic systems in which Canadian national parks operate, it has become apparent that the provision of viable ecological and experiential services will require the management of human use.

Human use management is the direction and guidance of people, their numbers, their behaviour, permissible activities, and necessary infrastructure. The objective of human use management is to allow people to enjoy a national park without damaging its ecological integrity; while it may require some restrictions, it should not be seen as limiting people's freedom. Alternatives for managing access and use will vary from relatively low-key approaches, such as better signage and education, to more active approaches such as closures, quotas, and permits. Our challenge in developing an effective human use strategy is to determine which combination of approaches will address both visitor and ecological needs in a manner that supports both. Currently, there is little direct management of human use in the Rocky Mountain National Parks.

Human use management involves two aspects of the visitor opportunity—supply and demand. Supply relates to the amount of use (determined according to activity types, locations, and timing) that can be provided in a park, subject to defined ecological and social objectives. Supply targets can be expressed in a number of different ways (user numbers, satisfaction rates, educational/knowledge change, etc.). Once the supply of the visitor opportunity is defined, demand can be managed to achieve a better balance between the two. Demand will have to be actively managed and will require the involvement of internal and external groups. Parks Canada has made advances in defining use relative to ecological objectives, but it is only at the preliminary stages of defining socially based supply targets or managing demand.

Acknowledging that human use management will be difficult, Parks must move forward to develop tools to help it

In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management; 1999 May 23–27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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meet that mandate. There must be an attempt to determine which ecological systems are sensitive to human activities and how they will be useful in guiding the management of these activities. It is proposed that large carnivores generally and grizzly bears (*Ursus arctos*) specifically represent a sensitive ecological system.

Within the Central Rockies Ecosystem (figure 1), including Yoho, Kootenay, Banff, Jasper, and Mount Revelstoke/Glacier National Parks, there have been considerable resources devoted to large carnivore research, monitoring, and management in the past decade. The conservation of large carnivores is an issue that transcends both geographic and administrative boundaries. The recognition of the social and ecological role that the species fulfills has made it the focus of many research and land use planning initiatives. Table 1 summarizes the current perspectives on conservation of grizzly bears.

Three case studies will be presented to illustrate how grizzly bears are being used to help define acceptable human use management strategies. The studies represent approaches taken at the landscape, watershed (landscape management unit), and sub-watershed (area planning) scale. Some of the work presented reflects work in progress, while others were recently completed.

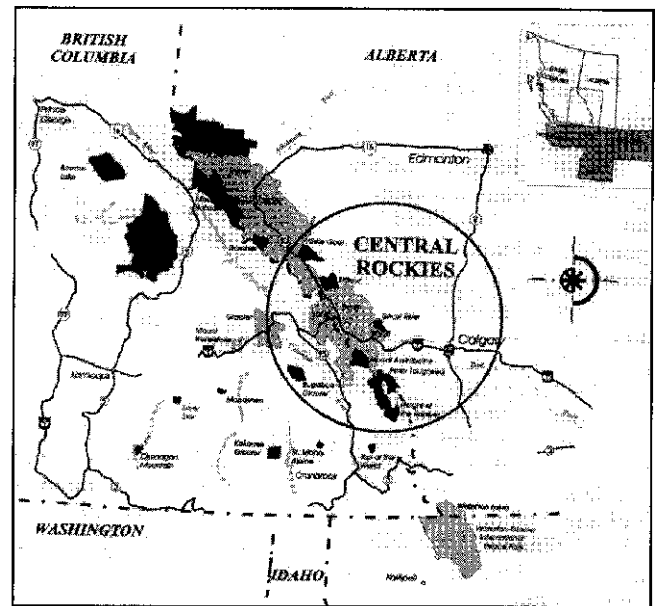


Figure 1—Central Rockies Ecosystem.

Table 1—Current thought related to the management of grizzly bears (Paquet, P., personal communication).

Issue	Conserving grizzly bears in human dominated landscapes	Human disturbance is the single largest threat to sustaining grizzly bear populations Developing innovative and cooperative strategies are key to improved conservation
Goal	Sustain the natural environment and meet human needs	Means reducing the potential for one seriously to encroach upon the other
Objective	Conserve free-ranging and self-sustaining grizzly bear populations	Implies conservation of all biological diversity and maintenance of ecological integrity
Problem	Ecological Social Management	How probability of persistence changes with habitat degradation, small population size, and population isolation What probability of persistence and environmental quality is compatible with economic goals, and acceptable to society How to achieve ecological and social objectives within constraints of legislation
Direction	How to progress toward sustainability	Require mechanisms to address pragmatic issues such as economic needs and conflicts that inevitably arise between humans and grizzly bears
Current regional problems	Conflicts	Spatial needs of grizzly bears and potential overlap with humans have generated social, political and environmental conflicts Heated political controversies, reduced public funding, and diminished management options Environmental concerns have been subsumed to commercial needs Human population pressures and associated land uses have supplanted large areas of natural habitat Conversion of extensive portions of habitat from optimal to unsuitable conditions Ongoing destruction of habitat is confining increasing numbers of grizzly bears into small and insular patches
Implications		Additional ecological impoverishment will occur because intensity of human activity is increasing

Case Study A: Regional Landscape

The landscape approach presents work being completed as part of a rewrite of the Park Management Plans for Yoho and Kootenay National Parks.

The Rocky Mountain National Parks have recognized that human use, which includes the direct physical use and the associated infrastructure and support services, is the single greatest stressor on the integrity of park ecosystems. In the absence of global economic collapse or major restrictions to international travel, historical growth trends in the tourism sector and the continued attractiveness of the Canadian Rockies as a tourism destination suggest that human use, if unmanaged, will continue to increase into the next millennium (Petersen 1999).

The need for integrated management is based on the fundamental principle when considering park zoning and visitor use in a management plan” (Parks Canada 1988). The basis for this management must be thorough consideration of both ecological and social objectives.

Yoho and Kootenay National Parks support and encourage sustainable human use and provide a range of visitor opportunities that enhance the opportunity to understand, appreciate, and respect the natural and cultural resources, while at the same time ensuring that the resource base is protected and allowed to function according to natural processes. Applicable strategic management goals include:

- To influence visitor expectations and manage human use aimed at enhancing the visitor experience, protecting ecological integrity, and supporting viable wildlife populations.
- To manage human use to ensure the ecosystem continues to support viable populations of carnivores (wolves and bears).

To develop facilities and services within a park that supports ecological integrity, it is critical that the objectives for human use and resource protection be coupled in a planning context. To accomplish this, Parks Canada has developed and implemented the following integrated planning approach for use in the development of their management plans.

The approach first attempts to visually present the relationship between the ecological and human use objectives from a landscape perspective. It then subdivides the park into smaller geographic planning units (landscape management units—LMU) in which actions to manage human use are proposed. These geographic units, originally defined as bear management units, are based generally on the home range of an adult female grizzly bear.

The second step of the planning process involves the completion of a situational analysis for each LMU. This task includes assessment of the existing issues related to a comprehensive listing of ecosystem issues developed for the park (wildlife, vegetation, aquatics, social, and cultural). This exercise revealed that the following ecologically based issues were the most important for an integrated planning approach: grizzly bear habitat effectiveness, wildlife

movement corridors, wildlife mortality, wildlife disturbance, and significant/rare habitats.

Carnivore Management Unit Habitat Effectiveness

Parks Canada has endorsed the application of the habitat effectiveness model as a tool for managing human use (Parks Canada 1997). Habitat effectiveness is a component of the Cumulative Effects Model (Gibeau 1998 as cited in Jalkotzy and others 1998, USDA Forest Service 1990, Weaver and others 1986) (figure 2). The analysis determines, for each of the units, the effectiveness of the habitat after human use impacts have been considered. For management purposes, a habitat effectiveness target has been defined and is used to guide future management of the type, nature, location, and intensity of human use.

Wildlife Movement Corridors

A fundamental requirement for maintaining viable populations of wide-ranging species is the opportunity for individuals and populations to interact and move throughout the landscape. These wildlife corridors are important for movements within the Parks, as well as for providing linkages to adjacent Provincial lands. Two areas in which aggressive action is required are:

- Pinch points—where corridors pass through a topographically constrained area in which there is a high level of human activity.
- Fracture zones—high use transportation corridors (Trans-Canada Highway, Canadian Pacific Railway) can block wildlife movements and must be mitigated to allow safe crossings for wildlife species.

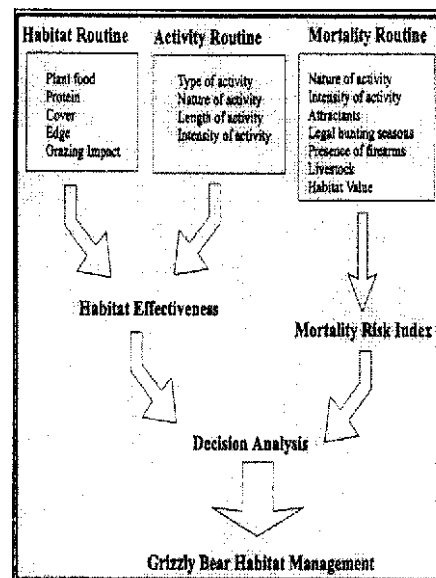


Figure 2—Cumulative effects assessment model.

Significant/Rare Habitats

These units are significant because they contain rare/endangered species or ecosystems, have limited geographical representation, or are critical to the life requirements of wildlife species.

Wildlife Mortality

Human-caused mortality has the potential to negatively impact the viability of wildlife populations (Benn 1998). This can be the result of direct mortality (highways) or indirect mortality from management actions taken in response to wildlife habituation to humans. Parks Canada has committed to reducing the number of grizzly bears killed as a result of human activity to less than 1% of the population annually (Parks Canada 1997).

Wildlife Disturbance

Increasing shoulder-season (fall/winter/spring) visitor use has the potential to disturb wildlife during critical and/or vulnerable stages of life cycle.

The visual representation of these issues and their objectives as GIS mapping layers provides a geographic sense of the constraints within which human use must be managed.

Social Context

The marketing position of Yoho and Kootenay has been expanded upon and now forms the basis for the social objectives of the planning units. Yoho and Kootenay are positioning themselves as a transition park, in which people can develop their skills and abilities to understand and participate in protected areas issues and related activities. The Parks will manage their internal and external visitor services to provide a range and progression of appropriate opportunities. To achieve this, all of the planning units have been rated according to the “experience level” that they are offering: from 1—where opportunities for trail activities and solitude are limited but all basic and essential services are provided—to 6—where solitude will be provided and infrastructure development will be minimal. It is envisioned that people can work their way through the opportunities in the Parks according to their existing skill levels or as their abilities advance. Levels of interpretation and infrastructure development should match the type of experience provided. Visitor surveys can then be used to detect whether people are being provided with the pre-trip information that directs them to appropriate areas and whether people are advancing their skills and understanding as they move through the various planning units within the Parks.

The units were rated against a series of social descriptors: visitor encounter expectations, motivation, degree of self sufficiency, level of infrastructure development, appropriate activities, trip duration, access, and substitution. These descriptors were selected from a more comprehensive list because they provided the best overview of the social conditions to be expected/provided in a unit. This step in the process is critical because it is through the acceptable match

between motivations/expectations and benefits that satisfaction is achieved.

Lessons Learned

What was learned during this integrated planning process was the following:

1. The management of summer human use became a focus largely in response to the availability of the grizzly bear habitat effectiveness model. There is a need to develop similar models for species (such as wolverine) that could provide direction for the management of winter human use.

2. The habitat effectiveness model is one of very few ecological models that provide clear direction regarding acceptable levels of human use. Unfortunately, within the model, the only significant use values are above or below a threshold of 100 users/events per month. This number is very restrictive and difficult to apply in areas with high current levels of human use (for example 1,000-10,000 users/events per month). However, for the management of critical grizzly bear habitat areas and for social objectives of wilderness/solitude, the model parameter of <100 users/events per month proved to be a useful planning tool. In these areas, it was easy to integrate social and ecological objectives.

3. Where grizzly bear habitat values are lower, and the realities of existing use would make it impossible to manage within the low use category (<100), the areas will be managed ecologically to minimize the potential for bear habituation and bear/human encounters, provide for movement corridors, minimize mortality, and provide access to critical habitats.

4. Habitat effectiveness model limitations include:

- The model does not accept habitat changes (such as artificial habitats created by ski hills).
- There needs to be additional research into the impacts on bears of various “disturbance event” management options.
- The model is useful to provide a feedback mechanism between management experiments and model results (that is to test changes in habitat effectiveness caused by implementation of management decisions).
- Ecological gains can be shown even when they are not reflected in the resulting habitat effectiveness values.

Case Study B: Landscape Management Unit

This case study will present work that is occurring within the Moraine Lake area of Banff National Park. It reflects work at a landscape management unit scale.

Moraine Lake is an important area that receives 500,000 to 600,000 visitors per year. With only one commercial overnight facility, the majority of the 8,000 visitors/day are there as day users. Many of the front and backcountry trails within the area have been subject to management closures during the period 1995 to 1998. These closures have been in response to the activities of both habituated resident and other migratory grizzly bears. Although the management actions were warranted and justified to ensure public safety

and the survival of the bears, there were resulting impacts to the visitor opportunities in the area. Therefore, Parks Canada determined in early 1999 that there was a need to define the issues and determine whether there were other ways to manage for ecological health and public safety while still maintaining public access to the visitor opportunities. A planning process was initiated and a working group, which included representation from internal and external interests (including environmental, business, natural resource, cultural, operational, and local communities), was assembled. Members of the working group prepared background papers that defined the issues from their perspectives. The background papers provided an opportunity for exploration of the issues and identification of linkages between the interest groups and enabled more comprehensive understanding of the current situation. The working group then identified short- and long-term actions to address the identified issues.

The issues coalesced around two central themes: "sense of arrival" and bear/human conflict. The former related to crowding of day use facilities during the peak season, congestion and conflicting patterns of use in the parking area, and impacts on staff/visitors/operators during closures of popular backcountry areas and trails. The latter issue related to the need to provide both safe visitor opportunities and for the ecological needs of the resident and transitory grizzly bears.

