RESEARCH ON SOIL AND VEGETATION IN WILDERNESS: A STATE-OF-KNOWLEDGE REVIEW

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ABSTRACT

This paper deals primarily with research on soil and vegetation impacts caused by recreational use of wilderness. Studies have documented the most obvious effects of trampling, described conditions on campsites and along trails, described the spatial distribution of impact, and documented changes in impact levels over time. Relatively little is known about the effect of trampling on soil biota and plant physiology or the impact of recreational stock. Lack of theoretical work and the short time frames of most studies leave us with little ability to evaluate the significance of most of the impacts that have been described. An increasing number of studies have examined the importance of factors that influence amount of impact. A considerable amount of research on the relationship between amount of use and amount of impact has consistently shown the relationship to be highly curvilinear. Impact generally occurs rapidly and can be severe even on lightly used sites. The exact nature of this relationship varies with type of use, environmental conditions, and impact parameter. Different environments vary considerably in their resistance and their resilience. Few studies have differentiated impact by type of user, however. There has also been a limited amount of applied management-oriented impact research, mostly in the areas of development of monitoring techniques and experimentation with techniques for rehabilitating damaged recreation sites. There is a need for more longitudinal studies, more specialized, detailed studies, more interdisciplinary approaches, and an expanded regional coverage. Current high-priority research needs are identified. Such studies are unlikely to be conducted until careers in the field become available.

INTRODUCTION

Wilderness soil and vegetation have been the subject of two rather distinct types of research. In the first type, natural soil and vegetation conditions and processes are studied. The primary objective of this line of research is to improve our understanding of the structure and function of natural ecosystems. The second type of research is focused more narrowly on human-caused changes in soil and vegetation conditions in wilderness and how best to manage these impacts. The primary objective of this second type is to help managers with the difficult task of preserving natural conditions in wilderness. Major agents of change in wilderness include air pollution, fire, nonrecreational grazing, mining, introduced plants and animals, and recreational use. Many of these research topics are discussed in detail in review papers on basic ecological processes, air, fire, and fish and wildlife; others have been the subject of very little research. I will only briefly comment on and cite a few examples of most of these lines of research and focus primarily on research on the impact of recreational use on soils and vegetation (fig. 1).

RESEARCH ON WILDERNESS SOIL AND VEGETATION CONDITIONS AND PROCESSES

Wilderness has been used as a natural laboratory for studying undisturbed soil and vegetation conditions and processes. (Throughout this paper, when I use the term "wilderness" I mean both legally designated wilderness and large nonroaded areas.) Due to their large size and a management regime that stresses preservation of natural conditions, wildernesses provide unique opportunities for basic biological research. Wilderness is particularly valuable for research that seeks to integrate complex elements of the ecosystem or that aims to classify and characterize natural plant communities. But wilderness is also an appropriate location for detailed, small-scale studies. For example, topics of papers presented at the Second Conference on Scientific Research in the National Parks ranged from extensive integrative studies (such as "Interactions Among Fluvial Processes, Forest Vegetation, and Aquatic Ecosystems, South Fork Hoh River, Olympic National Park, Washington") to small-scale studies (such as "Demography of *Salix setchelliana*—a Prostrate Willow of Alaskan Gravel Bars").

In most cases, studies are conducted in wilderness because wilderness provides the best opportunities for studying minimally disturbed soil/vegetation components. Undisturbed wilderness conditions also provide a baseline for evaluating the nature of human influences outside wilderness. The primary object of study is not the wilderness as such, but conditions that increasingly can only be found in wilderness. This is a major feature distinguishing this type of research from impact-oriented research. Such research can contribute to improved management of wilderness, however, even if this is not the primary goal. An improved understanding of the structure and function of natural ecosystems helps managers evaluate the significance of human disturbances.

Some of the best examples of basic ecological research that both capitalize on the unique characteristics of wilderness and also strive to make results applicable to management are the "pulse" studies that Jerry Franklin

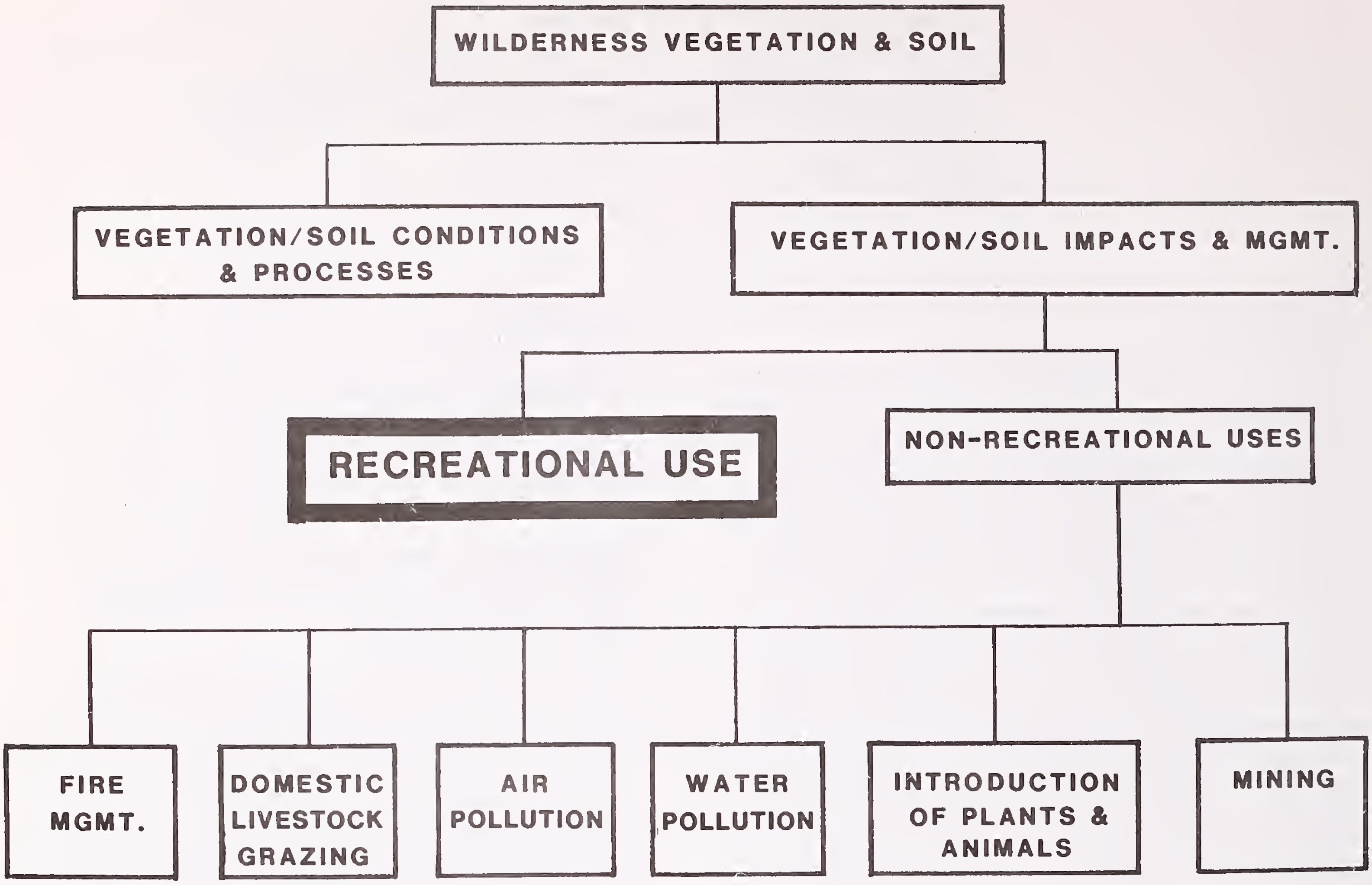

Figure 1.—Major types of research under the general heading of research on wilderness vegetation and soil.

has directed in Olympic and Sequoia/Kings Canyon National Parks. These studies bring together scores of specialists to measure, monitor, record, and then interact in such a way that linkages between ecosystem components are highlighted. Both studies have provided a wealth of basic ecological information. The Olympic study, for example, furthered the emerging recognition of the significant functions of woody debris—both on land and in stream channels—and provided a better understanding of disturbance agents in an environment that had been inadequately studied (Franklin and others 1982). A better understanding of the importance of woody debris as a seedbed and the effects of Roosevelt elk grazing on vegetation will help Park managers in their attempts to preserve natural conditions.

Most basic ecological research in wilderness could contribute to improved wilderness management. To maximize these benefits, however, communication between managers and researchers needs to be improved. Researchers can help by identifying potential management applications in the study design phase and by striving to highlight management implications. Managers can help by identifying information needs and communicating these to researchers.

Beyond these studies that contribute—often serendipitously—to management, all wildernesses need baseline and inventory information. Information is needed to (1) classify ecosystems, (2) map their distribution, (3) describe their basic characteristics, and (4) identify their dynamic properties and the factors affecting their dynamics (Franklin 1978). Information of this type is scarce. One of the few places where such information has been collected in a systematic manner is at Mount Rainier National Park (Moir and others 1979). A number of other National Parks have vegetation maps, data characterizing vegetation types, and permanent plots for monitoring trends, but other agencies with wilderness management responsibilities have little information of this type. The Forest Service, for example, is heavily involved in classifying vegetation in roaded areas and spends large sums of money every year for timber stand examinations and inventory. But little of this effort is directed toward wilderness, despite specific direction in the Forest Service Manual (FSM 2322.1) stating that wilderness management plans will (1) "describe the current condition of all resources and biotic associations"; (2) "describe the interrelationships of all resources, existing uses and activities, and highlight unique ecological situations"; and (3) "identify problems associated with maintaining an enduring wilderness resource and the reasons for the problem."

Of particular utility to management would be a classification system stratifying the land into units that respond similarly to disturbance agents that managers might want to control. Since susceptibility to disturbance varies greatly across the landscape, managers must understand and be able to communicate this variability. In an attempt to parallel current efforts by foresters to develop timber management guidelines for different habitat types (see, for example, Pfister and others 1977), Cole (1982a) classified the vegetation of a portion of the Eagle Cap Wilderness in Oregon. Drawing on observations, vegetation sampling along trails and in campsites, and analysis of stand structure, each vegetation type was characterized as to its suitability for trails and campsites, the likely effects of fire suppression, site rehabilitation potential, and other likely management problems. Although limited in scope, this attempt illustrates the potential value of a land stratification system for organizing and communicating information on how best to manage various land types.

Another attempt to incorporate knowledge about the capabilities of different land types comes from Sequoia/Kings Canyon National Parks. Meadow types were classified and then studied to determine their potential to support recreational grazing (DeBenedetti and Parsons 1983). As a result of this research, Park managers have proposed a recreational grazing plan that (1) is sensitive to annual variation in climate, (2) considers the nature of specific meadows, and (3) is sensitive to the inherent ability of individual species to withstand grazing and trampling. Other offshoots of the program that will become increasingly valuable with time are a monitoring program and the designation of selected meadows representative of each major meadow type to be protected from all stock use, to serve as benchmarks for comparison with grazed meadows.

Clearly there is much value in basic research into ecological conditions and processes—both to the advancement of biology and to the management of wilderness. The keys to maximizing value lie in analyzing the information needs of individual areas, promoting the use of wilderness for biological research, and increasing commitment for the basic inventory and monitoring needed to meet even the most fundamental wilderness goals.

RESEARCH ON SOIL AND VEGETATION IMPACTS

Across the wilderness system as a whole, the agents of change associated with human activities that have most affected natural vegetation and soil conditions are probably fire and grazing of domestic livestock. The effects of fire suppression, including longer intervals between fires, buildup of fuels, and unusually large catastrophic fires, are expressed across vast wilderness acreages. Future attempts to return fire to a more natural role, either through allowing selected fires to burn or through scheduled ignitions, will also affect vast acreages. Research into the natural role of fire, its effects, and its management has been one of the most active fields in wilderness research. Refer to the review paper presented in this conference by Bruce Kilgore for more detail.

Although not well documented, impacts of domestic grazing animals, particularly cattle and sheep, have been pronounced in many wildernesses, particularly in the West (Vale 1977; Vankat and Major 1978). Many areas supported huge herds of animals during the late 19th and early 20th centuries. Although herds were cut back dramatically following recognition of unacceptable levels of deterioration, livestock grazing still occurs in over 40 percent of wildernesses (Washburne and Cole 1983).

Literature on the relationship between domestic livestock grazing, vegetation, and soils is extensive. Most of it is prescriptive, however, and little of it applies directly to wilderness. Descriptions of grazing effects on natural vegetation have been of secondary importance, although they do exist—even for wilderness areas (Reid and others 1980). With few exceptions, range management studies have had objectives of maximizing sustained production, with little regard for the "naturalness" of conditions. Wilderness management could be aided greatly by the development of management prescriptions that optimize both production and "naturalness." Managers of about one-quarter of the wildernesses with grazing feel it is a problem; most of these areas are located in the Southwest (Washburne and Cole 1983). This is a major research gap.

A unique opportunity to study grazing in wilderness settings was provided in 1982 when, as part of a bill to prolong grazing in Capitol Reef National Park, Congress directed the National Academy of Sciences to study grazing and its impacts at Capitol Reef (Public Law 97-341). A select committee of eminent scientists, after reviewing the situation in the field, talking with concerned parties, and examining available literature, proposed a research program. To do the job correctly, they estimated a budget of $930,000, to be provided by the National Park Service. The proposal has been shelved. The interesting thing about this example is that it provides an objective measure of the level of funding necessary to research this subject. Examples of what it would cost to conduct wilderness research correctly (as opposed to how much funding is available) are almost as rare as funding for such projects.

Air pollution has also affected the vegetation and soils of wilderness areas. For example, ozone effects on vegetation, including increased mortality of ponderosa pine, have been documented in the San Bernardino Mountains of southern California (Cobb and Stark 1970), where several National Forest wildernesses are located. "Acid rain" effects on forests similar to those in many northeastern wildernesses are currently the subject of considerable research. Baseline data, to monitor future effects of air pollution, are being collected in other areas, from Sequoia/Kings Canyon National Parks in the West to the Shenandoah and Great Smoky Mountains National Parks in the East. Refer to the review paper presented in this conference by Kent Schreiber and James Newman for more detail on air quality and its management.

The effects of introduced animals on vegetation have been an active subject of research, but almost entirely in the National Parks. For example, 20 papers (about 5 percent of those presented) at the Second Conference on Scientific Research in the National Parks were on introduced animals, their effects, and their management. Considerable attention has been focused on European wild boars in the East, feral pigs in Hawaii, burros in the Southwest, and nonnative mountain goats in the Olympics. In several cases, such as burros in Grand Canyon, this research has led to aggressive control measures. Conceivably much of the knowledge generated from these studies can be applied to wilderness managed by other agencies, but with no comparable research program.

Although less studied in wilderness, vegetation change has also resulted from the introduction of plants, such as salt cedar in riparian areas of the Southwest (Robinson 1965), and of diseases or insect pests, such as Dutch elm disease (Karnosky 1979) and the gypsy moth (Marshall 1981). Elimination of species, such as large predators, has also affected the biota, although again these effects are not well documented. Wilderness management can probably benefit from active research programs, particularly related to insects and disease, outside wilderness.

A final controversial source of change is mining. Only about one-third of all wildernesses have mining claims or developments (Washburne and Cole 1983), and effects on vegetation and soil where mining does occur are likely to be highly localized. Transportation systems may be more disruptive than the mining itself, although improper disposal of tailings can have widespread effects. Much research, outside wilderness, has been directed at minimizing and rehabilitating impact. However, important constraints in wilderness that make direct application of nonwilderness research problematic include the importance of "natural conditions," the lack of mechanized equipment, and the prevalence of extreme environmental conditions that make regeneration more difficult. Research analogous to that being done outside wilderness, particularly on methods for rehabilitating mining disturbances, could be extremely helpful to wilderness management.

Of the significant causes of impact on vegetation and soils, the cause with at least a modestly developed research base that is not being dealt with elsewhere at this conference is recreation. Recreational impacts are a significant problem, at least locally, in most wildernesses. They are more subject to management control than air pollution, for example, and they are the subject of a discrete discipline—recreational ecology. This discipline—its development, its methodologies, the results it has provided, its application, and its future—will be the focus of the rest of this paper.

DEVELOPMENT OF RECREATIONAL ECOLOGY RESEARCH

Beginnings

The field of recreational ecology is now over 50 years old. The earliest study I could find was an examination of recreational impacts on the California Redwood State Parks (Meinecke 1928). Similar nonrigorous descriptions of impact in recreation areas typified the state of the field, particularly in the United States, until the mid-1960's. Most of this work was conducted in the National Parks; examples include the documentation of packstock impacts in Sequoia and Kings Canyon National Parks by a series of researchers from Armstrong (1942) to Sumner (1968), of subalpine meadow damage at Mount Rainier (Brockman 1959), and of impact problems in Grand Teton National Park (Laing 1961; Merkle 1963).

During this same period of time, academic interest in the ecological effects of recreation developed. Lutz (1945) studied soil changes on picnic sites in Connecticut. Several theses—Thornburgh's (1962) study of soil compaction and vegetation change on backcountry sites in Mount Rainier National Park and what is now Glacier Peak Wilderness, Hartesveldt's (1963) study of soil compaction and growth of giant sequoia in Yosemite National Park, Willard's (1963) study of recreational impacts on alpine tundra in Rocky Mountain National Park, and Wagar's (1964) treatise on carrying capacity—were also completed. These studies were more rigorous and quantitative; they provided estimates of amount of change by comparing recreation sites with controls. They also employed a wider variety of measurement techniques. This set the stage for efforts to more thoroughly describe the nature and significance of recreational impacts. It is worth noting that these studies examined a wide range of situations along the recreational spectrum from heavily used roaded parks and picnic areas to remote campsites in wilderness.

Europeans were taking a somewhat different tack at this time. In England, particularly, most concern was with the impact of informal countryside recreation—strolling and picnicking in a rural setting. Compared with the United States, use levels were high but not as highly concentrated at such destination sites as designated picnic grounds or campsites. This difference may explain why early European recreational ecology studies were less likely to be descriptions of conditions on recreation sites. Instead they tended to focus on the effects of a more generic activity—trampling. This focus on activity, as opposed to site conditions, placed more emphasis on process and may explain why the Europeans moved more rapidly into the experimental phase of impact study.

Pioneering work along these lines was done by Bates (1935), in England. He described the conspicuous vegetational gradient perpendicular to trails—from bare earth, through a short vegetation of trampling-resistant species, to natural vegetation—noting that changes in species composition reflected differential tolerance both to direct mechanical injury of vegetation and to the indirect effects of soil change. His greatest contribution was probably his use of experimentation to examine the relative importance of these two mechanisms of change. Although none of the European literature deals with situations analogous to wilderness or backcountry, much of it contributes to our understanding of wilderness impacts due to this emphasis on the effects of trampling.

Early Development

These beginnings served to define the magnitude of the problem and outline many of the components in need of examination. This led to a period of time during the 1960's and early 1970's when research in recreational ecology, supported almost entirely by governmental agencies charged with managing recreational and natural areas, intensified. Early in the 1960's the most noticeable developments were in the United States. Developed campsite conditions and management were studied by a number of Forest Service researchers—Magill and Nord (1963) in California, Wagar (1965) and his associates in the Intermountain West, Ripley (1962) in the southern Appalachians, and LaPage (1962) in the Northeast. Cooperators at the University of Minnesota began a series of projects examining wilderness campsites in the Boundary Waters Canoe Area (Frissell and Duncan 1965). The National Park Service also sponsored a study of conditions on developed campsites in Rocky Mountain National Park (Dotzenko and others 1967) and how damaged sites can be rehabilitated (Jollif 1969).

These studies improved knowledge in three general areas: (1) the importance of factors that influence amount of impact, such as amount of use or environmental conditions (LaPage 1962; Ripley 1962; Wagar 1964; Frissell and Duncan 1965; Dotzenko and others 1967; McCool and others 1969); (2) change in campsite conditions over time (LaPage 1967; Magill 1970; Echelberger 1971; Merriam and others 1973); and (3) methods for improving vegetation conditions on deteriorated campsites (Jollif 1969; Cordell and James 1971; Beardsley and Wagar 1971; Cordell and others 1974; Beardsley and others 1974).

In Europe, research activity intensified in the late 1960's. Two conferences in 1967, one sponsored by Great Britain's Nature Conservancy (Duffey 1967), the other by the International Union for the Conservation of Nature and Natural Resources (1967), focused attention on the ecological effects of recreation. Two active centers of research were the Nature Conservancy's work on problems resulting from new ski developments on Scottish mountain tundra (Watson and others 1970; Bayfield 1971, 1973, 1979) and the work of Conservation Course students from University College, London (Speight 1966; Goldsmith and others 1970; Burden and Randerson 1972).

