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GRASS SEEDING FOR WILDFIRE REHABILITATION:

SCIENCE AND POLICY

By

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B.S., Bates College, 1983

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Abstract

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This professional paper reviews both the technical and policy aspects of the Forest Service practice of aerially seeding grass to rehabilitate areas burned by forest fires. Assessment of the state of knowledge covers four main questions: Does seeded grass establish and grow? Does grass seeding reduce erosion? What impacts does grass seeding have on native flora and successional patterns? How might grass seeding affect site productivity? Recent literature suggests that the effectiveness of grass seeding is still in doubt, and that potential adverse consequences on natural processes are poorly understood. As a case study, postfire monitoring on the 1988 Canyon Creek Fire shows that even with a good growing season, establishment of seeded grass is highly variable across a wide range of site conditions. Seeded grasses produced the most cover on moist aspect sites, at lower elevations, and on the gentler slopes. An evaluation of the monitoring methods employed suggests many refinements: notably the need for paired treatment and control plots, and the need to integrate research into operational monitoring of land management practices. A critical review of the implementation and administration of emergency burn rehabilitation traces its fiscal and statutory foundations and current administrative framework. The main flaw in the policy is that it fails to provide for evaluation of its effectiveness in meeting its objectives. Recommendations include revision of the Forest Service rehabilitation Handbook to improve cost-effectiveness analyses, and establishment of coordination, technical assistance, and funding of operational monitoring studies across the Region.
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Introduction

Many of the large 1988 wildfires in the Forest Service's Northern Region were seeded with grasses for emergency watershed protection. The objective of emergency rehabilitation is "to help stabilize soil, control water, sediment, and debris movement" (USDA, 1989: FSM 2523.03). Aerial seeding is often the preferred technique for rapidly replacing burned vegetation and litter cover so that the soil surface is protected from erosion. Less erosion means that the productive topsoil stays on the slopes, and less sediment is delivered to stream channels. This in turn reduces the magnitude of sediment damages to lives and property downstream should flooding occur. The practice of aerial seeding has evolved continually over the last 50 years, but the pace has not kept up with many currently accepted concepts of fire ecology. Even so, seeding can be a valuable tool to reach land management objectives. In the Northern Region, little information is available to assist in making effective rehabilitation seeding prescriptions, and current administrative procedures supply little incentive to collect such information. Standards of review applied to more routine land management practices need to be applied to emergency burn rehabilitation seeding to ensure the efficiency of the technique in meeting its ultimate objective of land stewardship. In particular, systematic monitoring and evaluation procedures need to be implemented.
In the summer of 1989, one of my assignments for the Forest Service in monitoring the vegetative recovery of the Canyon Creek fire of 1988 was to assess "seeding effectiveness". At the time, effectiveness was not clearly defined. The spring and summer of 1989 were considered locally to be very good growing seasons with soil moisture maintained by almost weekly rains. These conditions should have fostered excellent establishment and growth of grasses seeded in the burned area, yet substantial variability in grass growth was observed. Later, in reviewing the literature on the subject, it became clear that post hoc definitions of effectiveness were common in the few evaluations of seeding that had been made. Seeding prescriptions, based on professional judgement, have been made in very broad terms, with only general expressions of the effect desired. Seeding was deemed effective if grass alone, or grass plus native plants, achieved an unspecified level of cover guessed to be adequate to reduce erosion. Despite a poor level of understanding of the variables influencing seeding success, few research projects or results from monitoring of practical applications have been reported. Furthermore, several recent papers witness a growing controversy over both the costs of ineffective seeding treatments and the potential adverse ecological effects of seeding. While there is substantial guidance in Forest Service policy regarding monitoring and evaluation of other vegetation management practices, burn rehabilitation has escaped systematic review.

The objective of this paper is to review the state of technical knowledge and agency policy of aerial seeding for rehabilitation of burned watersheds in the Northern Region. Insight gained from a case
study will be used to make further recommendations for changes in program implementation and administration to address questions of ecology and economy. The first section of the paper reviews the technical literature on the physical and biological processes of fire recovery and the influences of grass seeding. Results and critiques of seeding studies in other areas raise relevant issues that should be addressed. In the second section the emergency rehabilitation seeding of the Canyon Creek fire and monitoring of revegetation in the first postfire season is presented as a case study. Trends suggested by the Canyon Creek data illustrate several specific research needs, and possible improvements in experimental design and methodology for other evaluation studies. In a concluding section, an analysis of the implementation and administration of emergency burn rehabilitation will be used to identify some obstacles that hinder adequate information feedback.
Chapter 1.
The practice of Emergency Burn Rehabilitation seeding and its relationships with vegetative succession and site productivity.

Rehabilitation in practice.

The practice of supplemental seeding to aid vegetative recovery after wildfires apparently originated in the watersheds around the expanding suburbs of Los Angeles in the 1920's, when chaparral fires were followed by damaging floods and debris flows (Barro and Conard, 1987). Experimentation with seeding native shrubs, then with introduced forbs and grasses continued through the 1940's when Annual or Italian ryegrass (*Lolium multiflorum*) emerged as the species of choice (Ibid.). Annual ryegrass, a native of temperate areas in Europe and Asia, has several characteristics desired of species used for rehabilitation: it germinates and establishes rapidly, has a fibrous root system and a bunchgrass form, is short-lived where winters are cold, and seed is readily available and relatively inexpensive (Hafenrichter et al., 1968; Beard, 1973). Other species are selected for their particular temperature and moisture requirements, annual or perennial habit, persistence of litter, forage value, or competitive nature. Legumes are often added to a seed mix to provide nitrogen, and tree seeds may be included if no local seed source survived the fire. A recent review of the use of Annual ryegrass in burned area revegetation in California concluded that neither the efficacy of this practice in reducing erosion nor the long-term effects of introduced grasses on
chaparral ecosystems are known (Barro and Conard, 1987). Fewer studies have been made of other rehabilitation species or seed mixes, and there is very little documentation available on the results of seeding in the northern Rockies. Of the literature available, little is in refereed journals.

Over the years, experience with many fires has led to the acceptance of general procedures for assessment of burned watersheds and prescription of rehabilitation treatments. The discussion here is limited to the standardized procedures directed by the Forest Service Manual and Handbook, and these directives are discussed in greater detail in the last section of this paper. Rehabilitation of areas mechanically disturbed by fire suppression activities, such as dozer lines and camps, is outside the scope of this paper. When a wildfire is brought under control, an interdisciplinary team is assembled to survey watershed conditions. If the team determines that threats to lives, property, water quality or control, or site productivity exist, they may propose rehabilitation measures to abate the hazard. Several criteria are considered in selecting areas that may be stabilized by grass seeding, with the relative importance of each criterion dependent on site-specific factors. The main criteria are high burn intensity, steep slopes and high drainage density, a high percentage of a watershed burned, erosive soil types, areas subject to intense storms, understory vegetation unlikely to survive fire, and high probability or severity of flood and sediment damage to downstream lives and property. Areas where seeding would conflict with sensitive plant populations or designated research or wilderness areas get closer attention, but seeding is not
necessarily prohibited.

Costs, controversy, and questions.

Seeding is often only a fraction of a total emergency watershed rehabilitation bill, but it can be expensive. Costs depend on the price of seed, the cost of aerial application and contract administration, and the size of the fire. Seed of native grasses is generally much more expensive and less available than agricultural cultivars. Seeding Annual ryegrass in California is at the cheap end of the scale, at $2.00 to $7.00 per acre (Barro and Conard, 1987). Seeding various mixtures of grass species on 1988 fires in the Northern Region ranged from $9.50 to $51.42 per acre, with an overall average of $27.56 per acre (Putnam, pers. comm., 1990). In all, 44,913 acres of National Forest System lands in the Region were seeded at a total cost of $1,238,030. Many thousands of acres in other ownerships were seeded similarly, with other sources of funds. These figures are the highest of any year in the region, corresponding with the highest burned acreage in recent years, a high level of public awareness, and concern for cumulative effects of many large, unstable watersheds.

The controversial nature of grass seeding is widely recognized (Gautier, 1983; Barro and Conard, 1987; Ruby, 1989; Taskey et al., 1989; Goudey, 1989; Putnam, pers. comm., 1990). This controversy stems from inconsistent results of seeding projects, disagreements about economic efficiency, differing assumptions about the abilities of native plants to survive fire and maintain postfire site productivity, and differing values regarding human management of wildland landscapes. Conflicting
values are not likely to be resolved with improved factual understanding, but better data in other areas could both reduce conflicts and help make seeding treatments more efficient.

The impacts of seeding in Northern Rockies ecosystems may or may not follow patterns suggested in California, but because similar physical and biological processes are involved, the same questions are relevant. The outstanding questions identified in the chaparral area review are:

1. What conditions or processes constrain the establishment of seeded grasses?
2. Are introduced grasses effective in controlling erosion?

While erosion of topsoil is a critical factor in site productivity, another basic ecological question should be added to this list:

4. How are nutrient cycling and site productivity affected by grass seeding?

The following review of the literature will discuss each of these issues in turn.

1. Conditions and processes constraining establishment of seeded grasses.

While agronomists have studied and refined mechanical seedbed preparation for maximal establishment of grass seedlings, such control over conditions is not available in wildland fire rehabilitation. Grass seeds generally need at least shallow burial to provide moisture
conditions constant enough for germination and establishment of roots. Seedbed conditions in burned forest soils may be extremely variable. Establishment of seedlings is often poor if there is unburned duff remaining on the surface, particularly on droughty south and west aspects (DeByle and Packer, 1981; Cline and Brookes, 1979). The surface is often a layer of fluffy ash until it is disturbed by wind or precipitation. If seed can be broadcast before the crumbly surface is compacted, puddled, or crusted by rainfall, more seeds will land in microsites conducive to establishment. Freezing and thawing mixes the surface layer of the soil and may act to bury seeds, or uproot young seedlings. Mulch can enhance seedbed conditions by moderating soil surface moisture and temperature extremes, and by anchoring seeds and reducing wind erosion of ash and fine soil (Monsen and McArthur, 1984). Late fall or early spring seeding on wet snow can be successful, but many grasses require moderate temperatures for germination and growth. If seedlings are very cold-sensitive, sowing later will reduce frost losses (Ibid.). As both soil and climatic conditions are difficult, if not impossible to predict, seeding rates are generally much higher than would otherwise be necessary.

Various recommendations for seed application rates are available, but none were encountered that related seeding rates to rates of establishment based on seedbed or climatic conditions. Seeding prescriptions are traditionally made on a very broad scale, so that the typical actual application meets the general needs of a large and diverse area. But the average of poor seed establishment on some sites and overabundant establishment on others does not necessarily mean that
overall rehabilitation needs will be met. The Burn Rehabilitation Handbook suggests 20–60 pure live seeds per square foot, but it is neither expected nor desired that plants establish at this density (FSH 2509.13.25.13). Aerial seed application on several 1987 fires on the Siskiyou National Forest in western Oregon controlled erosion, but "stocking density on most areas was higher than needed to provide erosion protection (Gross et al., 1989). On a 1987 fire on the Shasta-Trinity National Forest in Northern California, seed traps and germination monitoring revealed 6 to 42 percent germination success of a mix of species applied at 50 seeds per square foot. Resulting ground cover ranged from 10 to 90 percent (Miles et al., 1989). According to a simple curve relating ground cover density to sediment yield, 10% cover is not effective in reducing erosion, and "ground cover density greater than 50% does not generally provide commensurate reduction in sediment yield" (FSH 2509.13.25.12).

