82

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VEGETATIONAL CHANGES ASSOCIATED WITH RECREATIONAL USE AND FIRE SUPPRESSION IN THE EAGLE CAP WILDERNESS, OREGON: SOME MANAGEMENT IMPLICATIONS

DAVID N. COLE

USDA Forest Service, Intermountain Forest and Range Experiment Station, Forestry Sciences Laboratory, Missoula, Montana 59801, USA

ABSTRACT

Construction and use of trails and campsites, grazing by recreational packstock, and suppression of fires have altered the vegetation of the Eagle Cap Wilderness Area in northeastern Oregon, USA. The amount and type of alteration has varied among vegetation types and with the kind of activity. Changes attributed to intensive recreational use are localised but severe, particularly in more densely forested areas. Suppression of fires has produced subtle but widespread changes. Current attempts to minimise these changes are often inadequate due to the lack of ecological information and the difficulties of implementing regulations. Management suggestions are offered.

INTRODUCTION

Most of the interest in preserving remnants of wilderness in the United States has focused on the allocation of land for this purpose. Although this effort has led to the establishment of numerous wilderness areas, little has been done to define and develop wilderness management goals and techniques. Few people have recognised that true protection demands more than the establishment of boundaries (Lucas, 1973; Hendee, 1974). As a result, dramatic increases in wilderness use and the persistence of ecologically disruptive land use and management policies are already eroding the quality of the very wilderness ecosystems intended for preservation. Deterioration will continue until enlightened management programmes are developed. Such programmes will require extensive research on the impact of human activities on wilderness ecosystems and the development of management techniques capable of minimising this impact.

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Much of this research effort must be concerned with the impact of human use and management strategies on wilderness vegetation (Heinselman, 1965; Stone, 1965). Relevant research, such as studies of vegetational responses to trampling, can be found scattered through the ecological literature (Liddle, 1975). The results of these studies need to be integrated and applied to the solution of wilderness problems. Furthermore, most of this research has been confined to one type of activity and its effect on a limited range of vegetational variability. There is also a need for studies that compare the impact of many different human activities on the diverse vegetation typical of most wilderness areas.

This paper attempts to address both of these needs. It provides an overview of human impacts on wilderness vegetation resulting from the construction and use of trails and campsites, grazing by packstock, and fire suppression. Relevant literature is briefly summarised in a general discussion of vegetational changes resulting from each of these activities.

The impacts of each of these activities on the vegetative structure and floristic composition of common vegetation types in the Eagle Cap Wilderness in northeastern Oregon is then described in more detail. The comparison of impacts resulting from each activity identifies the most ecologically disruptive activities and suggests which impacts need the most immediate attention. The comparison of changes in different vegetation types identifies those types that are most susceptible to each source of disruption.

Although wilderness management must be concerned with both preservation of natural conditions and provision of opportunities for wilderness recreation, this paper only discusses vegetative alteration. This does not imply that the human impacts described are unacceptable; that is a decision that managers must make after weighing up the goals of preservation and use. Instead, the intent is to provide ecological information that will contribute to a more comprehensive base of knowledge for wilderness decision-making.

REVIEW OF IMPACTS ON WILDERNESS VEGETATION

Recent increases in recreational use have caused changes in the vegetation of wilderness areas throughout the United States. Most of these changes occur along trails and in campsites where use is most intense. However, less frequent use of other areas, particularly streambanks and lakeshores, and trampling and grazing by recreational packstock have also affected vegetation.

The construction and use of trails

Trail construction and use affects vegetation in four major ways: (1) improved access increases trampling and grazing of the vegetation; (2) increased trampling and grazing alters soil conditions; (3) site manipulation associated with trail construction (e.g., brush removal and earth movement) removes vegetation and changes microclimatic and topoedaphic conditions; and (4) new vectors of plant dispersal are introduced.

Numerous studies of trampling effects on vegetation and soil conditions show that although low levels of trampling may increase vegetative cover (Bayfield, 1971), increased treading consistently reduces vegetative height, weight, and cover (Bates, 1935; Goldsmith *et al.*, 1970; Chappell *et al.*, 1971; Liddle, 1975). Few plants can survive the stress of recurrent trampling and a highly compacted soil. Furthermore, those individuals that do survive are unable to maintain the biomass typically attained on less disturbed sites (Liddle, 1975). Consequently, heavily-used trail corridors exhibit three conspicuous parallel zones: an essentially bare trail tread, a zone of reduced vegetation 1–2 m wide on either side of the trail, and essentially undisturbed vegetation beyond this zone (Bates, 1935; Bayfield, 1971; Burden & Randerson, 1972; Dale & Weaver, 1974).

The floristic composition of the trailside zone differs from that of undisturbed areas because trampling and grazing exert a selective force favouring species with particular morphological and physiological characteristics (Frenkel, 1970). Many researchers have noted that the graminoid growth-form consistently dominates heavily trampled areas (Bates, 1935; Davies, 1938; Burden & Randerson, 1972), especially where these plants grow in tussocks (Liddle & Greig-Smith, 1975), or where they reproduce vegetatively (Dale, 1973; Liddle, 1975). Small dicotyledonous species, with leaves that grow flat against the ground, are common in areas subject to moderate levels of trampling (Dale & Weaver, 1974; Liddle, 1975). Woody plants, in contrast, tend to be eliminated from consistently trampled areas (Wagar, 1964; Dale & Weaver, 1974).

In addition to removing trailside cover, site manipulation also affects the composition of plant communities where it alters topoedaphic and microclimatic conditions along the trail. Common topoedaphic alterations include creation or removal of bare rock faces, deposition of debris below the trail, importation of material to improve a trail surface, and superposition of a flat surface on a steep slope. Moisture conditions along trails are altered, directly where drainage systems are intercepted, and indirectly through tree cutting and brush removal. This latter action increases direct precipitation and light intensities and decreases evapotranspiration rates along the trail (Dale & Weaver, 1974).

Furthermore, floristic composition along trails responds to the introduction of new seed dispersal mechanisms, the human, his packstock and in some cases his dogs. Trail traffic increases the mobility of animal-dispersed plants and provides a means for rapid invasion by exotic species. The most abundant species along low elevation trails in Eagle Cap Wilderness, for example, is *Trifolium repens*, a Eurasian clover. The seeds of this species probably entered the wilderness in the stomach of packstock and germinated following excretion along the trail.