The working group identified numerous short-term actions to address the issues related to sense of arrival. The focus of the actions was on an effort to use enhanced communication to encourage a voluntary change in the behaviour of visitors to the area. The behavioural change was in the areas of transportation to/from the site and discontinuation of overflow parking. Communication products (brochures, signs, radio, Internet) will provide visitors with accurate expectations about the congestion, etc., that will be encountered if they plan to visit the Moraine Lake area during the peak periods. The communication also stresses the use of public transportation and car pooling as alternative access options. The goal of these initiatives is to reduce congestion through a voluntary change of public behaviour.

To manage bear/human conflict, the Park is attempting to pilot a new use management option that could be employed as a proactive measure to reduce the likelihood of an encounter. It is proposed that if bear/human conflicts are reduced, there will be less need for closures of trails, areas, and facilities. The new approach is a change to the existing bear management policy, which has only two options available — either a warning or a closure. A new "Restricted Access" option which requires that, in addition to regular enhanced bear safety precautions, hikers travel in a minimum group size of six and horse stock users in a minimum group size of two has been proposed for implementation between a warning and a closure. Mountain bike use is being restricted until scientific information is available to provide some direction for either an acceptable groups size or alternative risk management mitigative measures (such as continuous noise making device, trail sight lines, speed, etc.).

Although only a small number of grizzly bears may use the Moraine Lake area, current human use levels and patterns create considerable potential for it to become a mortality sink. If this were to occur, it would have serious implications for the grizzly bear population in a much broader area (Benn 1998).

The long-term issue for both the frontcountry and backcountry is that there is a need to define a use capacity target for the area based on ecological and social objectives. This will be a longer term planning issue that will be controversial and require considerable public input and consultation. In the short term, data gaps and required ecological (core security area and linkage zone analysis, habitat effectiveness, and bear risk assessment) and social (transportation, visitor satisfaction, and quality/nature of experience) information is being collected.

Lessons Learned

The short-term actions are being implemented for the 1999 visitor season. It is too early to determine how much enhanced communication convinced people to voluntarily change behaviour in the frontcountry. Similarly, any ecological gains achieved through the restricted access option will not be known until the strategy is evaluated in the fall of 1999. In the Moraine Lake case study, grizzly bear issues have provided direction for visitor access (type, timing, and amount), group size, and the management of risk related to bear/human conflicts.

For high-use areas such as the Moraine Lake study area, the grizzly bear habitat effectiveness model is of little utility. However, the species is still valuable in that, as illustrated, it can be used to assist with the development of more creative human use management options.

Case Study C: Area Planning

This example presents a completed project within the O'Hara Valley area of Yoho National Park (figure 3). A series of bear/human encounters in this area convinced park management to commission an independent bear hazard assessment of the area's trail network. In response to the report's recommendations for public safety and a mandated concern about general human impacts to the ecological requirements of a viable local and regional grizzly bear population, a number of indeterminant trail closures were effected. Park management was subsequently challenged, regarding both the science supporting the actions and the use of the closures themselves as a necessary and appropriate management response.

Consequently, a four year "Lake O'Hara socio-ecological research project" was undertaken. The collaborative project used ecological and social data in a computer-based decision support model to provide recommendations to park management on methods to resolve the land allocation issue between grizzly bears and humans.

The modelling components included: grizzly bear suitability (ecosite capability, habitat capability, and habitat linkage), bear encounter risk (noise, visibility, tread, use, rub trees, habitat suitability, and large mammal carcass), and human suitability (preference and use). To generate a final map layer for each of the models, principles of pairwise comparison, weighted valuation, and multi-criterion evaluation were used to analyse the data. The final maps were then overlaid and management recommendations based on the divergence and convergence of conflicting land use requirements.

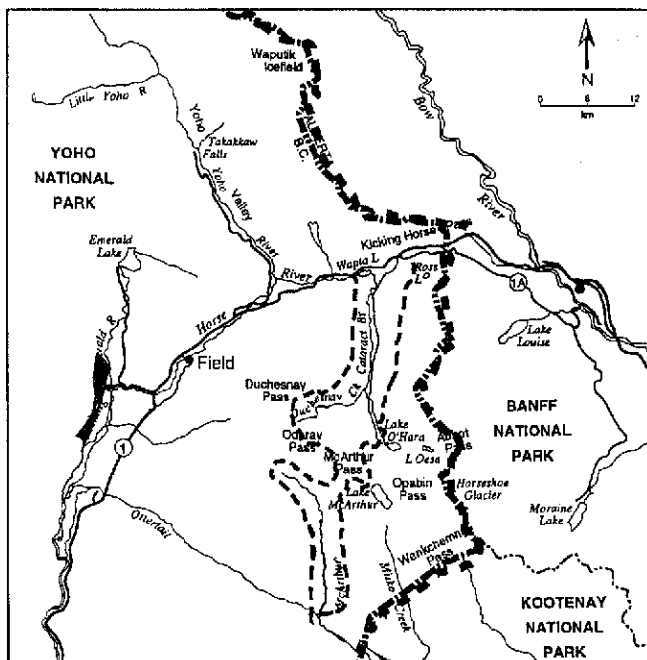


Figure 3—Geographical location of Lake O'Hara.

The specific objectives of each of the research components are detailed below.

Social (Wright and Kelly 1997)

The main objectives of the social component of the project were:

1. To learn more about the type of visitor experience hikers currently have in Lake O'Hara.
2. To identify the types of recreation features that are important to visitors.
3. To assess how visitors feel about trail closures and other management actions for managing bears and people in parks.
4. To examine the preferences and current patterns of use of trails in the Lake O'Hara area and how they may coincide with potential grizzly bear habitat.
5. To provide park management with recommendations to manage the Lake O'Hara area for the benefit of both humans and bears.

Through the integration of onsite visitor survey information, a trail level of use assessment and a visitor photography exercise, the project's central research questions were addressed.

Geographical Information System (GIS) (Donelon and Paquet 1998)

Two software packages were used to develop the GIS component of the decision support model. IDRISI for Windows Version 2.0 provided the primary software environment and was used for its spatially based decision support modeling modules, to develop raster layers used in the model and for graphic output, including a Digital Elevation Model, Slope Model, and Aspect Model.

MapInfo Professional Version 4.1 was the secondary software application and was used for digitization and spatial database queries and to create the graphic output for map files.

Ecological (McCrory and others 1999)

Data collected by the field researchers included:

1. Bear use/activity (hair sticks and direction lines, sand track pits, ground tracking, sightings, DNA hair collection, permanent bear habitat transects, camera/video installations, bear movement trails, access and egress points, and habitat use).
2. Vegetation (scat collection, scat decomposition rates, berry and pine nut phenology and productivity, vegetation transects, eco-site classification, and habitat microsites).

The methodology used for the Lake O'Hara study closely represents a "human dimensions" approach—efforts to make decisions that are more responsive to the public and that, in the long term, increase the effectiveness of decisionmaking (Knight and Gutzwiller 1995).

"People must have a shared understanding and be able to communicate clearly about the resources and issues in order to make decisions and reach agreement. Information systems that aid solving complex problems by augmenting the user's knowledge are called decision-support systems (DSS's). Supporting learning and communication are basic functions of a DSS" (O'Brien and others 1995). DSS will typically provide a set of tools that support the process of problem structuring, understanding the problem, producing alternative solutions, evaluating them, and facilitating group processes in decisionmaking (Gurariso and Werthner in O'Brien and others 1995).

Grizzly Bear Suitability Model—One of the first objectives of the decision support model (Donelon and Paquet 1998) was to develop a Grizzly Bear suitability model for the study area. Based on available information and data collected by researchers on the project, three separate criteria were identified for use in the development of the suitability model. These were ecosite capability, grizzly bear habitat classification, and habitat linkage (comprised of cost surface, slope, and distance to human features layers). Each of the suitability criteria were spatially mapped with a common classification scheme with classes of 1-10. The data collection and GIS analysis were also segregated into three temporal classes. Pre-berry season (to July), berry season (July and August) and post-berry season (September on). Figure 4 illustrates one of the bear habitat suitability model outputs.

A panel of grizzly bear experts were used to develop the final grizzly bear suitability model. The process involved the application of Satey's pairwise comparison matrix to develop linear weighted values of importance for each of the three criterion in the model (ecosite capability, habitat capability, and habitat linkage). These weighted values were then used to combine each of the criterion, through Multi Criterion Evaluation (MCE), to produce a final grizzly bear suitability map for the study area.

Bear Encounter Risk Model—The bear encounter risk model for the study area is comprised of seven criteria. These consist of three trail design features (noise, visibility, and

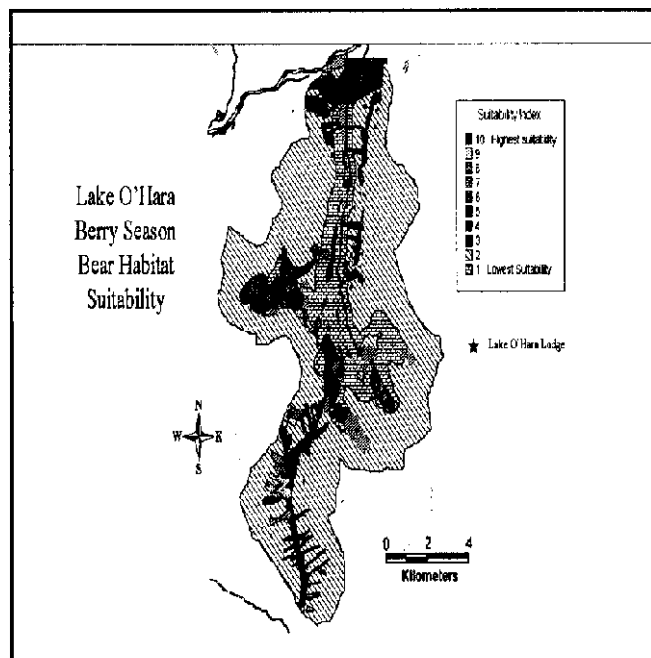


Figure 4—GIS model output of grizzly bear habitat suitability.

tread design), bear use, bear habitat suitability, availability of large mammal carcasses, and the occurrence of rub trees. The relationship between these criteria and risk management are contained within table 2.

Table 2—Encounter risk criteria.

Criteria	Comments
Noise	Amount of noise proximal to trail (affects potential for and severity of an encounter)
Visibility	Line of sight distance and amount and thickness of cover (affect likelihood of an encounter)
Tread design	Width and roughness of trail surface (less maintained trails require more attention when travelling upon and detract from ability to detect the presence of a bear)
Bear use	Index of use (bear movement occurrences) within 150 m buffer along each trail
Rub trees	Trail segments classified based upon the number of rub trees found within 150 m of the trail
Habitat suitability	Criterion from suitability model Boolean image isolating habitats within 100 m of trail segment Highest habitat value adjacent to trail segment selected for suitability value
Large mammal carcass	Likelihood of large mammal carcasses being present (proximity/availability of carcasses alters bear behaviour and affects nature and severity of potential encounter)

To create the final bear encounter risk map the bear experts again used Satey's pairwise comparison and MCE.

Human Preference—In the absence of a clear rationale for completing a pairwise comparison, it was determined that it was acceptable to develop the final human preference map by using equal weighted values for the two criteria.

Criteria	Origin	Comments
Use	Trail counters	Strong correlation between survey and trail counter use estimates
	Visitor survey	Counter data extended to trails without counters
Preference	Visitor survey	Illustrated preference relative to the respondents knowledge of the area

To conclude the decisionmaking process, a meeting was convened between Parks Canada management staff and the research project steering committee. The intent of the meeting was to expose managers to the process and content of the decision support system and to facilitate their discussions so that a final decision could be made about the management scenario to be implemented in the Lake O'Hara area. The format of the meeting was very informal to encourage questions and open discussion on the components to the model and the sensitivities and assumptions within it.

Lessons Learned

As a case study attempting to develop a new computer support model to aid in decisionmaking, it was a success. Although the cost of this test application was significant ($\pm \$150,000$), it was felt that when applied again considerable efficiencies could be achieved.

One of the advantages of the GIS environment to modeling and decisionmaking is that it is dynamic and thereby allows updates to the maps and background information at any time. This provided the opportunity to both segregate or combine model layers to better explain the results of the field research.

Park managers appreciated the visual format of the model and were able to come to a consensus decision on the management scenario that seemed to best meet the requirements of the mandate, while still providing for an acceptable level and quality of visitor experience.

Conclusion

It has been shown through the previous three case studies that grizzly bears can be used as a filter for managing human use. In the regional landscape case study, habitat effectiveness, movement corridors, significant habitats, disturbance, and mortality were all useful to define acceptable levels, types and timings of human activities. In the landscape management unit case study of Moraine Lake, the grizzly bear was used within a risk and ecological management framework and provided guidance for defining appropriate types of human use. The area planning case study of Lake O'Hara illustrated a fine-scale application of a decision support model that addressed the competing land uses

between grizzly bears and humans. Each application has built on the successes and corrected the failures of previous efforts.

Despite the accomplishments to date in taking a more integrated approach to planning and management, work will continue to more fully incorporate human dimensions research into the decisionmaking process.

Although human use management in the Rocky Mountain National Parks can be partially guided through application of grizzly bear related models and constraints, there is a fundamental question regarding an overall appropriate use threshold. Current research and monitoring within national parks is focused largely on the understanding of ecological systems and the assessment of ecological impacts resulting from existing levels of human use and development. Parks Canada will need to refocus its existing science program to begin to investigate the social and economic issues surrounding human use and the setting of capacity targets.