These studies were notable in their attempt to interject more rigor and quantification into recreational ecology. More elaborate experimental designs and techniques were developed, and there were some early attempts to use multivariate statistics. Liddle and Greig-Smith (1975), studying the effects of trampling on sand dune vegetation in Wales, followed in this tradition. They utilized experimentally controlled applications of trampling to develop mathematical models relating amount of trampling to consequent effects. On campsites in northern Michigan, Legg (1973) developed multiple regression equations that predicted level of impact on the basis of site factors—an extension of analytical techniques first used by Wagar (1964).

Another subject that received some work in the early 1970's was trail deterioration and its management. The vegetation on and along trails had been examined as early as Bates' (1935) study but primarily to better understand the effects of trampling. Ketchledge and Leonard (1970) provided an early estimate of erosion rates on trails in the Adirondack Mountains. More rigorous studies followed. Root and Knapik (1972) and Bayfield (1973), working in the Canadian Rockies and the Scottish Highlands, respectively, studied the relationship between trail condition and various site and design factors. Dale and Weaver (1974) looked at the relationship between trail width and depth and amount of use, whether use was by hikers only or horses and hikers, and whether the trail was located in meadow or forest.

Another research program that should be mentioned here is the series of studies undertaken between 1972 and 1975 by Parks Canada in the Canadian Rockies (for example, Landals and Knapik 1972; Landals and Scotter 1974). These reports were general assessments of impact problems, particularly on campsites and along trails, with recommendations for management. Although study methods generally lacked rigor and the reports were never published, this set of reports probably represents the most thorough assessment of impacts and their management available anywhere.

By the mid-1970's, then, the focuses of research concern had been established and a sizable quantity of research data had accumulated. The basic research questions being addressed were (1) what changes occur on campsites/picnic sites? (2) what changes occur on trails? (3) what changes result from trampling? (4) how do site conditions change over time? (5) how do impacts relate to use and environmental factors? and (6) how effective are certain management techniques at avoiding or rehabilitating impact? The first syntheses of the recreational ecology field also were compiled (Speight 1973; Goldsmith 1974; Liddle 1975a). With recreational use and consequent impact problems increasing greatly every year, it appeared that the field of recreational ecology was ripe for moving from its largely descriptive and uncoordinated adolescence into a more rigorous, focused, mature phase.

Recent Development

When we look at research activity in terms of number of publications, the late 1970's appear to be a "golden age" for recreation ecology. The total number of publications in the late 1970's was twice that of the early 1970's (fig. 2). However, this number was inflated by several conferences—two on scientific research in the National Parks and one on recreational impacts on wildlands. These three conferences generated almost one-third of the published papers in this period, and the vast majority of these papers offered only modified versions of data published elsewhere.

By the early 1970's, the Forest Service had terminated all of its earlier recreational ecology research programs. In fact, between the early 1970's and 1980, I could find

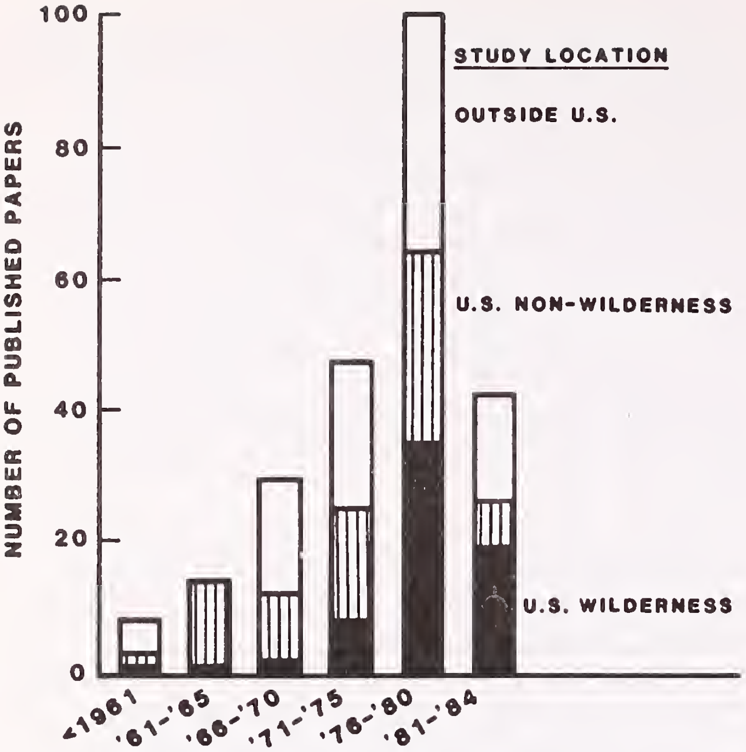

Figure 2.—Number of recreation ecology publications from various time periods. Papers in conference proceedings were considered publications, but theses were not.

published results from only two studies supported by the Forest Service—Helgath's (1975) study of trail erosion in the Selway-Bitterroot and Fay's (1975) test of fencing, fertilization, and liming as possible means of restoring vegetation to a backcountry camp in New Hampshire's White Mountains. The only other relevant papers are a haphazard collection of non-data-based papers on measurement and management techniques (Hendee and others 1976; Fay and others 1977; Leonard and Whitney 1977; Rinehart and others 1978; Frissell 1978). Most of the early researchers were still in Forest Service recreation research, but their focus had shifted to the social and planning aspects of recreation.

In contrast to the Forest Service, the 1970's was a period of growth in this field for the National Park Service. During this period recreational ecology papers resulted from supported work in Grand Canyon (Carothers and Aitchison 1976), Yosemite (Holmes and Dobson 1976; Malin and Parker 1976; Hecht 1976; Foin 1977; Lemons 1979), Sequoia/Kings Canyon (Parsons and DeBenedetti 1979; Parsons and MacLeod 1980), Denali (Stelmock and Dean 1979), Olympic (Schreiner and Moorhead 1976), Mount Rainier (Edwards 1979), and Great Smoky Mountains (Bratton and others 1978, 1979) National Parks, and Ozark National Scenic Riverway (Marnell and others 1978). Although somewhat more rigorous than the anecdotal work of the 1960's and before, these projects were largely descriptive. Objectives were usually to survey impacts over large areas. Although often quantitative in nature, only the most obvious impacts were examined, and generally neither measurements nor analyses were very sophisticated. This primarily represented an expansion of the regional coverage of impact studies. This increased coverage was valuable from the standpoint of being able to evaluate the general applicability of specific results; however, it did little to advance methodology or theory or to move the field toward a deeper plane of inquiry or understanding.

In addition to an expansion of regional coverage, Park Service studies looked, at least briefly, at two new sources of impact—campfires (Fenn and others 1976) and urine (Holmes and Dobson 1976).

Many of the papers published in the late 1970's were short-term, one-time studies by academics (for example, Hartley 1976; Coombs 1976; Jones 1978; Rutherford and Scott 1979). Presumably, students interested in the subject conducted these studies, with the assistance of a professor who found the subject an interesting change, and then both went on to do something else. New places were studied and new information was uncovered, but there was no building, coordination, or deepening of the field.

Many countries were involved in recreation ecology research by the late 1970's. Work continued in Great Britain but at a much reduced rate. Recreation ecology research was also done in Finland (Kellomäki 1977), the Netherlands (Boomsma and van der Ploeg 1976), Poland (Falínski 1975), Sweden (Byran 1977; Emanuelsson 1979), the U.S.S.R. (Rogova 1976; Kazanskaya 1977; Spiridinov 1979), Australia (Edwards 1977), and Canada (James and others 1979).

Generally reviewing the 1970's, we see expansion in the field of recreation ecology. Scientists from more countries were studying recreational impacts. Within the United States, increased interest on the part of the Park Service partially offset the demise of Forest Service research programs. All major lines of inquiry were advanced, and new ones, particularly the development and evaluation of management techniques and monitoring systems, appeared. The major problem was lack of continuity. The field remained dominated by short-term, one-of-a-kind studies that did not build on each other.

CURRENT STATE OF THE DISCIPLINE

In the 1980's, as managers of more and more wildernesses are struggling to deal with recreational impact problems, we would like to think that this field of inquiry is continuing to mature. Unfortunately, there is much to suggest that this is not the case. Productivity has dropped radically; from an average of 20 papers per year in the late 1970's, the rate in the early 1980's has dropped to 10 per year. Participation has dropped even more radically; of 44 papers I could find that have been published since 1980, 18 are by just two authors.

Within the Forest Service, currently about 20 professional scientists are doing recreation research. None of these researchers emphasize recreational ecology. Reviewing Forest Service recreation research publications between 1961 and 1982, I found only 33 out of 932 references (3.5 percent) dealing with recreation impacts and their management. This compares with 100 on visual resource management, 94 on user descriptions, preferences, and benefits, and 62 on how to assess use. The Forest Service does support recreational ecology research by outside cooperators; however, such research has been conducted in only five wilderness or backcountry areas.

A similar situation exists in the Park Service, although here the emphasis is on basic biology rather than social research. Of the papers presented at the Second Conference on Scientific Research in the National Parks in 1979, 5 percent dealt with impacts of recreation on the natural environment. This compared with 5 percent for water, 5 percent for air, 12 percent for social recreation research, 13 percent for fire, and over 50 percent for basic biology.

It is more difficult to assess participation outside these agencies. No one in the U.S. Department of the Interior, Fish and Wildlife Service or Bureau of Land Management appears to be doing recreation impact research. Moreover, research by the academic community is sporadic at best. I could find only 50 authors (or groups of authors) who have reported—in published or unpublished form—data-based research on recreation impacts in wilderness/backcountry settings. Only 10 of these 50 participated in more than one study.

This limited participation would be understandable if ecological impacts and their management were not a problem. However, this is clearly not the case. We conducted a survey of managers of all existing wildernesses and many likely additions to the wilderness system in 1980 (Washburne and Cole 1983). Our results corroborated those of an earlier sample of 35 wilderness managers (Godin and Leonard 1979), finding that recreational impacts, particularly on trails and campsites, were the most common wilderness management problem in the eyes of managers.

Outside the United States, the condition of the discipline is as bad—if not worse. Activity in the 1980's is less than at any time since the early 1960's. In 1983, I attended a conference on the ecological impacts of outdoor recreation in Europe and North America. Organized by the Recreation Ecology Research Group—a primarily British group that includes some of the first scientists to work in the field—the conference had the ambitious goal of bringing together most of the researchers in the field. From discussions with many of the attendees, it appears that only a couple of people in the world are currently able to pursue careers in recreational ecology research.

Perhaps the one bright point, from the perspective of this conference, is that a large proportion of the research in the 1980's has been in wilderness in the United States. Although recreational ecology is at its lowest ebb since the 1960's, more new studies in wilderness have been published since 1980 than in any other 5-year period. We now have published data on ecological impacts from nine National Forest wildernesses: Boundary Waters Canoe Area, Eagle Cap, Frank Church-River of No Return, Selway-Bitterroot, Lee Metcalf, Bob Marshall, Rattlesnake, Alpine Lakes, and Glacier Peak; from wilderness or backcountry in 13 National Park areas: Grand Canyon, Grand Teton, Great Smoky Mountains, Dinosaur, Mount Rainier, Denali, North Cascades, Olympic, Sequoia/Kings Canyon, Yosemite, Rocky Mountain, Guadalupe Mountain, and Ozark National Scenic Riverway; as well as from the Adirondack Mountains in New York, the Mission Mountains Tribal Wilderness in Montana, and the Green Mountains of Vermont (fig. 3).

Given that we now have about 90 million acres of wilderness, in over 450 different units, these 24 areas represent a meager start at even describing impact problems. The best represented parts of the system are, in order, the Boundary Waters Canoe Area (BWCA), the Northern Rockies, the Sierra Nevada, and the Pacific Northwest. Major gaps exist in the East, the Central States (except for the BWCA), the Southwest (except for the Sierra Nevada), and Alaska.

A problem that may be more significant than inadequate regional representation is the declining trend in research sophistication. An increasing proportion of papers are not based on data. There has been relatively little progress in the development of either theory or more sophisticated research techniques.

In a review of recreation ecology literature in Europe in 1976, Satchell and Marren stated:

> We found a great disparity between the relative abundance of data on recreational demand such as numbers, origins and attitudes of visitors, and the paucity of data on the ecological consequences of recreational activities .... We consider that the current level of research is not commensurate with the magnitude of the problem of reconciling the maintenance of amenity and conservation interest in areas used for public recreation with the demand for outdoor leisure pursuits in increasingly urban societies.

Figure 3.—Backcountry/wilderness areas in which recreational impact studies have been conducted.

Clearly these disparities are even worse a decade later—in Europe and in the United States.

In another review, Goldsmith (1974) wrote that most studies merely "record observations of a rather superficial nature and only a few describe specially designed experiments with detailed analysis of the resultant data." This situation has also not improved dramatically. Most research continues to merely document the obvious; time frames are short, theory is lacking, and few studies are comparable.

None of this is very surprising when a field is only the part-time interest of a few researchers. Even today there are essentially no careers available in recreational ecology. Only with the possibility of a career can we expect people to undertake the long-term work that is critically needed or to design complex studies that examine all the variables that influence impacts. Only then will studies build on one another or will students be able to get training in the field. Only then will it be possible to develop the critical mass necessary to stimulate creative thinking and the development of theory.

It has been suggested that the ecology of recreational impacts is known well enough and is seldom more than documentation of the obvious. This seems to me to be more an indictment of what the field has done than what it could do. Because support for this discipline has never solidified, the field is characterized by an ebb and flow of personalities all starting at a low level of sophistication and moving on to greener pastures before any significant advancement can occur.

RESEARCH METHODS

Study Designs

A variety of study designs have been utilized in recreational ecology studies. I choose to place them in four categories: descriptive surveys of recreation sites, comparisons of used and unused sites, before-and-after natural experiments, and before-and-after simulated experiments. Within each of these broad categories there can be tremendous variation in spatial and temporal scales, as well as the soil/vegetation parameters under study.

The simplest and, to date, the most common of these approaches is the descriptive survey of recreation sites (for example, Bratton and others 1978, 1979). Vegetation and soil parameters are measured or estimated to determine the current condition of the resource. Variability across an entire area can be assessed, and conditions can be compared to objectives to determine whether or not problems exist and management actions are required. Site conditions can be correlated with use and environmental conditions to suggest hypotheses of cause and effect. Conditions can also be followed over time to establish trends.

The two basic problems with this approach are that (1) there is no measure of impact or change and (2) it is difficult to evaluate cause and effect. Existing conditions are affected by a wide variety of environmental as well as use parameters. Fifty percent vegetation cover on a site may be perfectly natural, or it may represent a dramatic loss of cover. Without a comparison to undisturbed conditions, it is impossible to evaluate the effects of recreational use. Moreover, there is also the problem of very little information on what has actually happened to the sites under study. Recreational impact is variable from year to year, group to group, and site to site. Surveys of site condition, alone, are incapable of substantially improving our ability to predict the consequences of various types and amounts of use in various places.

The great value of these studies is in their immediate management utility. They can usually be completed relatively rapidly, without a high investment in training, and are useful in assessing the general condition of the resource and monitoring its change over time. Thus these studies can form the backbone of information needed to guide day-to-day management.

Another common research approach involves comparison of used and unused sites (for example, Cole 1982b). With this approach, ecological impact is estimated on the basis of a comparison between conditions on recreation sites and neighboring undisturbed sites (controls) with similar environmental settings. The implicit assumption is that the recreation site was similar to the control prior to use and, therefore, that any difference between the two sites is the result of recreational use. Once these estimates of change are obtained, it is a relatively simple matter to compare change on sites with different use and environmental characteristics—utilizing a cross-sectional design—in order to better understand how these affect amount and type of impact.

This is a very convenient and attractive approach because, in contrast to the previous approach, it does provide a measure of change. Moreover, changes—despite occurring over a long period of time—can be evaluated at one point in time; long study periods are not required.

However, two major sources of error exist that can make the results of this approach misleading. First, controls never perfectly replicate preexisting site conditions. Therefore some of the difference between recreation sites and controls is a result of environmental variability rather than recreational use. This problem becomes increasingly severe as local environmental variability and the uniqueness of sites selected for recreation use increase. In some situations it is also impossible to find controls that have not been affected by recreational use. Care in selecting controls and disqualifying sites without adequate controls will counter some of these problems. The environmental variability problem can be reduced with a large sample size, and its effect can be better interpreted with an evaluation of inherent variability between control sites.

Second, it is difficult to identify, let alone control, all the ecological and human-use variables that affect the amount of change occurring on a site. For example, a common objective is to assess the effects of amount of use on amount of impact. To do so requires accurate measures of use, which are seldom available. Even where current use can be assessed, there is little certainty that current use patterns reflect the past use history of the site. Moreover, the effect of amount of use is strongly modified by type of use, timing of use, and the resistance of the site—all of which are difficult to assess and virtually impossible to control perfectly.

The solution to these problems is not to abandon this approach; rather, the solution lies in controlling variability as much as possible, studying the influence of these other variables, sampling as many sites as feasible, not exceeding the limitations of data in the analysis phase, and qualifying the final results. For example, rather than treating 1 year's use data as interval level data and regressing it against some measure of impact, it is much more realistic to establish discontinuous classes of use and use analysis of variance (for example, Marion and Merriam 1985). The analysis may be less sophisticated, but the results are less likely to be misleading. Similarly, unless the magnitude of statistically significant differences is great, researchers would be prudent to recognize the importance of uncontrolled variability and play down the importance of these differences.

Most of these problems are more effectively dealt with by using the longitudinal before-and-after natural experiment approach (for example, Merriam and others 1973). In these studies, site conditions are measured before use of the site commences and then after it has occurred. Changes identified in this way are more accurate assessments of recreational impact because the error caused by assuming a site was identical to a control is eliminated. Ideally, conditions will be measured periodically and over a long enough period of time for the effects of recreational use to equilibrate. If this is not done, this approach loses value because it will not be possible to predict the consequences of long-term use, the primary objective of most research. It is also important to follow changes on control sites in order to incorporate nonrecreation-related changes into the final interpretation of results. Finally, it is imperative to accurately evaluate both amount and type of use on the site. If this is not done—and it is extremely difficult to do in backcountry—this approach loses many of its advantages over the cross-sectional approach previously discussed.

A variation on this approach is to evaluate the effects of a change in use or management or both by taking measurements before and after the change. Examples might include examining the effect of packstock on a trail that had never had stock use before or recovery on a campsite that was being closed to all use (for example, Stohlgren 1982). Such natural experiments can be enlightening. However, they need to be undertaken under a wide variety of situations before it will be possible to evaluate the general applicability of results.

To get beyond entirely site-specific conclusions, longitudinal studies need to utilize factorial designs. For example, the effect of amount of use on sites located in various environments could be evaluated by stratifying sites environmentally and examining sites receiving a wide range of use. However, this brings out the primary disadvantage of this research approach—its cost, both in terms of time and money. Sites need to be measured periodically, probably at least once per year (and preferably more frequently) for at least 5 years. Use measurements need to be taken continuously over this time period. In backcountry this requires a lot of field time. Moreover, results will not be in hand for a long period of time. Finally, it is often difficult to find the real-life situations needed for the factorial design or to even predict which sites will fall into which use category to ensure sufficient sample sizes in all cells of the factorial design.

The common solution to all of these problems is the simulated experimental approach (for example, Weaver and Dale 1978). The major difference between this and the preceding approach is that the investigator plans the application of recreational use. This makes it much easier to find the situations to be examined in a factorial experiment. It also makes use measurement easier and more accurate, and it makes it easier to eliminate variability in parameters not under study, such as whether trail use is by horse or hiker, or whether use occurs when soils are wet or dry. Occasionally, this approach has involved dropping an artificial "foot" or rolling a corrugated roller along the ground. More frequently, the investigators trample areas themselves, walking back and forth at known rates.

To be of most value, it is important to take measurements before and after each treatment. Changes also need to be followed on controls. Even where this is done, local environmental variability will interject some error into estimates of recreation-caused change because the effects of a given amount of use will vary with the tolerance different species have for impact. To predict the long-term consequences of continual use, it is best to both apply use and measure impacts over a period long enough for effects to equilibrate. This is costly and has never been done.

A major drawback to this approach is that it seldom truly simulates the type of recreation use of concern to managers. It is much more directly applicable to problems of trail deterioration and dispersed trampling than to campsite damage. It would be possible for investigators to camp different numbers of times on different vegetation types and measure responses. This would be very costly, however, and there are still variables, such as weather conditions, that cannot be controlled.

Each of these approaches, then, has inherent advantages and disadvantages. The more scientifically rigorous approaches are more costly and time-consuming and less likely to simulate "real" conditions. The other options have more error in their estimates of recreational impact and are less capable of unraveling cause and effect. All approaches can provide valuable information. The key is to match the approach to the objectives and situations at hand, while carefully designing a study that minimizes the problems inherent to any single approach. Finally, most light will be shed on the subject by utilizing several approaches simultaneously. For example, descriptive surveys can help managers assess their current situation and provide baseline data for a monitoring program. Cross-sectional studies, in a factorial design, can suggest what factors managers might want to manipulate to minimize impacts. Long-term simulated experiments can explore these hypotheses in more detail to "fine tune" future management, while longitudinal studies are useful for evaluating the effectiveness of new management policies taken to counteract problems identified in the descriptive surveys.