A review of seeding the 1987 Stanislaus Complex fire in central California was designed to address more detailed questions. On the Stanislaus, monitoring of seed application with sticky card seed traps revealed a percentage difference between intended and actual seeding rates of 29% for a brome-fescue mix, 34% for annual ryegrass, and 54% for a perennial mix (Janicki, 1989). Percent survival of these three actual seed applications was 14.5%, 8%, and 5% respectively, with a large variability between plots. Resulting cover provided by the seeded species ranged from 2% to 92%, with means of 16%, 20%, and 5% for the three treatments respectively. Chi squared tests at .05% level indicated that slope was a significant factor in seed establishment, but
elevation, aspect, and plant community were not. In retrospect, Janicki recommends increasing seeding rates on steep slopes to compensate for seed washed off in early storms, or seeding steep slopes with species that have hygroscopic awns that act to bury the seed. Monitoring studies of seed application such as this are critical to understanding the environmental factors controlling seed establishment. Detailed monitoring of seedling establishment by species within seed mixes is also needed.

2. Effects of seeding on postfire erosion.

Under many circumstances, grass seeding can reduce erosion in the early postfire years, and may add to the level of watershed stabilization achieved by native plants. The emergency burn rehabilitation team must make an overall assessment of the locations and causes of hazardous watershed conditions, then determine which are amenable to treatment. They must consider sediment storage in the drainage, and the relative contribution of hillslope runoff and sediment erosion to off-site damages when storms arrive. Water yields increase when forested watersheds burn because less water is lost through canopy interception and evapotranspiration. Unconsolidated alluvium and fine sediments in the stream channel may be mobilized by the higher postfire flows. If drainage densities are high or soil structure is conducive to gully formation, vegetative cover on the slopes may be especially important in reducing delivery of fresh sediment to the channel. Timing and intensity of storm patterns affect the relative importance of infiltration capacity of the soil and potential for overland flow. The
rehabilitation team must also weigh assumptions about the value of maintaining nutrient-rich surface soils on-site. If soils are shallow, little loss is tolerable.

The need for supplemental vegetation is a judgement call the rehabilitation team must make from experience and evidence on site. Rehabilitation is generally not considered necessary where moderate intensity fire leaves a mosaic of areas with litter or duff or if overstory vegetation remains to re-supply the litter layer (FSH 2509.13.23; Connaughton, 1935). If rocks, organic litter, and live plant crowns together account for at least 30% ground cover, emergency revegetation may not be warranted (FSH 2509.13.25). In addition to or instead of seeding, the rehabilitation team may recommend slope stabilization with mulch in particularly sensitive areas, or log erosion barriers, which are snags felled and staked down on the contour to trap sediment and halt rill formation. Miles et al. (1989) provide a discussion of applications, cost, risk, and effectiveness of various techniques.

The best indicator of potential native plant recovery is the nature of heating by the fire, which depends on available fuels and weather conditions during burning. Survival of vegetation depends on both the heat pulse upward into the canopy (intensity), and the heat pulse downward toward buried plant propagules (severity). Intensity is technically the rate of energy output per length of the flame front, estimated by flame length observed or the height of char or scorch left on trees. The term intensity is often used loosely to refer to both intensity and severity, but the two measures do not necessarily
correlate. Where the overstory is completely consumed, severity is the best indicator of understory plant survival. Soil heating depends on the residence time of fire on the site, and is estimated by observations of the degree of consumption of duff, size and depth of char on remaining fuels, and the color of ash on the surface (Ryan and Noste, 1985). If burning conditions would allow understory plant survival, one must then ask what plants were present before the fire. Patterns have been documented that relate understory plant composition and density to certain forest habitat types and stand conditions (Fischer and Clayton, 1983; Kessell and Fischer, 1981; Pfister et al. 1977). Stands with full canopy closure would likely have shaded out many shrubs and herbs, and if litter and duff accumulations were deep, many of buds and rhizomes of understory plants would not have been protected in the mineral soil. Survival mechanisms of understory plants are discussed in more detail below.

Water repellent soils present an unusual rehabilitation problem. This phenomenon is caused by the volatilization of aliphatic hydrocarbons in hot fires, then movement of these substances through a temperature gradient in the pore spaces of the soil, and condensation on soil particles as much as several inches deep (DeBano, 1981). Precipitation saturates the surface layer but cannot wet the hydrophobic layer beneath. The drastic reduction in infiltration capacity can cause excess precipitation to run off over the surface, carrying away topsoil. This hydrophobic layer is usually patchy, but may cover extensive areas where fuels were similar, and may persist for years (Dyreness, 1976). Hydrophobic layers may be broken up with chemical wetting agents or
mechanical mixing of the soil profile, or avoided by prescribed burning at lower temperatures (DeBano, 1981). Wetting agents may not be effective in field applications (Dyner, 1976) and may suppress certain forbs (DeBano and Conrad, 1974). Hydrophobic soils inhibit seed establishment by erosion of both soil and seed downslope, and by interrupting capillary rise of moisture from the subsoil. DeBano recommends further research to address the differences between native and introduced species in revegetation. Seeds of native plants are resistant to erosion as they tend to be buried deeper in the soil. Competition with introduced grasses early in the growing season could reduce the ability of the deeper rooted and longer lived natives to locate discontinuities in the hydrophobic layer and grow to maturity (Ibid.).

Wildfires in known landslide prone areas require special attention in rehabilitation. Most of this attention is focused on replacing large organic debris consumed in the fire to stabilize channels and scarp headwalls, and other channel treatments (Smith and Wright, 1989). On oversteepened slopes, tree roots may be an important component of soil shear strength, and as they decay after fire, landslide hazards may increase. Mass failures can cause greater concern than surface erosion after fire because they can deliver greater total volumes of sediment into streams. Debris flows and other high peak flows can also cause long-term destabilization of stream channels. Seeding hillslopes in these areas may reduce overland flow and sediment delivery, but it must be weighed against the need to re-establish deep rooted trees and shrubs which may be competitively excluded by grass.
Soil erosion depends on rainfall intensity and seasonal distribution, soil infiltration capacity and structural stability, slope length and gradient, and organic cover (Brady, 1974). Live vegetation, litter, duff, and humus all absorb the kinetic energy of rainfall, increase infiltration of water into the mineral soil, and obstruct the overland flow of water and sediment (Wright and Bailey, 1982). Models relating these factors to sediment delivery have been developed primarily for agricultural situations. Site conditions in forested, mountainous terrain are extremely variable, so local data should be used to calibrate surface erosion and sediment delivery models (Cline et al., 1981). Such data may not be available, especially in the particular case of fire as compared to other land-disturbing activities (Ibid.). Grass seeding is a commonly recommended mitigation, but its contribution to reduction in sediment production is rarely quantified. Models for sediment production from mass erosion or peak flows have not been attempted (Ibid.; Potts, pers. comm., 1989).

The timing of cover development relative to damage-producing storms causes substantial controversy in California, where the initial winter rains can arrive with enough force to wash seed off the slopes before they germinate (Barro and Conard, 1987; Ruby, 1989). In this climate, seeded species rarely achieve protective levels of cover until the second season (Ibid.). In the northern Rockies, summer thunderstorms cause the greatest concern, followed by rapid snowmelt or a rain on snow runoff event. On the North Hills fire of 1984, on the Helena National Forest in west central Montana, an intense convectional storm extinguished the fire and triggered surface erosion and massive
debris torrents that exceeded the level of the 100-year flood (Putnam, 1985). Evidence of high peak flows from thunderstorms was also noted on the 1985 Sandpoint fire on the Lewis and Clark National Forest in central Montana. Revegetation on this fire was inadequate the first year to control sheet erosion and gullying on side slopes. Once rills and gullies formed, continued low vegetative cover has failed to prevent similar flows for at least four years (Phillips, pers. comm., 1989). Vegetation recovery can be set back severely by a dry spring, stressing seedlings of both native and introduced species when they are most vulnerable.

"The success of seeding efforts are judged more often by the amount of grass established than by the amount of actual erosion controlled or flood damage prevented" (Taskey et al., 1989). In their review of current knowledge of the use of annual ryegrass seeding for emergency revegetation in chaparral ecosystems, Barro and Conard (1987) state:

We think that more studies are needed to evaluate the role of native annuals in slope stabilization and achievement of early cover of burned slopes. Few studies have compared the date of emergence of postfire annuals and grass, and none have compared their relative abilities to reduce erosion. Barro and Conard neglect the results of Gautier (1983), who reported that reduction in soil loss measured by vertical displacement correlated with increasing cover, regardless of native or introduced origin. In the first postfire season, average reduction of net soil loss with seeding was 31% across all plots. Gautier and many others note that Ceanothus shrub seedlings suffer heavy mortality in competition with annual ryegrass, which fact he develops into a long-term erosion and
sedimentation model for the chaparral suggesting that short term soil stabilization comes at the cost of later erosion increases (Ibid.). A more recent study in a chamise chaparral burn reported a significant increase in sediment production on ryegrass seeded plots compared to unseeded plots, despite greater total vegetative cover (Taskey et al., 1989). The difference was attributed to increased activity of pocket gophers (*Thomomys bottae*) in the treated plots. Gopher densities in the hand-seeded plots were similar to nearby aerially seeded areas. Taskey et al. mention that precipitation in the first postfire winter was near or below normal, but exceptionally gentle. Replication of a study like this one during more typical weather might put gopher impacts in perspective. The tectonically active chaparral country has borne the vast majority of both emergency rehabilitation projects and studies of rehabilitation. One should use caution in extending specific relationships established there to other areas, especially in regard to erosion.

Outside of the chaparral, and with other grass species, a few studies have attempted to measure sediment production onsite, with mixed success. In South Dakota, Orr (1970) quantified sediment runoff with a trough and collection tank at 8 plots in a burn in second growth ponderosa pine. Plots were selected in July and August of the first postfire year on sites with sparse cover and sites with relatively dense cover of seeded grass. The plots with dense cover produced less runoff and less sediment than those with sparse cover. Orr attributed sediment production to precipitation in excess of infiltration by regressing sediment with various precipitation intensities. He postulates that at
least 60% total ground cover is needed for runoff control and soil stability, and that this level of cover would not have been achieved within four years without artificial seeding. Unfortunately, Orr's design in selecting plot locations does not isolate grass establishment from inherent differences in site conditions. Photos in Orr's report show substantial disturbance of the ground for plot installation, including removal of burned trees. Hann (pers. comm.) has monitored sedimentation and vegetative recovery on two fires on the Helena National Forest for several years, but results are not yet available. Guidelines and techniques for measuring sediment need wider distribution so that impacts of land management can be evaluated. Further, this information then needs to be incorporated into sediment routing models.

Sediment yield data from streams draining burned watersheds is available from a few other studies. Roby (1989) monitored vegetation recovery for 5 years on seeded and unseeded subdrainages in a 1979 fire on the Plumas National Forest in the northern California Sierras. He also measured sediment volumes with channel cross-section measurements and from catchment basins. There were no significant differences in ground cover or sediment volume per area between the subdrainages. Roby recommended that sediment from channel sources should receive at least equal emphasis as upland erosion in planning rehabilitation treatments, and that research into selection of seed mixtures for site conditions was needed. On the Entiat experimental Forest in northwest Washington, researchers tracked sediment and nutrient losses and the effects of seed and fertilizer in three adjacent watersheds burned in the same event. Total vegetative cover in the first postfire year was only 5.6% - 10.8%.
of which the seeded species contributed only 18% - 32% across all treatments (Tiedemann and Klock, 1973). First year sediment production increased dramatically over prefire rates, and rapid snowmelt the second postfire spring caused several debris torrents which destroyed sediment sampling weirs (Helvey, 1980). Nutrient analysis of debris fans and stream water led Helvey, Tiedemann, and Anderson (1985) to conclude that:

Because the source area for nutrients lost by soil erosion and debris torrents is mainly the riparian zone, the productivity of areas outside the riparian zone is not expected to be severely affected by the nutrient losses reported here. Nutrients leaving the watersheds in solution (not reported) probably are more important to overall watershed productivity.