References

- Banff-Bow Valley Study. 1996. Banff-Bow Valley: At The Crossroads. Technical Report of the Banff-Bow Valley Task Force (Robert Page, Suzanne Bayley, J. Douglas Cook, Jeffrey E. Green, and J.R. Brent Ritchie). Prep. For the Honourable Sheila Copps, Minister of Canadian Heritage, Ottawa, ON.
- Benn, Bryon. 1998. Grizzly Bear Mortality in the Central Rockies Ecosystem, Canada. For MDP Committee, Faculty of Environmental Design, University of Calgary. Calgary, AB.
- Canadian Heritage. 1994. Guiding Principles and Operational Procedures. Minister of Supply and Services Canada, Ottawa, ONT.
- Donelon, S.; Paquet, Paul C. 1997. Development of a G.I.S. Based Decision Support Model for the Lake O'Hara Socio-Ecologic Study. For Parks Canada - Lake Louise, Yoho and Kootenay National Parks, BC.
- Guariso, G.; Werthner, H. 1989. Environmental Decision Support Systems'. In: O'Brien, Michael L. Visualization in Decision-Support Systems: the George Wright Forum 12(3):26-40, 1995.
- Jalkotzy, M.G.; Riddell, R.; Weirchowski, J. 1998. Grizzly Bears and Habitat Effectiveness in the Skoki, Baker, South Pipestone, and Lake Louise Bear Management Units, Banff National Park. For Parks Canada-Lake Louise, Yoho and Kootenay National Parks, BC.
- Knight, R.L.; Gutzwiller, K.J., 1995. Wildlife and Recreationists. Island Press.
- McCrory, W.; McTavish, C.; Paquet, P.C. 1999. Grizzly Bear Background Research Document. Unpublished. For Parks Canada-Lake Louise, Yoho and Kootenay National Parks, BC.
- O'Brien, Michael L.; Miller, Patrick A.; Skabelund, Lee R.; Phillips, Claudia G. 1995. Visualization in Decision-Support Systems: the George Wright Forum 12(3):26-40.
- Paquet, P. 1997. Personal communication. Conservation Biology Institute, Meacham, SK.
- Parks Canada. 1988. The National Parks Act. Minister of Supply & Services Canada, Ottawa, ON.
- Parks Canada. 1997. Banff National Park Management Plan. Parks Canada, Banff, AB.
- Petersen, Derek. 1999. Sustainability of Experiential Supply & Demand Flows in Protected Areas. Thesis Proposal, University of Calgary, Resources and the Environment.
- USDA Forest Service. 1990. CEM—A model for assessing effects on grizzly bears. USDA Forest Service, Missoula, MT.
- Weaver, J.L.; Escano, R.E.; Winn, D.S. 1986. A framework for assessing cumulative effects on grizzly bears. North American Wildlife and Natural Resource Conference. Vol. 52:364-376.
- Wright, P.; Kelly, D. 1997. Lake O'Hara Visitor Experience Study. For Parks Canada-Lake Louise, Yoho and Kootenay National Parks, BC. Unpublished. Centre for Tourism Policy and Research, Simon Fraser University, Vancouver, British Columbia.

The Development of the 1999 Management Plan for the Tasmanian Wilderness World Heritage Area (Australia)

Nicholas Sawyer

Abstract—This paper describes the multi-stage public consultation process and other aspects of the development by the Tasmanian Parks and Wildlife Service of the second (1999) management plan for the Tasmanian Wilderness World Heritage Area (Australia).

It describes the background to, and rationale for, the process used in developing the plan; it details the consultation process itself; and it critically examines the lessons learned in the course of developing the plan and considers how the effectiveness of such a process can be assessed.

Tasmania is one of the states of Australia. It is an island in the Southern Ocean, immediately south of mainland Australia. It has a cool, temperate, maritime climate, substantially different from most of mainland Australia.

Australia has 13 World Heritage Areas. The best known are Kakadu, Uluru (formerly known as Ayers Rock) and the Great Barrier Reef. The Tasmanian Wilderness is probably the best known of the rest. World Heritage probably has greater significance in Australia than in most other countries because land management is the responsibility of the states, and all Australian 'national parks' are actually proclaimed under state legislation. However, the World Heritage Convention is an international agreement, signed by the federal government. This gives the federal government a role, which it would not otherwise have, in the management of Australia's World Heritage Areas. Hence these areas, which are managed jointly by the state and federal governments, almost amount to a 'National' national park system. Management of the Tasmanian Wilderness is the responsibility of Tasmanian Parks and Wildlife Service, with limited oversight by the federal government. The area's World Heritage status also results in Tasmania receiving considerable federal funding for management of the area.

Tasmania is approximately 300 kilometres (200 miles) north to south and 300 kilometres east to west, and about 300 kilometres south of mainland Australia. Around 30% of the state's land is reserved under some category of conservation land tenure. The Tasmanian Wilderness covers

approximately 20% of the state. It comprises Tasmania's four largest national parks and several smaller areas of various other conservation land tenures.

Tasmania has a population just under 500,000. It has the weakest economy of all the Australian states, and tourism is seen as one of the few economic growth areas. Tasmania's tourism marketing promotes ecotourism based on the state's natural values; particularly those of the Tasmanian Wilderness. This puts considerable environmental pressure on the Tasmanian Wilderness even though most tourist accommodation is outside the boundaries and most tourism occurs at a few well-developed sites near the periphery of the area.

The Tasmanian Wilderness is an extensive, wet, temperate, wilderness area covering much of southern and western Tasmania. It is approximately 200 kilometres north to south and averages 70 kilometres east to west (120 by 40 miles). Although the highest point is only 1,600 metres (5,000 feet) above sea level and there is no year-round snow cover, much of the area is very rugged and contains the only extensive, recently glaciated areas in Australia. The last glaciation ended 10,000 to 12,000 years ago (Smith and Banks 1993).

The area was used for millennia by Aboriginals, who have left their signature on the area in the form of an ecology strongly influenced by their burning practices, as well as physical remains including middens and artwork. No Aboriginals now live permanently in the area, but some places are of great significance to the present-day Tasmanian Aboriginal Community.

Historically, the area was extensively explored and prospected during the 19th century, but the only economic activity in the area has been small-scale mining and logging, a limited amount of trapping (for furs) and, in a limited area, grazing, which continued until very recently. The area also contains one large and several smaller hydroelectric schemes. Apart from the hydroelectric impoundments, none of these activities have left much lasting trace. Hence there are extensive areas where there is little evidence of twentieth century 'civilisation'; wilderness by most definitions of the term.

The Tasmanian Wilderness contains no permanent human habitation, apart from a small amount of accommodation near the periphery. Few roads penetrate the area. The predominant use of the area is for recreation; it offers excellent opportunities for wilderness bushwalking (trips up to several weeks duration, on or off tracks). It is widely regarded as the 'Mecca' of Australian bushwalkers and has a growing international reputation. There is also a highly regarded trout fishery (introduced northern hemisphere species) in the Central Plateau lakes. Unlike most of the rest of the Tasmanian Wilderness, the Central Plateau section has a long history of use by local people. As well as fishing, some

In: Cole, David N.; McCool, Stephen F.; Borrie, William T.; O'Loughlin, Jennifer, comps. 2000. Wilderness science in a time of change conference—Volume 5: Wilderness ecosystems, threats, and management; 1999 May 23–27; Missoula, MT. Proceedings RMRS-P-15-VOL-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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hunting, horse riding, four-wheel-driving and associated hut use continues. These 'established' practices are seen by some groups to be at odds with achieving conservation outcomes.

The area was placed on the World Heritage List in two stages, in 1982 and 1989. The 1982 listing came in the midst of a political furore over the proposed construction of a major hydroelectric scheme (the 'Franklin Dam') within the area. Construction of the scheme did not proceed as a result of federal government intervention using authority obtained as a result of the World Heritage listing. The area was expanded in 1989 as a result of a decision to protect a major area of tall eucalypt forest from logging (the 'Helsham Inquiry'). Again, the area's World Heritage status gave the federal government the right to be involved, and reinforced the perception in some sections of the Tasmanian community that World Heritage listing was a ploy to give the federal government the right to intervene in land management issues which would otherwise be a matter for the state government alone.

There was also serious distrust of the Parks and Wildlife Service in some quarters, mostly dating back to when the Central Plateau was added to the Tasmanian Wilderness in 1989. Many established practitioners had been led to believe (not by the Parks and Wildlife Service) that all activities that had previously been permitted within the area would be allowed to continue after World Heritage listing. Soon after listing, some of their more environmentally unacceptable activities were restricted or banned to reflect the new status of the area (for example, several four-wheel-drive tracks into sensitive areas were closed).

This history resulted in a polarisation of strongly held views in the Tasmanian community on the future management of the area and, in some quarters, considerable antagonism towards the Parks and Wildlife Service. This legacy of ill-feeling was one of the obstacles to be overcome by the Parks and Wildlife Service in preparing plans for the area.

In 1990, planning for the area was still poorly coordinated. Only one of the four major national parks had a finalised management plan and, although plans were in varying stages of completion for several other parts of the Tasmanian Wilderness, the decision was made to prepare a single management plan for the entire area. Several stages of public comment, accompanied at times by considerable controversy in the local media, led to a very 'pro-wilderness' draft management plan. A series of last-minute alterations to the plan, following a change of state government and after the closure of public comment, diluted the 'pro-wilderness' nature of the plan and thereby antagonised the conservation lobby, but defused many of the strongly felt objections of 'established' users, some of whom had threatened civil disobedience in relation to some plan prescriptions. However, some of these stakeholders, particularly local communities adjacent to the area, felt that their input to the planning process had been ignored and remained fundamentally dissatisfied with aspects of the plan, which was finalised in September 1992.

Some aspects of the 1992 plan met with poor acceptance from 'established practitioners' from the start, and some other problems (such as the absence of a mechanism to assess new development proposals) became apparent as the plan was implemented. Nevertheless, it guided management of the area for the next seven years, two years longer

than its intended life. The Parks and Wildlife Service was determined to overcome a number of the ongoing issues from the 1992 plan so, in 1994, the decision was made to review the plan with the aim of having the new plan in place by September 1997. This deadline was not met for a variety of reasons, including state and federal elections that delayed key approval processes. The new plan took effect in March 1999.

The most controversial management issues dealt with in the development of the new plan were those related to tourism, 'established' practices and fire management; the key nature conservation question being whether land managers should actively use fire to maintain the diversity of the ecosystem.

Development of the new plan was the responsibility of the World Heritage Area Planning Team within the Policy and Planning Section of the Parks and Wildlife Service. The team was responsible for most of the policy development and drafting of the plan in consultation with various Parks and Wildlife Service specialists and field staff. Given the importance of the tourism industry to the Tasmanian economy and a push by successive Tasmanian governments to take a 'whole of government' approach to the promotion of tourism, we made a special effort to involve Tourism Tasmania (the state government tourism promotion agency) and the Tourism Council of Australia (the main industry lobby group) in the development of the plan. As well, there were discussions with all other relevant State Government agencies, who also got to comment on an early (pre-public-release) draft of the plan.

The decision to attempt a multistage public consultation process was made on pragmatic grounds. The planning team was very aware of the poor acceptance of some aspects of the 1992 Plan (as described above) and also of the major problems encountered with planning for two other Australian World Heritage sites. At Willandra Lakes a final draft plan of management was completed after ten years, but never adopted, due to a 'failure to account for local concerns' and a failure 'to engender, amongst local stakeholders, a sense of ownership for its strategies' (Corbett and Lane, 1997). In the case of the Wet Tropics the release of the management plan was delayed for several years, due largely to the failure of the planners to adopt a collaborative approach towards several key stakeholders both inside and outside of government (Lane, 1997).

The Tasmanian planning team was also aware of the general trend towards a transactive approach in both urban and natural area planning. They recognised that the public involvement in the development of the 1992 plan had not succeeded in gaining the support of some key stakeholders, despite having been done with the best intentions of consulting with and educating the public, and being a major advance over any similar process previously conducted by the Tasmanian Parks and Wildlife Service. They concluded that the only way to gain broad public acceptance of the new plan was to move beyond public consultation as an adjunct to rational planning to engage the critical stakeholder groups and create a consensus; the approach advocated by McCool and Stankey (1986).

Since this was the second plan for the area, the planners already had a very good idea of the key issues and stakeholders so they tailored the planning process to suit their

particular circumstances rather than follow the steps prescribed by any particular planning theory. For example, the Recreation Opportunity Spectrum 'has all too often been applied as a recipe rather than a set of principles' (Hamilton-Smith, 1999).

This attempt to gain the involvement and support of all stakeholder groups by means of the multistage public consultation process is described below.

The Community Consultation Process

There were three formal stages of public consultation during the preparation of the 1999 Tasmanian Wilderness World Heritage Area Management Plan. Simultaneously, but independent of this broad public consultation and with more restricted public input, two projects looked at Aboriginal management of the Tasmanian Wilderness and non-Aboriginal established practices in the Tasmanian Wilderness. While all this was happening, there were numerous public meetings and meetings with interested groups, and the issues were considered in detail by the World Heritage Area Consultative Committee, the main stakeholder advisory group for the World Heritage Area.

Stage 1—Issues Stocktake

The 'Issues Stocktake' was a 'blank sheet' approach: Respondents were asked to tell us what they considered to be the issues, and how they would like to see them managed. Copies were sent to everyone on an extensive mailing list of people who had made submissions on the previous management plan or previously contacted us on World Heritage matters. In addition, the process was widely advertised.

The 'blank sheet' approach was adopted in the hope of ensuring that all issues were raised at the earliest possible stage of the plan review, to get a broad range of stakeholders involved at an early stage and to avoid accusations of 'leading' public comment, which had been made during the development of the 1992 management plan. The Issues Stocktake successfully achieved all of these objectives but the analysis of the unstructured responses was very time consuming, especially as many respondents ignored the instructions which were intended to give some consistent structure to their submissions.

Responses were received from all of the main interest groups (and many individuals) who had shown an interest in management of the Tasmanian Wilderness in the past. There were no surprises among the issues raised, but some changes in the strength of feeling on particular issues were evident when compared to the consultation on the 1992 plan five years previously.