With few exceptions, recreational ecology studies have only examined obvious impacts and have operated at an intermediate level of resolution. There have been few detailed studies of the process whereby a change such as a shift in species composition occurs. Likewise there have been few studies of interrelationships between ecosystem components affected by recreational use. This neglect of both process and interrelationships at the ecosystem level makes it difficult to evaluate the significance of recreational impacts in anything other than esthetic terms.

Vegetation and Soil Parameters

When describing vegetation response to recreation, most attention has been given to description of the community and damage to trees. Descriptions of morphological and physiological responses are much less common. Vegetation is usually sampled along transects or in plots of varying size and configuration and then described in terms of percentage cover of total vegetation and each species. Many studies have used these data to characterize species composition, diversity, and life or growth form spectra. Fewer studies report frequency, density, or biomass. Population structure has been described, for *Juncus trifidus*, by Pryor (1985). Fletcher and Shaver (1983) studied plant demography of disturbed *Eriophorum vaginatum* populations.

In forested areas, investigators commonly report frequency of tree damage and root exposure and density of tree reproduction. A few also have reported diameter class distribution and have observed apparent declines in vigor and growth rates.

The most frequently reported morphological parameters are vegetation height, growth form, and bud location. Several studies have also reported number of flowers per plant. In perhaps the most indepth description of morphology, Liddle (1975a) reported data on leaflet area for *Trifolium repens* and on number of tillers per plant, length of live tillers, number of live leaves per tiller, dry weight per shoot, and dry weight per tiller for *Festuca rubra*. Goryshina (1983) presented data on leaf area and cell size and number for three forb and two moss species. He also described thallome thickness and size and number of algal cells and hyphae in two lichen species.

In keeping with the paucity of information on the impact process, there are very few data on physiological responses to use. The only physiological responses examined at all are seed production, germination and dissemination, seedling establishment, and changes in the carbohydrate content of roots. However, Blom (1976, 1977), who studied the effects of trampling and soil compaction on the emergence and establishment of four *Plantago* species, has shown the direction such studies could take if we wanted a more detailed understanding of the physiological basis for observed impacts on vegetation.

Descriptions of soil changes, if anything, have been even more superficial. Most studies report loss of organic horizons, usually as changes in the percentage cover of litter and/or exposed mineral soil. A few also report the depth of organic horizons. Considerable attention has been given to soil compaction, with investigators usually reporting either penetration resistance or bulk density of the soil. While bulk density has the advantage of not being subject to moisture conditions, penetration resistance is a more sensitive indicator of impact. Others have measured consequences of compaction, particularly changes in total porosity, pore size, and infiltration rates. There are also some data on moisture content, organic matter content, and evidence of erosion. Soil properties for which very limited data exist include pH, nutrient content, texture, aggregate stability, and soil biota.

A few studies have attempted to model recreational effects on erosional processes utilizing the Universal Soil Loss Equation (Kuss and Morgan 1980, 1984; Morgan 1985). Such theoretical work has potential, but has yet to be translated into practical application.

As with the evolution of the recreational ecology discipline as a whole and the development of theoretical structure, research methods have not advanced greatly since Bates' (1935) early work on trampling effects on vegetation. Experimental work is receiving more emphasis but is not fundamentally different from earlier work. Similarly, more advanced techniques and equipment are available for measuring soil and vegetation properties but are seldom used.

RESEARCH RESULTS

Over the more than 50 years of research on recreational impacts, a considerable body of information has developed. Much of it is highly site-specific, but many general conclusions can be drawn from this work. I will begin by discussing descriptive studies of the effects of recreational activities on different types of recreation sites and the spatial and temporal patterns these impacts exhibit. Following these, I will examine the factors that influence type and magnitude of impact and studies that evaluate the effectiveness of management techniques. I will conclude with a discussion of work on monitoring and site rehabilitation techniques. Research in wilderness will be emphasized, but I will draw on information gleaned from studies undertaken in more developed settings. Other syntheses of research results include Speight (1973), Liddle (1975a), Satchell and Marren (1976), Wall and Wright (1977), Manning (1979), Hart (1982), Kuss and others (1985), Price (1985), and Cole (1985a).

Descriptive Studies of Recreational Impacts

The three primary activities by which recreationists alter wilderness soil and vegetation conditions are (1) trampling by humans and packstock, (2) the collection and burning of wood in campfires, and (3) the confinement and grazing of recreational stock.

Before turning to a detailed discussion of the impacts associated with these three activities, several unstudied sources of change should be mentioned. The first of these is change caused by formal construction of campsites and trails. Campsite construction by managing agencies is rare but does occur. Generally this involves brush and tree removal and leveling of the ground surface. Trail construction, however, is common and probably has a greater effect on vegetation and soils than use of the trail by recreationists. During construction, vegetation is removed, soil is compacted, drainage is altered, and topography is rearranged.

Another unstudied source of change is the transportation of foreign substances into the wilderness. The propagules of exotic plant species are often brought unwittingly into the wilderness—on or in humans and packstock. Once inside they frequently establish themselves in disturbed areas. Garbage, soap, and waste water, particularly when scattered around campsites, can affect soil chemistry and biota; effects are likely to be highly localized, however. Finally, the effects of urinating on plants were investigated in Yosemite. This study concluded that although urine can lead to desiccation of plant tissues and increase the likelihood that a plant will be eaten, such effects are not pervasive enough to be significant (Holmes and Dobson 1976).

Effects of Trampling.—The most obvious effect of trampling is injury and destruction of ground-level vegetation. Injury results in reduced vigor and reproductive capacity in most species. At high levels of trampling or when fragile species are trampled, all plants may be eliminated. Some species may increase in abundance, however, and new species may be introduced. Increases are more likely to reflect reduced competition or a change in microhabitat than a positive response to trampling impact. For example, Pryor (1985) found that moderate levels of trampling increased the abundance of small *Juncus trifidus* plants, a species that germinates frequently on the bare and gravel surfaces that expand as a result of trampling. The final and most frequently documented results of trampling are loss of cover, reduction in stature, and shift in species composition.

The precise effects of trampling on plant morphology and physiology are poorly understood, as are the reasons some plants tolerate trampling better than others. Trampling reduces the area of individual leaves (Liddle 1975a). Goryshina (1983) demonstrated that this results more from inhibition of cell division than of cell elongation. Number of leaves and tillers, in grasses, is little affected by trampling (Liddle 1975a); however, number of leaves per shoot is reduced in orchids (Bratton 1985). Height is reduced and prostrate branching tends to increase as a result of frequent damage to terminal buds. Reduced height and leaf area decrease photosynthetic area; this depletes carbohydrate reserves (Hartley 1976) and can affect plant vigor. Physiological stress, such as reduction in photosynthetic area and carbohydrate reserves, along with mechanical damage to elevated buds, may explain reductions in flower density and seed production per flower (Liddle 1975a; Hartley 1976; Bratton 1985). However, few relevant studies have been conducted; this makes it difficult to evaluate whether these results are generally applicable or only apply to the species under study.

Trampling also compacts soils. Compaction reduces porosity, particularly the volume of macropores. This tends to reduce water-holding capacity in fine-textured soils and increase it in coarse-textured soils. Infiltration rates are universally reduced, leading to increased runoff and erosion potential. Other likely consequences that have only been studied in an agricultural context include oxygen shortages and changes in soil biota. Because these processes have never been studied in detail, we do not know with any surety what it is about compacted soils that makes vegetation reestablishment so difficult.

Many of these effects on soil are exacerbated by the abrasion and loss of organic horizons, a loss that becomes particularly pronounced following a reduction in vegetation cover. Loss of organic matter increases susceptibility to compaction and results in increased runoff. The soil biota is likely to be affected further, as is the germination capacity of species that prefer organic seedbeds. There is conflicting evidence regarding trampling's effect on organic matter incorporated within the mineral soil. Young and Gilmore (1976), Legg and Schneider (1977), Monti and Mackintosh (1979), Cole (1982b), and Marion (1984) found increases in soil organic matter content on campsites, while Dotzenko and others (1967), Dawson and others (1978), Rutherford and Scott (1979), and Stohlgren (1982) found decreases. This discrepancy does not appear to correlate with differences in vegetation, soil, climate, campsite age, amount of use, or measurement technique.

All of these changes are likely to affect germination, establishment, growth, and reproduction of plants. Compaction reduces the heterogeneity of soils and, therefore, the density of favorable germination sites (Harper and others 1965). Compaction increases the mechanical resistance of the soil to root penetration. This can reduce emergence of seedlings, although in some dry soils and for some species, this adverse impact is less important than the beneficial effect of increased water availability in compacted soils (Blom 1976). Once plants are established, low levels of compaction can lead to increased growth rates (Liddle 1975a). This probably reflects increased moisture again, and may not occur in soils where moisture is not limiting. At some level, compaction leads to oxygen shortage which, along with the mechanical resistance of the soil to penetration, inhibits root growth and, therefore, plant growth, vigor, and reproductive capacity. Effects on these processes are very poorly understood.

Impacts Associated with Campfires.—Ecologic impacts associated with campfires result from both the removal of wood, either dead or live, standing or on the ground, from large areas around campsites, and from the burning of this wood in campfires (Cole and Dalle-Molle 1982). Very little research has been conducted on this subject. The collection of firewood and associated trampling greatly enlarge the area affected by camping. In Great Smoky Mountains National Park, the area disturbed by firewood collection was typically more than nine times the size of the devegetated area around campsites. In this much larger area, number of live and dead trees, particularly those in smaller size classes, and woody fuels were reduced (Bratton and others 1982). Shifts in understory species composition—presumably a result of trampling—were also evident in areas disturbed primarily by firewood collection (Saunders 1979). In Grand Canyon National Park, where campfires are prohibited, the areal extent of disturbance around campsites is remarkably small given the heavy use many of these sites receive (Cole 1985b).

The significance of firewood collection to the entire ecosystem can only be suggested from studies undertaken outside the recreational context. Such studies suggest that the most serious impacts are likely to occur when woody debris larger than about 3 inches in diameter is removed. Decaying wood of this size plays an important and irreplaceable role in the ecosystem—in water and nutrient conservation and as a substrate for biological activity (Harvey and others 1979). Collection of wood that can be broken by hand is likely to have relatively little effect (Cole and Dalle-Molle 1982).

The effects of campfires are particularly severe for the areas actually burned. In one experiment, a single intense campfire eliminated 90 percent of the organic matter in the upper inch of soil (Fenn and others 1976). Fires are also known to lead to losses of certain chemical elements (nitrogen, sulfur, and phosphorus, for example), increases in pH and many cations, and decreases in moisture-holding capacity, infiltration rates, and mycorrhizal populations in the soil (Cole and Dalle-Molle 1982). However, such changes have never been studied in a recreational context. Significance has never been assessed in terms of effects throughout the ecosystem or difficulty of rehabilitating fire sites. Usually campfire impacts are highly localized, but where fire sites move around, continually being rebuilt after being removed by rangers or other campers, a sizable area can be affected.

Generally, the whole subject of campfire-associated impacts and their management has been neglected and is poorly understood. This is particularly unfortunate given the energy currently expended on educating people in proper use of campfires. Managers are developing educational messages, and social researchers are experimenting with delivery systems for these messages. And yet, the bottom line is that we have not studied the problem and its potential solutions enough to know what the message should be and how it should be tailored to different environmental and use situations.

Effects of Confining and Grazing Recreational Stock.—Recreational stock trample vegetation and soil, as hikers do. They also cause some unique types of impact resulting from their need to graze (unless grazing is prohibited and all feed is carried in) and to be confined when not in use. The primary effect of confinement is localized severe trampling damage. Because a primary means of confinement is tying stock to trees, other serious problems are root exposure, as soil is eroded from around the base of trees, and damage of tree trunks from the abrasive action of rope. Where stock are tied to small trees, girdling can kill the trees. In the Bob Marshall Wilderness, campsites frequented by parties with packstock have an average of 25 trees with exposed roots (Cole 1983a). Although this type of damage has not been well studied, the problem, its cause, and its solution are obvious; consequently, this is not a high research priority.

In contrast, the effects of recreational grazing—also not well studied—are subtle and complex, as are the pros and cons of various potential solutions. Most of what we know about grazing impact comes from range management studies outside wilderness, usually dealing with animals other than horses and mules and with management objectives that place little importance on natural conditions. Moreover, in the few studies of range condition that have been conducted in wilderness, it has not been possible to isolate the effect of recreational grazing because it is superimposed on the effects of earlier or, in many cases, ongoing grazing by domestic livestock.

Sequoia/Kings Canyon National Parks are the only wildernesses where the effects of recreational stock on meadows have been assessed in detail. Over 40 years of research have been conducted, from early qualitative surveys of meadow conditions (such as Armstrong 1942) to recent controlled clipping experiments. These studies have provided a basic understanding of the nature of problems, trends, and relationships between meadow conditions and important environmental and use variables. The payoff to management is a proposed stock management plan that is efficient (meaning management objectives are met without the imposition of unnecessary restrictions) and thorough (DeBenedetti and Parsons 1983). Implementation of the plan is currently stalled due to objections from some of the recreational stock users.

Early meadow surveys in the Parks identified problems of reduced vegetation cover, rill, channel, and gully erosion, and invasion of meadows by lodgepole pine and other "weedy" species. Trend studies showed that actions taken to deal with these problems, including closing certain areas to grazing and limiting numbers of animals and lengths of stay, were generally successful. However, more subtle changes, not readily identifiable in the early qualitative surveys, were occurring, and management success varied from place to place and year to year. Consequently, more intensive quantitative research was conducted in the 1970's. This research provided a better understanding of annual fluctuations in the productivity of major plant associations and identified the response of each association to different levels of herbage removal. A predictive index of susceptibility to trampling under different moisture conditions was produced for each association. This allows managers to predict the consequences of alternative use prescriptions for individual forage areas, even accounting for differing hydrologic years. In addition to managing use effectively, the plan also incorporates a monitoring plan and designates a network of representative meadows for each major association to be protected from stock use (DeBenedetti and Parsons 1983). Other areas would do well to emulate such a research and management program rather than ignore the problem or institute arbitrary actions not based on an understanding of the resource, its condition, and its variability.

Impacts on Campsites.—Although affected by some of the same activities, impacts on trails and campsites—the two major focuses of impact and management—are sufficiently different to be discussed separately.

Impacts on campsites result from trampling by humans and, sometimes, by stock; the collection and burning of firewood; the confinement and grazing of stock; pollution with garbage, soap, and other substances; and thoughtless or malicious acts, such as hacking trees. Of these, trampling is the only unavoidable activity. The others are optional, and effects can be eliminated through education or prohibition of activities. Trampling impacts and their management are more subtle and complex and will be covered in more detail here.

Currently we have research data on campsites from 15 wilderness/backcountry areas: Olympic National Park (Schreiner and Moorhead 1979), Eagle Cap Wilderness (Cole 1981a, 1982b), Sequoia/Kings Canyon National Parks (Dykema 1971; Simon 1978; Parsons and DeBenedetti 1979; Stohlgren 1982), Bob Marshall Wilderness (Cole 1983a), Lee Metcalf Wilderness (Frissell 1973), Mission Mountains Tribal and Rattlesnake Wildernesses (Fichtler 1980; Cole and Fichtler 1983), Selway-Bitterroot Wilderness (Ranz 1979; Cole and Ranz 1983), Frank Church-River of No Return Wilderness (Coombs 1976), Dinosaur National Monument (Jerry 1977), Grand Canyon National Park (Carothers and Aitchison 1976; Cole 1985b), Boundary Waters Canoe Area (Frissell and Duncan 1965; McCool and others 1969; Merriam and others 1973; Marion 1984), Ozark National Scenic Riverways (Sutton 1976; Marnell and others 1978), the Adirondacks (Rechlin 1973), and Great Smoky Mountains National Park (Bratton and others 1978, 1982; Saunders 1979).

Half of these are descriptive studies of campsite conditions without relation to controls. They portray the extent and distribution of campsites in varying condition, but do not describe what has happened to the sites in terms of recreational impact. Of those that did estimate amount of impact, this amount is related to amount of use by Frissell and Duncan (1965), Sutton (1976), Coombs (1976), Simon (1978), Fichtler (1980), Cole (1982b, 1985b), Cole and Fichtler (1983), and Marion (1984); to type of use by Cole (1983a); to age of the campsite by Marion (1984); to environment by Dykema (1971), Sutton (1976), and Cole (1985b); and to time since the campsite was closed to use by Parsons and DeBenedetti (1979), Ranz (1979), Cole and Ranz (1983), and Stohlgren (1982).

These studies universally report the loss of vegetation cover and species change that generally result from trampling. Differences in the magnitude of change reported often reflect incomparable study methods as much as real differences. Species change and cover loss are most pronounced close to the center of campsites. Stohlgren (1982), for example, found that the core area of campsites just closed to use in Sequoia National Park had lost over 90 percent cover (in comparison to controls), compared to about 40 percent loss on less used parts of the site away from the core. Species richness and vegetation height were also reduced more in the core than on the fringes of the site. The sharpness of this intrasite disturbance gradient and the magnitude of differences vary with both amount of use and environment. In the Grand Canyon, no vegetation survives in the core of all but the most lightly used campsites; however, the area immediately adjacent to this barren core typically loses only about 12 percent cover (Cole 1985b). When intrasite variability is disregarded, reported measures of cover loss on the entire campsite range from 96 percent in Eagle Cap spruce forests in Oregon to less than 20 percent on montane grasslands in Montana's Bob Marshall (Cole 1981a, 1983a).

Shifts in species composition have been pronounced in all but two cases—the Grand Canyon and Rattlesnake Wilderness (Cole and Fichtler 1983; Cole 1985b). In these two cases, an index of floristic difference between campsites and controls had a mean of only 31 percent and 27 percent, respectively, within the range of variability expected in undisturbed vegetation. This compares with a mean index value as high as 88 percent in the Boundary Waters Canoe Area (Marion 1984). Species change is highest when resistance of the natural vegetation is low and there are many trampling-resistant invaders to take the place of the original occupants. Close to 20 percent of the species found on Boundary Waters campsites were nonnatives.

All studies—with the exception of the Grand Canyon, where no change in composition occurred (Cole 1985b)—have found that grasses and sedges increase in relative importance on campsites. Although they too are damaged by use, they survive more frequently than other growth forms. Low shrubs are generally susceptible to damage, as are tree seedlings and lichens. Loss of tree seedlings has been nearly complete on most campsites studied and is an especially serious problem for the long-term maintenance of forested campsites. Large shrubs are relatively resistant to damage, particularly if they are widely spaced and thorny, as they are on Grand Canyon sites (Cole 1985b). Forbs and mosses are highly variable in response; as a group they are neither highly resistant nor highly susceptible.

Established trees, because of their size, are generally little disturbed by trampling. Even where the practice of tying horses to trees exposes roots, tree mortality is seldom a problem, unless girdling occurs. Some tree species do appear to grow less rapidly in trampled areas (LaPage 1962; Brown and others 1977; James and others 1979). In shallow soils trampling can also increase water stress (Settergren and Cole 1970) and lead to increased windthrow (Frissell and Duncan 1965).

Although large enough to be spared most trampling damage, trees on campsites are frequently felled and used for tent poles, hitchrails, and firewood. On campsites in the Eagle Cap and the Bob Marshall, the median percentages of trees that had been felled were 28 (four trees per campsite) and 15 (15 trees). This loss is in addition to trees with exposed roots—32 percent (three trees) in the Eagle Cap and 54 percent (28 trees) in the Bob Marshall (Cole 1982b, 1983a). Root exposure is even more prevalent in the Boundary Waters Canoe Area, where shallow soils aggravate the problem (Marion

1984). Further damage is inflicted pointlessly by jackknife artists and ax-wielders. Although most trees survive such injuries, thin-barked trees, such as aspen and paper birch, are often killed (Hinds 1976; Marion 1984). Typically, few trees on campsites show no evidence of damage. Moreover, tree damage, in contrast to trampling damage, is cumulative and even longer lasting.

Loss of organic horizons follows loss of vegetation. Many campsites lose vegetation without losing litter, but the reverse is seldom the case (Cole 1982c). Loss of organic horizons occurs as a result of both trampling and increased runoff and erosion. As with vegetation loss, litter loss is greatest around the center of the campsite. Stohlgren (1982) found litter loss, by weight, to be over 90 percent of control values in the core and just over 60 percent on the fringes. Reported estimates of decrease in the thickness of organic horizons for entire campsites range from 45 to 66 percent (Frissell and Duncan 1965; McCool and others 1969; Cole 1982b, 1983a; Marion 1984). Litter loss exposes more mineral soil and rock. Amount of increase is highly variable among campsites both within and between areas. Typical increases have ranged from 6 percent in the Mission Mountains (Fichtler 1980) to 59 percent in the Grand Canyon (Cole 1985b). Differences relate both to how much use sites receive and the thickness of organic horizons on undisturbed sites (Cole and Fichtler 1983; Marion and Merriam 1985).