The development and use of campsites

The plant populations on campsites respond to mechanisms of change similar to those which operate along trails, although trampling stress is usually more severe on campsites and site manipulation is less extreme. Additional impacts on campsites include the cutting and scarring of living trees. This often increases the incidence of decay and in some species, such as *Populus tremuloides*, can result in almost complete stand mortality in a few decades (Hinds, 1976).

One further problem on campsites and surrounding areas results from the collection and burning of firewood in campfires. This practice leads to trampling of the vegetation and also removes a nutrient source from large areas of forest, concentrating minerals around fire pits. The heat of campfires results in the loss of nitrogen and organic matter from the surface soil and often creates a water-repellent layer in the soil, thereby encouraging accelerated erosion (Fenn *et al.*, 1976).

In non-wilderness campsites and picnic grounds, researchers have documented decreases in the cover and density of herbs, shrubs, and tree seedlings (Lutz, 1945; LaPage, 1967; Merriam *et al.*, 1973; Brown *et al.*, 1977; Foin, 1977; Young, 1978), decreases in the diameter growth of some tree species (LaPage, 1962; Magill & Nord, 1963; Brown *et al.*, 1977), and decreases in organic matter at the surface and in the soil (Dotzenko *et al.*, 1967; Settergren & Cole, 1970; Dykema, 1971; Dawson *et al.*, 1978). They have also recorded increases in soil compaction and soil density, and decreases in soil pore volume, air capacity, and infiltration rates (Lutz, 1945; LaPage, 1962; Dotzenko *et al.*, 1967; Settergren & Cole, 1970; Merriam *et al.*, 1973; Young & Gilmore, 1976; Dawson *et al.*, 1978), increases in soil moisture stress (Dotzenko *et al.*, 1967; Settergren & Cole, 1970), and evidence of soil erosion, including hard-packed surfaces, gullying, and root exposure (Magill & Nord, 1963; Dotzenko *et al.*, 1967; Settergren & Cole, 1970).

Heavily used campsites (occupied 61–90 days/year) in a wilderness area in northern Minnesota lost 87% of their groundcover and 100% of their tree reproduction while campsites that received only light use (0–30 days/year) lost 80% of their cover (Frissell & Duncan, 1965). Other studies show similar insignificant differences between lightly and heavily used developed campsites in measurements of species composition, amount of organic litter, shrub cover, organic matter content, soil compaction, bulk density and soil moisture content (Dotzenko *et al.*, 1967; Young & Gilmore, 1976; Young, 1978). This suggests that beyond a relatively low threshold value increases in use cause little additional ecological impact. Therefore attempts to reduce impact by reducing use or redistributing it from heavy use areas to light use areas may not be very successful and may have other negative consequences, such as increased impact in other areas.

Grazing by packstock

Heavy grazing by packstock reduces the cover and alters the floristic composition of localised areas, particularly in meadows. Cover loss is a product of both the continual removal of biomass by grazing and the abrasion and compression of plants by trampling. Floristic composition changes because some species are better adapted morphologically to survive these stresses and because some species are more frequently grazed than others. In Eagle Cap Wilderness, this latter factor, palatability, appears to be the more significant. The highly palatable species that are morphologically adapted to survive grazing and trampling are less common than unpalatable species that exhibit fewer morphological adaptations. Thus many palatable grasses and sedges have decreased in abundance despite the superior adaptation of this growth-form to grazing (Daubenmire, 1974). Such species have usually been replaced by less palatable forbs.

The impact of fire suppression

Throughout most of the forests of the western United States, fire has been the principal determinant of stand structure, floristic composition, and vegetative pattern (Heinselman, 1971; Habeck & Mutch, 1973; and others). Periodic fires either thin or destroy the overstory, creating more suitable sites for the germination of shade-intolerant species. This balances the ever-present tendency toward canopy closure and selection for shade-tolerant species, thereby contributing to maximum species diversity (Loucks, 1970; Cole, 1977*b*). As the primary historic agent that destroys old stands and initiates new ones, fire also acts to maximise structural diversity (Heinselman, 1971; Habeck, 1972).

During the last 40–50 years, however, there has been an ever-increasing ability to suppress forest fires. In response to this decrease in fire frequency, many forests are experiencing increases in density and litter accumulation (Dodge, 1972). These changes favour species that can reproduce abundantly in heavy shade and deep litter and inhibit those that regenerate best in the open and in mineral soil. Lack of fire also removes the selective advantage of species adapted to survive fire. Thus, fire suppression is causing a change in the floristic composition of these forests.

THE CASE OF EAGLE CAP WILDERNESS

An area suitable for intensive study was selected within the Eagle Cap Wilderness in northeastern Oregon (Fig. 1). The study area included the contiguous drainage basins of Hurricane Creek and the West Fork of the Wallowa River. Fed by subalpine lakes, these streams flow in steep-walled canyons, which drain 15,000 ha on the north slope of the Wallowa Mountains. The glaciated topography is typical of many wilderness areas in the United States. Elevations range from 1400 m to 3000 m. The most common rock types are granodiorite, basalt and partially metamorphosed limestone and shale (Smith *et al.*, 1941).

Climatically the area is characterised by short, mild summers and long, cold winters. Immediately north of the study area, at Joseph, Oregon, mean maximum

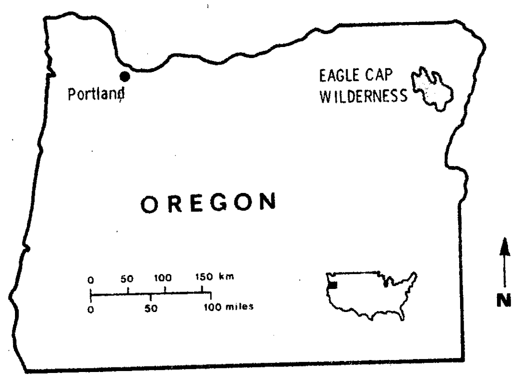

Fig. 1. Location of Eagle Cap Wilderness.

and minimum temperatures average 0 °C and −11 °C in January and 26 °C and 8 °C in July. The mean annual precipitation in most of the Eagle Cap Wilderness is probably 100–200 cm. Except for a dry period in July and August, precipitation is distributed equitably throughout the year (US Weather Bureau, 1965).