Stage 2—Issues and Options

This stage of consultation was designed to obtain a more detailed and informed response on a narrower range of issues than the Issues Stocktake. A series of 'Issues and Options' papers were written (most were two or three pages in length) to give background information on ten topics. The subjects were selected from the topics which had aroused the

most interest in the Issues Stocktake, but with the condition that they were matters for which public feedback could be useful and influence final policy. Every effort was made to present a balanced view. The Issues and Options kit included a set of questions specific to each paper.

The topics were:

1. Management Objectives and Zoning
2. Fire Management
3. Visitor Facilities and Tourism Development
4. Central Plateau Conservation Area Issues
5. Walking Tracks
6. Fishing
7. Recreational Vehicle Use
8. Hunting
9. Horse Riding
10. Aircraft Overflights

The analysis of this data was much simpler than the analysis of the Issues Stocktake because the respondents answered specific questions. We also had an ulterior motive in this stage of the consultation; several of the papers were published as much for their educational role as for the usefulness of the feedback from them.

The Issues and Options process gave a useful insight into the range of opinions on these issues, who held them, and the strength with which they were held (refer to further discussion under 'Analysis of Results'). Little of the information was new or unexpected, but it served a very useful role in confirming the policy directions to be taken in the new management plan.

Stage 3—Formal Public Comment on the Draft Management Plan

The formal public comment period was double the minimum required by law (one month) to give the public every opportunity to comment.

Only one minor new issue came to our attention, and the comment period was uneventful. This was a great improvement on the 1992 plan, which was wracked with controversy at this stage.

The range of comments generally reflected those already received in previous stages of consultation, so only minor changes were made as a result.

Feedback to Contributors

The provision of feedback was seen as essential if participants were to know that their opinions were being taken seriously and that the consultation was not just 'window-dressing.' Publishing a summary of the public comment at each stage also filled the valuable educational role of making the public aware of the range of views which the Parks and Wildlife Service had to reconcile (this point is discussed further under 'Broader Issues ... Set the Context' below). The summary of the previous stage of consultation was mailed out to all contributors at the start of the following stage of the process and the final summary of comments on the draft plan was mailed out at the time of the launch of the final plan.

Volume and Continuity of Comment Across the Three Formal Stages of Consultation

A total of 1,062 individuals or groups made one or more submissions at one or more of the three stages, an impressive total for a small state (few submissions came from outside Tasmania) and one which illustrates the level of public interest in management of the Tasmanian Wilderness.

The Issues and Options papers attracted the greatest response (578 submissions), followed by the draft plan (390) and the Issues Stocktake (300). The popularity of the Issues and Options probably reflects their content; they asked respondents specific questions about their area(s) of interest, a less challenging task than defining the issues (Issues Stocktake) or critiquing the draft management plan. The relatively low response to the draft management plan hopefully reflected a general level of satisfaction with it, but one possible cause was 'respondent fatigue;' many government processes were calling for public comment on a range of issues during the same time frame, and it is usually the same few individuals who get involved in many of these. Other possible causes were a feeling of having already commented, via the earlier stages of the plan review, or a perception that little was likely to change, regardless of what comment was made at this stage. Another possible factor was the timing of the release of the draft plan. It was launched in mid-November, with comment closing in late January, so the end of the comment period coincided with the main Christmas-summer holiday season in Australia.

The three formal stages of consultation were designed on the assumption that it would be basically the same audience responding to each stage of the process, and their comments would be informed by the feedback from the previous stage. However, only 2% of total respondents made submissions to all three stages, and only 16% of respondents commenting on the draft plan had been involved in either of the previous stages. Still, the majority of submissions to the final plan appeared reasonably well-informed on the issues, suggesting that respondents had been exposed to some relevant information source such as public meetings or newsletters from organisations or clubs.

Aboriginal Consultancy

This was the main form of consultation with the Aboriginal community and led to a negotiated partnership between the Aboriginal community and the Parks and Wildlife Service to manage the Aboriginal values of the area. The work was done by a consultant from the Tasmanian Aboriginal Land Council.

Established Practices Consultancy

Established Practices refers to long established-activities, primarily by people living adjacent to the area, such as horse riding, four-wheel-driving, hunting and maintenance of privately constructed huts. This consultancy was done by a social anthropologist from outside the Parks and Wildlife Service.

It resulted in considerable concessions for these activities, compared to the 1992 management plan, permitting them to

continue where they did not threaten the values of the area. This has reversed the attitudes of many of the local communities around the area from being vocal critics to actively supporting the new plan.

Tasmanian Wilderness World Heritage Area Consultative Committee

The Consultative Committee meets quarterly, with each meeting lasting several days. Meetings usually include a field trip to inspect some environmental issue in the Tasmanian Wilderness.

It is a combined scientific and community committee, comprised of 14 influential members of the Australian and Tasmanian community who represent a very broad range of scientific and general interests. Half are nominated by the state government and half by the federal government. They spent a total of 20 days debating the new plan, reaching consensus on almost all issues.

The committee is an extremely useful sounding board for the Parks and Wildlife Service and an invaluable mechanism for getting information back to the stakeholder groups that its members represent. They are also influential; when such a broadly representative group, containing such a range of experience and expertise, reaches an informed consensus, it is very hard for either the Parks and Wildlife Service or politicians to ignore them.

When the final plan was released, the editorial writers in the two major Tasmanian newspapers rang several members of the Consultative Committee. When they got generally similar comments and support for the plan from all members, the writers concluded that it must be broadly supported, thereby ensuring a positive and low-key coverage in the media.

Public Meetings

As well as several formal public meetings, we undertook to meet with every group, however small, that requested a meeting. This stemmed partly from a general determination to be as open as possible in the consultation, and partly from the acknowledgement that consultation based on written submissions discriminates against the less well-educated, and it was particularly important to include the country people who live adjacent to the Tasmanian Wilderness.

These meetings generally served to confirm the feedback received in other aspects of the consultation process and also helped to break down the 'faceless bureaucrat' stereotype.

The Need to Involve All Stakeholders

There is a need to actively ensure that all major stakeholders are involved. Local communities and the Aboriginal community are key stakeholders whose involvement is essential for a successful outcome, yet they are reluctant to participate or likely to be overlooked in a broad consultation process and need to be contacted directly. Conservationists and bushwalkers are generally well-educated and well-organised; they can readily make their point in an 'academic' written consultation process. In contrast, local and Aboriginal communities are among the sections of the

population least likely to respond to such an approach and specific initiatives, such as the meetings and consultancies described above, need to be made to get them involved.

Types of Media

As described above, the various stages of consultation mainly relied on verbal communication and the distribution of printed material. Little use was made of the electronic mass media or newspapers, except for a few items publicising the consultation process.

The Issues and Options papers were made available on the Internet as well as on paper; provision was made for people to respond by completing on-line forms, but only 11 responses were received electronically (August 1996). Since that time, the number of potential respondents with Internet access has increased significantly, and the effort required to produce attractive and effective Web pages has decreased dramatically. We were probably too close to the cutting edge of new technology at this time and perhaps failed to adequately advertise this opportunity.

Both the draft and final management plans have been made available on the Parks and Wildlife Service Web Site (www.parks.tas.gov.au/wha/whahome.html), along with a lot of other background and planning related information. Over 1,600 downloads of the draft plan took place. The on-line availability of the plan (in Adobe Acrobat™ format) obviously enhances information interchange around the world and enables anyone to search the entire document for a particular text string, which can be very useful to ensure that you are aware of all references to a particular issue. However, the file size of the entire plan, including maps, is over five megabytes, so its usefulness for some users is limited by the time required to download such a large file.

What Could Have Been Done Better?

Consistency and Coordination Within the Parks and Wildlife Service—A separate document requiring public comment, a discussion paper on permits for overnight walking within the Tasmanian Wilderness, was released almost simultaneously with the Issues and Options papers from a different Branch of the Parks and Wildlife Service. This put an additional burden on the many people who wished to respond to both, and some claimed that there were inconsistencies between the two documents.

Overestimating Our Ability to Deliver With Limited Resources—This led to an inability to follow through on some commitments; most notably an open day which was planned to discuss Central Plateau management issues on site. It did eventually take place, but on a much smaller scale than originally planned. At a different order of magnitude, it was realised quite early in the plan development process that it would not be possible to assess the Recreation and Tourism potential of the World Heritage Area at the desired level of detail without seriously delaying completion of the plan. This led to a decision to treat parts of this key issue at a relatively general level in the plan while committing the Parks and Wildlife Service to completing a detailed Recreation and Tourism Strategy within 12 months of finalisation of the plan itself. This avoided a major delay to the plan, but

at the cost of devolving many significant decisions to a subsidiary document, the Recreation and Tourism Strategy.

Broader Issues Relating to Public Consultation

Set the Context (Both the Legal Framework and the Range of Stakeholders' Views)

A recurring criticism during consultation is: 'I have already told you what I wanted; why haven't you done it?' There are usually two main reasons 'why we have not done it.'

We Are Not Allowed to Do It—There is a clear need to set the context for what is possible in the plan, to explain to stakeholders the legal and policy constraints on the Parks and Wildlife Service, that there are some matters that are beyond the scope of the planning process, however important they may be (for example, the boundaries of the area in question). In addition, there are some policies that we cannot change, regardless of what they tell us (for example, legal requirements) and some where we are unlikely to be able to change (for example, where we are clearly directed by government policy). It is also useful if stakeholders recognise that there are some issues where we are unlikely to be prepared to change (for example, where the activity in question is demonstrably causing significant environmental damage).

We Cannot Please Everyone—If one group of stakeholders says 'yes' and another group says 'no,' it is obviously impossible to satisfy everyone. As discussed above, the Issues and Options papers served a valuable educational role in making stakeholders aware of the range of views put to the Parks and Wildlife Service and making stakeholders realise that the best possible outcome for their group did not necessarily equate with getting everything that they wanted. Getting all groups together in one forum can also be a really useful mechanism to make them aware that we have to manage for all users, not just them. For example, one public meeting early in the process was attended by both hunters and conservationists. At the start of the meeting, the hunters were criticising us for not giving them more concessions. By the end of the meeting, they had realised how passionately the conservationists opposed hunting and were thanking the Parks and Wildlife Service for its support of any continued hunting in the Tasmanian Wilderness.

Analysis of Comment

Analyses of public consultation must acknowledge the limitations of the data on which they are based. In particular, submissions do give a good indication of the range of views among members of particular groups, but they do not represent public opinion, and they do not give much indication of the level of support for particular proposals.

Range of Views—The submissions do give a good indication of the range of views present among those members of the public who are really interested in the management of the Tasmanian Wilderness, and the range of views present

among members of particular groups, such as fishermen or bushwalkers.

Public Opinion—Over 1,000 public submissions do not constitute a statistically valid public opinion poll because the respondents are self-selected. The respondents have a far greater interest in, and also greater knowledge of, the Tasmanian Wilderness than the ‘average’ member of the Tasmanian public. If you want public opinion, you need to run a properly conducted public opinion poll on randomly selected members of the public, with the questions set at an appropriate level of detail. The Parks and Wildlife Service is also well aware that there are some substantial user groups, such as the tourists who form the majority of visitors, who are hardly represented at all in these submissions. These tourists would also be missed in a public opinion poll because they are mostly from interstate or overseas, so it is necessary to use an entirely different process, such as a visitor survey, to gauge their opinions.

Number of Submissions (1)—The number of submissions for or against an issue does not give a reliable indication of the level of support for a particular proposal, either in the general public or in particular user groups, because the respondents are self-selected. People have a complex range of reasons for choosing to get involved in a public consultation process; there is no justification for assuming that the relatively small numbers who make submissions are a representative sample of any larger group. For example, it is likely that the actual number of submissions reflects the enthusiasm with which the leaders of the various lobby groups encourage their supporters to get involved in the process, especially when groups attempt to ‘stack’ the process by encouraging their supporters to complete large numbers of ‘proforma’ responses. In the report on the first stage of public consultation on the 1992 management plan, the Parks and Wildlife Service implied undue significance to the number of submissions by reporting the numbers for and against each suggestion. This resulted in some lobby groups treating the second stage of consultation as a ‘pseudo petition,’ (Rando, 1992), with the main aim being to get as many signatures as possible, a futile exercise which did nothing to enhance understanding of the issues or the credibility of the consultation process.

Number of Submissions (2)—Another reason for not attaching much significance to the number of submissions received for or against an issue is that some come from individuals, while others come from a wide range of groups and organisations. A simple count would imply equal weight to submissions from a private individual (who may or may not have real knowledge/interest in the issue), an organisation or club (which may represent a very small or a very large membership), a commercial operator, an industry body or another government agency.

Number of Submissions (3)—There is also the question of logic and supporting information in submissions; one well argued submission for a particular point of view should count for more than any number of unsupported statements opposing it!

Estimating the Degree of Polarization on Particular Issues—Many of the questions in the Issues and Options Comment Guide included Likert Scale (tick box) questions on a scale of ‘strongly approve,’ ‘approve,’ ‘neutral,’ ‘disapprove,’ ‘strongly disapprove.’ The manner in which these data were used can best be demonstrated by a simple fictitious example:

- Suppose that a proposition ‘scored’ 100 ‘strongly approve’ (all from established practitioners) and 200 ‘strongly disapprove’ (all from conservationists).
- If these were simply added together, it would give a clear majority for ‘strongly disapprove,’ but this would only prove that more conservationists completed the question than did established practitioners — this adds nothing to our understanding of opinions on the proposition.
- However, two useful conclusions can be drawn from these data:
 1. The proposition was strongly approved by established practitioners and strongly disapproved by conservationists, and;
 2. There was strong polarization in the WHA stakeholder community on this issue because all opinions were ‘strongly ...’; there were no ‘approve,’ ‘neutral’ or ‘disapprove.’

This gauging of the strength of feeling on particular issues is very useful for identifying ‘hot’ issues which may deserve further attention and for briefing senior decision-makers about where they can expect significant criticism or support.