As was mentioned earlier, there are conflicting results concerning soil organic matter on campsites. On backcountry sites in the Eagle Cap Wilderness and Boundary Waters Canoe Area, Cole (1982b) and Marion (1984) found increases of 20 and 17 percent, respectively. On campsites in Sequoia National Park, in contrast, Stohlgren (1982) reported decreases of almost 80 percent in the core and almost 50 percent on the fringes. In neighboring Kings Canyon National Park, Simon (1978) found increases at one lake and decreases at another. No adequate explanation for such discrepancies exists. Such differences may be important to any site rehabilitation attempt, although rehabilitation appears to be difficult under conditions of both soil organic matter accumulation and depletion.

Experiments have shown that impacts on mineral soil are initiated even before vegetation cover starts to decline (Quinn and others 1980). Increases in penetration resistance and bulk density—both measures of compaction—have been identified in most campsite studies. Bulk density increases of 0.1, 0.1, 0.3, 0.3, and over 1.0 g/cm^3 were found by Simon (1978), Cole (1982b, 1985b), Marion (1984), and Stohlgren (1982), respectively. Both Cole (1985b) and Stohlgren (1982) found increases in bulk density outside the campsite core to not be statistically significant. Increases in penetration resistance—a more sensitive indicator of compaction—are statistically significant beyond the campsite core, suggesting that some compaction occurs even if increases in bulk density are not substantial enough to be accurately measured. On campsite cores, increases in penetration resistance of as much as 214 percent (Marion 1984) and 337 percent (Cole 1985b) have been recorded.

Changes in porosity—one of the more significant effects of compaction—have also been investigated. In a developed campground in Ontario, Monti and Mackintosh (1979) found a 60 percent loss of macropores. Along with frequent surface crusting (documented by increases in penetration resistance), loss of macropores reduces infiltration rates. Although reductions in infiltration on campsite cores were only about 30 percent in the Eagle Cap Wilderness, they typically exceeded 70 percent in the Bob Marshall Wilderness and Grand Canyon National Park (Cole 1982b, 1983a, 1985b). Reductions in initial infiltration rates were slightly less dramatic than reductions in saturated rates. Beyond the core of campsites in Grand Canyon National Park, infiltration rates were not significantly slower than on controls (Cole 1985b).

Where studied, soil moisture content has been lower on campsites than on controls. Reductions in soil moisture were more pronounced on campsites in Sequoia than Grand Canyon National Park (Stohlgren 1982; Cole 1985b). Core areas had reductions of 65 percent and 38 percent of control values, respectively. Reductions in fringe areas were 23 percent and 8 percent. In Kings Canyon National Park, Simon (1978) found a reduction of 19 percent of control values. Campsites in the Boundary Waters Canoe Area had only 7 percent less soil moisture than controls. Lower soil moisture should be the general case, although Liddle and Greig-Smith (1975) showed that in dry sand-dune soils experimental trampling could increase both soil water content and the amount of water available to plants. Blom (1976) has shown that under these same conditions (dry sand-dune soils) more seedlings of *Plantago major* emerged when soils were compacted. So, although soil compaction can be detrimental, there are at least some circumstances under which it can be advantageous.

Finally, changes in the pH and nutrient content of soils have been investigated by Cole (1982b) and Stohlgren (1982). In both cases pH was higher on campsites, but the other results are remarkable in their inconsistency. Cole found a doubling in the concentrations of both Mg and Ca on campsites, while Stohlgren found these elements to be reduced by factors of more than two. Cole found no statistically significant differences, between campsites and controls, in concentrations of NO_3, K, and total N, while Stohlgren found sizable reductions in all of these. Cole also found an increase in Na and no change in PO_4, while Stohlgren found decreases in P and NH_4. Results of similar analyses on developed sites generally corroborate Cole's results. Most nutrients increase while others are unaffected; none are greatly depleted (Young and Gilmore 1976; Rutherford and Scott 1979). Although results are confused, I hesitate to suggest a need for research until the significance of such changes is demonstrated. Suggested reasons for increases in nutrients include scattering of fire remnants, soap, and litter around the site, as well as reduced leaching due to decreased infiltration rates.

Erosion—aside from what occurs around the base of trees that have had stock tied to them—is surprisingly unimportant. In Great Smoky Mountains National Park, for example, erosion is evident on less than 1 percent of the average area of intensive disturbance (Bratton and others 1978). Water velocities are low on most campsites because they are essentially flat, and soil does not detach readily from compacted surfaces. These factors inhibit erosion and compensate for the increase in erosion potential that results from reductions in vegetation cover, litter cover, and soil permeability on campsites. In most wildernesses, however, certain poorly located campsites likely will have serious erosion problems. Erosion problems are also unusually pronounced on campsites in the Boundary Waters Canoe Area, where soils are shallow and steep slopes from canoe landings to campsites are heavily trampled and eroded (Marion 1984).

Impacts on Trails.—It is more difficult to define when trail impacts become problems because the majority of change is purposeful change caused by trail construction and maintenance. Purposeful changes include opening up the tree and brush canopies in forest and shrublands; creating a barren, compacted trail tread that may alter drainage patterns; and producing a variety of new microhabitats—where slopes are flattened, rock faces are incised, and so on. Clearing vegetation increases light intensities and reduces competition for species capable of surviving along the trail. Shifts in species composition result from increases in trampling and grazing pressure, the dissemination of propagules of exotic plants, increases in nitrogen from urine and manure, increases in sunlight, and increases in moisture—less precipitation is intercepted by trees, fewer plants are losing water through evapotranspiration, and the compacted trail tread is shedding water along its sides.

The nature of shifts in species composition has been described, in wilderness situations, by Dale and Weaver (1974), Hartley (1976), Cole (1978), Stelmock and Dean (1979), and Teschner and others (1979). The most pronounced shifts are increases in exotic species and species with characteristics making them resistant to trampling, and decreases in shrubs and caulescent forbs. Meadow species often replace forest species (Dale and Weaver 1974). This reflects both a positive response to increased light along trails and the inability—according to general theory proposed by Grime (1979)—of plants adapted to the stresses of a low light environment to also adapt to a high disturbance environment. Specifically, shade-tolerant plants, in contrast to heliophytes, have more supportive and conductive tissue, greater leaf areas, and thinner cuticles, cell walls, and stems—morphological adaptations that make them susceptible to breakage (Cole 1979).

Where drainage disruptions cause water ponding, increases in moisture-loving plants and even aquatics occur. Where level, soil-covered trails are built across talus slopes, plants can spread into naturally inhospitable habitats. Numerous other construction effects on natural vegetation and soil conditions occur wherever trails cross the wilderness.

Because most of this is planned by management and accepted by the visitor, trail alteration becomes a serious problem only where it is unusually obtrusive (for example, where parallel ruts scar an alpine meadow) or where deterioration of the trail makes use difficult and requires the expenditure of large amounts of money and manpower for maintenance. While this is probably a realistic definition, it reflects the extent to which we use anthropocentric criteria to define impact problems. This anthropocentric bias is made even more clear by the fact that usually more money is spent dealing with trail problems than any other wilderness management problem.

Four important types of trail problems can be identified: (1) excessive erosion, (2) muddiness (with or without lateral spread), (3) multiple parallel trails, and (4) development of impromptu trails at attraction sites (for example, fishermen's trails around lakes or trailing and trampling damage near a waterfall). Erosion is a localized problem, although it does appear to be considerably more troublesome in the East, where trails are often steeper and use is heavier than in the West (Ketchledge and Leonard 1970; Bratton and others 1979). Two studies of erosion over an entire trail system (as opposed to examination of purposefully selected problem segments) found that deposition on trails actually exceeds erosion (Fish and others 1981; Cole 1983b). Material does move about, entering and leaving the system, but accumulation is greater than depletion.

As far as ease of use is concerned, the trail-system-wide lack of erosion is irrelevant. The problems are those stretches where erosion is pronounced, particularly where trails develop an uneven rocky tread or are deep and narrow. Interestingly, wide and deep trails (where erosion loss is greatest) are less of a problem to users than narrow and deep trails. Although trampling can cause erosion of some trails, its principal effect is to make the trail surface more susceptible to erosion by churning up the soil, reducing infiltration rates, and removing vegetation. The primary agent of erosion is running water from streams, snowmelt, intercepted springs, and even intense precipitation (Root and Knapik 1972). Erosion is primarily a problem where trails channel water and water is not diverted off the tread.

Erosion of wilderness trails has been studied by Ketchledge and Leonard (1970), Dale and Weaver (1974), Helgath (1975), Bratton and others (1979), Teschner and others (1979), Summer (1980), Fish and others (1981), and Cole (1983b). Most of these researchers studied the relationship between trail deterioration and use and environmental variables, a topic that I will discuss later. Dale and Weaver (1974), Bratton and others (1979), and Cole (1983b) also measured trail width, depth, and erosion problems, extremely site-specific data. Trails in Great Smoky Mountains National Park were generally wider and less deep than those examined in the Northern Rockies. In terms of annual change—again a very site-specific measure—Ketchledge and Leonard (1970) recorded an increase in trail width and depth of 1 inch per year in the Adirondacks. Summer (1980), working in Rocky Mountain National Park, found annual increases in depth of 0.4 to 5 inches (mean of 2 inches) and annual increases in width of 0 to 59 inches (mean of 5 inches). On two highly eroded segments in the Selway-Bitterroot Wilderness, Cole (1983b) recorded annual cross-sectional area losses of 30 and 56 in^2. Such figures represent the worst examples and are highly variable at all scales from a few feet to nationwide.

Muddiness can be a significant problem. In the Bob Marshall, where trails are often calf-deep in mud, complaints about damaged trails increased sixfold between 1970 and 1982 (Lucas 1985). Unwilling to walk through the quagmire, hikers and stock skirt the problem, widening the trail and the quagmire greatly. In some cases, mud is a temporary problem caused by snowmelt or by intense rainfall on trails that have been churned to dust. Mud can also be a season-long problem in soils with high or perched water tables (Helgath 1975). About 9 percent of the maintained trail system in Great Smoky Mountains National Park was muddy (defined as having a soil surface that moved when wet) (Bratton and others 1979). Surveying 17 miles of trail in the Selway-Bitterroot, Cole (1983b) recorded mud at 17 percent of the observation points, but only 1 percent of the trail was muddy enough to make travel difficult. The sporadic problem segments—28 in all—averaged 56 inches wide, almost 2.5 times the Forest Service maximum tread width standard of 24 inches.

Multiple parallel trails are a troublesome problem in meadows, particularly at high elevations. Here the problem is primarily esthetic, as such stretches are not difficult for hikers or stock to negotiate if they walk on one of the less deeply incised treads. This aggravates the multiple trailing problem further and leads to an increase in both the number of treads and the width of disturbance. From my observations, total width and intensity of surface deterioration never reach the extremes they do with muddiness problems. Multiple trailing has not been specifically studied in the United States wilderness. However, Price (1981) examined the problem in alpine meadows adjacent to the Sunshine ski area in Banff National Park, AB, and Palmer (1979) described some experiments with various means of rehabilitating multiple trails in Yosemite National Park.

The final problem, impromptu trail development, is a catchall category for recreational impacts outside campsites, designated trails, and grazing areas. The most frequent impacts are "fishermen's trails" around lakes and along streams and trampling damage at scenic attractions, such as waterfalls and other scenic viewpoints. This situation is the one most analogous to the random trampling generally studied by British recreational ecologists. Although seldom mentioned in more than casual observations, this type of impact is probably most similar to what happens on campsites. Vegetation is lost; what survives is floristically, morphologically, and physiologically distinctive; organic horizons are removed; and soils are physically, biologically, and probably chemically altered.

Managers' Perceptions of Problems

Two studies have attempted to assess the prevalence and significance of impact problems by querying wilderness managers about various problems that make it difficult for them to achieve management objectives. In a survey of managers of selected wildernesses, 80 percent of the managers that responded to an open-ended question about important problems mentioned trail and campsite deterioration. The only other frequently mentioned recreation-related problem was user conflict (Godin and Leonard 1979). The conclusion that trail and campsite deterioration is, in the opinion of managers, the most prevalent recreation-related problem in wilderness was corroborated in a more extensive survey of managers of all units in the National Wilderness Preservation System (Washburne and Cole 1983). Trail deterioration and campsite deterioration were considered to be a problem by 76 and 72 percent of the managers, respectively. This compared with perceived problem frequencies of 51 percent for crowding, 36 percent for impacts on wildlife, and 22 percent for water pollution.

This finding that campsite and trail deterioration is the most frequent recreation problem in wildernesses, at least in the opinion of managers, suggests that ecological problems occur at lower use levels than social problems. Reviewing recreation problems in England, Munton (1972) also concluded that "the thresholds of the semi-natural biological system are exceeded . . . before the psychological threshold resulting from overcrowding."

In addition to being a problem in more wildernesses, campsite and trail deterioration are also the most widespread problems within individual areas. When asked if problems occurred in "a few places" or in "many places," the number of wildernesses (out of 152) with problems in many places were 51 for campsite deterioration, 39 for trail deterioration, 19 for crowding, 9 for wildlife disturbance, and 3 for water pollution (Washburne and Cole 1983).

Beyond assessing the prevalence of problems, one would also like to assess the importance of problems. This, however, is where the insufficient research base becomes readily apparent. In contrast to considerable research on the extent to which crowding detracts from visitor satisfaction (an important management objective in wilderness), there has been little attempt to assess the importance of vegetation and soil deterioration in terms of not maintaining natural conditions (the primary management objective in many people's opinion). We desperately need a better understanding of the significance of these prevalent ecological impacts on trails and campsites.

Spatial Distribution of Impact

One component to be addressed when considering the significance of impacts is the areal extent of impact. On this basis, one might conclude that impacts are not highly significant. Recreational use is highly concentrated along a few major trails and at a few popular destinations. This leaves the vast majority of wilderness essentially unvisited and therefore undisturbed by recreational use. In the Adirondacks, Rechlin (1973) estimated that only 0.01 percent of the area had been disturbed by camping. In the Great Smoky Mountains, where camping is allowed only on designated sites, a conservative estimate of camping disturbance was 0.06 percent of the backcountry (Bratton and others 1978). In two relatively popular drainages in the Eagle Cap Wilderness, where camping is allowed almost anywhere, Cole (1981a) estimated 0.2 percent was disturbed by camping. Finally, 1.3 percent of a very popular lake basin in the Eagle Cap had been disturbed by camping (Cole 1982c). Although this percentage is still small, it is concentrated in the places where visitors spend most of their time—so evidence of human impact is omnipresent—and certain types of environments (flat, rock- and brush-free locations) may all be disturbed by camping.

Other areas have been disturbed by trails and grazing. Trail disturbance varies greatly between areas. Results of a survey of wilderness managers (Washburne and Cole 1983) indicate that the highest trail densities are in the Sawtooth and Joyce Kilmer-Slickrock Wildernesses. Assuming a 9-foot-wide swath of disturbance, about 0.7 percent of these wildernesses has been disturbed along designated trails. Over an entire wilderness, trails and campsites are unlikely to disturb more than 1 percent of the area. Locally, however, they could disturb a much higher percentage of popular lake basins and lakeshore areas.

More significant, in many places, is the area disturbed by grazing. In part of the Eagle Cap Wilderness, this amounted to about 1.3 percent of the area—almost three times the area of trails and campsites (Cole 1981a). However, more important than areal extent is the fact that grazing often occurs on relatively rare ecosystem types. In some places all representative examples of some meadow types may be disturbed by grazing. This, without question, represents a serious problem and a change that is essentially irreversible. The frequency of such situations has never been assessed, but it argues for the wisdom of a policy, such as that proposed for Sequoia/Kings Canyon National Parks, prohibiting grazing of representative examples of most meadow types. The near-complete lack of research on grazing impacts and their management in wilderness is a serious research gap.

Concentration of use on a small proportion of the wilderness can and has been considered both a bad and a good thing. Some have argued that most impact problems result from excessive concentration of use (Stanley and others 1979). In their survey of managers, Washburne and Cole (1983) reported that the most frequent response to an open-ended query about management's most significant problem was impact as a result of concentrated use. Others suggest that this tendency for use to be concentrated is what has kept wilderness resources as undisturbed as they currently are (Cole and Fichtler 1983). Clearly, managers should actively manage spatial patterns of use and impact. There may be situations where excessive impact at concentrated use sites should be reduced through use dispersal; there are also likely to be situations where undisturbed areas should be preserved through maintenance of existing patterns of use concentration.

The major threats to the concentration of impact are cross-country travel and the proliferation of campsites in destination areas. Neither have been examined in much detail. Cross-country travel, if use is frequent enough, can lead to the development of undesired impromptu trails. Where this occurs, the only management options are either to restrict use or to designate official trails and confine impact to those trails. The campsite proliferation problem was documented by Cole (1982c) in the Eagle Cap Wilderness, where 221 campsites were found around two subalpine lakes. Where use levels are high, campsite proliferation can only be avoided by confining use—either through education or regulation—to a small number of established sites. Where use levels are low, both highly impacted sites and site proliferation can be avoided through use dispersal and promotion of minimum impact camping techniques. Management of both the impromptu trail problem and the campsite proliferation problem would profit from more research on the relationship between use and impact discussed below. Such research could identify, for major ecosystem types, use thresholds beyond which trail and campsite problems are likely to develop.

On a smaller scale there are distinctive impact patterns on individual campsites. We have already described differences in amount of impact between the campsite core and fringes of the site. Often this concept of radial impact, with impact decreasing with distance from the campsite center, breaks down. Campsites may be linear or L-shaped, and they often include undisturbed "islands" and disturbed "satellite" sites. McEwen and Tocher (1976) proposed recognizing three zones for developed campsites: "(1) impact zone, the corridors of heavy use between and around site facilities; (2) intersite zone, the relatively undisturbed areas between the corridors of heavy use; and (3) buffer zone, the rarely disturbed areas at the border of the site." They prescribed different management strategies for each zone, emphasizing concentrated use on a small impact zone, creating intersite zones, and preserving the buffer zone.

In wilderness, these impact patterns are less pronounced because unstructured campsites have a more diffuse zonation (Hart 1982). In heavily used areas, however, facilities such as fire grates and toilets are not uncommon, and they tend to structure zonation patterns. In 1980, 19 percent of wildernesses had open-pit toilets, 15 percent had enclosed outhouses, 12 percent had shelters, 10 percent had constructed fireplaces, 8 percent had tables, and 7 percent provided a potable water supply (Washburne and Cole 1983). Clearly, impact could be reduced by channeling use between such facilities through careful design during construction and maintenance. Policies regarding fire rings—whether they are broken up or left—will also affect impact patterns; impacts are likely to be more extensive where fire rings are frequently broken up and rebuilt in different parts of the site (Cole and Dalle-Molle 1982). There is also a strong tendency for campsites to expand in area over time (Merriam and others 1973). Amount of expansion is likely to be greatest in open vegetation types and in places where party size limits are high and where party members tend to seek privacy from each other (for example, on outfitted trips made up of numerous unrelated individuals or groups). It should be possible to confine expansion, channel use to selected satellite sites where necessary, and protect most of the buffer zone from indiscriminate expansion. Despite increased expenditures by managers on site maintenance, no research evaluating existing efforts or suggesting improved techniques has been conducted. The benefit/cost ratio of such work would be high.

Considerable effort is also going into rehabilitating campsites, usually after permanently closing them. Research on this subject will be described more fully later. However, closures are sometimes temporary and in some cases (for example, the Boundary Waters Canoe Area and Eagle Cap Wilderness) some rehabilitation is attempted without curtailing use. A particularly important goal of such efforts is to get some tree reproduction on sites where it has been eliminated by trampling. Designing impact zones and protected intersite "islands" is critical to successfully rehabilitating any site that will be used again.

Trails have a sharp disturbance gradient perpendicular to the trail. The central trail tread is usually devoid of vegetation and organic matter and is highly compacted. Disturbed strips, on either side, are usually vegetated, but vegetation stature is short, cover is often low, and composition is different from adjacent undisturbed vegetation. If in a forested zone, trees are absent along the tread and disturbed zones. Lack of an overstory affects microclimate along the trail. The width of these zones is quite variable, but from the perspective of the entire wilderness, they are always narrow. The combined width of tread and disturbed zones is typically about 9 feet but can be much wider, particularly in meadows and where the trail is muddy.

The lateral expansion of trails can be controlled. Stock, in particular, tend to walk on the downslope side of trails. This breaks down the outer edge of the trail so that more soil must be brought in to rebuild the trail. The result is a wide trail, a much wider zone of disturbance, and an ongoing maintenance problem (Whitson 1974). Placing boulders on the outside of the trail can force stock and hikers to walk on the inside of the trail. Other techniques, such as avoiding excessive trail brushing, locating the trail in rough terrain (Bayfield 1973), and bridging wet areas can be used to limit lateral expansion.