Lyon (1976) established permanent monitoring transects on the Sleeping Child burn of 1961 in Western Montana to track vegetal development in a lodgepole pine (Pinus contorta) forest. A mixture of annual and perennial grasses was seeded throughout the burn, but did not achieve its primary purpose:

Orchardgrass (Dactylis glomerata), chewing fescue (Festuca ovina var. duriuscula), and timothy (Phleum pratense) can be credited with major contributions to herbaceous cover on the burned area; but the fact remains that total cover was under 5% in the first year and only 18% in the second. During that period the soil surface was vulnerable and protection may have been minimal by any standard (Ibid.).

Evaluation of grass seeding was not an objective of Lyon's work, and no treatment controls or erosion measurements were made.

There are several other miscellaneous factors that may influence postfire erosion. In addition to gophers, introduced grasses may attract livestock and wild ungulates. If seeding is necessary,
particularly sensitive sites should be fenced or seeded with less palatable species to avoid concentrated soil disturbance from animal hooves. In chaparral soils, saprophytic fungi whose spores germinate after heat treatment may play a significant role in protecting the soil surface from detachment by raindrop impact (Dunn et al., 1982). These native ascomycetes thrive in postfire conditions, and in laboratory tests showed a 40% reduction in sediment dislodged by rainsplash. Innoculation of burned areas with additional quantities of this fungus may enhance production of stabilizing soil crusts (Ibid.). While "seeding" cryptogams may have potential in postfire erosion control, impacts of consequent alteration of seedbed conditions for other plants should be tested before broad application. Crane and Habeck (1982) suggest that seeded grass may have functioned as a nurse crop for mosses by moderating the ground layer climate the first few postfire years, but later suppressed the bryophytes under litter. While litter from native forbs may not be as persistent, bryophytes or grass litter would protect the soil surface, and have unknown effects on seedbed conditions for later colonizing species.

3 Impacts of seeding on postfire succession and fuel management.

Understanding patterns of natural postfire plant succession may help identify those areas where natural revegetation will be adequate to control erosion after fire. It may also suggest the nature of competitive impacts of introduced grasses on native plant communities and ecosystem processes. Depending on the resource management objectives for the land in question, not all of these impacts may be
desirable. Such long term goals must be identified to avoid obvious conflicts (FSH 2509.13.03). The following review of succession patterns, the adaptations of native species to fire, various mechanisms of plant competition, and concerns for weed control provide a groundwork in plant ecology concepts, and yet more questions about the application of grass seeding. Predicting seeding impacts requires a finer level of understanding of site-specific performance of seeded grasses than is currently documented.

Postfire succession.

Forests in the northern Rockies have evolved with the regular presence of wildfire, and both flora and fauna have adapted to survive fire or return to seral habitats. A long term research program on recovery from large, catastrophic wildfires at the Intermountain Forest and Range Experiment Station in Missoula has documented that:

practically all plants that survived the fire reestablished within the first year....Also, data suggest that virtually all species that contributed significantly to early vegetal cover were established the first postfire year (Lyon and Stickney, 1976).

These data contrast with the classic conceptual model of succession in which early seral plants modify the environment to their own exclusion and replacement by later colonizers. Instead, Lyon and Stickney support a model in which the more rapidly maturing and often shade-intolerant herbs are initially dominant but are succeeded by taller, slower-growing, and often more shade-tolerant shrubs, and then trees (Ibid.). Supplemental seeding potentially creates strong competition for establishment of all species in the initial year. The practice of
seeding grass to hasten vegetative cover seems more fitting under the old model. Failure of native species to continue to colonize a burned area as the introduced grasses die out might lead to a community depauperate in species and in cover. In the northern Rockies, this hypothesis is as yet untested.

A closer look at the mechanisms by which native species revegetate burns may suggest areas where the impacts of grass seeding might be more predictable. Stickney (1986) classified plants appearing in the first ten years after intense fires according to their origin on- or off-site and the timing of their establishment. Of four groups identified (survivors, residual or onsite colonizers, initial offsite colonizers, and secondary offsite colonizers) 60% of the initial flora was from onsite survivors or seedbanks. After ten years secondary colonizers outnumbered onsite species on half the plots. Species richness in secondary colonizers does not necessarily correlate with a dominant or persistent membership in the seral community. Of 28 species establishing on plots anytime after the initial postfire year, only three species achieved measurable cover (Ibid.). In a related study on logged and burned sites, Stickney (1982) found that plants surviving from underground parts or colonizing from an onsite soil seedbank formed the majority of the vegetative cover on south slopes. On north aspect slopes with similar treatments, colonizer species from both onsite seedbanks and dispersers from offsite contributed the most cover. More than half of the species inventoried before treatment were survivors in each case, but on north slopes the survivors are more often slow-growing, shade tolerant species.
The composition and abundance of colonizers in the postfire flora is much harder to predict (Ibid.). Seedbanks tend to reflect the type, intensity and frequency of disturbance, and may also contain species not represented in the immediate prefire flora (Archibold, 1989). Kramer and Johnson (1987) calculated constancies of buried viable seeds in mature forest of three habitat types in central Idaho, which may be used as an aid to predicting postfire composition. Work in progress by Stickney (pers. comm.) on species initiating succession after the 1988 wildfires will also aid in predicting natural regenerative potential of different forest types and elevations. A practical problem in predicting postfire plant survival is simply a shortage of inventory information available from prefire conditions. A skilled person can make good estimates of prefire habitat type after disturbance, and can be a great asset to an emergency rehabilitation team in predicting potential survivors. However, even with a good estimate of postfire floristic composition, the rate of recovery will remain a site-specific judgement. Continued documentation and evaluation of fire recovery is needed to improve this art.

Competition.

Competition is a major force driving changes in floristic composition. For lack of controlled studies, the impact of grass seeding on early postfire or later community composition in the northern Rockies is not known. Elsewhere, grass seeding has resulted in a decline in species diversity (Barro and Conard, 1987; Keeley et al., 1981; Schultz et al., 1955; Orr, 1970; Taskey et al., 1989). The consequences of
floristic change may persist beyond the lifespan of the grasses themselves.

Individual species have a suite of characteristics that enable them to exploit particular ecological opportunities. Competition for water, nutrients, or light may limit plant establishment or growth. Numerous studies have described a substantial competitive advantage of germination by even a day in advance of neighboring plants, although this advantage is density dependent and highly variable between species (Bergelson and Perry, 1989). Early germination may also be detrimental if seedlings are intolerant of cold. Annuals, with early establishment and rapid growth, develop extensive root systems that are better able to capture moisture and nutrients than many taprooted shrubs and trees that establish later and grow more slowly (Schultz et al., 1955).

Moisture is usually the most limiting of plant requirements after fire, as there may be little organic matter in or above the soil to hold moisture or moderate temperatures, and the black soil surface with no shade can reach very high temperatures. Even at low levels of relative cover, moisture competition can be significant. On the 1961 Sleeping Child burn in high elevation lodgepole, the rate of attrition of tree seedlings was five times higher where grass cover was 29% than where it was only 1%, but this could reflect other site conditions (Lyon, 1976). Where moisture is abundant, nutrients or light may become limiting, and again, those species with rapid growth are better able to capture nutrients and exploit both root space and canopy space than latecomers. The initial postfire spring is a window of opportunity for establishment under conditions unequalled at any other time in the fire cycle.
The competitive impact of seeded grass is most notable on those species with similar ecological adaptations. In a study of postfire succession in the chaparral, Keeley et al. (1981) described four early seral strategies: generalized herbaceous perennials would fit Stickney's (1982) survivor category, and generalized annuals, fire annuals and fire perennials would fit Stickney's onsite colonizer category. Ryegrass seeding in this study:

had no apparent effect on total herb cover since sites with poor Lolium establishment had as high or higher herb cover as sites with high Lolium establishment. Lolium success was at the expense of the native cover and this negative effect was greatest on the fire annuals (Keeley et al. 1981).

In the northern Rockies, the fire annual category is filled by at least two herbaceous species that seem to require heating to germinate soil-stored seed. Dragonhead mint (Dracocephalum parviflorum) and Bicknell's geranium (Geranium bicknellii) act as annuals or biennials and produce abundant cover and litter after fire, but disappear from the flora completely between fires, leaving no evidence of their potential to provide early cover (Stickney, 1982). Taskey et al. (1989) observed in planter box and field studies in the chaparral that species richness of annuals in particular was reduced in the presence of ryegrass, and that second season reproduction of nitrogen-fixing lupines was dramatically reduced. Native grasses are generally good fire survivors, but seeded grasses in bare areas between survivors may limit secondary recruitment of native grass seedlings. While seeding impacts on the diverse California flora may be higher in absolute terms, relative impacts on species diversity in the northern Rockies may be substantial.

Shrubs also may be reduced in number and diversity in competition
with introduced grass. While natural rates of attrition of shrub seedlings may be high, they can be driven higher by competitive grasses (Keeley, 1981; Taskey et al., 1989; Crane et al., 1983). Shrubs in the genus Ceanothus are adapted to fire by resprouting and recolonizing from long-lived seed stored in the soil that germinates after heat scarification. In addition to providing soil cover and forage, these nitrogen-fixing shrubs may have a more important role in restoring the nitrogen lost during the fire. Shrubs in the genus Alnus, or alders, are another group of non-leguminous nitrogen fixers, but their fire adaptation is less clear. On moist sites with good soils, competition with shrubs can impede reforestation, and in such cases grass seeding may be used to reduce this competition (McDonald, 1986). In some cases, herbicidal control of grass before replanting trees is more feasible than control of a broad spectrum of native shrubs and herbs.

Grass seeding can have both adverse and beneficial effects on tree regeneration. While trees species differ in their environmental requirements, competition for soil moisture is a critical factor in initial seedling growth and survival in the northern Rockies (Lotan, 1985). Direct competition of grass with tree regeneration is not well documented in this area, but as noted above, is capable of reducing tree seedling densities. If natural tree regeneration will lead to overstocking, as is common in burned Lodgepole (Pinus contorta) stands, grass competition may be a desirable thinning agent (McLean and Clark, 1980).

The persistence of seeded grasses varies by species and by site conditions. Many annual grasses fail to set seed in areas with short
growing seasons, and gradually die out in 2 or 3 years. Other species may set seed, but require seedbed conditions that are available for only a brief period after the fire (Hunter, pers. comm., 1990). Many agricultural cultivars are poorly adapted to persist with low nutrient availability, and gradually decline. Some perennial grasses such as orchardgrass (Dactylis glomerata) are long-lived, and may persist without increasing until the canopy begins to close. Perennial grasses are seeded where native plant recovery is expected to be slow and annual grasses will not persist long enough to supply erosion control (Phillips, pers. comm., 1989). Many perennial grasses establish and grow more slowly than annual grasses, so early erosion control is less. Mixtures of annuals and perennials are often used, but if annuals in the mix are very successful, the perennials may be suppressed. Evaluation of the performance of commonly seeded species or of different seed mixtures or application rates in the northern Rockies has not been adequate to support any general conclusions.

Weeds

There are two conflicting perspectives on the effects of aerial seeding on weed control. Many areas in the west are threatened by invasive and aggressive exotic plants, many of which are adapted to colonize disturbed sites. Rapid establishment of competitive grass cover on roadcuts, for example, can shorten the time that these sites are vulnerable to weed establishment. This argument has been applied to large burned areas (Ruby, 1989) but such an extrapolation should be qualified by consideration of the vectors of weed seed dispersal. On
the other hand, aerial seeding also carries the risk of being a vector itself for weed introduction (Christensen, 1989). Seed supplier contracts usually specify that the seed be certified free of noxious weeds, but there is still a risk that sampling will not detect weed seeds at low levels. Additional samples are usually taken when the seed shipment is received, to verify weed content, composition of the seed mix, and percent germination. Results of this sampling are generally not obtained until germination tests are complete, which can vary from one to three weeks. A burn in the Hells Canyon National Recreation Area was seeded in 1988 with grass seed later found to be contaminated with a very small percentage of the aggressive exotic yellow starthistle (*Centaurea solstitialis*) which will be very costly to control (Hells Canyon NRA, 1989). If tests for weed content are to be useful, they should be obtained before the seed is applied. A delay in seed application for two or three days for such preliminary analysis of seed samples is justified, especially if seeded areas are large or isolated.