Cost Effectiveness

This was the longest, most expensive consultation process ever undertaken by the Tasmanian Parks and Wildlife Service. Was it worth it?

Throughout the process, we were concerned that we were possibly spending a lot of money confirming the obvious, gathering a lot of information which we knew anyway or which could have been gathered from a less inclusive, less expensive process. There is an element of truth in this, in that only a minority of policies and prescriptions in the new plan changed as a result of this feedback. However, the formal consultation often served to confirm information that we had gleaned from other sources; the picture that you get from talking to several small groups and individuals can be very biased, but if it is confirmed from other source(s), such as several hundred written submissions, you can have a lot more faith in it.

The whole consultation process also had a major, but unquantifiable role in ‘selling’ the final plan. To be accepted, all planning decisions need to be transparent and accountable; stakeholders and the public need to understand how and why decisions were made, and the exposure received by most policies during the consultation process added greatly to their credibility.

There is also the need to not just consult, but to be seen to consult. After multiple stages of public consultation, far in excess of the minimum legal requirement, nobody could deny that consultation had occurred, and few could argue that it had not been done sincerely.

What Are the Criteria for Gauging the Success of Community Consultation?

Is this just an absence of controversy over the final plan or is it more than that?

The political reality is that the plan ultimately had to be approved by our state and federal ministers, and they would not endorse it without it being supported by the key stakeholders, particularly the tourist industry, the Aboriginal community and the local communities around the area. So we had to achieve consensus with all key groups. At the same time, it still had to be an effective plan; we did not want a 'lowest common denominator' plan, which everyone could agree with because it hardly said anything. This required something much better than a 'lowest common denominator' standard of community consultation. The reception of the final 1999 plan, which is supported by all major stakeholders, suggests that we achieved this.

But is this because we did it so much better than in 1992 or just because times have changed?

We Did Do It Better This Time—The consultation process was more inclusive and extensive, with some groups it included genuine negotiation rather than just consultation, and some participants could see changes as a result of their submissions. During this lengthy period of consultation, other Parks and Wildlife Service initiatives, including the way in which we implemented the 1992 Plan, also helped to restore the faith of the community in the Tasmanian Wilderness and its management. In hindsight, having so many stages of consultation was probably overkill, but the end result has been that many people who normally oppose any conservation measures actively support the new plan.

But We Also Need to Acknowledge That the World Has Changed—In 1992, the expanded Tasmanian Wilderness World Heritage Area was still new and was either wonderful or threatening, depending on your point of view; now most stakeholders have accepted the new status quo. Also the broader political landscape has changed; it is less polarized. Few people still see World Heritage as a threat, and even our critics recognize the Tasmanian Wilderness as a major drawcard for the tourism industry. The 1992 plan has been accused of treating wilderness as the overriding value of the World Heritage Area to the exclusion of all other values. The 1999 plan is more acceptable to more people because it recognizes not only the wilderness, but also the Aboriginal, established practices and tourism values of the Tasmanian Wilderness. As such, it reflects changes in community attitudes and government policy during the intervening period.

Conclusions

Acceptance by stakeholders is a crucial aspect of making a plan work, but ultimately a plan is only a means to an end;

the ultimate rationale for undertaking a planning process is not to produce a plan but to produce on-ground outcomes that enhance the management of the area. The overall objective of the 1999 Tasmanian Wilderness World Heritage Area Management Plan is 'to identify, protect, conserve, present and, where appropriate, rehabilitate the world heritage and other natural and cultural values of the World Heritage Area, and to transmit that heritage to future generations in as good or better condition than at present.'

The process of developing the plan has already made some progress towards this objective. It has resolved a number of troublesome issues and enhanced all stakeholders' understanding of many other issues, but ultimately the success of the plan and all the effort that went into its development can only really be judged by the state of the Tasmanian Wilderness at the end of the plan's lifetime, and the plan specifies how we intend to assess that.

Acknowledgments

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References

- Corbett, A. and Lane, M. 1997. The Willandra Lakes Region World Heritage Property: A Planning Phoenix? *Environmental and Planning Law Journal*, volume 14, number 6, December 1997.
- Hamilton-Smith, E. 1999. A Holistic Approach to Maintaining Quality in Environmental Management and Visitor Experience, in Kyle, G. (Ed.), in press, *Proceedings of the 1999 Northeastern Recreation Research Symposium*. USDA Forest Service Northeastern Research Station.
- Lane, M. 1997. The Importance of Planning Context: The Wet Tropics Case. *Environmental and Planning Law Journal*, volume 14, number 5, October 1997.
- McCool, Stephen F. and Stankey, George H. 1986. Planning and Social Change: Responding to the Revolution in Recreation Demand. In: *Proceedings of the 18th IUFRO world congress division 6*; 1986 September 7-21; Ljubljana, Yugoslavia. Vienna, Austria, IUFRO World Congress Organising Committee.
- Rando, S. 1992. Lessons from a Large Community Participation Program. Unpublished internal report, Department of Parks, Wildlife and Heritage, Tasmania.
- Smith, S.J. and Banks, M.R. (editors) 1993. *Tasmanian Wilderness – World Heritage Values*. Hobart, Tasmania. Royal Society of Tasmania.

Wilderness Management Planning in an Alaskan National Park: Last Chance to Do It Right?

Michael J. Tranel

Abstract—Like many wilderness areas, Denali National Park and Preserve faces a variety of challenges in its wilderness management planning. As an Alaska conservation unit that has been significantly expanded by the Alaska National Interest Lands Conservation Act of 1980 (ANILCA), Denali faces the additional responsibility of acknowledging that its management of controversial issues affects how other wilderness areas are managed throughout the state. Advocates of managing Denali as wilderness in its purest sense encourage the park to see its wilderness management planning as the “last chance to do it right.” Other individuals and organizations advocate activities such as continued motorized uses in Denali wilderness. As a result, Denali’s backcountry management plan addresses such issues as aircraft overflights and landings, snowmachine use, other motorized uses, and commercial and recreational uses. Wilderness management planning in Denali requires proper interpretation of ANILCA and accurate definition of types and levels of use. Success requires working with the public to develop innovative approaches to allocating uses, minimizing conflicting uses, and protecting remote yet accessible backcountry resources.

The National Park Service initiated a backcountry management plan for Denali National Park and Preserve in 1998, gathering information on levels and types of use in the backcountry and on the legal parameters for planning. Based on this initial data collection, the agency determined that additional scientific information is essential to the planning effort, as is the need to deal with potential threats and continue with studies and monitoring. Public understanding of planning constraints determined by laws, regulations, and policies is also needed. Questions for the planning process include:

1. What are the legal parameters for planning, and what range of management options should be considered?
2. What are the most appropriate and effective methods for public involvement?
3. How does the plan proceed if scientific information is limited?

Establishing the legal parameters for planning sets the context for discussing potential alternatives and management options in the public arena and helps prevent legal

challenges. Identifying the highest priority data needs and addressing these first can be a strategy to overcome limitations in previous studies. Since Denali National Park receives considerable attention from the public in Alaska and in the context of environmental issues nationwide, public involvement strategy is crucial.

Educating, involving and enlisting the support of the public is essential to successful backcountry management planning for the Park. Protected areas in Alaska have been viewed as a “last chance to do it right” by environmental organizations. However, other groups view the large protected areas in the state that are relatively new to the landscape, at least in terms of political boundaries, as viable opportunities for continued resource extraction and expanded tourism. The resulting controversy affects the planning process for the wilderness and backcountry of Denali.

Background

Denali National Park and Preserve is located in south central interior Alaska and includes over six million acres, of which approximately two million are designated wilderness. (See location map.) The Park is slightly larger than the state of New Hampshire. Development is limited to visitor facilities, maintenance and administrative support facilities and an employee housing complex near the entrance area of the Park at mile 237 of the George Parks Highway. The Parks Highway connects Anchorage and Fairbanks, Alaska’s two largest cities. Additional visitor facilities exist at several locations along the 90-mile Denali National Park Road that extends from the Park entrance to Wonder Lake and the former mining community of Kantishna. Lodges and a campground are located in the Kantishna and Wonder Lake area near the end of the park road. Automobile traffic on the park road is restricted beyond the Savage River at mile 14.8. The primary access into the Park’s interior is on a tour bus, visitor transportation shuttle bus system, or by bus to a Kantishna area lodge. This controlled access system has been in place since 1972 after the George Parks Highway was completed. Controlled access is a significant factor in protecting resources, especially wildlife, and the visitor experience in Denali.

Denali National Park and Preserve is an internationally significant protected area that has been proclaimed a biosphere reserve under the United Nations Man and the Biosphere program. Wilderness is a fundamental value identified with Denali at its establishment, and this value has been reaffirmed throughout the administrative history of the Park. The philosophy and policies for managing the wilderness and backcountry areas of the Park are intertwined with and have constantly influenced the

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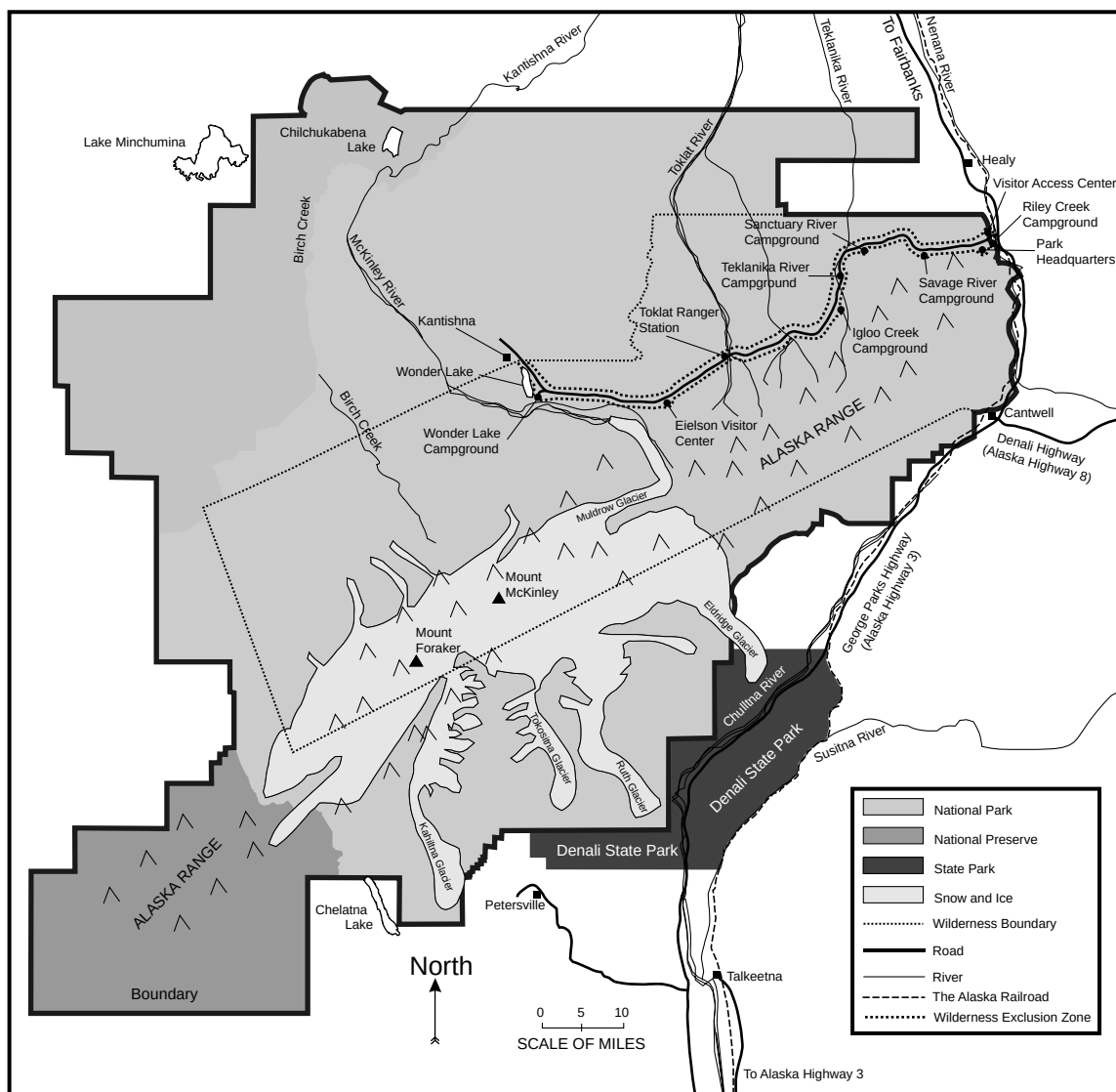
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management of the more developed and heavily visited regions of the Park. Denali still exemplifies the intent of the 1964 Wilderness Act and provides an opportunity for the public to experience wilderness values.

The Park (fig. 1) contains large areas where trails and evidence of human use are minimal to nonexistent. Approximately one-third of the Park is designated wilderness. Of the other four million acres, most is proposed for wilderness

designation, and almost all of it is suitable. National Park Service policy mandates that it be managed as designated wilderness.

The purposes of Denali are specified in the enabling legislation for the original Mount McKinley National Park and in ANILCA. The Park's purpose is also tied to the traditions of the other national parks and preserves added to the system through ANILCA. Denali includes several administrative



PARK/REGION
Denali National Park and Preserve
 U.S. Department of the Interior • National Park Service

Figure 1—Denali National Park and Preserve.

subsets with different legislative histories and legal mandates (original national park, national park additions, national preserve and designated and proposed wilderness). It is a place where special uses related to subsistence and a frontier-type way of life continue, subject to regulation to ensure they do not jeopardize the integrity of park resources.

Denali's administrative history clarifies its purposes. The Park's origins are loosely linked to the large, Western national parks established during the first two decades of this century, since the original Mount McKinley National Park was established in 1917 and since early development included railroad access and a hotel. Because of its early designation within the National Park System, Denali has evolved to become one of the most well-established national parks. Because of its outstanding natural resources and accessible wilderness, Denali has become one of the most heavily visited of the national parks in Alaska. Still, development and use have been limited because of the Park's remote location (compared with the lower 48 states) and by management decisions and park plans to achieve its legislative purposes.