Coniferous forests occupy most of the study area. The most conspicuous tree species, below 2000 m, is *Pseudotsuga menziesii* var. *glauca* (nomenclature follows Hitchcock & Cronquist, 1973). Higher elevation forests are dominated by *Abies lasiocarpa* and *Picea engelmannii*, species that also descend to lower elevations along valley bottoms. *Pinus albicaulis* joins these species on exposed subalpine sites and extends upward in elevation as the dominant timberline tree species. *Pinus contorta* and *Larix occidentalis* are common in some areas, particularly on sites formerly burned in wildfires, and *Pinus flexilis* is prominent on calcareous substrates. Non-forested sites include valley bottom and subalpine meadows, where the water table is perennially high, talus and avalanche slopes and ridges which extend above timberline (Cole, 1977*a*).

Prior to designation of the Eagle Cap Wilderness Area in 1940, livestock grazing was the major human impact on the vegetation. The aboriginal Indians entered the area to hunt game and gather foodstuffs, but were only seasonal visitors, and their numbers were always small (Josephy, 1965). They accidentally and intentionally set fires (Cole, 1977*a*), but this merely augmented the effects of lightning fires to which the vegetation was presumably adapted.

The settlers entering the Wallowa Valley in the 1870s had recognised the potential of the mountain meadows for summer pasture and by the end of the 19th century more than 380,000 sheep were utilising this resource annually (USDA Forest Service, 1974). This intensive grazing led to changes in species composition, loss of vegetative cover, and accelerated erosion, damage which was documented in range management studies of the early 20th century (Sampson, 1909). Sheep populations were subsequently reduced so that although there is still some sheep grazing in parts of the Eagle Cap Wilderness, there is none in the study area. Evidence of this previous use is still visible throughout the area, however.

Recently the types of human impact have shifted dramatically. Burning, which had been practised by the Indians and the sheepherders, has been discontinued and in recent decades both natural and human-caused fires have been effectively suppressed before they can burn a sizeable area. There has also been a dramatic increase in use by summer visitors. Between 1950 and 1977, visitor use in the wilderness area increased threefold to a 1977 use level of approximately 83,000 visitor days (a visitor day is a stay of 12 h by an individual; Cole, 1977*a*; Werner, 1978). Beyond the general exclusion of motorised equipment common to all wilderness areas, the only regulation on trail and campsite use is a prohibition on camping within 70 m of lakes. Essentially all existing campsites were built by recreational visitors. Most of the trails were built by early sheepherders, firemen, and recreationists, but many of these have been reconstructed by the Forest Service in recent years (USDA Forest Service, 1974). About 22 % of these visitors bring packstock, which have replaced sheep as the most abundant non-native grazers in the study area. Several trails and lake basins have been closed to stock use or overnight stays with stock and grazing is generally prohibited within 70 m of lakeshores.

The use and construction of trails

Vegetative responses to the construction and use of trails in Eagle Cap Wilderness are revealed in measures of cover loss and floristic change along trails. Measurements taken along trails in different vegetation types were previously reported in more detail in Cole (1978). The measurement of cover reduction used was:

$$CR = \frac{(C_2 - C_1) \cdot 100}{C_2}$$

where C_2 is the percent cover in ten, $0{\cdot}5 \times 1$ m quadrats located 10 m from the trail and C_1 is the percent cover in similar quadrats located immediately adjacent to the trail. Floristic change was evaluated with a measure of floristic dissimilarity between trailside and undisturbed undergrowth,

$$FD = 0{\cdot}5 \sum |P_1 - P_2|$$

where P_1 and P_2 are the importance values of the same species in the trailside and undisturbed vegetation sample.

Depending upon the vegetation type, trailsides had lost between 12 and 73 % of the cover present in the undisturbed vegetation (Table 1). This loss of cover was most extreme (53–73 %) in the four most densely forested vegetation types. These forests have undergrowth dominated by shrubs or lush, erect forbs (e.g. *Physocarpus* *malvaceous*, *Vaccinium scoparium*, and *Thalictrum occidentale*). Low light intensities at the forest floor encourage the development of morphological characteristics which are poorly suited to the survival of trampling. In comparison with plants which grow in the open, plants growing in shade tend to have greater leaf areas, thinner cuticles, cell walls, and stems (Daubenmire, 1974), and more supportive and conductive tissue (Treshow, 1970). These growth-forms are highly susceptible to breakage and are quickly eliminated along trails.

The four open forest and meadow vegetation types had lost less than 38% of their cover. The undergrowth dominants in these types are usually graminoids, many of which either are rhizomatous (e.g. *Calamagrostis rubescens*) or exhibit a tussock growth-form (e.g. *Stipa occidentalis*). These plants have protected meristematic tissues, as well as tough and flexible vegetative parts. Although they decrease in cover along trails, they survive trampling more frequently than the undergrowth dominants in forests.

TABLE 1

MEAN COVER REDUCTION AND FLORISTIC DISSIMILARITY VALUES ALONG TRAILS IN COMMON VEGETATION TYPES OF EAGLE CAP WILDERNESS

Vegetation type	*CR*(%)[a]	*FD*(%)[b]
Pseudotsuga menziesii/*Physocarpus malvaceous* forest	73	82
Picea engelmannii/*Thalictrum occidentale* forest	64	64
Abies lasiocarpa/*Vaccinium scoparium* forest	57	67
Pinus contorta/*Vaccinium scoparium* forest	53	41
Pinus albicaulis/*Vaccinium scoparium* open forest	22	62
Pseudotsuga menziesii/*Calamagrostis rubescens* open forest	37	36
Stipa occidentalis grassland	38	44
Carex-forb subalpine meadow	12	37

[a] Cover reduction (CR) is the decrease in the vegetative cover of quadrats along trails as a percentage of the cover of undisturbed quadrats.
[b] Floristic dissimilarity (FD) is a measure of the floristic difference between trailside and undisturbed quadrats.

Floristic composition has also changed along trails in the Eagle Cap Wilderness. The dissimilarity between trailside and undisturbed vegetation varied with vegetation type, from 36% to 82% (Table 1). Again, alteration is greater in the denser forests. This reflects both a more complete elimination of the native dominants along trails and a greater increase in light intensity following trail construction. Increased light levels along the trail encourage the invasion of species that cannot survive in the shade of the undisturbed forest (Dale & Weaver, 1974). The light regime of open forests and meadows, in contrast, is not significantly altered by trail construction.

The loss of vegetative cover and change in floristic composition in response to trail use and construction occurs in a zone usually less than 3–4 m wide (Dale & Weaver, 1974; Cole, 1978). Both of these types of vegetative change are more pronounced in dense forests, presumably because shade-tolerant undergrowth is more susceptible to elimination by trampling and because the microclimate of these forests is more drastically altered by trail construction. Invasion by alien species is most pronounced along lower elevation trails.