Reburn

Several years after wildfire, burned areas can again pose a substantial fire hazard, and a second burn can affect plant composition more drastically than the first. Trees killed in the first fire begin falling as their root systems decay, leaving large areas with heavy fuels close to the ground. Such fuels may cause severe soil heating. Shrub and herbaceous vegetation can create a continuous layer of fine fuels that increase rate of spread, carrying the fire over a large area. Grasses in particular cure out and remain standing for much of the fire
season, and in dense swards create an extremely flashy fuel. Seeding in
swaths on the contour would reduce this hazard by interrupting fuel
continuity (Gross et al., 1989). The relative persistence or
flammability of litter from grasses compared to native forbs has not
been documented, but for erosion control, Orr (1970) observed that
gasses provided more evenly distributed cover and more persistent
litter than native plants. Trees may be eliminated from a site if
reburning occurs before a seed crop can be produced, and plants that
recolonize from a seedbank in the soil would be depleted (Griffin, 1982;
Archibold, 1989). To enhance this effect of reburning where conversion
of forest or shrubland to herbaceous vegetation is desired, optimal
seeding rates and grazing management strategies have been identified
(Schultz et al., 1955). Better information from studies of
rehabilitation would contribute to the predictability of seeding in
other management applications.

4. Effects of grass seeding on site productivity

The most basic level of understanding of ecosystems is arguably
the mass-balance approach: measuring the inputs, sinks, outputs, and
rates of flux of elemental substances. There are at least three ways
that grass seeding could change the rates and pathways of natural
processes that control site productivity. Grass seeding may help retain
nutrient-rich topsoil on site, it may capture nutrients in soil solution
and thereby reduce leachate losses, and it may influence populations and
vigor of symbiotic nitrogen-fixing shrubs and legumes. Seeding prescriptions should assess the relative importance of each of these influences on a site-specific basis.

The nutrients bound in organic matter are either volatilized or mineralized during burning, and the mineralized forms are deposited in ash in highly bioavailable forms (Wright and Bailey, 1982). These forms may be highly soluble, and the solubility of some ions further enhanced by the rise in pH associated with fire. Heat sterilization of soil microflora may also release nutrients in mobile forms (Raison, 1979). The potential for loss of nutrients via leaching depends on the amount of water moving through the profile and the cation exchange capacity of the soil, and the rate of uptake by plants and microflora.

The importance of nutrient uptake by seeded grass or other plants may be great if cation exchange capacity is low and precipitation is high. Cation exchange capacity may be reduced by combustion of soil organic matter in severe fires, but inorganic exchange sites may be adequate to retain the majority of newly mobilized nutrient cations. On broadcast burns in western Montana, on soils developed in argillites and quartzites with a thin andic loess mantle, DeByle and Packer (1981) found that CEC was unchanged and remained adequate to retain mobilized nutrients within the rooting zone. Through lysimetric studies on a wildfire in north central Washington, Grier (1975) determined that mineralized cations leached rapidly from ash into the upper 7.5 inches of soil and were retained there. For grass seeding to capture soluble nutrients in excess of cation exchange saturation, should this occur, timing of plant growth is critical. Stream water quality studies
following the Redbench fire in northwestern Montana showed substantial peaks immediately after the fire and again with spring runoff, but not in response to summer thunderstorms (Spencer and Hauer, 1990). Grass seeding would not have been able to capture the nutrients mobilized by snowmelt runoff. In western Montana, Stark (1977) analyzed the composition of soil water, ash extracts, and postfire vegetation and fungi and determined that the net loss of nutrients from prescribed burning would be replaced by weathering and atmospheric input. Kimmins (1987) notes that soil structure also plays a role in nutrient retention, as permeability through macropores in coarse soil may move nutrients from the ash layer through and beyond the rooting zone.

Of the macronutrients nitrogen (N), phosphorus (P), and potassium (K), N is the most volatile and large amounts are lost to the atmosphere during fires. Rock weathering and atmospheric fallout resupply P, K, and most micronutrients lost during burning, but N is mostly replaced via symbiotic N fixation or in soil microbial reactions (Waring and Schlesinger, 1985). Estimated losses of N of around 700 pounds per acre (750 – 850 kg/ha) have been reported from slash and wildfires in coniferous forests (Grier, 1975; Wells et al., 1979). Ceanothus species have been reported to fix 21 to 49 pounds per acre per year (24–55 kg/ha/yr, Tiedemann, 1981); and up to 112 lbs/ac/yr (100 kg/ha/yr) on burned Douglas-fir sites in Oregon (Waring and Sclesinger, 1985; Wells et al., 1979). Ceanothus could therefore restore lost N in 8 to 35 years (Wells et al., 1979). Alder may produce 10.7 – 267 lbs/ac/yr (12–300 kg/ha/yr) (Tiedemann, 1981). Contributions of N fixing soil bacteria and blue-green algae may be significant (Ibid.) but are poorly
understood (Raison, 1979). Legumes such as clover or vetch are sometimes added to rehabilitation seed mixtures to restore nitrogen, but their performance in field conditions has not been documented. In absence of this information, it would be wise to assess the competitive impacts of grass seeding on native N fixers. Supplemental seeding of native N fixing species may be a worthwhile area for research.

A comparison of the risks to site productivity from topsoil erosion without seeding to the potential reduced capacity for N fixation with seeding would best be made specific to a site, to account for the probability of each of these mechanisms operating as expected. Soil erosion is often estimated with the Universal Soil Loss Equation (USLE), but this is probably inappropriate on many forest slopes, vegetation patterns, and precipitation regimes (Cline et al., 1981; Trieste and Gifford, 1980). However, the USLE concept of tolerable soil loss as a function of the rate of soil formation seems valid. Klock (1976) estimated that replacement of nutrients lost by erosion of surface soil would be on the order of several hundred years. In some systems, grass seeding may reduce short term soil erosion but increase it in the long term (Barro and Conard, 1987; Smith and Wright, 1989). A comparison of risks by cost of replacement may show the benefits of grass cover to outweigh negative impacts to natural mechanisms of maintaining site productivity.

In summary, despite decades of practical experience, the effectiveness of grass seeding remains in substantial doubt. Potentially adverse impacts on native floristic diversity and site
productivity have barely begun to be explored. In this relatively specialized branch of applied science, information sharing and technology transfer is critical, and improving. The most effective training for rehabilitation team members would be personal field experience in reviewing rehabilitation goals and effectiveness on a wide range of burns, but this is severely limited by feasibility. There is a high natural variability in site conditions that makes assessment of treatment effectiveness and impacts on natural vegetative succession difficult without controlled experiments. Rehabilitation prescriptions that are more refined will carry higher costs of implementation, but may be justified by the benefits.
CHAPTER 2.

Postfire Monitoring on the Canyon Creek Fire: a Case Study.

Rehabilitation of the Canyon Creek Fire included the largest aerial seeding project of all the 1988 wildfires in the Forest Service Northern Region. Monitoring the postfire recovery of vegetation on this fire makes use of an unusual opportunity to observe effects of similar fire treatment across a wide range of site conditions. Two other rehabilitation seeding projects in the Region in 1988 have been evaluated with quantitative methods, but these were smaller in scope and no results are yet available. In retrospect, it seems that a little more time or planning on the part of Forest personnel could have made the difference between simple documentation monitoring and defensible experimental designs and methods that could have resulted in reliable information for future applications. The administrative constraints on monitoring and evaluation of rehabilitation projects are discussed in the next chapter. While the design and methods used here may be less than ideal, documentation of this effort suggests more appropriate scales of resolution for assessment of the effects of grass seeding across environmental site variables. Trends evident in the preliminary results presented here may also help refine hypotheses for future evaluations of rehabilitation seeding.
Fire and Rehabilitation Chronology

The Canyon Creek fire originated from a lightning strike on June 25, 1988, in the Scapegoat Wilderness in west central Montana. It burned 51,200 acres as a prescribed natural fire under the Scapegoat - Danaher Fire Management Plan before it was declared a wildfire on August 30. During this period, drought conditions prevailed and the fire made intermittent small advances and occasional larger runs up mountain valleys. Suppression efforts to contain the fire within the wilderness boundary failed on August 29. On Sept. 6, winds in excess of 50 miles per hour pushed the many burning fronts over an additional 180,000 acres overnight. The fire was contained on Sept. 18 at a total perimeter acreage of 240,600 acres. This includes approximately 40,000 acres of Bureau of Land Management, State, and private lands, several structures, and considerable timber, livestock, fences, and hay. On the East side of the Continental Divide, nearly 70,000 acres of Forest Service land burned, including large portions of the Elk Creek, Smith Creek, and upper Dearborn River watersheds (Lewis and Clark National Forest, 1988b).

Immediately following the fire, the Lewis and Clark National Forest convened an emergency watershed rehabilitation team, in accordance with the FS Handbook title 2509.13. The objectives of emergency rehabilitation are "to minimize, to the extent practicable: 1. loss of soil and onsite productivity. 2. loss of water control and deterioration of water quality. and 3. threats to life and property onsite and offsite" (U.S.D.A., 1986). The interdisciplinary,
interagency rehabilitation team conducted a reconnaissance of the eastern portion of the burn and concluded that vegetation in areas of severe fire would not recover rapidly and that rehabilitation measures would be necessary. The team concluded that little could be done to reduce increased peak flows, but treatment would help protect the watersheds from erosion and sedimentation and reduce the threats to downstream values more quickly than natural vegetation recovery alone. Downstream values included roads and bridges, several houses and other buildings, agricultural improvements, and important trout spawning habitat in the Dearborn River. In the assessment of costs and benefits of the proposed treatment, maintenance of soil cover for site productivity on the burned area was also considered to carry extensive economic value (Lewis and Clark National Forest, 1988a).

The team evaluated several alternative courses of action with varying levels of seeding, log erosion barriers, dry channel sediment traps, and off site flood protection. The Forest Supervisor and the State Soil Conservationist approved the team's recommended plan to seed 28,600 acres which had been severely burned, had little understory vegetation, and had a low percentage of surface rock. Of this total, 18,050 acres are on the National Forest, and the remainder under State, private, and Bureau of Land Management ownership. No supplemental revegetation was proposed for the Scapegoat Wilderness. Two seed mixtures were used: On proposed wilderness areas in the Dearborn drainage, a mixture of slender wheatgrass (Agropyron trachycaulum) and annual ryegrass (Lolium multiflorum) was seeded at 13 pounds per acre, or 58 live seeds per square foot. Slender wheatgrass is a native, short
lived perennial, and annual ryegrass is alien, but expected to decline and die out in 3 to 5 years. In the non-wilderness drainages of Elk and Smith Creeks, a mixture of the perennials orchardgrass, (Dactylis glomerata) slender wheatgrass and intermediate wheatgrass (Agropyron trachycaulum and A. intermedium), smooth brome (Bromus inermis) and white dutch clover (Trifolium repens) was seeded at a total rate of 7 0 pounds per acre, or 43.9 live seeds per square foot. The rationale for using perennials was based in part on poor success of natives and seeded annuals and continuing erosion on the 1985 Sandpoint Fire in the Lost Fork Judith drainage in central Montana. There was a perceived need for supplemental cover for a longer time than annuals could be expected to persist, given this experience with similar fire intensity, soils, and climate (Lewis and Clark National Forest, 1988a).

Aerial seeding of the burn and rehabilitation of firelines were completed before winter. Random samples of the seed mixes were sent to the Montana State Seed Laboratory for analysis of purity, germination, and weed content. No noxious weed seeds were found.