The legislative mandates and administrative history of Denali place the Park with others that can be characterized as wild, rustic and expansive. Denali rests somewhere between the extremely remote, lightly-used Alaskan national park units and the large, wilderness parks of the lower 48 states that are highly accessible and more developed. This blend of largely pristine conditions and an intense focus on use and access in a relatively small but critical portion of the Park, coupled with the unique provisions of ANILCA, creates unusual management challenges and is often at the core of most controversial issues (Brown 1993).

Backcountry management planning in Denali National Park and Preserve involves many similar challenges to wilderness management planning in other protected areas. Because of its importance to the tourism industry in Alaska and its symbolic importance as a wilderness park, Denali receives considerable attention in the media and is often at the forefront of park management issues in Alaska. Decisions made in Denali may affect wilderness and backcountry management planning in other parks in Alaska and elsewhere.

Legislative Mandates

An understanding of fundamental park purposes from the Park's enabling legislation and ANILCA is critical to determining appropriate alternatives in management plans. In 1917, Congress established Mount McKinley National Park to "set apart as a public park for the benefit and enjoyment of the people . . . for recreation purposes by the public and for the preservation of animals, birds, and fish and for the preservation of the natural curiosities and scenic beauties thereof . . . said park shall be, and is hereby established as a game refuge" (39 Stat. 938). ANILCA contains language defining the broad purposes of the new national parks and preserves in Alaska as well as the specific purposes of each conservation unit including Denali.

The enabling legislation from 1917 and the Park purposes under ANILCA are referenced in management plans for Denali National Park and Preserve and provide the basis

from which vision statements and strategic planning goals are derived. Along with the Park's administrative history, legislative mandates set the course for the backcountry management plan.

The Alaska National Interest Lands Conservation Act of 1980

The Alaska National Interest Lands Conservation Act of 1980 (ANILCA) doubled the size of the area administered by the National Park Service, adding several new units and extensive areas of designated wilderness throughout the nation's largest state. A total of 104.3 million acres of national parks, national wildlife refuges and other protected units were designated by ANILCA (Williss 1985), and more than 56 million acres were added to the National Wilderness Preservation System (Landres and Meyer 1998). The former Mt. McKinley National Park was expanded from two million acres to six million acres and renamed Denali National Park and Preserve. Almost all of the former Mt. McKinley National Park was designated as wilderness.

Many aspects of backcountry management planning in Denali are unique to the Alaska conservation units that were created or significantly expanded by ANILCA. The primary purposes of the new and enlarged national parks and preserves in Alaska are included in Section 101:

- Preserve lands and waters for the benefit, use, education, and inspiration of present and future generations.
- Preserve unrivaled scenic and geological values associated with natural landscapes.
- Maintain sound populations of, and habitat for, wildlife species.
- Preserve extensive, unaltered ecosystems in their natural state.
- Protect resources related to subsistence needs.
- Protect historic and archeological sites.
- Preserve wilderness resource values and related recreational opportunities.
- Maintain opportunities for scientific research in undisturbed ecosystems.
- Provide the opportunity for rural residents to engage in a subsistence way of life.

ANILCA also includes language specific to Denali National Park and Preserve:

- To protect and interpret the entire mountain massif and the additional scenic mountain peaks and formations.
- To protect habitat for, and populations of fish and wildlife including, but not limited to, brown/grizzly bears, moose, caribou, Dall sheep, wolves, swans, and other waterfowl.
- To provide continued opportunities, including reasonable access, for mountain climbing, mountaineering, and other wilderness recreational activities.

ANILCA includes provisions for subsistence use, which continues in national park additions and preserves throughout Alaska regardless of wilderness designation. Motorized uses not traditionally associated with wilderness are also permitted by Section 1110 (a):

Notwithstanding any other provision of this Act or other law, the Secretary shall permit, on conservation system units, national recreation areas, and national conservation areas, and those public lands designated as wilderness study, the use of snowmachines (during periods of adequate snow cover, or frozen river conditions in the case of wild and scenic rivers), motorboats, airplanes, and nonmotorized surface transportation methods for traditional activities (where such activities are permitted by this Act or other law) and for travel to and from villages and homesites. Such use shall be subject to reasonable regulations by the Secretary to protect the natural and other values of the conservation system units, national recreation areas, and national conservation areas, and shall not be prohibited unless, after notice and hearing in the vicinity of the affected unit or area, the Secretary finds that such use would be detrimental to the resource values of the unit or area.

This section of ANILCA has been interpreted as an “ANILCA-guaranteed right of access” by some advocates of motorized use (Gauna 1999). Extensive debate between motorized use groups and environmental organizations has ensued over terms such as “traditional activities,” “reasonable regulations” and “detrimental to resource values.” Defining these terms has been critical to managing the uses specifically mentioned in the law. Management actions such as the regulations prohibiting snowmachine use in the designated wilderness in Denali National Park include definition of these terms.

Planning for the Backcountry and Wilderness of Denali National Park and Preserve

The need for a comprehensive backcountry management plan for Denali National Park and Preserve rises from the exponential growth in motorized uses during recent years, the rapid increase in proposed commercial activities and the accelerated use of areas such as the Ruth Amphitheater on the south side of the Park for individual recreational activities. ANILCA does not include direction for dealing with these types of changes, and there is evidence in the legislative history indicating such changes were not anticipated. A 1979 U.S. Senate report stated that:

The transportation modes covered by this section are float and ski planes, snowmachines, motor boats, and dogsleds. The adverse environmental impacts associated with these transportation modes are not as significant as for roads, pipelines, railroads, etc. both because no permanent facilities are required and because the transportation vehicles cannot carry into the country large numbers of individuals. (U.S. Senate, 1979)

Establishing the legal parameters for management alternatives is another essential component of the Denali National Park and Preserve backcountry management plan. ANILCA does not replace the NPS Organic Act, which directs the agency to:

...promote and regulate the use of the Federal areas known as national parks, monuments, and reservations...by such means and measures as conform to the fundamental purpose of said park, monuments and reservations; which purpose is to conserve the scenery and the natural and historic objects and wildlife therein and to provide for the

enjoyment of the same in such manner and by such means as will leave them unimpaired for the enjoyment of future generations.

The Organic Act was amended by the Redwood National Park Expansion Act of 1978, in which Congress explained that the promotion and regulation of the National Park System shall be consistent with the protection of park resources, and shall not be exercised in derogation of these values except as may have been specifically provided for by Congress (Bader 1999).

The challenge at Denali is to provide for backcountry uses consistent with the resource protection goals in the Organic Act, the Park’s enabling legislation and ANILCA. Major issues in the backcountry management plan include:

1. Levels and types of use: individual uses, group size, commercial uses
2. Visitor experience
3. Research and resource protection
4. Facility development, use and maintenance
5. Administration of backcountry management program
6. Coordination with other land management agencies, cross-boundary issues, land exchanges
7. Access

The most contentious issues that are expected to arise in backcountry management planning discussions relate to aircraft overflights and landings, snowmachine use, other motorized uses and commercial and recreational uses. The planning process requires accurate interpretation of ANILCA, following established procedures for interpreting legislation (Meyer 1999), and defining appropriate types and levels of use. Developing a comprehensive plan that can be effectively implemented will require working with the public to come up with innovative approaches to allocating uses, minimizing conflicting uses and protecting remote yet accessible backcountry resources.

Case Law Affecting Backcountry Management Planning

Two primary concepts emerge from an analysis of case law involving the National Park Service that have a direct bearing on how issues in backcountry management planning are to be addressed: (1) the allocation of recreational uses and (2) the National Park Service responsibility to act affirmatively to protect resources. These concepts were fundamental in a recent finding supporting closure of most of the designated wilderness in Denali National Park and Preserve to snowmachine use (National Park Service 1999).

Allocation of Recreational Uses

The administrative discretion granted to the National Park Service for managing national parks allows for allocating limited recreational opportunities among competing user groups. For example, in *Bicycle Trails Council of Marin v Babbitt* (1996), the Ninth Circuit Court upheld the NPS action of prohibiting bicycle use on 36% of the recreational trails in the park. Bicycles were allowed on the remaining trails so were not excluded from the Park. The court ruled that there is nothing in the Organic Act that

requires the National Park Service to allow unfettered use of a unit if that use is inconsistent with other recreational uses (Bader 1999).

This case may apply to Denali National Park and Preserve in that permitting motorized uses in winter in all parts of the Park is inconsistent with other types of recreation, such as dog mushing and cross-country skiing, where natural sounds may be an important part of the experience. If motorized use were not restricted, other uses would be compromised.

National Park Service Responsibility to Plan and to Manage Proactively

In carrying out its preservation mission, the National Park Service need not wait for actual damage to occur before taking action to protect wildlife and other natural attributes. The National Park Service decision was upheld in *Wilkins v Department of Interior* (1993), a case involving Carlsbad Caverns National Park. The National Park Service had removed deer to study whether they were a potential threat, and the agency's decision was upheld by the court (Bader 1999).

The National Park Service may also plan proactively for potential threats (*New Mexico State Game Commission v Udall*, 1969). In *Kleppe v New Mexico* (1979) the agency decision was upheld after removing an exotic species—wild horses—because of a *potential* threat to ecological integrity (Bader 1999). Language in ANILCA providing for regulation of access such as “...Secretary finds that such use would be detrimental to the resource values of the unit or area” is consistent with the theme of planning proactively identified in the above case law.

The Public Process

While the National Park Service has been given broad discretion by the courts in determining types and levels of uses of park resources and in allocating recreational uses, what happens in the public and political arenas is also crucial to park management. Recent progress in planning efforts at Denali National Park has been possible because the agency exceeded the public disclosure requirements in the National Environmental Policy Act. The public scoping process is critical to successful wilderness management planning and should include numerous informal meetings with agencies, organizations and individuals affected by proposed management alternatives. Formal scoping meetings provide an additional forum for discussions. Meeting with known and potential adversaries can help ensure that there are no surprises in public documents that result in unfavorable headlines. Denali National Park receives considerable support from Alaskans and other interested individuals throughout the United States and the world. The Park must continue finding new ways to enlist this support for meeting its mandates to provide for an outstanding visitor experience and to protect its internationally significant resources.

Public expectations of Denali National Park and Preserve are determined from visitor surveys and unsolicited visitor comments. Information on desired visitor experiences in protected areas is essential to conducting the National Park

Service Visitor Experience and Resource Protection program (VERP), Limits to Acceptable Change (LAC) or other methodology to deal with carrying capacity. Addressing carrying capacity is now required in NPS general management plans and will be included in the backcountry management plan for Denali. Information on visitor experience is equally important to scientific information on wildlife and other park resources. For example, many of the comments addressing the desired level of traffic on the park road during a 1996 planning process mentioned visitor experience instead of or in addition to wildlife concerns as a reason to hold traffic at existing levels (NPS, 1997a; Miller and Wright, 1998).

Conclusion

While ANILCA presents unusual challenges for wilderness management in Alaska, it also outlines the need for land managing agencies to “do it right” by protecting the integrity of the outstanding resources recognized by that law. Planning for the backcountry along with other management actions affecting Denali National Park and Preserve follows the guidance provided by the fundamental purposes of ANILCA. The park has developed a vision statement consistent with the general purposes of ANILCA:

Denali National Park and Preserve is a vast area that provides visitors of all abilities with opportunities for superlative, inspirational experiences in keeping with its legislative mandates. Over the long term, preservation of the wilderness and its continually evolving natural processes is essential to providing the opportunity for outstanding resource-based visitor experiences. (National Park Service, 1997)

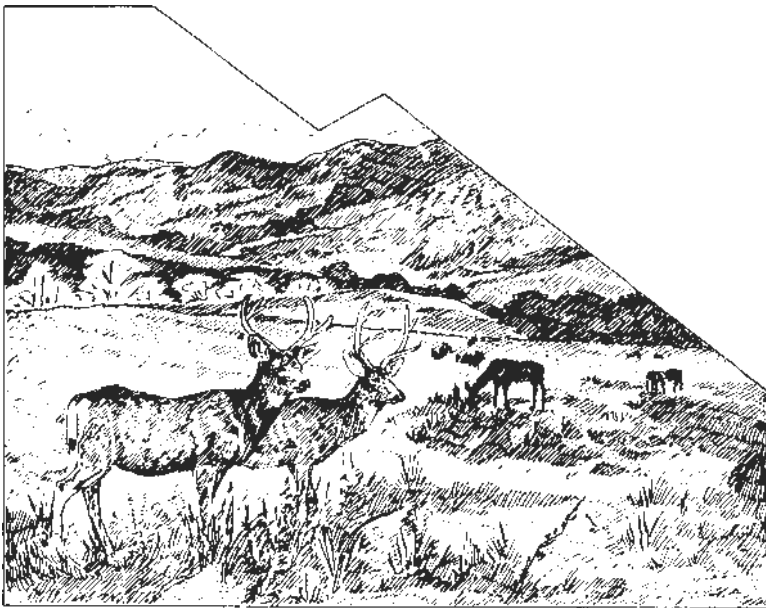
The backcountry management plan for Denali National Park and Preserve will follow this general vision and the direction of ANILCA to continue the tradition of providing for outstanding opportunities to experience wilderness established early in the 20th century. The National Park Service will encourage public involvement at every step during this planning process, recognizing that informed debate on controversial issues often results in creative solutions to difficult challenges.