Temporal Aspects of Impact

The few studies that have examined rate of deterioration on newly opened sites show that impacts occur very rapidly, even with moderate use, during a break-in period that seldom lasts more than several years (LaPage 1967; Merriam and others 1973; Legg and Schneider 1977). On new campsites in the Selway-Bitterroot Wilderness, vegetation loss approximated that on older campsites within 5 years. Loss of organic matter and resulting exposure of mineral soil occurred more slowly, but after 8 years was as pronounced as on older sites (Cole and Ranz 1983). On more heavily used sites, these changes can occur more rapidly. Merriam and others (1973) recorded near maximum levels of soil compaction after just 2 years' use on Boundary Waters Canoe Area sites. LaPage (1967), working on developed campsites, found that vegetation cover reached minimum levels after the first year of use. Where use levels are low, however, one study documented very little impact caused by camping (Leonard and others 1983).

After the break-in period comes a period of dynamic equilibrium (Hart 1982) in which seasonal and annual fluctuations predominate, in contrast to the unidirectional deterioration that occurs during the break-in period. On developed sites, Echelberger (1971) and Magill (1970) found little deterioration and some improvement over 5 years of use on long-established campsites. In the Boundary Waters Canoe Area (Merriam and Smith 1975; Merriam and Peterson 1983), the most pronounced change over time on sites at least 5 years old was an increase in campsite size. There also may be increases in tree damage, which is cumulative, in contrast to trampling damage, which after a few years should fluctuate around an equilibrium level (Hart 1982). Without management intervention, the dynamic equilibrium period will be followed, on forested sites, by a dying forest; sites must either become nonforested or be abandoned. The time it takes to reach this deteriorated state is determined by the longevity of tree species; it will usually be measured in hundreds of years, so we have few examples. However, the susceptibility of aspen to canker diseases following mechanical injury from recreationists makes the life span of developed campsites in aspen about 30 years (Hinds 1976). Maroon Lake Campground, an aspen campground in Colorado, is one of the first documented examples of a campground beyond the dynamic equilibrium phase (Johnson and Hinds 1977).

The first study to seriously question the dynamic equilibrium concept was Marion's (1984) recent examination of conditions on campsites in the Boundary Waters Canoe Area. He examined differences in amount of impact on campsites in three age classes: 5 to 10 years old, 11 to 13 years old, and more than 13 years old. There was little difference between the 5- to 10-year-old and 11- to 13-year-old sites, but the more-than-13-year-old sites had more root exposure and tree damage; they also were larger, and had experienced greater increases in bulk density and shifts in species composition. The larger area and greater root exposure and tree damage were as expected; however, the other changes challenge the notion of equilibrium. Differences in amount of impact are not pronounced; after statistically controlling differences in amount of use, an overall index of impact showed that 5- to 10-year-old sites had experienced 86 percent as much change as the over-13-year-old sites. However, this does suggest that trampling damage may increase slightly over time. Unfortunately, in addition to being older, these sites are also unique in their use history and location. They were developed by campers, prior to restrictions on party size and camping techniques, in prime locations. The younger sites were developed by the Forest Service, with concern for design and durable locations, and did not go through the "frontier lifestyle" phase the older sites did. It is impossible to separate these factors from campsite age and conclude that today's young sites will, over time, become like today's old sites. This is a good illustration of how lack of longitudinal studies forces reliance on cross-sectional studies that cannot, by themselves, provide definitive answers to questions such as how impacts change over time. For more detail on campsite changes over time, refer to the paper by Cole and Marion presented elsewhere at this conference.

Two factors contribute to the tendency for trampling damage to equilibrate. The first is that many types of impact reach a maximum limit. For example, bulk density will eventually reach a limit above which much heavier loads would have to be applied to cause an increase. The second factor is the existence of homeostatic controls or negative feedback loops. A pattern of impact and recovery, such as erosion of leaf litter every summer and accumulation of leaf litter every fall, tends to keep litter cover fairly constant. In a few cases, the impacts of recreation use actually decrease the likelihood of further deterioration. On trails, and to a lesser extent on campsites, soil compaction following trampling can stabilize the soil surface, making it less prone to further erosion (Malin and Parker 1976). It is still a highly altered environment, inhospitable for vegetation, but less likely to erode than it was in the phase between initiation of development and the formation of a compacted surface.

In a similar manner, trampling disturbance may promote the invasion of trampling-tolerant species. On newly opened developed campsites, LaPage (1967) documented an initial rapid loss of vegetation cover. After this initial drastic loss, however, vegetation cover during the second, third, and fourth years of use actually increased as trampling-resistant, often exotic species, invaded the site. The resultant ground cover, being relatively tolerant of recreation use, resists further cover loss. These homeostatic controls are much less effective in environments where recovery processes are extremely slow or where there are few "weedy" invader species. Alpine ecosystems are deficient on both of these counts and, therefore, deterioration can be unusually severe once use exceeds tolerance thresholds.

Studies of recovery rates on recreation sites are almost as rare as studies of deterioration over time. Longitudinal studies of 3 years' recovery on campsites were done in Sequoia National Park (Stohlgren 1982) and the Selway-Bitterroot Wilderness (Cole and Ranz 1983). Cross-sectional studies in the same areas estimated recovery over 15 years (Parsons and DeBenedetti 1979) and 5 years (Ranz 1979). Willard and Marr (1971) investigated recovery of tundra over a 2-year period near parking areas in Rocky Mountain National Park. Others have investigated the effectiveness of cultural means of increasing recovery rates (a topic discussed below) and have followed recovery after artificial trampling experiments for periods up to 8 years.

Recovery periods are universally long, but they have only been examined in harsh environments where low resilience would be expected. Assuming a linear recovery rate in the future, a cross-sectional study of 5 years' recovery allowed Ranz (1979) to predict recovery of predisturbance vegetation cover in 16 years. However, a followup study indicated recovery rates were not constant or linear functions (Cole and Ranz 1983). Moreover, core areas of campsites require much longer recovery periods than fringe areas of campsites. Stohlgren (1982) estimated recovery periods for vegetation cover—again assuming linear rates—of 56 years in the core and only 5 years in the periphery. Recovery periods for bulk density were 36 years in the core and 11 years in fringe areas. In a cross-sectional study in the Boundary Waters Canoe Area, estimated recovery periods were 20 years for vegetation cover, 50 to 60 years for a return to normal species composition, and 30 to 40 years for bulk density (Marion 1984). After 15 years without use, litter accumulation and soil penetration resistance on campsites in Kings Canyon National Park had reached inferred predisturbance levels, but fuel accumulation was still low and tree mutilations, social trails, and vegetation deterioration were still evident (Parsons and DeBenedetti 1979).

Two major implications of rapid deterioration and slow recovery are that impacts are inevitable with use and that they are unlikely to be managed efficiently through adoption of a rest-rotation system. Rest-rotation, an often-suggested management practice in which sites are periodically closed and then reopened after recovery, would lead to many closed sites for each open site, as long as recovery periods are longer than deterioration periods. Since all research shows recovery periods to be many times greater, rotation appears to be generally impractical.

There is some indication that recovery can be relatively rapid if use only occurs for 1 year (Willard and Marr 1971 and several experimental trampling studies). Some wildernesses, such as Shenandoah National Park, where resilience appears to be high, have been trying variations of rest-rotation systems. Research evaluating the relationship between duration of impact period and recovery rates in more resilient ecosystems and the success of management systems such as the one at Shenandoah would expand our understanding of how recreation sites change over time.

On trails, some segments are relatively stable while others deteriorate rapidly. Generally, most change occurs during initial construction of the trail. Subsequent "problems," such as erosion, probably also occur rapidly. Old, traditional trails in Great Britain, for example, do not seem to be getting wider with age (Bayfield 1985). Recovery rates, unless assisted, are likely to be even more lengthy than those on campsites. In many cases, trail erosion is likely to continue even after all use has been curtailed. All one has to do is look at the slow recovery of trails abandoned after rerouting to see this.

Recovery rates on trails are highly variable, however. Experimentally trampled trails in the southern Appalachians were almost completely revegetated after just 1 year (Studlar 1983), while vegetation cover was only 24 percent of that on controls after 6 years of recovery in dry alpine meadows in Glacier National Park (Hartley 1976). Rates are highly variable even within the same general area (Leonard and others 1985). For example, 5 years after being experimentally trampled by horses, vegetation cover of a grassland (*Festuca-Poa*) was 100 percent of normal; cover in a forest (*Pinus-Vaccinium*) was only 26 percent of normal (Weaver and others 1979). In addition to faster recovery in nonforested areas, observations suggest that wet places revegetate more rapidly than dry places. Natural infilling of severely eroded trails may never occur, except in depositional locations. Extensive rehabilitation work is often the only feasible means of bringing about recovery.

Factors that Influence Amount of Impact

Perhaps the greatest increase in understanding over the past 10 to 15 years has come from investigations of factors that influence amount of impact. This probably reflects the fact that such studies, while requiring more sophisticated research designs and analyses than descriptive surveys, can still be completed over a short period of time using cross-sectional techniques. Understanding the factors that influence impact is of paramount importance because managers can manipulate these factors to control amount and type of impact. This subject is the ecological analog to the so-called carrying capacity research by social scientists in which factors affecting crowding and visitor satisfaction have been studied.

The principal factors that have been identified are (1) amount and frequency of use, (2) type and behavior of users, (3) season and time of use, and (4) environmental conditions where recreation use occurs. While these are the factors that determine intensity of impact at any point of use, the areal extent of impacts is also affected by the spatial distribution of use, particularly its level of channelization. Although seldom the direct subject of research, it is clear that, everything else being equal, greater concentration of use leads to disturbance of less of the wilderness. When developing management strategies, both the factors that influence intensity of impact and those that influence level of use concentration need to be considered.

Amount and Frequency of Use.—Amount of use, strictly defined, should refer to the total number of people that have ever used a site or trail. However, the effect of a given number of users is likely to be modified by the number of years over which that use is spread. One would intuitively expect that, for a given number of users, spreading use over more years would cause less impact because this offers more opportunity for recovery. This hypothesis has never been adequately tested, however. If true, it would justify the tendency to use amount of use per year—a frequency measure—instead of total use as an independent variable.

Even when using measures of amount of use per year, there is some question about the importance of how that use is distributed over the year. Here some very limited data from trampling experiments suggest that, for a given amount of use, the distribution of that use over the year is a relatively unimportant factor (Cole 1985c). For example, the effect of 50 parties of two (100 total visitors) should be about the same as two parties of 50. While this seems reasonable on trails, it is not a reasonable assumption on campsites, where party size makes a difference, particularly to campsite size. Generally, recognizing considerable ignorance of the importance of frequency, studies of trail impact should use measures of total visitors per year (hopefully comparing trails of about the same age and that have not experienced major recent increases or decreases in use). In campsite studies, party-nights per year is probably as valid a measure as visitor-nights per year; both are probably important. In reality most studies have had to rely on ordinal use estimates of occupancy rates because these are all that are available. However, each of these measures has different implications that should be kept in mind.

Conventional wisdom has often held that amount of use is the most important factor influencing amount of impact. Such thinking has been supported by calling deteriorated sites "overused" and by proposing that solutions can be found by prescribing a "carrying capacity." Research shows that such thinking is oversimplified at best, and erroneous at worst. The importance of amount of use varies between environments, between activities, with impact parameter, and with the range of use levels being examined. In addition, effects differ depending on whether concern is with rate, intensity, or areal extent of change.

Research on the relationship between use and impact began in the early 1960's, with Frissell and Duncan's (1965) cross-sectional analysis of Boundary Waters Canoe Area campsites and Wagar's (1964) experimental trampling study in a recreation area in Michigan. Both studies examined the effect of different use levels on loss of vegetation cover. Frissell and Duncan (1965) found that the most lightly used campsites (0 to 30 nights per year) had lost 80 percent of their inferred original cover, while heavy use sites (60 to 90 nights per year) had lost only 7 percent more—87 percent. Wagar (1964) also found that although impact does increase as use increases, small amounts of use cause substantial amounts of impact. This relationship between amount of use and loss of vegetation is shown in figure 4. The curves are asymptotic, rather than linear. They have a single inflection point that separates the rapidly rising segment of each curve from the segment where increase in amount of impact is small.

The shape of this curve and the location of the inflection point have important implications for management. Attempts to limit vegetation damage by keeping use levels low will only be effective where use levels can be kept substantially below the use thresholds that correspond to the inflection points—those use thresholds are points X(a) and X(b) on the X axis. This may be impossible to do in portions of wildernesses that receive even moderate levels of use. Several studies in subalpine forests suggest that use levels of only five nights per year exceed these threshold levels (Cole and Fichtler 1983). In such places, concentration of use on a small number of sites appears to be the most appropriate strategy for minimizing vegetation loss.

Most studies have examined only sites with use levels beyond those that correspond to the inflection points on the curve. Consequently cover differences are not substantial and, generally, not statistically significant. One exception is Coombs (1976), who found that very lightly

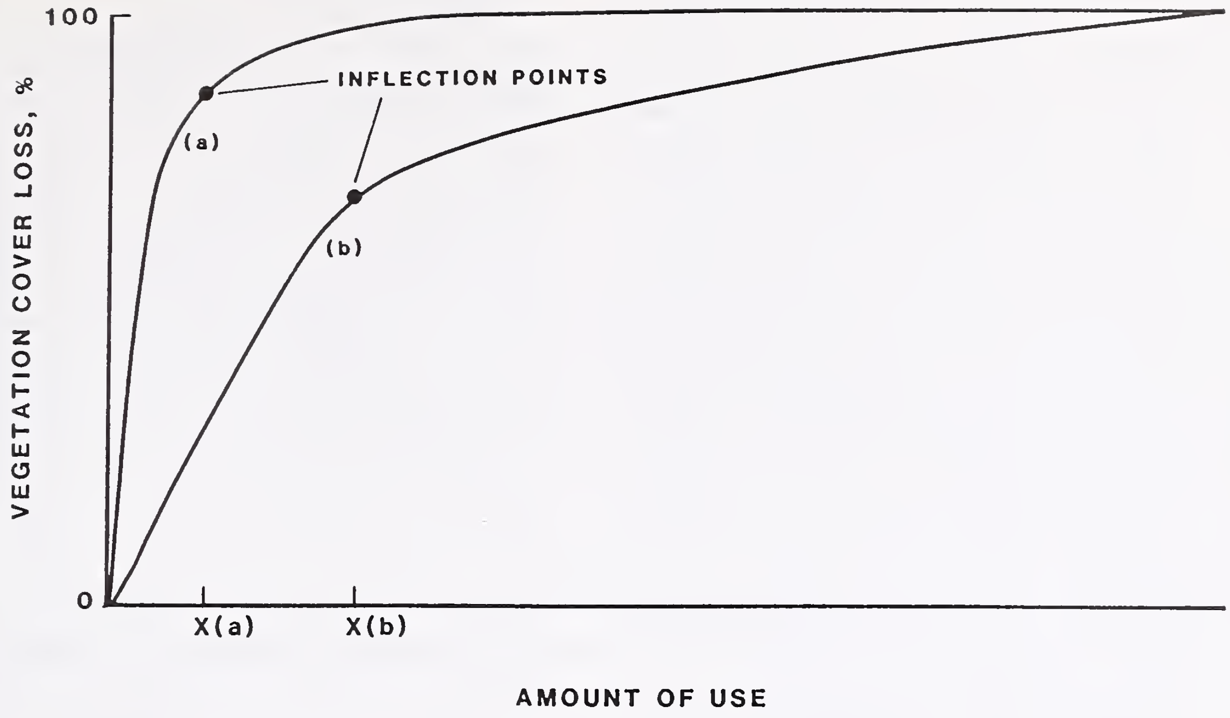

Figure 4.—The general relationship between amount of use and loss of vegetation cover for (a) a fragile vegetation type and (b) a more resistant type.

used sites, in what is now the Frank Church-River of No Return Wilderness in Idaho, had experienced less than 40 percent as much vegetation loss as heavily used sites.

Below use thresholds, even slight reductions in use can reduce vegetation loss substantially. This suggests a real opportunity to minimize impact in remote, lightly used places. In portions of wilderness where use levels can be kept well below these thresholds, it appears most appropriate to spread people out over a very large number of sites so that use levels are as low as possible. For this to work, visitors must be taught to use unused rather than lightly used sites, to practice low-impact camping, and to try to eliminate all traces of their stay when they move on.

Identification of inflection points and use thresholds offers tremendous potential for management because they can be used to choose between the opposing strategies of concentration and dispersal. Virtually every wilderness in the country could profit from employing both of these strategies in some part of their area.

Curvilinearity and the location of inflection points vary with differences in type of use and differences in environmental durability, however. In figure 4, curve (a) depicts the use/impact relationship for a quite fragile vegetation type, while curve (b) depicts the same relationship for a more durable type. Curve (b) is not as strongly curvilinear, and the use threshold associated with the inflection point—X(b)—comes at a higher level of use. If managers can get people to camp in type (b) rather than type (a), about twice as much use can be absorbed before the need to adopt a concentration strategy arises.

In one of the few studies undertaken on resistant vegetation types, Dunn and others (1980) examined developed campgrounds in the Atlantic Coastal Flatwoods region of South Carolina. They found no significant loss of vegetation cover except on heavy use sites. As in most other studies, inadequate measures of use make it impossible to establish use thresholds for these South Carolina campsites. More research, employing better use estimates and controlled experiments, could enable us to establish use thresholds for important environments across the country.

Hylgaard and Liddle (1981) hypothesized a rather different shape to the use/impact curve. They fitted cover loss, following experimental trampling, to a logistic equation. Logistic curves have three segments separated by two inflection points. At the very lowest use levels differences in amount of use have little effect on cover loss; vegetation can tolerate a certain amount of use before plants are killed. Above the first inflection point, cover loss increases rapidly with increasing use until the second inflection point is reached; beyond this point increases in use have less and less effect on vegetation cover. While such a model is intuitively appealing, there are few data to evaluate it.

Data from an experimental trampling study being conducted in Montana are presented in figure 5. Data are for two vegetation types that had been trampled for 2 successive years. A pass is a one-way walk down a trampling lane. The use/impact relationship for the spruce forest exhibits the highly curvilinear form of a fragile vegetation type—with an inflection point at around 200 passes per year. That corresponds to about 2 nights of camping use per year by an average party (Cole 1985c). In this forest, once use levels reach 2 nights per year, further increases in use cause little additional vegetation loss. Unless use levels can be kept at or below 1 night per year, it seems most appropriate to concentrate use on a few of these forested sites.

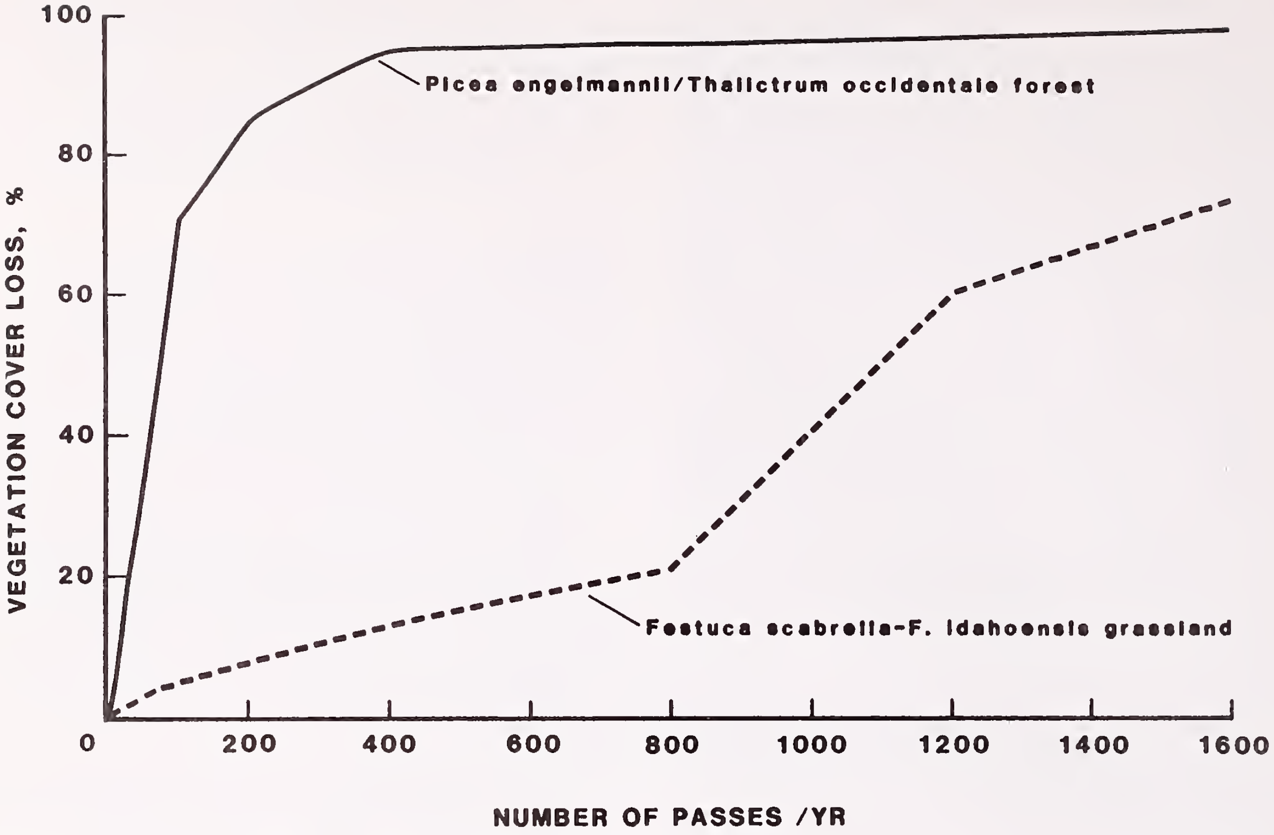

Figure 5.—The relationship between number of passes per year and vegetation loss (as a percent of original vegetation cover, adjusted for changes on controls) for two vegetation types in western Montana. Trampling was applied for two seasons.