Vegetation Monitoring

After the emergency watershed rehabilitation projects were completed, the Forest Service turned its attention to longer term recovery of the burned area and evaluation of the rehab treatments. Several projects were developed to monitor fire effects on wildlife, fish, water quality, and vegetation recovery. These studies were
initiated to provide a baseline for determination of recovery rates, to
document site specific changes in resource values, and to contribute to
a better understanding of fire effects in similar ecosystems. The
vegetation monitoring begun in 1989 can help answer many questions about
the nature of plant succession in the local area, response of plant
species and communities to fire, postfire wildlife habitat values,
future fuel loading for fire managers, and some of the results of the
seeding effort. The primary consideration in the design of the
monitoring was that it should provide a baseline for successional
studies using prefire vegetation data. In the first season, however,
the plots provide an inventory of vegetative recovery across several
site and treatment variables. This analysis of grass seeding is
secondary, and should not be considered complete after only one season.
The present paper reports the organization of the monitoring, some of
its limitations, and a descriptive summary and observations on the first
season of fire recovery.

Study Area

On the east side of the Continental Divide, the landforms burned
over by the Canyon Creek Fire are mainly derived from overthrust slabs
of paleozoic limestones and interbedded calcareous shales (Mudge et al.,
1984). Hillside soils are often developed in glacial drift and
colluvium with a mantle of volcanic ash. Precipitation in the
mountains ranges from 20 to 50 inches, with about 50 percent falling as
snow (Holdorf, Martinson, and On, 1980). Winds commonly redistribute
the snowpack on exposed areas. Chinook winds are also prominent at
times as western frontal systems descend from the mountains to the plains. Continental and Pacific weather systems alternate dominance over the study area, resulting in extreme and rapid temperature changes. Snowpack limits the growing season, and frosts have been recorded every month in the nearby Danaher basin west of the Continental Divide (Gabriel, 1976). Small thunderstorms are common in the summer, and can produce intense local precipitation. Elevations range from about 4000 feet at the town of Augusta to 9200 feet at the summit of Scapegoat peak. Plot elevations ranged from 4920 to 6600 feet.

Fire History

Many forest ecosystems in the northern Rockies have been strongly influenced by fire, and this area is no exception. One of the best early records for the area is the account of H.B. Ayres of the U.S. Geological Survey. His map and observations of timber and soils conditions and prospects for development in the Lewis and Clark Forest Reserve continually mention the role of fire in shaping the availability of timber.

Where fires have run they have been so severe that over large areas no seed trees and no seeds have been left. In fact, on most of these burns the humus has been consumed. ...Of the 1600 square miles within the reserve nearly 600 have been seriously burned within the last 40 years. Besides this severely burned area there are many lightly burned areas that now have some dead trees killed by fire, but are principally wooded. There are also many areas of old burns that have been restocked (Ayres, 1900). Ayres further describes the fire of 1889 as covering about 530 square miles in a very dry year, killing the canopy in most places and consuming the remains of previously burned stands. Of these areas he
found only 6 percent restocked with trees, and often only "scanty" cover of other plants, especially in dry or high altitude sites (Ibid).

Gabriel's (1976) dendrochronology work in a relatively isolated valley in the southern part of the Bob Marshall Wilderness supports his speculation that wildfires occur in two cycles: Long return period fires reinitiate a sequence of successional communities, and short return period fires are of low intensity and maintain communities. Gruell (1983) reprints many of Ayres' photographs, and others of his era, alongside recent photos of the same areas. Despite the addition of other land disturbances, Gruell's comparisons document return of forest cover in most vegetation types. Fires such as the Canyon Creek Fire are not unusual in this landscape, and the soils and vegetation reflect this coevolution.

Sample Stratification

The Forest Service was interested in assessing the effects of this fire across a wide range of site variables. An attempt to find pairs of plots with similar site conditions to compare vegetative response with and without seeding was not successful. To discover what environmental site factors control performance of seeded species as well as native plant recovery, samples were selected across an array of several independent variables. These variables, fire intensity, landtype, habitat type, slope, aspect, elevation, and prefire canopy closure are not truly independent of each other, and no statistical treatments make the assumption of independence. The dependent variables in all cases are the identity and vegetative cover of the native and seeded species.
are the identity and vegetative cover of the native and seeded species on each site, and estimated surface erosion.

Mapped information on the independent variables was inconsistent, and in most cases required ground-truthing. Logistical considerations (lack of accurate mapping) prevented the development of a distribution of samples across combinations of these variables in advance of the field season, so the stratification was, by default, erratic. Given the scale of the fire, the broad range of effects, and limitations of the monitoring budget, sampling for statistical validity was not attempted. Consequently, there was very little replication of samples even within broad grouping of site types. There were no untreated control plots established prior to the aerial seeding, apart from areas that did not meet the criteria for rehabilitation. Only plots within the wilderness were completely free of seeded species. In many cases it was not possible to distinguish areas that had been seeded deliberately but suffered poor establishment from areas that received only drifted seed. While grass seed was supposedly applied only to specific areas, redistribution of seed by winter winds may have contributed to the inaccuracy of seeded area maps. Monitoring of the actual application of seed to the areas designated for seeding in the rehab prescription was very limited. In the analyses below, plots with over 2% cover of seeded species were arbitrarily classed as having been seeded. Summary statistics of seeded cover may, therefore, underestimate the establishment and growth of the introduced species.

Pre-fire vegetation data is rarely available in the case of wildfires, but the Canyon Creek Fire coincidentally ran over vegetation
that had recently been sampled and described for other purposes. Over a hundred vegetation plots were measured in 1986 and 1987 as part of a project to map grizzly bear habitat from remotely sensed spectral reflectance values. This information provides an unusual opportunity to document fire succession on stands of known composition. Fire effects on ecosystems on the dry east side of the continental divide have received much less attention than on west side forests, so resampling of these plots dominated the study design. The sampling strategy used here benefits the interpretation of successional patterns more than the effects of grass seeding. The present paper concerns only the initial year of sampling, and focuses on the implications of grass seeding in forested areas with high intensity, stand replacing fire.

The biases of the 1986 and 1987 vegetation sampling are not fully known, but they preclude the assumption that post fire resampling is a random representation of fire effects for this area. While there was no preconceived bias in sampling many of the independent variables, neither was there any deliberate randomization. Recovery of nonforested sites and of lower intensity burns was of less interest, so resampling of these areas was minimal. The results presented below therefore should not be taken as representative of individual site types or of the entire range of variation across the burn.

Sampling Methods

Plot sampling was needed for detailed description of the various combinations of site and treatment variables and vegetative response. A standardized methodology for integrated resource inventory and
monitoring has been developed recently by the ecosystem management group in the Forest Service Northern Regional Office. "Ecodata" combines widely accepted sampling techniques with paper forms and data entry and storage programs to make resource inventory and analysis projects more convenient, efficient, and consistent throughout the Forests in the Region. Ecodata is also supported by several data analysis programs to cross check for logical errors, compile summary statistics, ordinate and classify plant communities, and other analyses. The developers of ecodata aimed to promote interdisciplinary information transfer and correlation by providing for specific plot location records including a variety of site parameters. Sampling methods for several levels of detail are available to help fit the package to particular project needs. Additional documentation can be found in the Ecodata Handbook (Hann, et al, 1988).

The vegetation plot data from the 1986 and 1987 mapping effort was not keyed to any permanent markers, so there is an element of subjectivity in resampling. I relocated pre-fire plots by following marked topographic quad maps to a general location, then searching for slope and aspect and stand conditions to match the prefire data. Once within the stand, I selected a plot center to represent the age structure and species composition typical of the stand. This follows the technique advocated by Mueller-Dombois & Ellenberg (1974) termed subjective without preconceived bias. Where possible, I also placed the plot near a distinctive boulder or twisted snag for ease in future relocation. Plots were permanently marked with a short length of steel rebar painted orange and surrounded with a small rock cairn.
photos taken after the fire were pinpricked and labelled for each plot. Each plot center was photographed and a sketch map of local features included with the plot sampling form.

Once a plot was located, site environmental features were recorded, including landform, slope, aspect, elevation, surface erosion, ground cover, tree, shrub, graminoid, and forb cover, and evidence of recent disturbance. Slope and aspect were measured with a clinometer and compass to within 5 degrees, and other parameters were estimated. Measures of fire intensity and severity as suggested by Ryan and Noste (1983) were added to the ecodata forms. Fire intensity was rated by six classes of scorch height, with additional information recorded for horizontal and vertical variation in canopy damage. Fire severity was scored by estimating the percentage of ground surface in each of four classes of depth of ground char. Habitat types were taken from pre-fire data where possible, as many understory indicator species are not good fire survivors. Where pre-fire data was not available, habitat type was estimated from the remains of trees, site characteristics, and survivor species while bearing in mind that cover would be reduced and that many fire-sensitive indicator species would be absent. Canopy closure of the pre-fire stand was estimated from the remaining snags.

Inventory plots, called Ocular macroplots by the Ecodata handbook, consist of visual cover estimates for all species on a 1/10th acre circular plot. The plot is marked by temporary flags at a 37 foot radius from the center cairn, then systematically searched and all plant species identified. Nomenclature of plant species followed Hitchcock
Canopy cover is estimated for each species, and recorded in classes of trace (<1%), present (1-5%), or one of ten classes with midpoints of 10%, 20%, 30%, etc. Mean heights and age-size classes, and comments on phenology are recorded. These attributes are also recorded for each dead tree species.

Visual estimates of cover classes were calibrated by scoring plots by a more objective method occasionally throughout the field season. For these plots, replicated measures of 25 systematically located microplots within the same 1/10th acre macroplot were summarized. Ocular estimates of ground cover and canopy cover were plus or minus one cover class at the time of sampling. However, the sampling season covered much of the growing season, so interpretations of these data should include a larger margin of error. Figures for total cover of introduced species were calculated from the sum of individual seeded species, and where cover was low, cover class breaks lead to inaccurately high total cover. The magnitude of this inaccuracy is very likely less than the unavoidable inaccuracy caused by sampling throughout the growing season. In some stands, inconsistent growth forms and complete combustion of cones and bark led to difficulties in distinguishing among whitebark pine, lodgepole, and limber pine, particularly on harsh sites. Conifer seedlings were not consistently identifiable to species. Vegetative characters of *Lolium* and *Agropyron* species are very similar, which prevented positive identification until later in the season. Relative success of different grasses in the seed mix was therefore not determined.

Sheet erosion can be difficult to measure as deposition from
upslope can be equivalent to downslope losses. Erosion depth was estimated on each plot by observing the extent and height of pedestals formed under pebbles and downslope from plant crowns. Uniformity of erosion was noted as a percent of plot surface area affected. Very little rill formation was noted, probably because soils are very stony. Spring runoff in 1989 produced very little sediment. Most surface erosion on the plots is the result of summer thunderstorms.

RESULTS

Fire effects can be assessed at several different scales. The stratification of independent variables used here encompasses a broad range of resolution, at which some level may correspond to the scale of variability in cover and species richness in a meaningful way. The results may suggest the best level on which to concentrate efforts to refine seeding prescriptions. Land managers have already classified the natural variability in forest ecosystems for various purposes. At the coarse end of the scale, landtypes are based on broad classes of landform, aspect, soils, and vegetation, and are intended for general land use planning (Holdorf, 1981). Fire Groups are narrower categories, developed as an interpretive tool for understanding the ecological role of fire (Davis et al 1980). Each fire group is composed of several habitat types based on the fire responses of their major tree species.
and their successional dynamics. Again, suggested successional pathways and fire management implications are intended for more general planning uses, not as site-specific predictions (Fischer & Clayton, 1983).