This suggests that, in the study area, vegetative alteration following trail construction can be minimised by routing trails through more open vegetation types, especially where these types have a dense graminoid groundcover. Forests with undergrowth dominated by brittle shrubs and lush, erect forbs should be avoided because these plants are particularly vulnerable. Where trails must be routed through dense forests, there should be as little brush removal and tree cutting as possible. This will confine visitors to the bare trail tread, reducing the lateral extent of alteration, and will also minimise increases in light intensity.

These suggestions may or may not be useful in other locations, although studies elsewhere imply similar conclusions (Foin, 1977; Weaver & Dale, 1978). Even when applying these results to the study area, one must be aware that some of the conclusions are based on percentage of cover loss rather than absolute cover loss. Therefore the visual impact of trails may be greater in open areas where the original cover was dense. Moreover, managers must make certain that action to minimise vegetation alteration, the primary concern of this paper, does not increase the alteration of some other ecosystem component. Along trails, vegetative alteration may be less significant than some other impact. In many subalpine areas, for example, trail erosion is much more severe in meadows than in forests, despite the lesser amount of vegetation alteration in meadows. In these cases the manager must decide which ecological change is most undesirable and locate trails in the areas least susceptible to that particular change.

The development and use of campsites

Undergrowth in campsites located in different vegetation types was evaluated by comparing the vegetation of heavily used parts of campsites with adjacent less disturbed vegetation. Soil compaction, measured with a soil penetrometer, was used to estimate use (Ward & Berg, 1973). In each campsite, a heavily impacted zone was identified where the unconfined compressible strength of the soil was greater than $3{\cdot}5\ kg/cm^2$. Although variable, this zone was typically circular and had a radius of 5 to 10 m. The vegetation of this zone was compared with the vegetation of adjacent areas in which soil compaction values were less than $2{\cdot}0\ kg/cm^2$. Measures of cover reduction and floristic dissimilarity in campgrounds are presented in Table 2.

In the study area, campsites had lost 40–96 % of their original cover, considerably more than is lost at the edge of trails. This reflects the heavier trampling that occurs in campsites. Again the forests suffer a more complete loss of groundcover than the non-forested vegetation types. Forested camps lost 90–96 % of their cover, leaving only 2–5 %. Non-forested camps lost 40–64 % of their cover, with 25–45 % vegetated even in the intensively used parts of these campsites.

Change in floristic composition was also more pronounced in campsites than along trails, but the differences between forested and non-forested sites were not as

TABLE 2

MEAN COVER REDUCTION AND FLORISTIC DISSIMILARITY VALUES IN CAMPGROUNDS IN COMMON VEGETATION TYPES OF EAGLE CAP WILDERNESS

Vegetation type	*CR*(%)[a]	*FD*(%)[b]
Picea engelmannii/*Thalictrum occidentale* forest	96	72
Abies lasiocarpa/*Vaccinium scoparium* forest	90	82
Pinus albicaulis/*Vaccinium scoparium* open forest	94	77
Stipa occidentalis grassland	64	89
Carex nigricans alpine meadow	56	50
Phyllodoce empetriformis heath	43	59
Carex-forb subalpine meadow	40	60

[a] Cover reduction (CR) is the decrease in the vegetative cover of quadrats in campgrounds as a percentage of the cover of undisturbed quadrats.
[b] Floristic dissimilarity (FD) is a measure of the floristic difference between campsite and undisturbed quadrats.

distinct as along trails. Comparison of campsite and control showed high floristic dissimilarity values (72–82 %) in all forested vegetation types, although the highest value (89 %) occurred in the *Stipa occidentalis* grasslands. The other non-forested types had lower values (50–60 %).

As with trails, forest floor vegetation is generally more highly altered by camping than is the vegetation of open sites. Use of forested campsites removes most of the dominant shrubs and erect forbs and creates a compacted surface, which is invaded by a few trampling-resistant species (e.g. *Taraxacum officinale*, *Carex rossii*, and *Juncus parryi*). Less than 5 % of the surface remains vegetated, however. Soil erosion is encouraged by the bare, compacted, mineral soil, and tree roots are often exposed (Fig. 2), weakening the overstory which, along with the lack of appreciable tree regeneration, may forecast the eventual deforestation of these sites.

Fig. 2. Exposure of tree roots is a frequently observed consequence of erosion on campsites.

A significant loss of groundcover also occurs in non-forested campsites. Many existing plants are eliminated in camps, but some survive and are joined by trampling-resistant invaders. This results in pronounced shifts in floristic composition, but permits the maintenance of a moderate vegetative cover. Consequently, soil erosion should be less severe and the site should not deteriorate as irreversibly as a forested campsite.

These results suggest that vegetative alteration might be reduced by confining campsites to non-forested areas. Similar conclusions have been reported elsewhere (Landals & Scotter, 1973; Leney, 1974; Lesko & Robson, 1975; Kellomaki & Saastamoinen, 1975). Unfortunately, most visitors prefer camping in or near to forests, where there is a plentiful supply of firewood and a sparse, easily eliminated groundcover. Even if campsites were established in open areas, campers would still trample neighbouring forested areas in search of firewood. Therefore, camping will probably continue to have a pronounced impact on forested areas.

In many areas, particularly where the location of campsites is not regulated, restrictions on the type of use that occurs may be most successful in reducing impact on vegetation. The prohibition of wood fires would help conserve nutritional resources in the immediate vicinity of the campsite and eliminate the trampling that occurs during the search for firewood. This action has already been taken by some heavily used wilderness areas, particularly in areas close to the timberline.

Limiting party size could reduce the areal extent of impact, because small groups would confine most of their impact to a relatively small camping area. In heavily-used areas this could lead to the creation of additional campsites, however; so the consequences of such an action should be very carefully considered. The elimination of packstock from camping areas would reduce vegetative damage, which is often severe in camping areas frequented by packstock. In a Montana wilderness area, for example, Frissell (1973) found that horse camps were 10 times as large and had 7 times as much bare ground as hiker camps.

Grazing by packstock

In Eagle Cap Wilderness, two types of meadow are frequently grazed by recreational packstock, the montane valley-bottom meadows and moist subalpine meadows (Cole, 1977*a*). The montane valley-bottom meadows are found below 2100 m on gently sloping aggraded, valley-bottom sites. Studies of range trend in eastern Oregon suggest that, in their pristine condition, these meadows were dominated by *Deschampsia caespitosa*. Due to the history of overgrazing, however, this species has been largely eliminated, bare ground has increased, and perennial grasses have been replaced successively by grass-forb, perennial forb, and annual forb communities (Reid & Pickford, 1946).