The grassland, however, exhibits the form of a logistic curve with two inflection points. Below 800 passes per year, further reductions in use have little effect. In other words, there is some initial resistance to vegetation loss before further increases in use cause a substantial loss of vegetation. Apparently, only the most resistant vegetation types have this form.

Another complication to our original simple model comes from the fact that the shape of the curve varies with the type of impact being studied. Figure 6 shows some data from a recent study of campsites in the Boundary Waters Canoe Area (Marion 1984). For each parameter, amount of change on low-, moderate-, and high-use sites has been expressed as a percentage of change on the high-use site. Degree of curvilinearity and use thresholds differ between parameters. Just as vegetation types vary in their susceptibility to any type of impact, the susceptibility of a site to different types of impact is also variable. Those parameters, like vegetation cover, for which a highly curvilinear relationship exists, include bulk density, penetration resistance, macropore space, infiltration rate, changes in soil chemistry, loss of tree seedlings, and tree damage (Young and Gilmore 1976; Legg and Schneider 1977; Dunn and others 1980; Cole and Fichtler 1983; Marion 1984). Loss of organic horizons, exposure of mineral soil, severe root exposure, and site enlargement are all changes related to use in a less curvilinear manner (Coombs 1976; Young 1978; Cole and Fichtler 1983; Marion 1984). There is more inherent resistance to these types of change; inflection points and use thresholds are usually higher. Changes in these parameters are easier to limit through manipulation of use intensities on campsites.

The final complication is that even the susceptibility rankings for these parameters do not apply universally. For example, in Grand Canyon National Park, exposure of rock and mineral soil increased as rapidly with increasing use as vegetation loss and more rapidly than increase in penetration resistance (Cole 1985b)—a reversal of the order in the Boundary Waters Canoe Area.

Research does suggest a strong relationship between amount of use and rate of vegetation loss. For example, in an experimental trampling study on alpine meadows in Mount Rainier National Park, vegetation cover was reduced to 50 percent of control values in 3 weeks when trampled at 75 passes per week. At 18 passes per week, it took 8 weeks of trampling for cover to be reduced to 50 percent of controls (Singer 1971). The areal extent of vegetation loss is also strongly related to amount of use (Bratton and others 1978; Cole 1982b). The finding that, at all but very low use levels, increased use has little effect on intensity of vegetation loss and a pronounced effect on area of loss, suggests the value of concentrating and channeling use on a small proportion of any area (Cole 1981b).

On trails, vegetation cover, bulk density, penetration resistance, and trail width relationships are highly curvilinear, as they are on campsites (Dale and Weaver

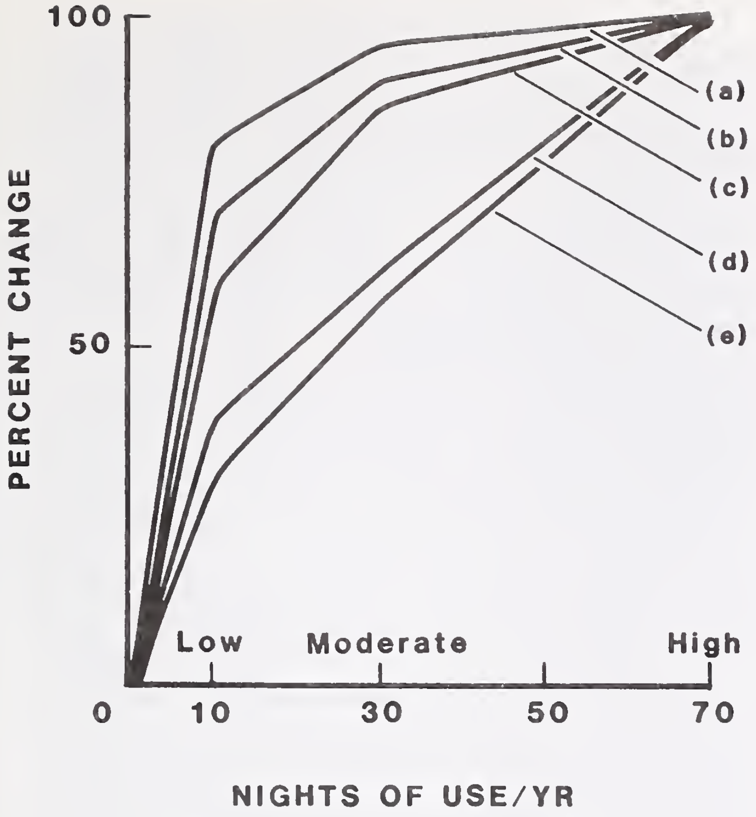

Figure 6.—Relationship between amount of use and (a) tree damage; (b) loss of vegetation cover; (c) increase in penetration resistance; (d) increase in exposed roots, mineral soil, and rock; and (e) campsite area on campsites in the Boundary Waters Canoe Area (Marion 1984). Change is expressed as a percentage of change on high use sites. Numeric use levels are estimated from ordinal classes of low (0 to 12 nights per year), moderate (20 to 40 nights per year), and high use (>60 nights per year).

1974; Crawford and Liddle 1977). Trail depth and the frequency of problems, such as muddiness, are generally not related to amount of use (Dale and Weaver 1974; Helgath 1975; Cole 1983b). Such situations relate more to location and design features, although they obviously must be triggered by some use or construction.

In sum, these results suggest that there is little value, in terms of reduced impact, in limiting use of constructed trails. On campsites, limiting use is only likely to be effective if use levels can be kept very low. This is possible in the majority of most wildernesses, but not in popular destination areas. In popular areas, counteracting the tendency for increased use to increase the areal extent of impact—through channeling and concentrating use—is one of the most effective means of minimizing impact. Because the tipping point for each of these opposing strategies—dispersing use to keep levels low or concentrating use to minimize areal extent—varies greatly between environments, use thresholds need to be identified for major ecosystem types.

Type of Use and Behavior.—The current vogue is to look to visitor education for solutions to wilderness management problems. This reflects a feeling, certainly true but never tested, that parties of the same size vary greatly in the damage they inflict. If all parties can be taught to minimize their impact, then impact problems will be reduced. There are certain problems (such as tree mutilation) that could be entirely eliminated through education. Much trampling damage would still occur, however, although at reduced levels.

Currently, over one-half of all wildernesses have educational programs (Washburne and Cole 1983); a number of studies of how to effectively communicate information to users have also been conducted (for example, Fazio 1979; Roggenbuck and Berrier 1981). In contrast, there has been very little research into what users should be taught. All published papers suggesting educational messages have been personal opinions not based on research. Although many ideas have merit and are not controversial, others are contradictory and oversimplified. Few take into account the need for varying behavior in different environmental and use situations. One exception is an article by Cole and Benedict (1983) that, following a review of literature, suggests different user responses to campsites in various stages of deterioration. It is wasteful to initiate major educational programs without investing some research in what the educational message should be.

One seemingly obvious statement that has frequently been made is that lug-soled boots are much more destructive than tennis shoes or shoes with flat soles. In this case, some research on the subject has been done. Three studies—all done on trails in the East—concluded that, at least under the conditions studied, lug soles are not substantially more destructive than other types of footwear (Whittaker 1978; Saunders and others 1980; Kuss 1983). Kuss, for example, found no significant difference in the volume of soil eroded from stretches of trail trampled by lug-soled and corrugated rubber-soled boots. This lack of difference was found despite increases in soil yield, after 600 and 2,400 trampling passes, that amounted to 1.4 and 1.7 times, respectively, the yield from undisturbed trail. Kuss suggests, however, that differences might be significant if soils were wet. Similarly, perhaps lug soles are more destructive of vegetation on campsites. We will not know unless more conclusive research is undertaken.

Research on the effects of various types of use has also been limited. In wilderness, differences related to party size and mode of travel (hikers vs. stock) are particularly important and amenable to control. The effects of party size have never been formally studied. There is little reason to suspect that large parties will have a more serious impact on trails than a number of small parties. On campsites, however, party size differences could be important.

In response to the need of large parties for more space, campsite area and the size of devegetated areas are likely to increase where sites are used by large parties. However, the severity of impacts in the central part of the campsite is unlikely to increase substantially. Per capita consumption of firewood is less with large parties (Davilla 1979), and reducing party size limits could have the undesirable effect of increasing the number of campsites. Most of these likely outcomes are speculative; they have not been researched. It is fairly safe to conclude that the current common party size limits of 15 or 25 have had little effect, one way or the other, on recreational impacts.

Differences between impacts caused by hiking parties and parties with stock have received a little more attention, but research has been surprisingly limited. Three studies have compared impacts using experimental trampling (Nagy and Scotter 1974; Douglas and others 1975; Whittaker 1978; Weaver and Dale 1978), two have compared impact on campsites (Frissell 1973; Cole 1983a), and one has compared impact on trails (Dale and Weaver 1974). In addition there are papers by McQuaid-Cook (1978) and Summer (1980) on horse trails and by Strand (1979) on horse effects on experimental trails, as well as the limited research, discussed earlier, on the effects of stock on grazing areas.

Many of the impacts caused by stock are similar to those caused by hikers except that they are more pronounced. Experimental trails produced by 1,000 horse passes were 2 to 3 times as wide and 1.5 to 7 times as deep as trails produced by 1,000 hiker passes. Bulk density increased 1.5 to 2 times as rapidly on horse trails. One-half of the vegetation was lost after 1,000 hiker passes and 600 horse passes in a grassland and after 300 hiker passes and only 50 horse passes in a forest (Weaver and Dale 1978). In a grassland in Waterton Lakes National Park, experimental trampling by horses destroyed vegetation cover four to eight times as rapidly as trampling by humans (Nagy and Scotter 1974). This suggests that multiple trailing and the development of impromptu trails will occur much more rapidly with stock use than with hiker use. Moreover, the trails created will be wider, deeper, more compacted, and less vegetated.

On existing trails, horse use caused more pronounced increases in trail width, trail depth, and litter loss than hiker use. While hiker use tended to stabilize the trail surface, horse use loosened the soil, making it more prone to erosion (Whittaker 1978). McQuaid-Cook (1978) suggested that it is this tendency for shod hooves to loosen soil that leads to the more pronounced incision of equestrian trails documented by Dale and Weaver (1974). The tendency for stock to walk on the downslope side of the trail probably explains the greater width of equestrian trails. Although never studied, equestrian trails are "brushed out" to a greater height and width, resulting in a wider swath of vegetation alteration, and they frequently require more elaborate engineering and more frequent maintenance.

On campsites, quantitative differences between hiker and stock impacts are even more pronounced. More significantly, there are more qualitative differences in type of impact. Although such differences are readily obvious, they have seldom been documented. Frissell (1973) found a sample of sites used by stock parties in what is now the Lee Metcalf Wilderness, MT, to be 10 times as large and have seven times as much exposed mineral soil as backpacker-only sites.

In a more detailed study by Cole (1983a) in the Bob Marshall Wilderness, stock sites were six times as large as backpacker sites. They had over four times as large a devegetated area; they had 11 times as many damaged trees and 25 times as many trees with exposed roots. Stock sites had been more extensively invaded by exotic plants, had lost more of their organic horizons, were more compacted, and had slower infiltration rates. These differences result primarily from the much greater trampling force of horses' hooves and the need to keep horses confined, usually by tying them to trees. Additional impacts result from grazing.

Season of Use.—Theoretically, there are many reasons to expect impact to vary with the time of year use occurs. In particular, soil moisture levels and the phenology of plants when trampling occurs should influence amount of impact. The effect of soil moisture level on the magnitude of trampling impacts on soils was investigated experimentally by Jones (1978). Impacts, particularly loss of macropores, were generally greater at high soil moisture levels. Vegetation damage inflicted by horse trampling was also greater and lasted longer in a wet meadow than in a dry meadow (Strand 1979).

The effect of season of trampling on vegetation loss has been examined experimentally in several studies. Singer (1971), working in an alpine meadow at Mount Rainier, found that vegetation cover was not significantly affected by the time of summer (July 3-August 13) at which trampling occurred. In 10 vegetation types in Waterton Lakes National Park, Canada, Nagy and Scotter (1974) found no consistent difference between trampling in early season (early June-early July) and midseason (late July-early August). In both of these studies, however, no pretreatment measurements were taken, and recovery periods between treatment and measurement were longer for the early season treatment. In a subalpine meadow in Yosemite National Park, Holmes and Dobson (1976) studied differences in the effect of midseason (early August) and late season (early September) trampling on the cover of 14 species. Late season trampling was substantially more damaging to nine species and slightly less damaging to one species; no seasonal difference could be detected for the other four. They attribute this higher vulnerability in late season to reduced plant vigor and drier, more brittle plant parts. There may, however, be little relationship between cover loss immediately after trampling and cover at some time in the future. Early season defoliation of forbs and grasses, for example, has a particularly severe effect on carbohydrate reserves (Donard and Cook 1970). This will affect vigor, reproductive success, and, therefore, long-term vegetation conditions.

Our understanding of the importance of seasonality, then, is extremely limited. Effects appear to be most important on trails and in grazing areas. Most trail management problems can probably be dealt with—without significant research input—through maximizing the advantages of good location and incorporation of design features. Grazing management plans, however, would profit from area-specific research programs, similar to that of Sequoia/Kings Canyon National Parks, that evaluate seasonal differences in vulnerability (DeBenedetti and Parsons 1983).

Environmental Conditions.—Environments vary greatly in their ability to tolerate recreational use. Understanding this variability is difficult, however, because any one place may be resistant to one type of impact and susceptible to another. Moreover, one characteristic of a site, such as good drainage, may raise a site's tolerance level while another, such as a closed canopy, decreases it. Finally, there may be little congruence between resistance—the ability to tolerate use without changing—and resilience—the ability to recover from changes that do occur. More research done outside the field of recreational ecology can usefully add to our knowledge about this topic than any other, with the possible exception of site rehabilitation. Therefore, the following discussion will, of necessity, be more selective than others.

Environmental parameters that affect tolerance can conveniently be divided into vegetation characteristics, soil characteristics, and topographic characteristics. At a higher level of generalization one can also assess the influence of broad ecosystem-level characteristics. Macroclimate also affects tolerance but cannot be influenced by management and, therefore, is not discussed here.

Vegetational characteristics of import are the resistance of individual species, the floristic composition of the vegetation, vegetation cover, and vegetation structure or physiognomy. Of these, most work—starting with Bates (1935)—has been on the resistance of individual species to trampling (Speight 1973; Holmes and Dobson 1976). Morphological characteristics that generally make a plant more tolerant include:

1. A procumbent or trailing, rather than erect, growth form.
2. A tufted growth form.
3. Arming with thorns or prickles.
4. Stems that are flexible rather than brittle or rigid, particularly if they are woody.
5. Leaves in a basal rosette.
6. Small, thick leaves.
7. Flexible leaves that can fold under pressure.
8. Either very large or very small structure.

Physiological characteristics that increase tolerance include:

1. Ability to initiate growth from intercalary as well as apical meristems.
2. Ability to initiate seasonal regrowth from buds below the surface.
3. Ability to reproduce vegetatively and sexually.
4. A rapid growth rate.

Native species that have been identified as particularly resistant to trampling damage in United States wildernesses are listed in table 1. Species have been omitted when evidence is contradictory. For example, *Hieracium gracile* was considered resistant by Coombs (1976) and sensitive by Schreiner (1974), Hartley (1976), and Cole (1982b). Nonnative species are also common on recreation sites in U.S. wildernesses. A list of these species can be found in the paper by Marion, Cole, and Bratton presented at this conference. Some species, such as *Poa pratensis* and *Trifolium repens*, are remarkably widespread and prominent on disturbed sites. In their native habitat in England, they were originally identified as trampling-resistant species by Bates (1935).

Attempts to generalize about the relative tolerance of classes of plants can be useful, although the inevitable exceptions provide considerable ammunition for those who prefer to emphasize the site-specific nature of plant response. Even the response of different individuals within a species varies in response to phenotypic and ecotypic differences (Leney 1974). Survival varies with seasonal cycles, with changes in conditions such as cell turgor, and with differences in associated species. For example, in subalpine meadows in Yosemite, survival rates of a given species were generally about three times greater in mixed communities than in pure stands (Holmes and Dobson 1976).

As mentioned earlier, mature trees and graminoids are generally resistant, mosses are neither highly resistant nor highly sensitive, and lichens and tree seedlings are highly sensitive. Shrubs vary from quite resistant to moderately sensitive; heaths can be particularly susceptible (Emanuelsson 1985). Forbs vary from moderately resistant to highly sensitive. These guidelines, along with consideration of the morphological and physiological characteristics influencing tolerance, can be used to assess the relative durability of different vegetation types. Those types with more abundant resistant plants will generally be more resistant.

Quite a few analyses of the relative resistance of entire species assemblages have been made. The ones that can be most readily compared are the experimental trampling studies. Table 2 ranks 38 plant communities on the basis of their resistance to trampling. Unfortunately, these studies only apply to the initial resistance of plant communities because only damage following one season of trampling was evaluated. Resistance to sustained trampling and resilience are not assessed. Vegetation types for which resistance is relatively high but resilience is low, such as many alpine vegetation types, are rated as resistant even though they may be poor sites for particular types of recreation use and facilities. Also, some very fragile types, such as lush forest/forb types, recover very rapidly after disturbance, if use is not continuous. For example, the "fragile" *Pinus contorta/Thalictrum occidentale* type (table 2) had much more vegetation cover 1 year after trampling than the much more "resistant" *Dryas octopetala* type (Douglas and others 1975). Generally, those vegetation types with ground cover dominated by graminoids and matted forbs are more resistant than those dominated by shrubs and tall forbs. Studies of change on campsites and along trails in different vegetation types confirm this (del Moral 1979; Cole 1981a).

The effect of amount of vegetation cover on durability is complex. Some communities with a dense ground cover are highly resistant (for example, *Carex nigricans* meadow, Cole 1982b), but some sparsely vegetated types are also resistant (for example, *Pinus albicaulis/Juniperus communis*, del Moral 1979). Perhaps the most important characteristic is the amount of vegetation cover that can survive trampling, because dense vegetation reduces erosion potential.