Habitat typing is a widely used and useful method of classifying sites according to their ecological potential (Daubenmire 1952). Through this classification, indicator species present through mid-serial and later stages are used to key out the potential natural community that would develop on the site in the absence of disturbance (Pfister et al. 1977). Ultimately, the site-specific responses of a plant community to fire depend on both the responses and interactions of the individual species present, and the growing conditions on that site.

To some extent, all these classifications incorporate the most basic environmental variables by using vegetation as an indicator of particular patterns of conditions. If research could discover a correlation of these mapped classifications with postfire recovery of native or seeded species, it could greatly improve the efficiency of rehabilitation treatments. In the following section, bar graphs illustrate the relationships of seeded and total cover to gradients in the basic independent variables, and tables illustrate the relationships of seeded and total cover to the above classifications. Data is also presented for species diversity relationships, and erosion. For all but the first comparison, of fire intensity effects, the analyses include only those plots that burned at high intensity.

Erosion at different burn intensities.

As discussed in the previous chapter, fire survivors are generally
the most important component of postfire vegetation, and fire survival is determined by the intensity and severity of the fire. Low and moderate intensity burns typically show low levels of erosion, as the soil surface is protected by surviving vegetation and a mulch of fallen scorched needles. Patchy variations in fire intensity in these areas also allow the survival of less fire-tolerant herbs, and the remaining tree canopy and mulch help moderate microclimatic growing conditions. In high intensity burn areas, the heat treatment is generally more uniform, and there is often little cover left to ameliorate growing conditions. Fire intensity can also vary with prefire vegetative condition, burning hotter and longer where more fuel has accumulated. Fire intensity may be indirectly dependent on soils, as better soils can support a greater biomass and hence, fuel load.

Figures 2, 3, and 4 illustrate the correspondence between postfire vegetative cover and erosion at three fire intensities. Erosion is scored here as an index, calculated by multiplying average erosion depth in tenths of an inch and the percentage of plot area eroding. This index is not intended as a reliable absolute measure of erosion, but is adequate to compare relative differences between plots. Erosion was greatest on high intensity burn sites, and less where burning was cooler or patchy. Within all fire intensity classes, the relationship of cover to erosion is less clear. Mean erosion depth for seeded and unseeded plots was very similar at .45 and .43 inch respectively. On the 5 plots with seeded cover over 20%, my conservative guess at the cover level that could affect erosion, the mean erosion depth was 0.50 inch. The local and sporadic nature of thunderstorms and the small sample of plots
Figure 2. Postfire vegetative cover ranked by erosion index for the nine plots sampled in low intensity burn areas. Vegetative cover is the sum of individual species' cover in percent, and the erosion index is the average depth of erosion on a plot in tenths of inches, multiplied by the percent of plot area eroded. Recovery of native vegetation was evident, averaging 58% cover. These areas were not intentionally seeded.

Figure 3. Postfire vegetative cover (in percent) for the 19 plots sampled in moderate intensity burn areas, ranked by erosion index. These areas were not intentionally seeded, but many apparently received seed drift. Average total vegetative cover was 54%.
Figure 4. Postfire vegetative cover (in percent) for 30 plots sampled in high intensity burn areas, ranked by erosion index. Many plots showed substantial erosion despite moderate levels of cover. Average total cover was 33%. The relationship between erosion and vegetative cover is weak, with no obvious correlation to the cover produced by seeded grasses. Note the change in scale from previous figures.
with effective seeded cover prevents any valid conclusion on the ability of seeding to reduce erosion.

The causality of the relation between seeded cover and erosion cannot be inferred from this sampling design. Control and treatment plots on the same site conditions and close enough to experience the same precipitation events would be necessary to attribute a reduction in erosion to the seeding treatment. In addition to this design change, more objective and repeatable methods for scoring erosion are needed. Small numbers of samples across other variables limit the interpretation of fire effects within intensity classes.

Variability in vegetative response within the high fire intensity class may have other contributing factors. High intensity burn treatments, as were observed on much of the Canyon Creek Fire, are not necessarily accompanied by deep or prolonged soil heating. Relatively light fuel loadings in young stands and high winds during burning probably contributed to the moderate and low severity ratings. Drought conditions during 1988 may have predisposed many plants to better fire survival. Early senescence and storage of energy reserves in underground parts would give perennials a more vigorous resprouting response than had they been burned while still actively growing.

Effects of site characteristics on vegetative recovery:

1. Slope

Steep slopes may affect initial fire response by increasing freeze-thaw soil movement, which can uproot shallow rooted seedlings. Surface erosion is also greater with increasing slope and adversely
affects seedling establishment. One would therefore expect resprouting natives to have a competitive advantage in restocking steeper sites. In the high intensity burn plots, slopes sampled ranged from 5 to 60 percent. Figure 5 plots cover of seeded species, native species, and total cover against slope classes in percent. Dividing the distribution into two classes above and below the median slope shows that both native cover and seeded cover are greater on gentler slopes. Seeded species produced more cover relative to natives on seeded plots on the gentle slopes, and relatively less on the steep slopes.

2. Elevation

High elevation affects vegetation recovery primarily through lower temperatures and a shorter growing season (Arno and Hammerly, 1984). However, there are many synergistic factors that could affect species composition and growing conditions. High elevation sites receive more precipitation, but also more wind and higher evapotranspiration rates. Soil moisture at higher elevations is not considered to limit forest growth (Ibid.), but coarse textures may inhibit establishment. High intensity plots ranged in elev from 5060 to 6600 feet. Figure 6 shows that most of the seeded plots are at lower elevations, where both seeded and native species achieved greater cover.

3. Aspect

Solar radiation varies with aspect according to the season. Soil temperatures, surface evaporation, and evapotranspiration potentials correlate to the duration and angle of incidence of sunshine. In the
Figure 5. Postfire vegetative cover (in percent) of 30 high fire intensity plots ranked according to slope steepness (in percent). Fewer steep slopes were intentionally seeded, and absolute cover of the seeded species was greater on gentle slopes.

Figure 6. Postfire vegetative cover (in percent) of 30 high fire intensity plots ranked according to elevation (in feet). All of the plots with substantial cover of seeded species were at lower elevations.
winter, aspect affects snowpack longevity and its relative contribution to soil moisture recharge. On the Rocky Mountain front, east aspects receive additional moisture in drifts deposited by prevailing west winds. In general, northern and eastern aspects are cooler and moister than southern and western aspects. Moisture stress in the shallow rooting zone of young plants is often the critical factor affecting survival (Haeussler and Coates, 1986). Figure 7 displays cover of seeded species and total understory cover by aspect. All the plots with seeded cover over 20 percent were on north or east aspect slopes. Cover of native species was also greater on these sites, but the increment of difference over other aspects was less.

4. Prefire Canopy Closure

In addition to abiotic site variables, prefire vegetation conditions may help explain some of the variability in postfire floristic response, in both quantity and composition. Overstory canopy closure often corresponds with a decrease in understory cover (Lyon and Stickney, 1976). As a stand matures, competition for moisture, light, and nutrients favors those species best adapted to site conditions, and the diversity of early seral generalists is reduced (Huschle & Hironaka, 1980). Drought and shade intolerance are probably eliminated at different rates depending on the site. Archibold (1989) notes that stand age is relevant to seedbank species only if a site is reburned before the obligate seedbank species produce seed. Stand age is also irrelevant to those species that colonize from wind dispersed seed. In most cases canopy closure would be a more robust estimator, and a more
Figure 7. Postfire vegetative cover (in percent) of 30 high fire intensity plots arrayed according to the aspect of the plot in degrees. North to East aspects (355 to 115 degrees) supported the most abundant growth of seeded grasses, while South and West aspects (140 to 350 degrees) supported primarily native plants.
practical indicator, of native vegetative recovery than stand age.

Figure 8 charts postfire understory cover against prefire canopy closure. There is clearly a wide variation in cover within canopy closure classes. Table 1 presents classes of prefire canopy closure in the high intensity burn plots with their stand age, understory cover, and species richness. Among the high intensity fire plots the number of native species per plot generally increased with decreasing prefire canopy cover. This relationship might be clearer if growing conditions were equivalent. Among the high intensity burn plots, prefire stand age did not necessarily correspond to canopy closure, reflecting variable site conditions. The correspondence of canopy closure to understory cover and species richness is confounded by two factors: open stands were intentionally not seeded, and where seeded cover is high, competition may have eliminated some native species. Again, without designed treatment and control plots on adjacent sites, such relationships must remain speculative.

5. Landtypes

There are two landtype classifications covering the area sampled. The Land System Inventory of the Scapegoat delineates and describes land units with similar response primarily to fire management, but also considering wildlife habitat, watershed behavior, and wilderness recreation (Holdorf, Martinson, and On, 1980). For the more intensively managed front country, the Soil Resource Inventory includes more specific ratings of soil stability hazards, productivity, and suitability for grazing, timber, and roads (Holdorf, 1981). Plot
Figure 8. Postfire vegetative cover (in percent) of 30 high fire intensity plots ranked according to prefire tree canopy closure in percent. Within classes of canopy closure, plots are ranked according to percent cover of native plants. There is no apparent pattern of response by native species, but seeded species grew best where prefire canopy closure was 60 to 70 percent. Mean cover for native species and seeded species by canopy closure class is given in Table 1.
Table 1.
Relationships of postfire cover and species diversity with prefire tree canopy closure.

<table>
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<tr>
<th>Plot</th>
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Diversity of native species in the first year after intense fire was greater where prefire stands were more open; but there was little difference in the quantity of postfire vegetative cover. Grass seed was applied mostly where the fire consumed dense forest, but whether for lack of seed or poor establishment, many plots produced only trace cover of seeded species. Mean seeded cover figures include only plots with at least 1% cover. In plots where cover of seeded species was high, both cover and diversity of native species may have been suppressed. Stand age is given to indicate variability in site quality within classes of tree canopy closure.
landtypes were determined directly from maps, but mapping units often contain inclusions of up to 15% of other landtypes with dissimilar characteristics.

Table 2 lists total understory cover and seeded cover for the landtypes sampled, along with site parameters that gauge the fit of the classification. Fire intensity is included in this table to help explain the variation in cover of surviving vegetation. Descriptions of the landtypes and relevant management implications can be found in Appendix A. Comparison of site parameters on many plots showed a poorer fit with the landtype definition they were mapped to. Small numbers of samples within each landtype, and the interference of other variables prevent the detection of any clear pattern in the relation of landtype to postfire vegetative recovery. Landtypes may exhibit a stronger correspondence to effects of less intense fires, but this was not tested.