References

- Bader, Harry R. 1999. A Review of Judicial Decisions Affecting Management Planning in the National Parks of the United States. Unpublished Report for Denali National Park and Preserve. Department of Forest Sciences, University of Alaska-Fairbanks. 49 p.
- Brown, William E. 1993. Denali, Symbol of the Alaskan Wild. Alaska Natural History Association, Denali Park, Alaska. 223 p.
- Gauna, Joe 1999. Snowmachine Rights in Park are Clear. Guest Opinion Column in Fairbanks Daily News-Miner, January 23, 1999. Section A, p. 4.
- Landres, Peter; and Meyer, Shannon 1998. National Wilderness Preservation System Database: Key Attributes and Trends, 1964 Through 1998. General Technical Report RMRS-GTR-18, Rocky Mountain Research Station, Forest Service, U.S. Department of Agriculture. 11 p.
- Meyer, Shannon 1999. The Role of Legislative History in Agency Decision-Making: A Case Study of Wilderness Airstrip Management in the United States. Proceedings: Wilderness Science in a

- Time of Change. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station (in press).
- Miller, Craig A. and R. Gerald Wright, 1998. Visitor Satisfaction with Transportation Services and Wildlife Viewing Opportunities in Denali National Park and Preserve. Wildlife Management Institute, University of Idaho. 35 p.
- National Park Service, 1986. General Management Plan/Land Protection Plan/Wilderness Suitability Review, Denali National Park and Preserve, Alaska. Denver Service Center, National Park Service, U.S. Department of the Interior. 231 p.
- National Park Service, 1997a. Final Entrance Area and Road Corridor Development Concept Plan and Abbreviated Final Environmental Impact Statement, Denali National Park and Preserve, Alaska. Denver Service Center, National Park Service, U.S. Department of the Interior. 358 p.
- National Park Service, 1997b. Strategic Plan: Implementation of the Government Performance and Results Act. Denali National Park and Preserve, Alaska. 54 p.
- National Park Service, 1999. Statement of Finding: Temporary Closure of the Former Mt. McKinley National Park Area of Denali National Park and Preserve to the Use of Snowmobiles for Traditional Activities. Denali National Park and Preserve, Alaska. 33 p.
- U.S. Public Law 64-235. 1916. An Act to Establish a National Park Service, and for Other Purposes. 39 Stat. 535.
- U.S. Public Law 88-577. 1964. The Wilderness Act. 78 Stat. 890.
- U.S. Public Law 96-487. 1980. Alaska National Interest Lands Conservation Act. 94 Stat. 2371.
- U.S. Senate, 1979. Report of the Committee on Energy and Natural Resources of the United States Senate, Together with Additional Views to Accompany H.R. 39, Alaska National Interest Lands. Report No. 96-413, 96th Congress, 1st Session. U.S. Government Printing Office, Washington. p. 248.
- Williss, Frank G. 1985. Do Things Right the First Time: The National Park Service and the Alaska National Interest Lands Conservation Act of 1980. National Park Service, Denver Service Center.

7. Dialogue Session Summary



Naturalness and Wildness: The Dilemma and Irony of Managing Wilderness

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Abstract—This paper summarizes a dialogue session that focused on two concepts that strongly influence nearly all wilderness management: wildness and naturalness. The origin and value of these concepts are discussed, as well as the dilemma and irony that arises when wilderness managers contemplate manipulating the environment to restore naturalness at the risk of reducing wildness. To illustrate this irony, a case study of a proposed large-scale manipulation to stop the loss of cultural resources in the Bandelier Wilderness is discussed. It is concluded that large scale wilderness restoration based on manipulating the environment will always cause a dilemma and entail the irony of balancing wildness against naturalness. One of the biggest hurdles facing wilderness policy-makers and managers today, as well as the concerned public, is how to reconcile these views and manage wilderness for both wildness and naturalness.

Two independent but related concepts are intertwined in the idea of wilderness. In the 1964 Wilderness Act, wilderness is defined in Section 2(c) as "...an area where the earth and its community of life are untrammeled by man, where man himself is a visitor who does not remain." Later in this same section, wilderness is further defined as an area "retaining its primeval character and influence...which is protected and managed so as to preserve its natural conditions." The key words in these quotes are *untrammeled* and *natural*. When the Wilderness Act was passed, these key words undoubtedly were intended to be complementary because untrammeled areas were certainly natural. Today, however, we are witnessing regional ecological impacts to areas that are untrammeled in every other way, as well as new understanding of the long-term ecological consequences of natural resource management. As a result, we now have divergent philosophical views of what wilderness is and what it should be. These views are encapsulated by the

words untrammeled and natural in a way that was likely unforeseen by wilderness proponents as they crafted legislative wording. This dialogue session explored the management dilemmas and social ironies resulting from these divergent views and presented a case study that brings these diverging views into sharp focus.

Terms and Concepts

In one of the first and clearest explanations of the word untrammeled, Zahniser (1956) wrote "...there is in our planning a need also to secure the preservation of some areas that are so managed as to be left unmanaged—areas that are undeveloped by man's mechanical tools and in every way unmodified by his civilization." Synonyms for untrammeled include unimpeded, unhampered, uncontrolled, self-willed and free. We suggest that the word "wildness" strongly connotes this sense of an area free from human control or manipulation. Use of this word is also supported by Zahniser's statement before a committee of the New York State legislature in 1953 that "We must remember always that the essential quality of the wilderness is its wildness" (Zahniser 1992). Synonyms for natural include native, aboriginal, indigenous and endemic, and we suggest that the term "naturalness" be used to capture this biological sense of wilderness.

While these concepts of wildness and naturalness differ from one another, both are essential elements of wilderness (Aplet 1999; Barry 1998; Worf 1997) and are highly valued in our society (Cordell and others 1998; Manning and Valliere 1996). As shown in figure 1, wilderness is the idea and place where the concepts of wildness and naturalness reach their highest expression. These concepts strongly influence, either directly or indirectly, virtually all of the decisions and actions taken in wilderness management.

An Emerging Dilemma

Traditionally, wilderness management was largely concerned with human-caused impacts to wilderness recreation experiences and to the plants and soil directly affected by this recreation, principally in campsites and trails. To mitigate these biophysical impacts, wilderness managers generally have few compunctions about closing a campsite or rerouting a trail. These actions take place over a relatively small area and don't violate most visitors' notion of wilderness.

In contrast, wilderness managers today face a set of problems likely unforeseen by those who wrote and debated

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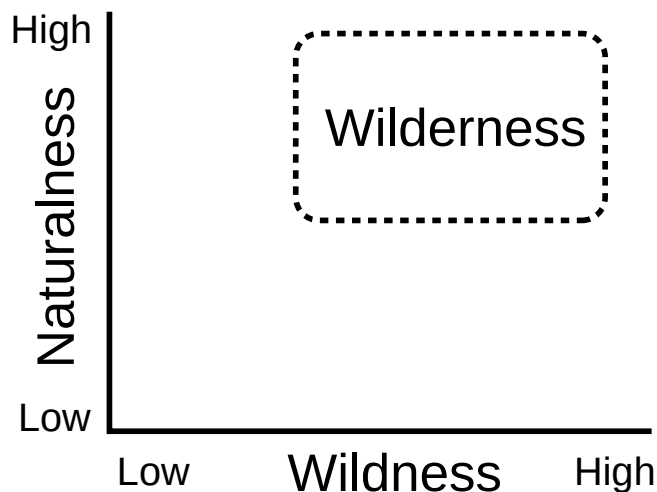


Figure 1—Naturalness and wildness are two related, but independent aspects of wilderness. Wilderness is the place and idea where the concepts of naturalness and wildness reach their highest expression.

the 1964 Wilderness Act (Brunson 1995). These problems are largely the result of broad-scale ecological impacts that pose significant long-term impacts to wilderness. Decades of fire suppression, for example, have increased fuel loads and allowed dense undergrowth of trees in areas where frequent, low-intensity fires were the norm, placing widely spaced old-growth trees at risk. The proposed solution is usually mechanical reduction of fuels, the use of management-ignited fire, or both to restore the natural fire regime. The widespread occurrence of exotic plants alters native plant and animal communities in wilderness, and the use of herbicides is often proposed to restore native plant communities. Acid deposition throughout the eastern United States and in certain areas of the western United States has significantly altered aquatic systems in several wildernesses. Liming these aquatic systems has been proposed to counter the acidity and restore these systems. The exotic white-pine blister rust has caused widespread mortality of high-elevation whitebark pine, and establishing forests of whitebark pine seedlings that have been genetically altered to be rust resistant has been proposed to restore these forests.

In each of these cases, the naturalness of the area has been compromised by broad-scale human actions, and some form of manipulation of the environment is proposed to restore this naturalness. The crucial issue this raises is whether large-scale manipulation, however undesirable, should be used to restore natural conditions, thereby sacrificing wilderness for naturalness (Cole 1996). In these situations, where human-caused impacts have caused wholesale changes to the wilderness environment, should the wildness of present-day wilderness be compromised to restore naturalness? In other words, should an undesirable means, such as manipulation of wilderness, be used to achieve a desirable end, such as restoration of natural conditions in wilderness?

Different people hold strong views on this issue, which goes to the heart of whether wilderness is, or should at least remain from this point on, wild or natural. Some people think the provision in the 1964 Wilderness Act that "...these [areas] shall be administered...so as to provide for the

protection of these areas, the preservation of their wilderness character..." is a clear mandate for restoring natural conditions in wilderness to overcome a myriad of human-caused insults. Indeed, restoration of these areas is often expressed in terms of an obligation and responsibility to correct human-caused problems (Windhager 1998). Others, citing the Wilderness Act definition of wilderness as "...an area where the earth and its community of life are untrammelled by man," claim that the fundamental character of wilderness is to be free of human manipulation (Worff 1997). Here, wilderness is the one and only place on our ever more crowded planet that is left free from our conscious manipulation, and these areas yield important and vital benefits to people and society because they are untrammelled.

The Central Dilemma of Wilderness Management: When to Take Action?

Deciding when to take action in wilderness was described by Landres and others (1998) as the central dilemma in wilderness management. Proposals to manipulate ecological conditions in wilderness to restore naturalness bring this dilemma to new heights, as well as raise significant and difficult questions: Does manipulation compromise the very values that are protected and preserved in wilderness? Is there sufficient technical knowledge to use large-scale manipulation to restore wilderness landscapes? What are the consequences and risks of taking action versus not taking action? Does the public sufficiently trust the agency to allow such large-scale actions? Does the desire to restore the ecological value of naturalness outweigh the social value of wildness? How much trammeling is necessary and tolerable in wilderness? Is it appropriate to even define a target for desired future ecological conditions in wilderness? Must we accept, albeit reluctantly, the human "gardenification" of wilderness, as suggested by Janzen (1998)?

Separating the concepts of wildness from naturalness helps clarify and partially resolve this management dilemma of when to take action. A two-way matrix of wildness and naturalness (figure 2) illustrates when a proposed action is not appropriate, when it is appropriate and when it entails weighing wildness against naturalness. Briefly, some proposed management actions, such as manipulating habitat to increase a wildlife species' density above natural levels, decrease both wildness and naturalness and should not be pursued. Conversely, proposed actions that support wildness or at least do not reduce it while increasing naturalness should be pursued. Closing and restoring a campsite, for example, doesn't manipulate the environment in a way that impedes wildness on a large scale, and restoring native plants increases naturalness.

Management dilemma and irony can be seen when either wildness or naturalness must be compromised to enhance the other (figure 2). For example, in forests where the natural fire regime is frequent, light, surface fires, a decision not to mechanically reduce fire suppression-caused buildup of fuels supports wildness, but it may decrease naturalness if the forest becomes more susceptible to catastrophic fire. Alternatively, reducing built-up fuels with mechanical thinning or management-ignited fire decreases wildness,

		WILDNESS	
NATURALNESS		Decrease	Support
	Decrease	NO ACTION	DILEMMA & IRONY
	Increase	DILEMMA & IRONY	ACTION

Figure 2—A two-way matrix showing suggested outcomes when proposed management actions support or decrease wildness and increase or decrease naturalness. Proposed actions that both decrease wildness and naturalness should not be considered, while actions that both support wildness and increase naturalness should be considered. Proposed actions that compromise either wildness or naturalness create management dilemmas and social irony forcing wildness to be weighed against naturalness.

but it may increase survival of the forest. The appropriate course of action in either of these cases is not clear, any judgment needs to be based on the spatial and temporal scale of the proposed actions and their effects, and how well-defined the target conditions are. If the degraded area and restoration actions are localized, if the actions taken today will allow managers to reduce their interference with the “will of the land” in the future, and if there are good reference sites to know what the undisturbed condition is, manipulative actions are probably justified. In contrast, if restoration actions are being considered over a large area and there is uncertainty about the effects of these actions or about the target conditions, much more caution and scrutiny is warranted. Each of these criteria—spatial scale, temporal scale and knowledge of undisturbed conditions—span from small (for example, a small area, a short time frame and a small amount of knowledge) to large, and there are no rules or guidelines about how small or how large is sufficient to warrant taking action or not.

Understanding the differences between wildness and naturalness doesn’t provide a definitive answer to solve this central dilemma of wilderness management. These concepts do help clarify when proposed actions are clearly inappropriate and when they are appropriate. Furthermore, they clarify what issues need to be discussed and weighed in determining whether proposed manipulative actions should be taken.

Understanding and Reconciling the Social Irony

Wilderness was established by Congress to uphold the social values of wildness and naturalness. As discussed above, wilderness managers now find themselves in the ironic situation of choosing between wildness and naturalness. In this section, we describe the social origins and implications of this irony. We suggest that differing

philosophical views led us to see nature and culture as dichotomous or convergent, that the 1964 Wilderness Act codified the dichotomous view, and that two recent movements—ecosystem management and ecosystem restoration—have arisen from a re-emergence of the convergent view. Finally, we discuss how perceptions of risk and uncertainty in natural systems influence the outcomes of this irony.

Fine (1997) identified three overarching philosophical views of the relationship between nature and culture that have predominated over the course of human history. The first of these is the “utilitarian” perspective, in which nature is seen primarily as a storehouse of goods that can meet human needs. In this view, nature and culture are seen as two separate entities, with nature existing primarily for the benefit of culture. The utilitarian view is often said to represent the traditional Judeo-Christian idea about nature; while that is surely an oversimplification, it certainly was a dominant philosophy during the Industrial Revolution and era of American expansion (Nash 1967).