Table 1.—Resistant native plant species in U.S. wildernesses

Species	Citation(s)
Agave utahensis	Cole (1985b)
Agoseris glauca	Cole (1983a)
Agrostis idahoensis	Thornburgh (1962)
Agrostis lepida	Stohlgren (1982)
Agrostis scabra	Cole (1983a)
Antennaria alpina	Holmes and Dobson (1976)
Antennaria dimorpha	del Moral (1979)
Antennaria lanata	Thornburgh (1962), Cole (1977, 1982b)
Arabis lyallii	del Moral (1979)
Arenaria capillaris	Singer (1971), Schreiner (1974), Hartley (1976), del Moral (1979)
Arenaria congesta	Cole (1985a)
Arenaria obtusiloba	Singer (1971), del Moral (1979)
Aster alpigenus	Singer (1971), Holmes and Dobson (1976), Cole (1977, 1982b)
Aster ciliolatus	Marion (1984)
Bernardia incana	Cole (1985b)
Bouteloua gracilis	Cole (1985b)
Carex aestivalis	Saunders (1979)
Carex bigelowii	Stelmock and Dean (1979)
Carex exserta	Lemons (1979)
Carex microptera	Cole (1982b)
Carex nigricans	Hartley (1976)
Carex phaeocephala	Schreiner (1974)
Carex podocarpa	Stelmock and Dean (1979)
Carex proposita	del Moral (1979)
Carex rossii	Coombs (1976), Cole (1977, 1982b, 1983a)
Carex scopulorum	Holmes and Dobson (1976), Cole (1982b)
Carex spectabilis	del Moral (1979), Cole (1982b)
Chrysothamnus nauseosus	Cole (1985b)
Coleogyne ramosissima	Cole (1985b)
Cowania mexicana	Cole (1985b)
Deschampsia atropurpurea	Thornburgh (1962), Schreiner (1974)
Deschampsia caespitosa	Marion (1984)
Ephedra spp.	Cole (1985b)
Eriogonum pyrolifolium	del Moral (1979)
Fallugia paradoxa	Cole (1985b)
Festuca idahoensis	Schreiner (1974)
Festuca scabrella	Cole (1985a)
Fragaria vesca	Marion (1984)
Geum rossii	del Moral (1979)
Haplopappus acradenius	Cole (1985b)
Juncus castaneus	Hartley (1976)
Juncus drummondii	Cole (1977)
Juncus parryi	Coombs (1976), Holmes and Dobson (1976), Cole (1977, 1982b), Stohlgren (1982)
Juncus tenuis	Sutton (1976), Marion (1984)
Lepidium lasiocarpum	Cole (1985b)
Lewisia columbiana	del Moral (1979)
Luetkea pectinata	Thornburgh (1962)
Luzula hitchcockii	del Moral (1979), Cole (1982b)
Muhlenbergia filiformis	Lemons (1979), Holmes and Dobson (1976), Cole (1982b)
Opuntia erinacea	Cole (1985b)
Oryzopsis kingii	Holmes and Dobson (1976)
Penstemon confertus	Cole (1983a, 1985a)
Penstemon davidsonii	del Moral (1979)
Penstemon procerus	del Moral (1979), Cole (1983a, 1985a)
Phleum alpinum	Hartley (1976)
Phlox diffusa	Singer (1971)
Plantago purshii	Cole (1985b)
Poa alpigena	Stelmock and Dean (1979)
Poa cusickii	del Moral (1979)
Poa epilis	Holmes and Dobson (1976)
Polygonum cilinode	Saunders (1979)
Rhus trilobata	Cole (1985b)
Rosa acicularis	Marion (1984)
Rubus strigosus	Marion (1984)
Schizachne purpurascens	Marion (1984)
Senecio pauperculus	del Moral (1979)
Senecio resedifolius	Hartley (1976)
Sibbaldia procumbens	Dale and Weaver (1974), Cole (1977, 1982b)
Smilax herbacea	Saunders (1979)
Solidago missouriensis	Cole (1983a, 1985a)
Solidago multiradiata	Cole (1977)
Stipa occidentalis	Cole (1983a)
Tauschia (Hesperogenia) stricklandii	Thornburgh (1962)
Trisetum spicatum	del Moral (1979), Stelmock and Dean (1979)
Xerophyllum tenax	Cole (1983a, 1985a)
Yucca baccata	Cole (1985b)

Table 2.—Relative resistance of plant community types to trampling damage

Plant community types[1]	Resistance to: Light[2] trampling	Heavy[3] trampling	Both[4]
1. *Pinus contorta/Thalictrum venulosum* (lodgepole pine forest)	VS	VS	VS
2. *Populus tremuloides/Heracleum lanatum* (aspen forest)	VS	SS	VS
3. *Populus tremuloides/Symphoricarpos albus* (aspen forest)	VS	SS	VS
4. *Vaccinium membranaceum* (subalpine huckleberry shrubland)	SS	VS	VS
5. *Phyllodoce glanduliflora* (subalpine heath shrubland)	SS	VS	VS
6. *Carex rostrata-C. aquatilis* (sedge marsh)	VS	SS	–
7. *Cassiope mertensiana* (subalpine heath)	SS	VS	SS
8. *Abies lasiocarpa/Luzula hitchcockii* (subalpine fir forest)	SS	SS	SS
9. *Aster alpigenus-Phlox diffusa* (alpine cushion community)	SS	SS	SS
10. *Picea engelmannii/Arnica latifolia* (Engelmann spruce forest)	SS	N	SS
11. *Valeriana sitchensis* (subalpine forb meadow)	SS	N	SS
12. *Picea engelmannii/Thalictrum occidentale* (Engelmann spruce forest)	SS	N	SS
13. *Pseudotsuga menziesii/Symphoricarpos albus* (Douglas-fir forest)	SS	SS	N
14. *Picea glauca/Vaccinium uliginosum* (boreal spruce forest)	SS	—	–
15. *Antennaria lanata-Carex nigricans* (alpine snowbank community)	SS	N	N
16. *Holcus lanatus-Agrostis stolonifera* (acid grassland)	N	SS	–
17. *Deschampsia flexuosa-Holcus lanatus* (acid grassland)	N	SS	–
18. *Xerophyllum tenax* (subalpine beargrass meadow)	N	SS	N
19. *Larix occidentalis/Linnaea borealis* (western larch forest)	N	N	N
20. *Lupinus lepidus-Carex phaeocephala* (alpine stone-stripe community)	N	N	N
21. *Anemone occidentalis-Trollius laxus* (subalpine forb meadow)	N	N	N
22. *Antennaria lanata-Hieracium gracile* (subalpine forb meadow)	N	–	–
23. *Phlox diffusa-Carex phaeocephala* (subalpine cushion community)	N	–	–
24. *Empetrum nigrum* (sand dune heath)	N	N	SR
25. *Arrhenatherum elatius-Holcus lanatus* (neutral grassland)	N	SR	–
26. *Pinus contorta/Vaccinium caespitosum* (lodgepole pine forest)	N	SR	SR
27. *Trollius laxus-Aster foliaceus* (subalpine forb meadow)	N	SR	SR
28. *Luetkea pectinata* (subalpine mat plant community)	N	SR	SR
29. *Calluna vulgaris-Deschampsia flexuosa* (heath-grassland)	SR	N	–
30. *Pinus albicaulis/Vaccinium scoparium* (whitebark pine forest)	SR	N	SR
31. *Arctostaphylos uva-ursi-Carex eburnea* (heath-grassland)	SR	–	–
32. *Dryas octopetala* (alpine cushion community)	SR	SR	VR
33. *Aster alpigenus-Festuca idahoensis* (subalpine meadow)	VR	–	–
34. *Pinus contorta/Xerophyllum tenax* (lodgepole pine forest)	VR	VR	VR
35. *Festuca scabrella-F. idahoensis* (grassland)	VR	VR	VR
36. *Festuca scabrella-Danthonia intermedia* (prairie grassland)	VR	VR	VR
37. *Poa pratensis-Festuca idahoensis* (grassland)	VR	VR	VR
38. *Carex nigricans* (subalpine sedge meadow)	VR	VR	VR

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[1] Sources are as follows: Nagy and Scotter 1974 (1,2,3,6,8,10,18,27,32,36); Landals and Scotter 1974 (4,7,11,28,38); Landals and Scotter 1973 (5,21); Singer 1971 (9); Schreiner 1980 (14); Bell and Bliss 1973 (15,20); Harrison 1981 (16,17,25,29); Schreiner 1974 (22,23,33); Hylgaard and Liddle 1981 (24); Weaver and Dale 1978 (30,37); Bowles and Maun 1982 (31); Cole 1985c (12,13,19,26,34,35).

[2] The index for resistance to light trampling is the number of passes required to reduce cover to 50 percent of original conditions. Classes are as follows: very susceptible (VS) (0 to 25 passes); somewhat susceptible (SS) (26 to 100 passes); neither susceptible nor resistant (N) (101 to 250 passes); somewhat resistant (SR) (251 to 500 passes); very resistant (VR) (more than 500 passes).

[3] The index for resistance to heavy trampling is relative or percent cover after 800 passes. Classes are as follows: VS (0 to 3 percent); SS (4 to 7 percent); N (8 to 15 percent); SR (16 to 35 percent); VR (more than 35 percent).

[4] The index for both is essentially the mean relative or percent cover across the range from 0 to 800 passes. Classes are as follows: VS (0 to 15 percent); SS (16 to 25 percent); N (26 to 35 percent); SR (36 to 50 percent); VR (more than 50 percent).

The most significant effect of vegetation structure relates to degree of canopy closure. Many studies have observed a tendency for more open communities to be more resistant than closed forests. For example, in a recent study in the Boundary Waters Canoe Area, Marion (1984) found that campsites with 75 to 100 percent tree cover lost 77 percent of their vegetation cover, while sites with 0 to 25 percent tree cover lost only 43 percent. This finding is all the more striking in that vegetation loss did not differ significantly among the seven forested plant communities studied.

This relationship between canopy cover and vegetation impact has a strong theoretical basis. Grime (1979) hypothesized that no plants are well adapted to environments characterized as both high in stress (in this case low light intensity) and high in disturbance. Adaptations to low light intensities make a plant particularly susceptible to trampling damage (Cole 1979). Moreover, the susceptibility of dense forests has been demonstrated in many places—the Pacific Northwest (Schreiner and Moorhead 1979; Cole 1981a), the northern forests (Marion 1984), and the Appalachians (Ripley 1962).

Important soil characteristics that influence amount of impact are soil texture, soil structure, organic matter, moisture, fertility, and depth. Regarding soil textures with the fewest limitations for campsites and trails, Montgomery and Edminster (1966), Epp (1977), and Fay and others (1977) recommend medium-textured soils—sandy loams, fine sandy loams, and loams. Such soils generally have good drainage, are not highly erodible, and have high potential for plant growth. Their major drawback is that their wide range of particle sizes makes them particularly susceptible to compaction (Lull 1959). Coarse soils generally resist water erosion because large particles are neither easily detached nor easily moved. However, their structural instability makes them vulnerable to trail widening and their low water-holding capacity and cation-exchange capacity make them relatively impoverished environments for plant growth. Such drawbacks are likely to be more serious for trails than for campsites. Nevertheless, coarse soils are clearly a better alternative than fine-textured soils. Silts and fine sands are highly erodible because soil particles are both readily detached and moved (Baver 1933). Moreover, silt soils are prone to needle ice formation when wet and become dusty when dry. The permeability of soils high in clay is greatly reduced when compacted. This promotes increased runoff and erosion. Although clay particles resist detachment, they are readily moved by running water. Clays also have little ability to support loads because they deform readily when wet. They are generally sticky when wet and they dry slowly (Leeson 1979).

The role of stones and rocks in soil is complex. Leeson (1979) suggested that they are desirable up to a volume of 25 percent of the soil because they inhibit compaction and increase the resistance of soil particles to entrainment. Above this volume, they make footing difficult and construction and maintenance costly. Bryan (1977) noted that rocks are generally advantageous unless trail degradation has advanced to the point where soil coherence is completely destroyed. Once stones are loose on the trail, rocks increase erosion potential because they increase the turbulence of running water and, when moved, corrode the trail themselves. Summer (1980) suggested not categorically removing stones from trails because this frequently leads to erosion of underlying fines, exposing more rocks. Many of the most severe erosion problems are in stone-free, homogeneous-textured soils (Root and Knapik 1972; Bryan 1977). This textural limitation may explain the problems with trail incision and development of multiple trails in mountain meadows.

Well-developed soil structure promotes drainage; this is generally good. Trampling disrupts structure, particularly when soils are high in clay and when trampling occurs when soils are wet. This loss of structure makes soils more prone to erosion. Generally soils with a granular structure and a high proportion of water-stable aggregates have the fewest limitations for recreational use (Leonard and Plumley 1979). Because trampling destroys both, it is not clear how important these properties are. Soil pans, such as the iron pan described by Bryan (1977), impede drainage and can become more heavily cemented and impenetrable in compacted soils, but, once exposed, their resistance can also serve to inhibit further incision.

The advantages and disadvantages of organic matter are also complex, varying with amount, type, and associated soil characteristics. Organic soils—as opposed to mineral soils—have particularly low bearing strengths when wet. Consequently, they are highly vulnerable to puddling (Bryan 1977) and quickly become wide, muddy quagmires where crossed by trails. Thick surface organic horizons, under forest, shield the mineral soil from compaction (Legg 1973; Marion and Merriam 1985) and inhibit runoff and erosion (Lowdermilk 1930). They also inhibit the germination of most plant species, leading to reduced vegetation cover. Incorporated into the mineral soil, organic matter promotes structural development, which enhances drainage (Leonard and Plumley 1979), inhibits compaction (Dotzenko and others 1967; Marion and Merriam 1985), helps resist dispersion and detachment of soil particles, and promotes plant growth due to its positive influence on water-holding capacity and nutrient availability (Leonard and Plumley 1979).

Soil moisture, as with most other soil parameters, is best in moderate quantities that promote plant growth but do not cause the problems common to poorly drained, wet soils. Moisture decreases the load-bearing capacity of many soils, making them more prone to compaction, puddling, and muddiness problems. Moisture problems are most severe with fine-textured soils and are most likely to cause problems on trails. Moisture also exacerbates the damage inflicted by stock on grazing areas (Strand 1979). Probably a majority of trail problems, in the West at least, result from locating trails in areas that are poorly drained or have high water tables (Root and Knapik 1972). It is often possible to identify vegetational indicators of the soils with high moisture content that are likely to develop muddiness problems unless avoided or bridged (Cole 1983b).

Limited data suggest that vegetation on moderately fertile soils is more resistant than that on highly fertile soils, with vegetation on infertile soils being least tolerant (Ripley 1962; Papamichos 1966; Kellomäki and Saastamoinen 1975; Harrison 1981). There are insufficient data, however, to evaluate how generally applicable or significant such a tendency is.

Finally, deeper soils are often better suited to recreational use than shallow soils (Fay and others 1977). This is primarily a reflection of the erodibility of very shallow soil and the vulnerability of vegetation established in pockets of thin soil. This vulnerability, attributable to very thin soil, is likely to explain the common assumption that mosses are fragile. Mosses, growing on bare rock or in shallow soils, are susceptible to being dislodged. In deep soils, trampling experiments, almost without exception, have shown them to be relatively resistant. The most resistant sites of all are bedrock.

Table 3 summarizes, in very general terms, how these soil properties influence tolerance. Although these general patterns have been identified, much more needs to be learned about how significant these factors are to the maintenance of desirable conditions. The most serious problems, or at least those that have received most attention, occur where trails are located on soils with homogeneous textures (deeply incised, braided trails in meadows) or on wet mineral or organic soils (wide, muddy quagmires).

Locational characteristics include slope steepness and position, topography, aspect, and elevation. On campsites, slopes are generally negligible. On trails, the angle of slope, both along and across the trail, and the position of the trail—close to the top or bottom of a slope—all influence potential for deterioration. These variables all affect the amount and velocity of water running down the trail.

As slope angle along the trail increases, so does erosion potential. Coleman (1981) developed regression models that related path width and depth to the square of the path slope. Her data suggest little problem until slopes exceed 12 to 13 degrees; beyond this, problems increase exponentially. On relatively flat trails, Cole (1983b) found that path depth increased significantly as slope increased, but slope only explained 8 percent of the variation in depth and was not related to path width. Water drainage devices and frequent maintenance can minimize trail problems, even on steep slopes. Moreover, trails with no slope at all have their own problems, primarily a result of poor drainage.

Trails oriented parallel to the slope channel water directly downslope and often deteriorate more dramatically than trails oriented perpendicular to the slope (Bratton and others 1979). Trails located high on slopes have smaller watersheds and, therefore, less erosion potential than trails close to the base of slopes. In valleys covered by complex glacial till deposits, trails near the base of slopes frequently intercept perched water tables (Helgath 1975).

Quite variable results are available for the effect of elevation on site durability. In Great Smoky Mountains National Park, Bratton and others (1978, 1979) reported positive correlations between elevation and both campsite and trail deterioration. Although no data are available, Fay and others (1977), working in the Northeast, suggested that deterioration problems increase with elevation. In the Sierra Nevada, Dykema (1971) found campsite alteration to be greater at both low and high elevations than at moderate elevations. An examination of the tolerance of the plant communities in table 2 shows no relationship between durability and elevation.

The complexity of factors influencing environmental tolerance makes it unlikely for a variable like elevation to relate strongly to tolerance. Perhaps the most consistent effect of increasing elevation is a decrease in length of the growing season. This, along with locally variable factors such as frequent high winds and needle ice, often makes resilience lower at high elevations. Alpine vegetation, for example, is frequently quite resistant (Grabherr 1982 and table 2), but resilience is low. In desert regions, however, higher moisture levels at higher elevations may compensate for a shorter growing season and lead to greater resilience at high elevations. Other characteristics that increase vulnerability and are more common at higher elevations include organic soils, the homogeneous fine-textured soils that are associated with certain types of glacial deposits, areas with poor drainage, and steep trail pitches. Such problems do not increase with elevation in all wildernesses, however, and at almost any elevation sites vary greatly in their level of tolerance.

The effect of aspect is similarly complex. One of the most frequent aspect-related problems occurs high in the mountains, where late snowmelt on northerly aspects contributes to trail widening and erosion through the medium of trampling of water-saturated soils and subsequent channeling of meltwater down the entrenched trail (Price 1981). Under droughty, low-elevation conditions, however, one might expect the more mesic northerly aspects to be more tolerant. In Iowa, Dawson and others

Table 3.—Relationships between soil characteristics and susceptibility to impact

Soil property	Level of susceptibility		
	Low	Moderate	High
Texture	medium	coarse	homogeneous; fine
Organic context	moderate	low	high
Soil moisture	moderate	low	high
Fertility	moderate	high	low
Soil depth	none	deep	shallow

(1974) found that trails on north-facing slopes were less compacted and lost less ground cover than trails on flood plains or south-facing slopes. In northern Utah, vegetation resistance to simulated trampling was greatest on steep northeast slopes, at low elevations (Cieslinski and Wagar 1970). While such generalizations might provide useful guidelines within localized areas, their general utility is questionable.

At the ecosystem level, some have proposed that tolerance increases with primary productivity (Liddle 1975b) and with more advanced successional stages (Goldsmith 1974). Liddle felt that productivity summarized potential for regrowth and the overall hospitality of the environment. Data from five studies generally support the relationship, provided that only the productivity of the ground flora in woodland is considered. However, when quite different physiognomic types are compared, the relationship breaks down. Desert shrub, for example, is notably unproductive and yet it is more resistant to low to moderate levels of recreation use than coniferous forest vegetation (Cole 1983a, 1985b). Perhaps productivity might relate more to resilience than to resistance.

Goldsmith also noted—once he stated that earlier stages in succession were more fragile—that this principle must be qualified by differences in the tolerance of growth forms and by limiting environmental variables. Thus, a seral grassland stage may prove to be more resistant than a climax forest because grasses are more tolerant of trampling than broad-leaved forbs. The theory behind the importance of successional stage stems from the concept that diversity begets stability—that more diverse environments are less susceptible to disturbance. Because later successional stages tend to be diverse, they should be more tolerant of recreational pressure. This concept is generally considered to be an oversimplification and this is certainly true for its application to recreational tolerance. *Carex nigricans* meadows, for example, have low diversity, and yet they are very resistant after soils dry out (table 2).

In sum, it appears that vegetation resistance is probably most highly dependent on the growth form of the constituent species. As this can be highly variable locally, resistance can vary greatly over short distances (Cole 1985c). Resilience, in contrast, is more dependent on environmental factors such as soil fertility, length of the growing season, sunlight levels, and moisture levels. Therefore, it may be possible to demonstrate reduced resilience at higher elevations and with lower primary productivity and diversity. Further work might lead to better theoretical development along these lines.

There is little doubt that an improved understanding of how environmental factors influence impact problems would be tremendously useful to management. By reducing impact through improved site selection, more people can enjoy the wilderness with less impact. Given the complex variables involved and great variability from place to place in the significance of different variables, it is important for wildernesses to develop their own guidelines concerning durable and fragile locations. Many of the best examples of site-specific analysis of impact problems and how they can be avoided are Canadian (for example, Lesko and Robson 1975; Root and Knapik 1972; Landals and Knapik 1972; Landals and Scotter 1974). These studies assess the nature of problems, their causes, how they relate to use and environmental variables, and provide management guidelines to avoid future problems and correct existing problems. In this country, four examples of site-specific research to guide management are: (1) Fay and others' (1977) guidelines for locating and designing overnight facilities in the Northeast, (2) Summer's (1980) ratings of erosion potential and likely trail deterioration problems for major landform units in Rocky Mountain National Park, (3) Cole's (1982a) descriptions of design and locational considerations for major vegetation communities in the Eagle Cap Wilderness, and (4) DeBenedetti and Parsons' (1983) meadow type-specific management recommendations for grazing. Management of most other resources benefits from knowledge about environment variability and its importance to management; wilderness should be no different.

Evaluations of Management Techniques

Remarkably few attempts have been made to evaluate the effectiveness of different techniques that are or could be used to minimize ecological impacts. Few natural experiments have assessed the consequences of management actions that have been taken, and even fewer papers have reviewed existing data and theory for their application to management. This contrasts with the social-psychological side of recreation research where natural experiments have frequently been used to evaluate the effectiveness and acceptability of such techniques as use rationing (Stankey 1979) and the use of information to disperse use (Roggenbuck and Berrier 1981).