The subalpine meadows are found between 2100 and 2400 m, on flat areas which are perennially moist. They are particularly common around lakeshores and on the sites of former lakes. *Carex scopulorum* and *Deschampsia caespitosa* appear to have been the most common dominants under pristine conditions (Cole, 1977*a*), but with grazing these species tend to be replaced by less palatable forbs.

The impact of recreational packstock on these meadows is superimposed upon the previous more drastic effects of sheep grazing. Consequently the following discussion is not a comparison of grazed and ungrazed areas but areas with and without current packstock grazing; all areas experienced heavy sheep grazing in the past.

Seven meadows were each sampled with ten 1 × 1 m quadrats, in which the cover of each species was estimated. Of the three montane valley-bottom meadows sampled, two are at major trail junctions and are currently grazed heavily; the other is essentially ungrazed by packstock. Of the four moist subalpine meadows sampled, two are located adjacent to heavily used lakes and two are on similar sites away from heavily used areas.

The unused montane valley-bottom meadow (Table 3, Site 1) has a complete vegetative cover (104 %) which is composed primarily of graminoids. The ratio of

TABLE 3

CHARACTERISTICS OF LIGHTLY AND HEAVILY GRAZED MEADOWS OF THE MONTANE VALLEY-BOTTOM AND MOIST SUBALPINE TYPES

Meadow-type	*Montane valley-bottom*			*Moist subalpine*			
Site	*1*[a]	*2*[b]	*3*[b]	*4*[a]	*5*[a]	*6*[b]	*7*[b]
Total cover (%)	104	73	63	105	103	101	102
Graminoid cover (%)	81	30	19	90	80	35	40
Forb cover (%)	23	43	44	15	23	66	62
Forb cover/graminoid cover	0·28	1·43	2·32	0·17	0·29	1·89	1·55
Deschampsia caespitosa cover (%)	23	—	—	8	40	5	4
Carex scopulorum cover (%)	—	—	—	70	4	5	8
Annual species cover (%)	—	6	2	—	—	—	—
Alien species cover (%)	2	1	20	—	—	—	—

[a] Lightly used by recreational packstock.

[b] Heavily used by recreational packstock.

forb to graminoid cover is only 0·28. *Calamagrostis canadensis* is the most abundant grass, but *Deschampsia caespitosa* provides 22 % of the total cover. The current condition of this meadow, then, is quite similar to inferred pristine conditions, although the presence of alien species and the reduced cover of *Deschampsia caespitosa* provide evidence of some disturbance.

The vegetation of the heavily used montane meadows (Sites 2 and 3) has been noticeably depleted; total cover is only 73 and 63 %, and most of the remaining plants are unpalatable forbs (e.g., *Penstemon globosus* and *Potentilla gracilis*), so that forb to graminoid cover ratios are 1·43 and 2·32. *Deschampsia caespitosa* provides only a trace of cover and the most abundant graminoids are *Carex hoodii* and *Poa pratensis*. Furthermore, on Site 2, 20 % of the species are annuals and on Site 3, alien species constitute 32 % of the total cover.

The lightly used moist subalpine meadows (Sites 4 and 5) have a dense vegetative cover (105 % and 103 %) and are dominated by graminoids; forb-to-graminoid cover ratios are only 0·17 and 0·29. Furthermore, the species assumed to be the dominants under pristine conditions, *Carex scopulorum and Deschampsia caespitosa*, are still the most abundant species, providing 40–75 % of the total cover. This suggests that either early sheep grazing did not cause as much floristic change in these meadows as it did in the montane valley-bottom meadows or that recovery after the removal of sheep has been more rapid.

The heavily used moist subalpine meadows (Sites 6 and 7) have a cover similar to that of the lightly grazed meadows, but they are floristically different. *Carex scopulorum* and *Deschampsia caespitosa* contribute only 10–12 % of the total cover, a reduction attributed to grazing, since both species are highly palatable to livestock (USDA Forest Service, 1937; Hermann, 1970). Forbs, particularly *Potentilla flabellifolia*, currently dominate these meadows. Forb-to-graminoid cover ratios are 1·55 and 1·89. There has been no invasion of annual or alien species, however.

Where grazing by recreational packstock continues, meadows are being maintained in the degraded state inherited from the era of sheep grazing. Sedges and grasses, the original dominants, have been selectively removed so that graminoids are now subordinate to forbs. In the montane valley-bottom meadows, grazing has reduced vegetative cover, creating large bare areas. These lower elevation meadows have also experienced a more complete elimination of the originally dominant species and a greater increase in cover by alien and annual species. This suggests that the montane meadows have been more highly altered by grazing than the moist subalpine meadows, a situation also noted in the Sierra Nevada of California (DeBenedetti & Parsons, 1979).

The easiest way to reduce this alteration is to eliminate the use of packstock entirely. This would also reduce the trail and campsite deterioration problems mentioned earlier. In most wilderness areas of the western United States, however, many visitors consider packstock a traditional and accepted element of the wilderness experience. Consequently, restrictions on packstock use are usually confined to severely overgrazed areas and places with insufficient forage to support stock.

In areas that experience heavy packstock use, impact on meadows can be reduced by requiring visitors to bring in their own feed, by carefully regulating means of confining stock, and by limiting the amount of packstock use. Use of pelletised feed or weed-free hay reduces direct grazing and trampling effects and removes the selective advantage of unpalatability of some species, allowing meadows to return to a more natural composition. Prohibiting particularly destructive means of restraining stock, such as tying them to trees, or confining them to a small area, eliminates the most severe impacts of trampling. Limitations on party size and amount of use reduce both grazing and trampling damage. Some areas have had such restrictions for 15 years (USDI, National Park Service, 1963) and other areas are becoming increasingly restrictive each year.

The impact of fire suppression

As is the case in most other forested areas in the western United States, the frequency with which forests in the study area burn has declined during the recent period of fire suppression. Effects of this change vary with vegetation type, because some types burned more frequently or more intensely than others. Although no fire history data were collected in this study, enough information exists on fire regimes in similar stands in the northern Rocky Mountains to relate successional trends in the study area to changes resulting from fire suppression documented elsewhere.