The second view, the “preservation” perspective, also holds nature and culture to be separate. But in this view, nature is seen to exist *in spite of* culture, and the best role for nature is to be protected from the influences of humanity. Fine (1997) calls this the “strong environmentalist” position. Some adherents equate it with non-Western cultures, which they see as being more biocentric than our own, but it is more properly identified with the romantic philosophies of Rousseau and Thoreau, which have found their fullest expression in post-war Europe and America.

The third view is the “organic” perspective. Fine (1997) points out that this is both the oldest and newest orientation toward nature—characteristic of many pre-industrial cultures, as well as the modern sustainable development movement, among others—in which the natural world and human world are integrated and even inseparable. The appropriate role for nature in this view is that it is one sphere of human action.

The Wilderness Act, passed at the beginnings of the modern American environmental movement, when our society was just beginning to recognize the full extent of environmental degradation caused by modern industrial expansion, is legislation born of dichotomy between nature and culture. The preservationist view is seen clearly in its description of wilderness as a place “...where man himself is a visitor who does not remain.” Wilderness management has solidified this dichotomous perspective, as required by the language of the act itself, by distinguishing between natural and human-caused influences. Thus, for example, lightning-ignited fires typically are allowed to burn, but human-ignited fires are not, even if their ecological benefits to the health of wilderness ecosystems would be identical. Or bare ground may be mitigated if attributed to humans or domestic livestock but not wild ungulates.

Since passage of the Wilderness Act, however, other movements have begun to try to close the gap between nature and culture, even to inject culture into nature in order to redress some of the “sins” of culture. The dilemma over management action in wilderness today is born of our recognition of these later movements, which represent a re-emergence of the ancient holism seen in some pre-industrial views of humans in nature.

The first of these movements is ecosystem management, which acknowledges human dependence on biotic integrity and seeks to blur the boundaries between social and biotic systems (Yaffee 1999). The second movement is that of ecological restoration, which represents a recognition of society's ethical responsibility to try to "make things right" in our relationship with nature (Gobster and Hull 1999). Some thinkers such as Jordan (1985) have tried to create a "participatory ideal," in which restoration is best when it meets a wide range of human needs. Restoration is *not* simply fixing things and then leaving them alone, but rather a continued community action. The convergent view of nature/culture relationships has also made its way into wilderness management through adoption of the Limits of Acceptable Change planning process, which explicitly acknowledges that humans will be part of wilderness systems (as required under the Wilderness Act) and then gives society the responsibility for determining how extensive that role in wilderness is allowed to be (McCool and Cole 1997).

The dilemma we face—whether to err on the side of wildness by stressing the nature/culture dichotomy, or to err on the side of naturalness by restoring nature whenever possible — is rooted in the ongoing ambiguity of a wilderness policy and other environmental policies that are rooted both in the preservationist and organic views of nature and culture. Where we fall on the spectrum from dichotomy to convergence is often rooted in our view of risk and uncertainty: Do we dare trust science? Do we dare not? If we trust scientists to make wise, informed judgments about what "nature" would be without human intervention, we are more likely to approve of manipulations intended to produce those conditions. Alternatively, if we're concerned about the possibility of restoration going awry, we may be too risk-averse to allow restoration in wilderness.

Seen another way, if we believe that wild nature is doomed, we may be more likely to want to restrict further manipulation in order to save whatever's left in the least "damaged" condition possible. Alternatively, we may believe that leaving things alone will only make matters worse, as may be the case in systems we've simplified through fire suppression, so that the only justifiable action is to try to reverse the trends.

Our trust is not only in science, however, but in the people who apply it: scientists and managers. When people oppose manipulative restoration, is it the science they distrust or is it us? These are questions that we need to confront if we are to make reasoned decisions about whether to allow restoration of naturalness or protect wildness at all costs.

Case Study: Proposed Manipulation in Bandelier Wilderness

Bandelier National Monument was established in 1916 under authority of the 1906 Antiquities Act to protect the cultural resources left by ancestral Puebloan peoples in north-central New Mexico. Among National Park Service lands, Bandelier has one of the highest concentrations of cultural resources, with an estimated 3,500 archeological sites. In 1976, nearly 71% of the monument, 23,267 acres, was designated as the Bandelier Wilderness.

Approximately 70% (about 2,500) of the Monument's archeological sites are believed to be located in pinon-juniper

woodlands within the Bandelier Wilderness. The woodland soils are 100,000 years old and, until the early part of this century, supported a dense herbaceous ground cover, which limited the rate of soil erosion and associated archeological site disintegration. Frequent surface fires through the abundant herbaceous fuels prevented widespread establishment of pinon and juniper trees. With the introduction of the railroad in the 1880s, livestock grazing increased dramatically and continued until the early 1940s. This grazing caused the loss of the herbaceous ground cover and precipitated severe ecological change, including the loss of fire in the ecosystem. Tree density has increased dramatically in the past century in the absence of frequent fires, setting up a positive feedback cycle that is exacerbating competition for scarce water and soil nutrients and decreasing herbaceous cover and diversity (Gottfried and others 1995). The herbaceous ground cover has dropped below a critical threshold (Davenport and others 1998), initiating an ongoing cycle of severely accelerated erosion that will strip most of the soils from these areas in 100-200 years (Wilcox and others 1996a,b). This modern, human-initiated, accelerated erosion is currently affecting at least 80 percent of the recorded archeological sites in the pinon-juniper woodlands. In one rainstorm during 1995, for example, 1,040 cultural artifacts were washed into a sediment trap from a 0.1-hectare study watershed.

The question facing managers at Bandelier is how to break this positive feedback cycle, increase herbaceous ground cover to pre-livestock grazing levels, restore fire as a viable ecological process and stop the accelerated soil erosion that is demolishing both the natural and cultural resources. Although livestock grazing officially ended in 1932, and feral burros were removed in about 1980, there has been no recovery of herbaceous ground cover because physical processes now dominate in the barren, desertified interspaces between trees. Research done in the Bandelier Wilderness and adjacent areas has demonstrated that thinning trees and leaving them on-site produces a two to seven-fold increase in herbaceous cover and significantly reduces soil erosion (Jacobs and Gatewood 1999).

To break this positive feedback cycle and set in motion changes to restore herbaceous cover and natural fires, as well as to reduce soil erosion and slow the loss of cultural resources, the management staff at Bandelier is considering thinning some of the pinon and juniper trees over portions of 8,000 acres in wilderness. Such action would require the use of chain saws and leave clear signs of human presence for about two decades—perhaps longer. The dilemma now facing these managers is whether to intervene to restore sustainable wilderness conditions and stop extreme soil erosion and concomitant wholesale loss of cultural resources for which the monument was established, or to take no action so that the "hand of man" is not imposed on this wilderness. Either choice has significant consequences.

In developing management direction in the face of this dilemma, managers are considering the following questions:

- Does the Monument's enabling legislation (or the NPS Organic Act) reign supreme and, if so, at what cost to other resource values, including wilderness values, recognized later in the Monument's history?
- Should federal land managers intervene if wilderness ecosystems are degraded and unsustainable due to

federally sanctioned overgrazing and fire suppression over the past century?

- Can the “natural range of variability” be restored, and will it be sustainable?
- If restoration is possible, what should the goal or target conditions be in federally designated wilderness?
- While current erosion conditions within the Bandelier Wilderness warrant urgent management attention, are drastic restorative measures justified?
- Is it appropriate to conduct large-scale ecosystem restoration work in wilderness?
- If managers start manipulating wilderness, when and where will management intervention end?

Faced with this dilemma and after considering each of these questions, the managers at Bandelier are evaluating options through the NEPA process to temporarily compromise the value of wildness for the longer term sake of natural ecological conditions and cultural resources (Sydoriak and others, this volume). While most wilderness managers do not face the added burden of complying with enabling legislation that emphasizes cultural resource protection, they may well have to confront the wider issue of whether to take actions that will may shift conditions toward the natural range of variability.

Conclusions

Large-scale wilderness restoration based on manipulating the environment will always cause a dilemma and entail the irony of balancing wildness against naturalness. In one way, this dilemma is good because it forces us to carefully consider our actions and their consequences. “Doing the right thing” for wilderness used to be fairly straightforward. Today, with our increased knowledge of regional-scale human impacts, coupled with our desire to restore areas known to be degraded, “doing the right thing” is no longer a simple path because it is based on a philosophical choice between wildness and naturalness. Two people or groups may differ, sometimes strongly, about what they perceive is “right” for wilderness, and both views are valid. If there are significant doubts about a proposed action, one view would err on the side of protecting wildness, while the other view would err on the side of naturalness. One of the biggest hurdles facing wilderness policy-makers and managers today, as well as the concerned public, is how to reconcile these views and manage wilderness for both wildness and naturalness.

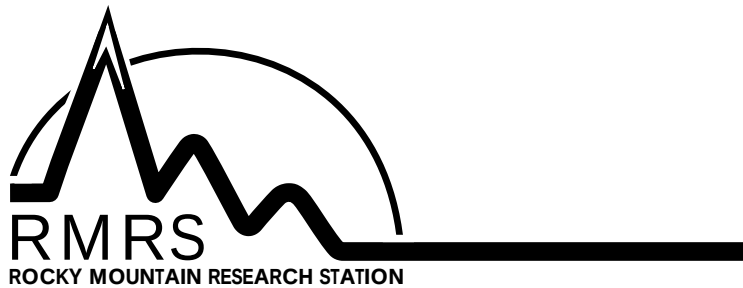
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References

Aplet, G.H. 1999. On the nature of wildness: exploring what wilderness really protects. *Denver University Law Review* 76:347-367.

- Barry, D. 1998. Toward reconciling the cultures of wilderness and restoration. *Restoration & Management Notes* 16:125-127.
- Brunson, M.W. 1995. The changing role of wilderness in ecosystem management. *International Journal of Wilderness* 1(1):12-16.
- Cole, D.N. 1996. Ecological manipulation in wilderness: an emerging management dilemma. *International Journal of Wilderness* 2(1):15-19.
- Cordell, H.K., M.A. Tarrant, B.L. McDonald, and J.C. Bergstrom. 1998. How the public views wilderness. *International Journal of Wilderness* 4(3):28-31.
- Davenport, D. W., D. D. Breshears, B. P. Wilcox, and C. D. Allen. 1998. Viewpoint: Sustainability of pinon-juniper ecosystems—A unifying perspective of soil erosion thresholds. *Journal of Range Management* 51(2):229-238.
- Fine, G.A. 1997. Naturework and the taming of the wild: the problem of “overpick” in the culture of mushroomers. *Social Problems* 44:68-88.
- Gobster, P., and B. Hull. 1999. The restoration and management of nature. *Ecological Restoration* 17:44-51.
- Gottfried, G. J., T.W. Swetnam, C.D. Allen, J.L. Betancourt, and A. Chung-MacCoubrey. 1995. Pinyon-juniper woodlands. Pages. 95-131 in *Ecology, Diversity, and Sustainability of the Middle Rio Grande Basin* (D.M. Finch and J. A. Tainter technical editors). USDA Forest Service General Technical Report RM-GTR-268, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.
- Jacobs, B. F., and R. G. Gatewood. 1999 (in press). Restoration studies in degraded pinon-juniper woodlands of north-central New Mexico. Pages 294-298 in *Proceedings: Ecology and Management of Pinyon-Juniper Communities within the Interior West* (S.B. Monsen, R. Stevens, R. J. Tausch, R. Miller and S. Goodrich, editors.). USDA Forest Service Proceedings RMRS-P-9. Ogden, UT.
- Janzen, D. 1998. Gardenification of wildland nature and the human footprint. *Science* 279:1312-1313.
- Jordan, W.R. 1985. On the imitation of nature. *Restoration & Management Notes* 3:2-3.
- Landres, P.B., P.S. White, G. Aplet, and A. Zimmermann. 1998. Naturalness and natural variability: definitions, concepts, and strategies for wilderness management. Pages 41-50 in *Wilderness & natural areas in eastern North America: research, management and planning* (D.L. Kulhavy, M.H. Legg, editors). Center for Applied Studies, College of Forestry, Stephen F. Austin University, Nacogoches, TX.
- Manning, R.E., and W.A. Valliere. 1996. Environmental values, environmental ethics, and wilderness management. *International Journal of Wilderness* 2(2):27-32.
- McCool, S.F., and D.N. Cole, compilers. 1997. *Proceedings—Limits of Acceptable Change and related planning processes: progress and future directions*. USDA Forest Service General Technical Report INT-GTR-371, Intermountain Research Station, Ogden, UT.
- Nash, R. 1967. *Wilderness and the American mind*. Yale University Press, New Haven, CT.
- Wilcox, B. P., J. Pitlick, C. D. Allen, and D. W. Davenport. 1996a. Runoff and erosion from a rapidly eroding pinyon-juniper hillslope. Pages 61-77 in *Advances in Hillslope Processes, Volume 1* (M. G. Anderson and S. M. Brooks editors.). John Wiley & Sons Ltd., New York.
- Wilcox, B. P., B. D. Newman, C. D. Allen, K. D. Reid, D. Brandes, J. Pitlick, and D. W. Davenport. 1996b. Runoff and erosion on the Pajarito Plateau: Observations from the field. Pages 433-439 in *New Mexico Geological Society Guidebook, 47th Field Conference, Jemez Mountains Region*.
- Windhager, S. 1998. Biotic citizenship: a response to Barry. *Restoration & Management Notes* 16:128.
- Worff, B. 1997. Response to “Ecological manipulation in wilderness” by Dr. David Cole. *International Journal of Wilderness* 3(2):30-31.
- Yaffee, S.L. 1999. Three faces of ecosystem management. *Conservation Biology* 13:713-725.
- Zahniser, E. 1992. Where wilderness preservation began: Adirondack writings of Howard Zahniser. North Country Books, Inc., Utica, NY.
- Zahniser, H. 1956. The need for wilderness areas. *The Living Wilderness, Winter-Spring (1956-1957)*:37-43.



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