Several studies have evaluated the effectiveness of campsite closures as a means of reducing impact. Temporary closures in the Selway-Bitterroot Wilderness were generally ineffective. Little improvement in condition occurred over 8 years of closure. Meanwhile, new campsites developed near the closed ones; after 8 years, deterioration on the new sites was almost as pronounced as on the old sites. Poor success reflected slow natural recovery rates, the lack of assisted recovery, and poor compliance with closures (Cole and Ranz 1983). This suggests that rest-rotation is not a practical means of reducing impact. Slow rates of recovery have also been documented in Sequoia/Kings Canyon National Parks (Parsons and DeBenedetti 1979; Stohlgren 1982).

Limited data from Kings Canyon National Park suggest that implementation of a 1-night length-of-stay limit and a ban on wood fires has reduced campsite damage. One currently little-used site has recovered much of its litter and duff cover and woody fuels following imposition of these regulations (Parsons 1983). Finally, a series of meadow condition assessments, also from Sequoia/Kings Canyon National Parks, demonstrate the effectiveness of a number of actions taken to reduce stock damage (Sumner 1968).

Cole (1981b) reviewed the literature for implications about the likely success of reducing campsite impact through use dispersal and campsite closures. He concluded that dispersal is likely to aggravate impact problems, except in lightly used areas, that temporary closures are seldom effective, and that permanent closures should be evaluated on an individual basis, as opposed to blanket policies such as 200-foot setbacks from water. Craig (1977) advanced his opinions about certain campsite management techniques, recommending containing use to designated sites in heavy-use areas, limiting party size, zoning, hardening sites and providing waste-handling facilities where necessary, and restoring sites. Fay and others (1977) provided a useful summary of location and design criteria for overnight facilities, most applicable to the Northeast. Cole and Dalle-Molle (1982) put together a handbook for planning for and managing campfire impacts.

Trail management is less complex and generally better understood than campsite management. Proper location, design, and maintenance are more important than visitor management (Cole 1983b). Proudman (1977) provided a good summary of how to design, build, and maintain trails in such a way that damage is minimized. In my opinion, little further research on trail management is needed, although individual areas must develop area-specific methods for dealing with their unique situation and problems. This may require correlating trail problems with environmental conditions and monitoring the effectiveness of alternative designs for handling problems (Cole 1983b).

Site Rehabilitation

Significant effort is being expended in attempts to rehabilitate highly impacted backcountry sites. Such efforts are necessary due to the long recovery periods required when recovery is not assisted. A considerable body of information relevant to rehabilitation is accumulating, although it is widely dispersed and most is not directly applicable to wilderness. Perhaps the best initial sources of information on rehabilitation of wilderness sites—although both are now over 5 years old—are Cole and Schreiner's (1981) bibliography and the proceedings of the Recreational Impacts on Wildlands Conference (Ittner and others 1979). A tremendous amount of experience is not being shared because documentation and dissemination of results are poor.

Most of the rehabilitation work in backcountry that has been written up (but not published) is from the Pacific Northwest. Rehabilitation programs at North Cascades, Olympic, and Mount Rainier National Parks have been in the forefront. For this region, considerable progress has been made toward identifying suitable species for either seeding or transplanting (Miller and Miller 1976; Schreiner 1977). Greenhouse propagation techniques for producing transplants have been developed by Miller and Miller (1979). Other papers provide hints about the mechanics of transplanting, from guidelines concerning transplant size to suggestions about watering and fertilization (Miller and Miller 1976; Scott 1977). Two excellent reports (Dalle-Molle 1977; Miller and Miller 1977) discuss how to handle specific problems, such as frost heaving and keeping people off transplants. In the Pacific Northwest, transplanting is generally more successful than seeding, watering is important, and the need for fertilization is questionable and unresolved. One study in the Northeast found fertilization and liming to be ineffective in increasing vegetation cover, although the study's conclusion was to increase amounts, not discontinue the practice (Fay 1975).

One of the few documented experiments testing means of rehabilitating trails occurred on a stretch of multiple trails in Yosemite National Park (Palmer 1979). Of 22 techniques tried, the most successful involved cutting off the sod ridges between trails at the level of the trail tread and stacking them in the shade. The soil beneath both trails and ridges is dug up and sand is added to bring the area up to the level of the surrounding meadow. Finally, the stacked sod is divided into transplant plugs and planted. Considerable progress in rehabilitation could be made if those working with rehabilitation would invest more energy—as Palmer did—in documentation, experimentation, monitoring, and communication of results.

A lot of work was done by Forest Service researchers, in the 1960's particularly, on rehabilitation of developed campgrounds. In several studies involving attempts to establish grass, shrubs, and small trees on campsites in the Southeast, the most conclusive finding was that reducing overstory canopy cover greatly increases grass production (Cordell and Talhelm 1969; Cordell and James 1971; Cordell and others 1974). In Idaho and Utah, watering, seeding, and fertilizing maintained high levels of nonnative grass on campsites (Beardsley and Wagar 1971; Beardsley and others 1974). In Maryland state parks, mulching also contributed to revegetation success (Little and Mohr 1979). All studies stress the overriding importance of designing traffic flow to keep people off vegetated parts of the site.

Research on rehabilitation of nonrecreational sites varies greatly in its applicability to wilderness. Revegetation of high-altitude lands has been addressed in several places—a series of conferences on the topic (such as Kenny 1978), a Forest Service research program focused on the Beartooth Plateau (Brown and others 1978), and a history of work on depleted rangelands (for example, Hull 1974). Such studies help in the identification of factors limiting success, choice of species for rehabilitation, and, to a lesser extent, rehabilitation techniques. Because mining disturbance and rangelands are concentrated in the West, more research is available there than in the East.

Some of the most interesting work is basic research on the factors that limit revegetation success. Perhaps the most notable program here is the work of Marchand and coworkers (Marchand and Roach 1980; Marchand and Sproul 1981; Roach and Marchand 1984) on *Arenaria groenlandica*, *Juncus trifidus*, and *Potentilla tridentata*, common native species found in alpine areas along the Appalachian Trail in New Hampshire. Detailed autecological work has provided more understanding about seed production, dissemination, and germination and the early growth and survival of these species. Major limitations to recolonization by these species are lack of vegetative reproduction, poor seed dissemination, and frequent mortality by frost heaving.

Schreiner (1982) studied the autecology of a native (*Poa incurva*) and a nonnative (*Poa pratensis*) colonizer of disturbed sites. He concluded that traditional site restoration techniques, such as watering and fertilization, would favor *Poa pratensis* at the expense of native species. Research on mycorrhizae is also beginning to reveal their role and importance in rehabilitation (Reeves and others 1979).

The research on revegetation is too extensive to review adequately in several pages. However, much more is needed if we are to increase the cost-effectiveness of the rehabilitation work already underway. The top priority, as mentioned earlier, should be better documentation of what is being done and its success. Major issues that should be addressed include (1) advantages and disadvantages of transplanting versus seeding, (2) the value of scarification, (3) the value of fertilization, (4) the value of and recommendations for mulches, (5) how to reduce frost heaving problems, and (6) whether or not there should be any role for nonnative plants.

In addition, more basic research could improve our understanding of factors that limit the establishment and spread of vegetation. Our poor understanding of the impact process and the importance of documented impacts to ecosystem functioning is a serious problem. We need to understand why many transplants survive year after year, but do not spread. We need to understand more about the importance of mycorrhizae, and we need more autecological studies like those of Marchand and Schreiner.

Impact Monitoring

One final area of research is the development of monitoring systems. Most effort has gone into systems for identifying change on campsites. If replicable, the sampling procedure of any campsite impact study could be the basis of a monitoring system. A number of studies have proposed procedures specific to the task of monitoring, however.

Perhaps the earliest monitoring research was Walker's (1968) exploratory attempt to use panoramic photographs, monoscopic photographs, and stereophotogrammetry. On the basis of his suggestions, 360-degree panoramic photo mosaics, taken from permanent points located near the center of campsites, were incorporated into a monitoring system used by the Selway-Bitterroot Wilderness. While gross changes—fallen trees, new fire rings, and so on—could be identified, it was difficult to identify ground cover changes and any changes beyond the closest trees. Although photographs are unlikely to completely replace the need for field measurements, they can help identify sites for future reassessments, record campsite features not measured in the field, and provide a visual supplement to data collected in the field. Brewer and Berrier (1984) provide useful guidelines for the use of ground-based photography.

In open areas, aerial photography can be used to monitor impacts. Boorman and Fuller (1977) and Price (1981) provided examples of trail deterioration analysis based on air photo interpretation. In Grand Canyon National Park, devegetated trails and camping areas on beaches along the Colorado River have been monitored with large-scale, low-elevation aerial photographs.

Field measurements taken at campsites vary greatly, particularly in level of precision and consumption of time. Hendee and others (1976) proposed a Code-A-Site form for inventory and monitoring. The only site condition information they asked for was an undefined judgment as to whether prior impact had been extreme, heavy, moderate, or light. Some precision was added to this system when managers of the Selway-Bitterroot provided written definitions for these four classes. This was also the approach advanced by Frissell (1978). He proposed a system of five condition classes based on extent of vegetation loss, litter loss, tree root exposure, erosion, and tree mortality.

More elaborate systems have been suggested by Parsons and MacLeod (1980) and Cole (1983c). These systems quickly rate campsites on the basis of such impact parameters as vegetation loss, change in species composition, campsite area, area of barren core, campsite development, campsite cleanliness, loss of organic matter, social trails, tree mutilation, and tree root exposure. Compared to "condition class" systems, these have the advantage of more information, at little additional cost, and disaggregated data on individual impact parameters. Thus, they are more flexible, more generally useful, and more likely to provide insight into exactly how a site is changing over time.

The major alternative to such rapid estimation techniques is a measurement system for campsite area and devegetated area first developed by Schreiner and Moorhead (1979) for use at Olympic National Park. Mount Rainier National Park adapted Schreiner and Moorhead's system of radial measurements to produce reasonably accurate sketch maps to monitor campsite changes. Although quite time consuming, such a system will detect more subtle changes than rapid estimation systems.

Generally, research on campsite monitoring is sufficient for most needs. The next step is transfer of technology and adaptation of existing systems to meet local needs.

TECHNOLOGY TRANSFER

It is difficult to evaluate how extensive technology transfer is—both because it is impossible to keep track of all applications and because it is seldom clear where the idea behind an action came from. The best opportunities for technology transfer are provided in situations where the researcher and the manager have a shared interest in a project and cooperate from the beginning. This situation is most common in the Park Service, although cooperative examples elsewhere do occur. Transfer also becomes easier—or at least is more easily measured—the more tangible the research product. Thus, probably the foremost readily traceable example of successful technology transfer has been the recent spread of campsite monitoring systems.

Following the development of the Code-A-Site form (Hendee and others 1976), quite a few wildernesses inventoried their campsites using the form. Common problems that arose were insufficient information on campsite condition and uncertainty about what to do with the information collected. In Olympic National Park, more relevant information was developed for use with the form and systems for data analysis and periodic reassessments of conditions were developed (Schreiner and Moorhead 1979). Neighboring Mount Rainier National Park borrowed from and modified this innovation.

A compromise between the time-consuming Olympic-Mount Rainier type of system and the low-information Code-A-Site system was developed for use in Sequoia/Kings Canyon National Parks (Parsons and MacLeod 1980). It is also being used in neighboring Yosemite National Park. Cole (1983b) modified the Sequoia/Kings Canyon multiple estimated parameters system by incorporating new parameters, improving estimation techniques, and keeping data disaggregated. Cole's paper also linked the campsite monitoring system to the Limits of Acceptable Change (LAC) planning framework for wilderness management—an attempt to base management on specific objectives (Stankey and others 1985). The combination of a tangible product that is simple to apply, not time consuming, and fulfills a recognized management need led to rapid spread. Within the Forest Service, the Pacific Northwest Region (Oregon and Washington) is adopting the system. It has spread widely in the Northern Rockies (for example, Bob Marshall, Mission Mountains, Rattlesnake, Lee Metcalf, and Frank Church-River of No Return Wildernesses), and a few areas in the Southwest (for example, Superstition Wilderness) are experimenting with it. Through Marion (1984), the system spread first to the Boundary Waters Canoe Area and then to National Park areas along the Delaware River in the East. Through Cole's (1985b) work at Grand Canyon, a similar system is also being applied there.

Another example of successful research application is the management program at Sequoia/Kings Canyon National Parks. This success reflects close cooperation in the Park Service between research and management. At Sequoia/Kings Canyon, management has profited from research on campsite recovery (Parsons and DeBenedetti 1979; Stohlgren 1982), research on meadow condition and response to clipping (DeBenedetti and Parsons 1983), and extensive use of data on use permits and campsite monitoring forms. Particular achievements include a rational use limitation program (Parsons and others 1981), prescription of a variety of actions to deal with problems at a very high-use lake basin (Parsons 1983), and proposal of a comprehensive stock management program (DeBenedetti and Parsons 1983).

Unfortunately, these successes are far outweighed by the lack of research application. Three principal barriers to effective technology transfer are: (1) an adequate research base has not been developed, (2) too little attention has been paid to drawing management implications out of research results, and (3) managers have not always looked to research for solutions to their problems.

Currently, the problem of an insufficient research base is the most serious and basic of these barriers. Without a research base to start with, the other problems are moot. Current allocation of resources between management and research—with very little funding in research—may be adequate for dealing with current "brushfire" problems but is not cost-effective in the long-term because little investment is made in improving future management. The current allocation of recreation management research almost entirely into social research also seems unbalanced. It ignores the primary goal of wilderness management—maintenance of natural conditions—and the fact that ecological impact problems are less reversible than social problems.

The current situation is well illustrated by how resources are being allocated in minimum impact education programs. In 1980, over one-half of all wildernesses were investing management time and funds in such programs (Washburne and Cole 1983), and there are undoubtedly many more areas that now have such programs. Investment in research, both on what to teach and how to teach cost effectively, has been minimal, however. Moreover, what has been done is entirely on how to teach rather than what to teach—in my view, a classic case of getting the cart before the horse. So now we are in a position of investing scarce wilderness management funds in a curriculum based entirely on personal judgments and opinions (many of them good, I should say). Even a minor investment in what to teach and how to do it would more than pay for itself in little time.

Much of the recreational ecology research that has been done provides little basis for management application. A lack of careers in the field has meant that studies have generally been limited to short-term studies at one place and time. Consequently, results are generally superficial, there is no opportunity to gain the insights that come with experience, and there has been little chance to compare results from a number of places to assess the general applicability of conclusions. This helps explain the lack of theoretical development in the field. Because most studies are done by students at the master's level or as one-time projects, there has been a strong tendency to undertake relatively simple projects generally cut from the same mold. Consequently, we have numerous studies documenting changes in vegetation and soil conditions on recreation sites with little idea of significance.

Designing research so that it is most useful to management is also difficult given the current situation. Cooperation between research and management is limited. Aside from a few Park Service researchers, no agency researchers work in the recreational ecology field. In a few situations, academicians have worked closely with managers (for example, Marion 1984), improving the likelihood of future application, but all too frequently this is not the case. Short-term involvement in the field gives little opportunity for developing the kind of rapport that facilitates the technology transfer process.

Lack of careers also means less attention is given to the management of ecological impacts in recreation management programs in colleges and universities; recreation departments are staffed almost entirely by social scientists. With little training in management of ecological impacts, it is difficult for managers to search effectively for solutions to impact problems. This problem is aggravated by the schedules of most wilderness managers; they are sufficiently busy to preclude much time searching the literature for innovative management solutions.

If careers in recreational ecology were possible, we could build a more substantial research base and improve training of recreation professionals. Over a career, a researcher could gain insights from experience, develop the management implications of research, and build rapport with managers. The challenges of designing research to deal with critical management questions and encouraging the application of results would remain, but the outlook would improve significantly.

RESEARCH GAPS AND FUTURE DIRECTIONS

Problems with maintaining natural vegetation and soil conditions in wilderness are serious. They undermine the intentions of the Wilderness Act and, in many cases, are irreversible. Many impacts are readily obvious even to untrained observers, and simple solutions, such as closing sites to use, are usually available. However, detection of subtle changes and development of innovative solutions that minimize elimination of recreational opportunities and avoid simply moving problems around demand research beyond the "documentation of the obvious" that has characterized the recreational ecology field.

In my opinion, the primary reason for the poor state of the art is lack of support for such research. Despite allocation of almost 90 million acres to wilderness, a land classification with a primary goal of maintaining natural conditions, no continuing program exists in wilderness impact research. Consequently, available research is generally confined to a large number of descriptive studies that lack much time frame, theory, or comparability. Unless support for a critical mass of researchers is forthcoming, there is little to suggest that the current situation will improve appreciably.

To improve, changes need to be made in research design and approach. There is a critical need for more longitudinal studies. The current reliance on cross-sectional studies provides us with a perspective confused by spatial and temporal variability. Cause and effect are difficult to unravel, and there is little to suggest process or the significance of long-term processes. Study periods, for both before-and-after studies and experimental studies, need to be lengthened. Long-term studies will be possible only if researchers have long-term support.

More specialized, detailed studies of the impact process are also needed. Researchers have always had to be "jacks of all trades" to deal with the wide variety of impact problems. Now we need researchers with specialized knowledge, both in subject matter and methodologies. We need to go beyond measurements of soil compaction, for example, to studies of effects on more basic soil properties, such as aeration, micro-organisms, and mycorrhizae. At an even more fundamental level, we need to understand relationships between these altered soil characteristics and plant establishment, growth, and reproduction. Then we will be in a better position to mitigate and rehabilitate impacts.

There is also need for more interdisciplinary approaches. The simultaneous consideration of more elements of the recreational environment would contribute to the development of synergistic insights and more realistic perspectives on management. We need both greater specialization within the natural sciences and more cooperation between natural and social sciences.

We need to expand regional coverage to parts of the country that have seldom been studied (almost any place other than the Sierra Nevada, Pacific Northwest, Northern Rockies, and Boundary Waters Canoe Area). Some of this research needs to be designed to be comparable with studies undertaken elsewhere. This would improve our current ability to assess the general applicability of research results. Other research should remain site specific. A good example would be research, modeled after the grazing research at Sequoia/Kings Canyon National Parks, to develop similar grazing management programs adapted to the unique situation of the area where the research is being done.

Many topics stand out as high-priority research needs. We clearly need an improved knowledge of wilderness ecosystems and their dynamics. This is important from the standpoint of evaluating our actions outside wilderness as well as improving wilderness management. Such information is important to the wilderness manager as a planning tool for evaluating likely consequences of management actions and as a picture of baseline conditions to be maintained.

Another major research need is information to improve management of both recreational and nonrecreational grazing. Neither is well understood within the context of an area with nature conservation goals like wilderness. Although both are permitted uses that detract from certain management goals, such conflicts can be minimized through research designed to tailor management to a given use and environmental situation. The Sequoia/Kings Canyon program (DeBenedetti and Parsons 1983) provides a good recreational grazing example, but no comparable domestic livestock example exists.

Along the lines of more traditional recreational ecology, top priorities include identification of use/impact relationships for varied ecosystem types, improved campsite design and/or maintenance practices, improved site rehabilitation techniques, improved recommendations for minimum impact camping, and more evaluation of management practices. Improved understanding of use/impact relationships could help identify use thresholds above which impacts become unacceptable. This information is needed to decide between requiring use of designated sites and allowing at-large camping, to impose use limits, and to attempt to direct use to resistant sites. This is particularly important in managing impacts in low-use portions of wildernesses where management emphasis is on dispersal and low-impact use.

Where use is concentrated on a few designated high-impact sites, design and maintenance techniques are needed to channel use and avoid site expansion. "Intersite zones" need to be created and maintained, to be used as nurseries to replace overstory trees as they die. Experimentation with means of doing this should be a high priority. Improvement of site rehabilitation techniques is a closely related research topic. It is not clear, at this point, how much of this work can be generally applicable and how much is site specific or how much should be done by researchers as opposed to managers. What is clear is that we need both general and site-specific work and improved cooperation and communication between researchers and managers as well as between managers of different wildernesses.

I have mentioned the need for research on appropriate minimum-impact techniques a number of times in this paper. Although the value of recommendations such as not cutting down trees cannot be questioned, the advisability of certain fire building techniques is debatable. Moreover, the appropriateness of techniques varies between environments and between high-use and low-use situations. Improved knowledge, through research, could avoid problems of recommending inappropriate behavior and could tailor user behavior to different situations.

Finally, we should take advantage of the countless natural experiments that are taking place whenever management or use patterns change. We need to monitor change and evaluate the effectiveness of alternative techniques for minimizing impact.

The ultimate payoff of recreational ecology research to management is in efficient achievement of objectives. Impacts will never be eliminated, but both their distribution and their severity can be controlled and kept within acceptable limits. Actions taken to achieve objectives can be tailored to particular use and environmental situations that vary throughout the wilderness. To do this, however, we will need to finally make a serious investment in recreational ecology research, as we have in most other resource management disciplines.

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