Prior to recent suppression, open forests of *Pinus ponderosa*, *Pseudotsuga menziesii*, and *Larix occidentalis* experienced frequent surface fires fueled by the abundant grasses and tree regeneration of the forest floor. These fires, which usually occurred at intervals of 5 to 20 years (Weaver, 1974; Arno, 1976), consumed seedlings and the groundcover, but usually left the mature trees unharmed. This maintained open, uneven-aged stands of relatively shade-intolerant species.

In the densely stocked *Pinus contorta* stands, fire was less frequent but usually more intense. In the northern Rocky Mountains, fires in these stands occurred at average intervals of 17–50 years, with high-intensity, stand-replacing fires occurring where fire-free intervals were long and lower intensity surface fires occurring at more frequent intervals (Arno, 1979). In Eagle Cap, most *Pinus contorta* stands tend to have an even-aged, monospecific overstory, suggesting nearly exclusive recolonisation by *Pinus* following intense crown fires.

The dense forests dominated by *Abies grandis*, *Abies lasiocarpa*, *and Picea engelmannii* are not highly flammable and are often not susceptible to burning until large quantities of litter have accumulated and the old trees have been decimated by insect or fungal attack. In the northern Rockies, fires typically occurred in these stands every 70 to 150 years or more, and were usually intense, stand-replacing fires of varying size (Arno, 1979). At lower elevations, large *Pseudotsuga menziesii* and *Larix occidentalis* trees occasionally survive these fires and provide an immediate seed source. *Pinus contorta* is also an abundant early seeder after fire. At higher elevations, however, *Abies* and *Picea* recolonise these stands directly (Habeck & Mutch, 1973). This process maintains a mosaic of structurally diverse stands, which contain both long-lived, shade-intolerant species and abundantly reproducing, shade-tolerant species.

Pinus albicaulis and *Pinus flexilis* stands occur close to timberline or on very rocky sites, where fuel levels are low. In these stands, fire is usually infrequent and limited in area. Consequently, the role of fire in stand maintenance is relatively unimportant.

Successional trends were identified by analysing the population structures of the most prominent tree species. Where floristic composition is stable, the number of individuals of each species should decrease exponentially from the smallest to the largest size class. If there are fewer individuals than expected in the smaller size classes, the species is declining in importance. If there are more individuals than expected in the smaller size classes, and they indicate an ability to grow to maturity, then the species will probably become increasingly abundant in the future (Daubenmire, 1968). This type of analysis, augmented by considerations of the autecology, particularly the shade tolerance of each species, reveals successional changes following recovery from disturbance. In the areas sampled, this disturbance was presumably fire, because sites prone to other types of disturbance (e.g. floods or landslides) were avoided.

Five vegetation types, each having distinctive physiognomies are discussed in detail below:

(1) *Pseudotsuga menziesii/Calamagrostis rubescens* open forests, which have a luxuriant grass cover.
(2) Dense *Picea engelmannii/Thalictrum occidentale* forests, which have a dense forb undergrowth.
(3) Subalpine *Abies lasiocarpa/Vaccinium scoparium* forests, which have a sparse understory of low shrubs.
(4) Densely stocked *Pinus contorta/Vaccinium scoparium* forests.
(5) Open, timberline *Pinus albicaulis/Vaccinium scoparium* stands (Cole, 1977a).

In each vegetation type, at least five stands were sampled. Each sample consisted of a 10 × 20 m quadrat, within which the species and diameter at breast height (1·4 m) were recorded for each tree. The diameters were grouped into 9 diameter classes: <2 cm; 2–10 cm; 11–20 cm; 21–30 cm; 31–40 cm; 41–50 cm; 51–60 cm; 61–70 cm and >70 cm (Fig. 3).

In the *Pseudotsuga menziesii/Calamagrostis rubescens* open forests, *Pinus ponderosa* and *Larix occidentalis* have not been reproducing in recent years (Fig. 3a). Moreover, the reproductive success of *Pseudotsuga menziesii* has declined in 5 of the 8 sampled stands. Lack of fire inhibits the establishment of these species by allowing increases in litter accumulation and understory density and decreases in light intensity at the forest floor (Fowells, 1965). *Abies lasiocarpa*, which can reproduce under these conditions, has experienced a recent acceleration in its rate of reproductive success. This suggests that a prolonged period without fires would allow *Abies lasiocarpa* to dominate many of these stands. *Pseudotsuga menziesii* would probably remain as a long-lived seral species, but *Pinus ponderosa* and *Larix occidentalis* would be eliminated.

The dense *Picea engelmannii/Thalictrum occidentale* forests burned infrequently, so that 40 years of effective fire suppression has had less impact. Seedling establishment by both *Larix occidentalis* and *Pseudotsuga menziesii* has declined sharply in recent years, however (Fig. 3b). *Picea engelmannii*, which is the most abundant large tree in these stands, also shows a decline in normal seedling