6. Habitat Types and Fire Groups

Habitat typing, pioneered by Daubenmire (1952), is now widely accepted as an appropriate land classification system for most forest management and research applications, and for communication between these disciplines. As a classification of the ecological potential of a site, it acknowledges climatic and edaphic constraints that affect the successional sequence of communities on a site through time. The 30 high intensity burn plots sampled 16 different Habitat Types. Of these, the greatest replication of sampling was in the Subalpine fir / Beargrass type with 5 plots, and in the Subalpine fir / Pinegrass type.
### Table 2 continued

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<th>Land Plot</th>
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<th>Cover slope aspect</th>
<th>Elev</th>
<th>Fire Intensity</th>
<th>Habitat Type</th>
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<td></td>
<td>H Able/Vite</td>
<td>7</td>
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</tr>
</tbody>
</table>

**Wilderness landtype associations:**

| III       | 55     | -          | 70.5             | 5    | 20             | 5770          | H Able/Vite  | 7     |
| 58        | -      | 33.5       | 10              | 65   | 5980           | H Able/Mef  | 9         |       |
| 59        | -      | 3.5        | 10              | 270  | 5990           | H Able/Mef  | 9         |       |
| 60        | -      | 21         | 10              | 360  | 6000           | H Able/Vane  | 7         |       |
| 61        | -      | 4          | 15              | 40   | 5550           | H Able/Vite  | 7         |       |

| IIIA      | 52     | -          | 60.5             | 50   | 207            | 6060          | H Piece/Juco | 6     |

| V         | 51     | -          | 31              | 42   | 25             | 6220          | H Able/Vite  | 7     |
| 53        | -      | 14         | 30              | 350  | 6200           | H Able/Mef  | 9         |       |
| 54        | -      | 33.5       | 30              | 350  | 6200           | L Able/Mef  | 9         |       |
| 56        | -      | 33.5       | 50              | 186  | 6830           | L Able/Vite  | 7         |       |
| 57        | -      | 4          | 50              | 190  | 6420           | H Able/Vite  | 7         |       |
| 63        | -      | 4          | 5               | 185  | 6500           | H Able/Vite  | 7         |       |

| VI        | 66     | -          | 30.5             | 15   | 140            | 6660          | H Able/Vite  | 7     |

| VIII      | 62     | -          | 50.5             | 40   | 265            | 6480          | H Piece/Caru | 5     |
| 64        | -      | 21         | 55              | 225  | 6400           | H Piece/Cage | 5         |       |
| 65        | -      | 50         | 50              | 230  | 6400           | H Piece     | 5         | young Piece on grassland |
| 67        | -      | 50         | 55              | 210  | 6220           | H Piece/Caru | 5         |       |
with 4 plots. For both of these Habitat Types, sampling was in stands with 50 to 80 percent canopy cover and stand ages of 150 to 200 years. Three plots in the Subalpine fir / Pinegrass type supported at least a few plants of the seeded grass species, but their cover was less than 2 percent. Unfortunately, there was no way to verify whether these sites received only seed drift or were deliberately seeded and suffered poor germination and establishment. The other plots were in wilderness and were not seeded. Site parameters and summary statistics for cover of native and seeded species are presented in Tables 3 and 4. Both of the two most common habitat types sampled had highly variable cover of native species with average cover of each type similar to the average across all plots, and consistent poor performance of seeded species. A broader classification of site conditions might be appropriate to explain patterns of postfire vegetative response.

Fischer and Clayton (1983) used the classification of Forest Habitat Types of Montana (Pfister et al. 1977) as a basis for a summary of fire ecology and management considerations for forests east of the Continental Divide. To explain general patterns of fire behavior, fire effects and postfire successional pathways, Fischer and Clayton (1983) lumped habitat types together into Fire Groups based on the characteristics of their typical tree species and fuel loads, as well as common seral cover types. Descriptive definitions for the seven Fire Groups sampled are given in Appendix B. Table 3 illustrates several parameters of the plots belonging to each fire group sampled. Summary statistics for native and seeded cover are given in Table 4. Fire groups 4, 5, and 6, all Douglas-fir series Habitat Types, produced the
Table 3.  High intensity fire plots by Fire Group

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<th>Fire group</th>
<th>plot</th>
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<th>prefire canopy</th>
<th>stand age</th>
<th>Habitat Type</th>
<th>first year % cover</th>
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<td>02</td>
<td>Y</td>
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<td>120</td>
<td>Abla/Libo</td>
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<td>Abla/Libo</td>
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<td>70</td>
<td>170</td>
<td>Able/Nafe</td>
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<td>T P 3 3</td>
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Table 3. Performance of seeded grasses by fire groups for 30 high intensity burn plots. Descriptions of Fire groups are appended. Cover is listed for S = shrubs, G = graminoids, and F = forbs. Cover classes are abbreviated T (trace) = 0-1%, P (present) = 1-5%, 1 = 5-15%, 2 = 15-25%, 3 = 25-35% etc. Cover of seeded species is included in total cover, and also listed separately. See Table 4 for summary statistics.
Table 4.
Cover and Species Richness by Fire Groups and Habitat Types
Canyon Fire High Intensity Plots

<table>
<thead>
<tr>
<th>Fire Group</th>
<th>Native species cover% mean</th>
<th>S.D.</th>
<th>C.V.</th>
<th>Seeded species cover% mean</th>
<th>S.D.</th>
<th>C.V.</th>
<th>Species richness mean</th>
<th>S.D.</th>
<th>C.V.</th>
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<td>11.0</td>
<td>36.2</td>
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<td>(N = 1)</td>
<td>30.3</td>
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<td>5</td>
<td>40.6</td>
<td>17.0</td>
<td>41.9</td>
<td>0</td>
<td>(N = 2)</td>
<td>26</td>
<td>5.3</td>
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<tr>
<td>6</td>
<td>39.2</td>
<td>24.0</td>
<td>61.1</td>
<td>47.5</td>
<td>33.0</td>
<td>(N = 2)</td>
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<tr>
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<td>23.1</td>
<td>81.6</td>
<td>3.3</td>
<td>2.5</td>
<td>(N = 3)</td>
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<td>8</td>
<td>27.4</td>
<td>24.7</td>
<td>90.1</td>
<td>1.2</td>
<td>0.3</td>
<td>(N = 3)</td>
<td>18.8</td>
<td>8.0</td>
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<td>13.1</td>
<td>55.5</td>
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<td>7.3</td>
<td>(N = 3)</td>
<td>19.4</td>
<td>10.7</td>
<td>55.1</td>
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</table>

Habitat types:
Abla/Caru 33.5 24.5 73.1 1.25 (N = 2) 16.0
Abla/Xete 33.6 23.9 71.3 0 19.0

Table 4. Summary statistics for Cover and Species richness by Fire groups and habitat types for 30 high fire intensity plots. Fire group descriptions are appended. Abla/Caru is Subalpine fir / Pinegrass habitat type and is included in fire group 8. Abla/Xete is Subalpine fir / Beargrass habitat type and is included in fire group 7. S.D. = standard deviation, C.V. = coefficient of variation, or S.D./mean. N = number of plots in that fire group that had at least 1% cover of seeded species. Species richness includes only native species.
greatest cover of native species. Of the plots that had obviously been seeded, fire groups 6 and 9, moist Douglas-fir and moist lower subalpine fir Types, produced by far the greatest cover of seeded species, and average cover of seeded species exceeded that of native plants.

There is a distinct possibility that cover of seeded species interfered with the amount of cover produced by native species particularly on these moister sites. The two unseeded plots in Fire Group 6 produced relatively high cover, while on the two seeded plots, seeded species produced over half the total cover. While this constitutes a very small sample, it suggests that these sites can support relatively high plant biomass and it will belong to the earliest or most vigorous species. In Fire Group 9, two unseeded plots had relatively low cover, and three seeded plots had relatively high cover, most of which was seeded grasses. On these sites, it would appear that grass seeding adds to the total cover produced. Further research is needed to determine the circumstances under which the cover contributed by seeded grasses is compensatory or additive to the cover of native plants. It would seem likely that there may be a threshold of postfire seedling density at which this relationship would reverse.

The variability in cover within and between habitat types is compared to the variability in Fire Groups in Table 4. The similar coefficients of variance in these two scales of resolution may indicate that either scale is appropriate for an assessment of vegetative recovery, or that these particular habitat types are coincidentally near the center of the overall range in variability. The comparison is weak because of small sample sizes and no assurances that distribution of
variability is normal.

Classification of postfire flora.

Habitat Typing owes its utility to the concept of classifying site potential on the basis of currently existing 'indicator' plants. This idea of using the vegetation itself as the integrator of site conditions was applied in an attempt to classify initial postfire plant communities. If the community composition on certain plots was found to share a high degree of similarity when considered without the seeded species, this could effectively isolate the seeding variable. If species composition is very similar, differences in relative cover of native and introduced species might then be attributable to competitive effects. While ordination of vegetation data typically involves some measure of the abundance of each species, it is also amenable to presence / absence data. In highly heterogenous communities (such as the first postfire season) abundance may have a lesser value as an indicator than presence (Pielou, 1964). Ignoring abundance also removes the need to compensate for phenological development of cover through a long sampling season.

There are, however, some difficulties with this approach. If competition was severe, some species could be completely eliminated even before they could be noted in first season sampling. At some point a threshold level of similarity must be selected, and if there are no obvious break points in the numerical clustering, it would be arbitrary. Ordination of data is not amenable to statistical testing because the occurrence of a species is not necessarily independent from the
occurrence of any other species (Ibid). A good classification requires an adequate representation of community composition throughout the gradients sampled. A small database such as the 30 plots tested here may not be adequate to characterize real groups.

There are two procedures involved in building a classification or an ordination. The first step is measuring the similarity between plots and the second step is measuring the similarity between clusters of plots. An appropriate method must be selected for each kind of measurement.

For similarity of plot composition based on presence/absence data, Jaccard's index of similarity is one of the simplest: the ratio of common species to the total number of species in a pair of plots. Sorensen's index, the ratio of common species to the average number of species, is often preferred because it includes a term for the probability of a species' occurrence (Mueller-Dombois and Ellenberg, 1974). But Pielou (1984) argues that dissimilarity is a more appropriate measure of the multidimensional distance between plots for presence/absence data. Jaccard's index is converted to a measure of dissimilarity by taking its complement, or the ratio of unique occurrences to the total number of species in a pair of plots. Pielou argues further that this measure is better suited to the mathematical requirements of certain ordination and classification routines than dissimilarity as measured by the complement of Sorensen's index. It also has the advantage that species absences are given less weight than presences.

The next step in grouping plots according to their species
composition can be started from either of two directions. A divisive classification splits the entire group into successively smaller subgroups beginning with attributes of the entire data set (Pielou, 1984). An agglomerative classification begins by selecting the most similar plot pair and then successively lumping it with the next plot pair with attributes most similar to the combined pair. This involves the calculation of new dissimilarity indices for each cell in those rows and columns of the matrix at each step. For the purposes of the classification desired here, the agglomerative clustering is the more direct routine.

The matrix of indices of dissimilarity between plots is presented in Appendix C. Dissimilarity was calculated as the complement of Jaccard's index for all the plot pairs within each fire group. This was done as a preliminary step to filling the entire matrix (420 cells) because it was expected that the most similar plots would be at least partially within these groups. Fire groups 4 and 5 have the greatest proportion of dissimilarity figures below an arbitrary 50 percent cutoff, and these are the fire groups that also have the lowest coefficients of variation in species richness and native cover. In examining the seven plot pairs with at least 50 percent similarity, few appeared to be reasonable pairs for comparison of seeding effects. Four pairs had the same seeding treatment, and two other pairs had seeded cover of 1 percent or less. Only one plot pair, in fire group 7, offers the prospect of a viable isolation of seeding effects. Of the plots with at least moderate cover of seeded species, all had relatively low similarities with other plots within their respective fire groups.
Continuation of the classification routine beyond this point was not warranted.

Summary

The design of the Canyon Creek Fire vegetation monitoring study is much more conducive to description of the range of fire effects than to site or type specific responses. An appreciation for this range of variability should assist in designing future monitoring and studies of seeding effectiveness to an appropriate scale. Fire intensity variation can overwhelm most all other factors, so this should be a primary stratification. Landtypes are too coarse in resolution to correspond to fire effects patterns without much more extensive sampling. Fire groups and habitat types appear to be nearly equivalent in relation to early postfire succession trends at the intensity of sampling used here. If fire effects information is desired for a narrower range of site types, habitat types would be a more appropriate stratification. Slope, aspect, and elevation are at least partially included in site type classifications, which in effect integrate the myriad ecological interactions which contribute to community composition. On sites with similar ecological potential, canopy closure and stand age will be closely related, and should closely correlate with prefire understory flora and fuel loading. Throughout any sampling stratification, a paired plot design to isolate site variables and control the dependent variables would add immeasurably to the interpretability of results. Replicated sampling within plots will add confidence and allow for tests
of validity.

The criteria used by the rehabilitation team for selection of burned areas to be seeded were not systematically tested. To validate these criteria, monitoring of seed catch on target sites during the seeding operation would be needed. More refined evaluations of fire severity hold the most promise for improving seeding prescriptions. There may be significant relationships between soil textures, erosivity, and establishment of colonizing species, but they were not investigated here. On the better soils, those more highly developed and supporting most rapid tree growth, the vegetative potential of native seedbank species seems underestimated.