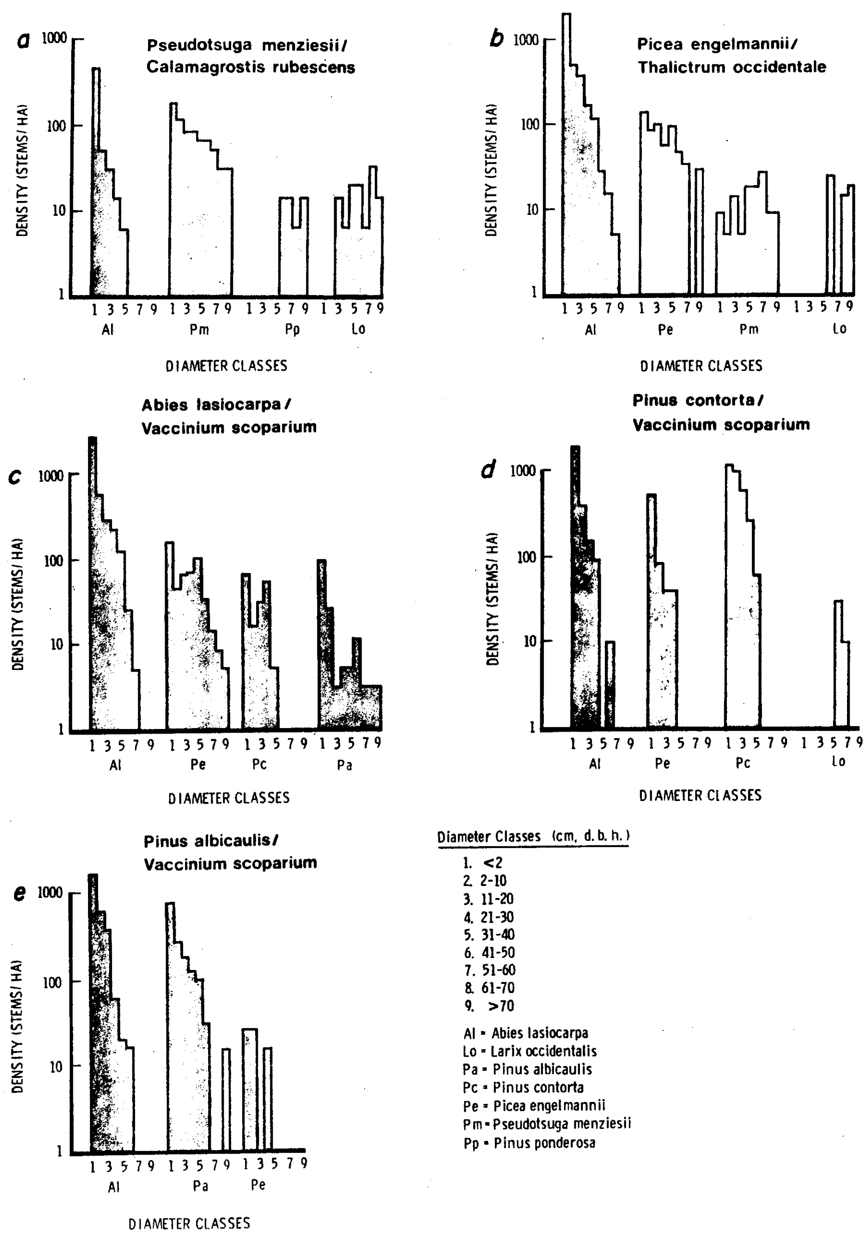

Fig. 3. Diameter class distributions for common tree species in five vegetation types.

increment. Again *Abies lasiocarpa* appears likely to become the most abundant species in the overstory. This shift from *Picea* to *Abies* apparently results from *Abies*' higher rate of survival on organic seedbeds and at low light intensities, conditions which become increasingly limiting to other species with time since the last fire (LeBarron & Jemison, 1953; Fowells, 1965).

Changes in the subalpine *Abies lasiocarpa/Vaccinium scoparium* forests are even more subtle. *Abies lasiocarpa* exhibits a stable population structure, with an approximately exponential increase in the density of successively smaller size classes (Fig. 3c). *Pinus albicaulis* reproduces erratically, but in sufficient quantities to maintain itself. However, the low density of *Picea engelmannii* and *Pinus contorta* in the smaller size classes indicates that the populations of these species have started to decline. These species often germinate in abundance after fire, but with decreasing frequency thereafter. Several centuries of fire postponement could eliminate *Pinus contorta*, which usually lives only about 150 years.

The overstory of *Pinus contorta/Vaccinium scoparium* stands consist of both *Pinus contorta* and *Larix occidentalis* (Fig. 3d). However, *Pinus contorta* is the most abundantly reproducing tree species in only two of the five stands sampled, and *Larix* has not been reproducing for years. In the other three stands, most of the seedlings are *Abies lasiocarpa* or *Picea engelmannii* (Cole, 1977*a*).

In the timberline *Pinus albicaulis/Vaccinium scoparium* stands, there are no apparent responses to fire suppression. *Picea engelmannii* establishment is erratic, but both *Pinus albicaulis* and *Abies lasiocarpa* enjoy consistent reproductive success (Fig. 3e). The discontinuous vegetative cover discourages frequent or large fires. When a stand does burn, recovery takes a long time, but the stand is recolonised from seed sources in neighbouring stands and there are no pronounced floristic shifts.

Each of these forest types varies in its response to fire suppression. Floristic changes are most pronounced in those types which were previously kept open by frequent fires (e.g. *Pseudotsuga menziesii/Calamagrostis rubescens* open forest). Types that burned less frequently (e.g. *Abies lasiocarpa/Vaccinium scoparium* forest) show more subtle changes, which would only become obvious after centuries of effective fire suppression. Types that burned infrequently and only in localised areas (e.g. timberline *Pinus albicaulis/Vaccinium scoparium* forests) appear to have been essentially unaffected by fire suppression.

The species most affected are also those adapted to frequent fires. *Pinus ponderosa* and *Larix occidentalis* survive frequent fires because they have thick bark, but they seldom germinate in the deep litter and heavy shade of unburned stands and no longer establish seedlings. *Pseudotsuga menziesii*, *Pinus contorta*, and *Picea engelmannii* are found in areas burned less frequently. They are still establishing seedlings, but in less than 50 % of the areas where they are prominent members of the overstory (Cole, 1977*a*).

The most prolific reproducer in recent years has been *Abies lasiocarpa*. Many stands dominated by other mature tree species support a nearly pure understory of young *Abies lasiocarpa*. Mature *Abies lasiocarpa* is at present dominant only in moderately high elevation forests, on moist, protected sites, but it is the most abundant seedling and sapling over approximately 70% of the forested area.

Maps showing where each tree species is the most abundant mature tree and where it is the most abundant seedling have been presented in Cole (1977*a*). A map of the areas in which currently dominant tree species are no longer the most abundant species in the smaller size classes was produced by superimposing these two maps. This map (Fig. 4) identifies the areas where floristic response to fire suppression is most pronounced.

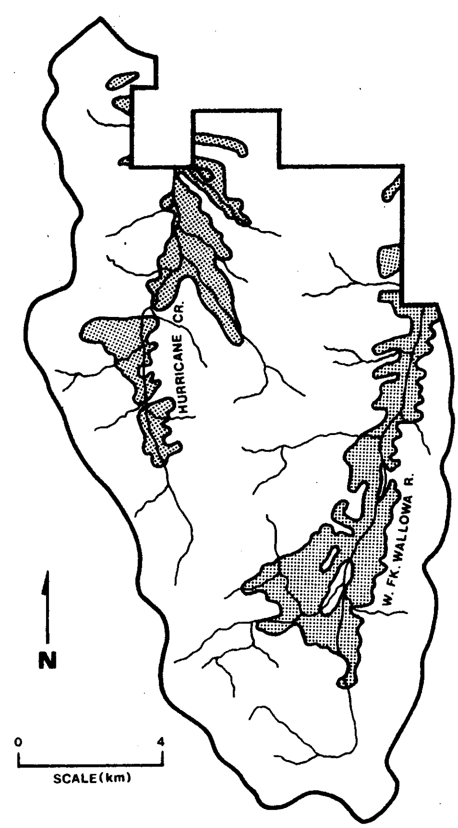

Fig. 4. Areas of rapid successional change resulting from fire suppression. These are forested areas where the seedlings and saplings (< 10 cm diameter-at-breast-height) are predominantly different species from those that dominate the overstory.

Most of this change is occurring in the valley bottoms and on lower slopes, below 2100 m. Inferences from other areas suggest that such stands burned frequently, keeping the forest open and preventing *Abies lasiocarpa* from achieving dominance. Forty years of fire suppression have allowed stand density to increase and *Abies lasiocarpa* to become increasingly prominent. If this trend continues, *Pseudotsuga menziesii*, *Larix occidentalis*, *Pinus contorta*, and *Picea engelmannii* forests will become increasingly rare.

Along with this increase in overstory density, one would expect corresponding changes in undergrowth. My observations of understories in stands of different ages (Cole, 1977*a*) suggest that there is a parallel increase in the areal extent of a mesophytic, shade-tolerant undergrowth association comprised principally of *Thalictrum occidentale* and *Arnica cordifolia*. This apparent convergence on a single association of shade-tolerant overstory and undergrowth species represents a drastic loss of floristic diversity.

This loss coincides with an equally dramatic loss of physiognomic diversity. As dense, heavily shaded stands become increasingly prominent, both open stands and the distinctive *Pinus contorta* stands are being areally constricted. Furthermore, as more stands reach a mature, many-aged population structure, young stands and even-aged stands are also becoming increasingly rare.

These changes result from a simplification of the effects of fire. In the past, fire frequency and intensity varied with differences in topography, vegetation, other environmental factors, and chance. This contributed to a floristically and structurally diverse landscape. The recent efficiency in fire suppression has universally decreased fire frequencies and resulted in a homogenisation of these stands. Reversal of these changes will require allowing fire to return to its more varied role. Although this is fraught with many physical and political difficulties, management plans which allow fire to play a more natural role have recently been developed in many wilderness areas (Despain & Sellers, 1977; Parsons, 1977; Vankat, 1977).

CONCLUSIONS AND IMPLICATIONS

Results of this study reveal the extent of vegetative alteration occurring in Eagle Cap Wilderness Area, despite an intention to preserve the 'natural features' of this area. This situation is typical of most high-elevation wilderness areas in the United States. Both structural and floristic characteristics of the vegetation are changing in response to recreational use and the effects of prolonged fire suppression. The areal extent and degree of alteration, however, is not uniform. Certain human activities cause more widespread and intense alteration than others, and certain vegetation types are more susceptible to these changes.

Although it is difficult to compare the impacts of fire suppression and recreational use, the latter causes immediate loss of cover and dramatic shifts in vegetational composition in localised areas; fire suppression results in more general and widespread shifts in stand structure and floristic composition. Recreational use also causes more irreversible damage, such as soil erosion, which is accelerated by the increased soil compaction and decreased vegetative cover of trampled sites.

Cover losses and changes in floristic composition resulting from recreational use are more pronounced on campsites than along trails or in heavily grazed meadows. Campsites also suffer from the effects of campfires, the depletion of wood supplies, the mutilation of standing trees and from particularly severe soil compaction and erosion. This makes recovery extremely slow. In contrast, abandoned trails, and meadows which are no longer grazed, show evidence of more rapid recovery.

Significant effects of recreational use, however, will continue to be confined primarily to a few major routes and destinations. In the study area, trail alteration occurs along 94·7 km of trail, in a zone approximately 4 m wide; this is only 0·3 % of the study area. Similarly, there are about 336 campsites, with an estimated mean area of intense alteration of 706·5 m^2 (r = 15 m). The areal extent of campground alteration, then, is 0·2 % of the study area. The area frequently utilised for grazing of recreational stock is about 1·3 % of the study area.

Fire suppression, on the other hand, has resulted in significant vegetative change throughout most of the study area. The effects of 40–50 years of suppression have been most pronounced in lower elevation forests, but prolonged fire suppression will eventually affect essentially all vegetation types in the Eagle Cap Wilderness.

In terms of providing a 'quality wilderness experience' for visitors, reducing recreational impacts could be considered more important than changes due to fire suppression, because the former are more obvious and are concentrated in the areas where visitors spend most of their time. In terms of maintaining 'natural conditions' in wilderness, however, the first priority should be a change in the current policy of fire suppression. If fire could be restored to its natural role, then significant alteration could eventually be confined to less than 2 % of the total area. This is a difficult task, however, because of problems with fuel accumulation, and the possibility of burning adjacent non-wilderness lands. Adverse public reaction has been a surprisingly minor problem where natural fires have been allowed to burn (Despain & Sellers, 1977).

Where wilderness managers have experimented with 'natural fire' policies, natural fires often continue to be suppressed in the lower elevation vegetation types (Chapman, 1977; Vankat, 1977). The results presented above and elsewhere (Habeck, 1972; Habeck & Mutch, 1973) suggest that the most dramatic responses to fire suppression occur in these lower elevation forests, particularly in the open forests. Therefore, immediate attention should be given to restoring fire successfully to its natural role in lower elevation montane forests.

In many wilderness areas, management responses to vegetation deterioration along trails have primarily involved rerouteing trails away from meadows and into forests (Snyder, 1966). Similarly, visitors are often advised to camp in forests and not in meadows. In both cases, trails and camps are more aesthetically displeasing in meadows, where bare ground and trampled vegetation are more visible. However, the results presented in this study indicate that percentage of cover lost and floristic change are more pronounced in forests. Managers must decide which is more important, aesthetics, vegetation change, or some other type of human impact not yet documented (e.g. erosional susceptibility or impacts on wildlife) and locate facilities accordingly.

In order to do this, more research on ecological change in wilderness areas is obviously needed. In terms of maintaining natural vegetation, current management responses have not always been directed toward the most significant problems or the most susceptible vegetation types. In the case of eliminating fire suppressional effects, this results from the difficulty of the task. In the case of recreational impacts, this appears to result from the common assumption that ecological change is most severe where human impact is most visually obvious. This reliance on intuitive assessments has nothing to do with lack of competence or good judgement; it merely reflects the lack of basic information on environmental responses to human activities. Even the changes documented in this study, primarily losses in cover and shifts in floristic composition, are superficial changes which may not be significant in the long run. Only with a major research effort, detailing human impact on all facets of wilderness ecosystems, can managers even approach truly preserving the remaining wildlands of the United States.

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