Future studies should seek to refine the threshold of plant density at which competition between natives and seeded species begins, so that seed application rates can be more efficient. This critical level will vary according to the site moisture regime. Evaluation of seeding treatments should continue over several years to document the persistence of seeded annual and perennial species, and to track long term influences on community composition. Measures of plant frequency will provide better information on vegetation dynamics across seasons. Soil chemistry studies are needed to determine the dynamics of nutrient capture by postfire successional vegetation.

The least predictable and most important factor in revegetation of stand-replacement fires is probably the weather in the first postfire growing season. In areas where our knowledge of severity effects on native species recovery is poor, the addition of supplemental colonizer species may be justified. If surface erosion is excessive early in the
growing season, both native and introduced colonizer species may be buried or washed away. The margin of improved protection gained by seeding species that establish rapidly depends on the relative rate of establishment of natives. These differences are poorly documented.

Assessments of risk will remain a critical element in designing fire rehabilitation prescriptions. Interpreting fire severity and its relative potential to kill resprouting species has a risk of error. There is a risk that grass will fail to establish or perform as expected. The risks of sediment damage from an untreated site where severity was underestimated must be balanced with the risks of weed introduction and possible suppression of native species where seed was overapplied. Improved evaluation and monitoring is essential to provide this information.
Chapter 3.

Implementation of the Emergency Burn Rehabilitation program and suggestions for policy refinements.

Actions that affect public land are inherently a matter of public policy. Wildfire rehabilitation is infrequent and has a very short planning horizon, and perhaps for these reasons escapes the standard of review applied to more routine projects. The above review of the technical literature shows that the information base is insufficient to accurately predict the effects of fire or of rehabilitation treatments. Administrative procedures in the rehabilitation program are also in need of improvement. The additional information needed could come from post-project monitoring, from research, or from the public involvement process. Public policy goals of the Forest Service may not be served efficiently by opening up the issue for debate at this juncture, but flaws in current administration need to be addressed. This section explains the administrative framework and procedures for developing rehabilitation prescriptions, critiques the process, and suggests remedies.

Statutory Framework

Forest Service activities are guided by several statutes which are codified in federal regulations, and interpreted and issued to all units
of the National Forest System. The Forest Service Manual (FSM) is a series of policy directives issued from the national headquarters. It contains the legal authorities, responsibilities, delegations, and general standards, criteria, and guidelines to plan and carry out programs Service-wide (36 CFR 216.2). A corresponding series of Forest Service Handbooks (FSH) contain the technical procedures and instructions for on-the-ground implementation. Supplements to these directives are developed as necessary by Regional Offices or individual Forests for their specific needs. Within these guidelines, operations on each Forest are directed by their respective Land and Resource Management Plan, known as the Forest Plan. This document spells out how the Forest intends to provide an integrated program of multiple use and sustained yield of goods and services. Forest Plans are prepared in compliance with the Forest and Range Renewable Resource Planning Act (RPA), the National Forest Management Act (NFMA), and the National Environmental Policy Act (NEPA), and other applicable statutes.

Budgets

The Forest Plan guides project-level resource management activities through area allocation, standards, schedules, and monitoring requirements, but the outputs of goods and services and the rate of plan implementation are determined by the annual budget process (Lewis & Clark N.F., 1986). Determination of Forest budgets is a very involved process, driven by national targets for outputs based on Administration and Congressional priorities (Ibid). The planning process is continuous, with amendments and adjustments made to reflect better data
or changes in assumptions and public needs. The annual budget proposals are likewise updated within the scope of the Plan, but the Forest is accountable for the targets set in the Congressional appropriation. Limited funds for wildfire suppression and rehabilitation are appropriated annually to a separate pool, not attached to Forest Plans but administered under Forest Service Manual and Handbook direction. This fund is known as Fighting Forest Fires or "FFF". Similar emergency rehabilitation funds are available for non-Forest System lands under the authority of the Emergency Watershed Protection assistance program administered by the Soil Conservation Service (SCS, 1975).

The Rehabilitation Team

As soon as a fire is controlled, an interdisciplinary team is mobilized to assess watershed conditions. The Burned-Area Emergency Rehabilitation Handbook (FSH 2509.13) describes pre-season planning and training for rehabilitation teams. Training generally consists of familiarization with the procedural content of the handbook, with potential team members assembling the references and other materials they will need to complete an evaluation and burned area report (Ibid). Cooperative agreements are made in advance to include other agencies where fires may affect multiple ownerships. While the Handbook recommends that several disciplines, e.g. hydrology, soil science, engineering, silviculture, be represented on the team, there are no rigid stipulations that particular skills must be included. Where seeding treatments may be anticipated, the team should include a plant ecologist or other person familiar with fire effects on understory
species. There is a heavy reliance on personal and professional experience in burned area assessment, as there is little time to collect and analyze data. To meet the 'emergency' criterion in the Handbook, the assessment report and request for funds must be made within two days of control of the fire (FSH 2509.13.21). On very large fires, this stipulation may result in prescriptions based on very coarse information (Hunter, pers. comm., 1990).

Assessment of a burned area and development of a rehabilitation prescription is a complex and site-specific task. First, a reconnaissance survey uses information from maps, photos, and/or overflights to obtain an overall perspective and identify subareas for more detailed investigation. Hazardous watershed conditions and areas that will be relatively homogeneous in response to treatment are delineated and refined with on-the-ground observations by team members (FSH 2509.13.22). The Handbook lists potential problems, techniques for inventory of postfire conditions, and treatments eligible for funding (FSH 2509.13.25). There is no mandate that any particular rehabilitation treatment be used. Seeding has evolved to be one of the more frequently used tools because it can quickly treat large land areas at a lower cost than many other options. Where controversial issues are involved, such as proposed actions within Wilderness areas, consultation with concerned groups is recommended (Putnam, 1985). Public involvement will be discussed in more detail below.

Development of the Rehabilitation Prescription

After the assessment of rehabilitation needs, the team develops
several treatment alternatives, including a no action alternative. Three separate indices are developed to assess the economic, environmental, and sociopolitical values of implementing a rehabilitation project, which are then summarized in an overall cost-effectiveness index (FSH 2509.13.30). The cost/benefit index is derived from quantified and discounted costs of the project and expected benefits over the life of the project. The present net value of treatment benefits is also calculated. Sullivan et al (1987) criticize this approach to assessment of economic efficiency for failing to consider the probability of treatment success. If rehabilitation is less than fully successful, costs remain the same but projected treatment benefits are reduced. The expected value of benefits could be adjusted by weighting according to the probability of treatment success (Ibid.). These risks would of course be estimates, as are most of the figures for resource values. It is difficult to assess the sensitivity of this economic analysis to quantification of resource values, but inclusion of a risk analysis would likely be an improvement. Evaluation of the accuracy of benefit/cost estimates is not included in the Handbook chapter on monitoring project effectiveness. Such an amendment to the Handbook should be considered.

In addition to the economic efficiency criterion, overall cost-effectiveness must weigh environmental and social indices. These indices are derived from the difference between subjectively weighted benefits with and without the proposed rehabilitation treatment. While construction of the latter two indices is highly subjective, it may also benefit from incorporation of a risk factor. Again, evaluation of the
accuracy of these estimates after project implementation would improve future projects. The three indices of economic, environmental, and social benefits are considered together to determine the overall cost-effectiveness index, and the final 'go' or 'no go' decision. At least two of the indices must be significant or favorable for a 'go' decision, except for the condition in which environmental benefits are significant while social and economic indices are not significant or marginal, which also yields a 'go' decision (FSH 2509 13.37). Once the alternatives have been prepared, the Forest Supervisor selects the most cost-effective alternative, requests funds, and oversees implementation.

While the handbook provides direction that cost-effectiveness should be calculated for all alternatives, including the no action alternative, the 1979 administrative review found that often only a single calculation was made after the selection of the preferred alternative (Tracy, 1980). The suggested remedy was additional training for rehabilitation teams to stress the need for documented analysis and justification of land treatments.

Funding Rehabilitation

Use of FFF funds for wildfire rehabilitation was first authorized in 1975, with separate accounts available for rehabilitation of suppression damages and of fire damages to watersheds. Several stipulations in the official directives aim to ensure that only true emergency hazard situations and effective treatments that are environmentally and socially acceptable are funded (FSM 2523.03, FSH 2509.13.03). For example, requests for funds will likely be denied if a
seed mixture contains species that are valued primarily for forage rather than soil protection, or if a seeding mix exceeds a certain cost per pound (Schmidt, 1989). The Handbook also stipulates that no administrative studies or research on burned areas shall be financed with FFF funds (2509.13.63).

Monitoring and Evaluation

As with projects carried out under Forest Plans, rehabilitation treatments should be monitored and evaluated to see that they are carried out as intended, that the results meet the objectives of the treatment, and that the initial data and assumptions are valid. In the Manual, this point echoes through the line of authority: "Regional Foresters shall: establish Regional procedures for evaluating the effectiveness of applied rehabilitation measures" (FSM 2523.04b). "Forest Supervisors shall: conduct post-treatment evaluation of projects to determine if emergency rehabilitation measures have met the planned objectives..." (FSM 2523.04c). "District Rangers shall: monitor conditions on burned-over areas..." (FSM 2523.04d).

In 1979, the Washington Office (WO) conducted a review of the Emergency Burn Rehabilitation Program, motivated by "concern for the quality and effectiveness of the emergency rehabilitation measures installed on-the-ground (Tracy, 1980). Among four "significant areas needing improvement" was the following:

Regions have not supplemented National direction in Manuals and Handbooks. There is a need for Regions to develop Manual supplements covering procedures and standards for evaluating the effectiveness of rehabilitation measures, and to issue instructions to comply with established direction regarding accomplishment reporting (Tracy, 1980).
The response to this finding was a "planned action" to amend the Manual and Handbook "to require the Burn Area Survey team to devise an evaluation and monitoring plan as part of the rehabilitation plan..." (Ibid.). Ten years later, such an amendment has not yet been made.

Current direction in the Handbook is general and permissive. At the discretion of the Regional Forester or Forest Supervisor, project analyses are encouraged to document the rehabilitation activity, study its effectiveness, and provide information to improve future projects (FSH 2509.13.62). According to the Handbook, "one day with a competent team should generally be sufficient for covering any of these phases of analysis" (Ibid.). A single day would hardly allow for a credible, quantified evaluation of rehabilitation effectiveness alone, unless such a team was extremely numerous and well organized. In the area of land treatments, "conditions that should be monitored include ... quality and quantity of water leaving the burned area and location and causes of problems, rate of recovery of vegetation, and effects of resource utilization and restoration activities and emergency rehabilitation activities on each other" (FSH 2509.13.61). In directions for prescribing vegetation treatment, the Handbook suggests that treatment "should prescribe: ...a post-treatment maintenance schedule — plan for it, fund it, and do it; and At least one method of post-project evaluation to be conducted on a specific date" (FSH 2509.13.25.13). The intent here is clear, but these minimal suggestions likely result merely in documentation to satisfy procedural requirements, and insufficient information to evaluate treatment effectiveness or confirm underlying assumptions.
Managers at any level of authority want to be good stewards of the land, and they understand the need for feedback on their activities. Several reasons have been suggested why getting this information has failed to be a priority. The most obvious obstacle is finding the money to conduct the monitoring. The 1979 review found that there was confusion about who was responsible for monitoring rehabilitation projects (Tracy, 1980). The perspective of the Washington office is clearly that responsibility resides in each Region or Forest. Few if any Forest Plans specifically mention fire rehabilitation as part of their planned monitoring of watershed disturbances, so such monitoring is not included as a hard target in Plan implementation. The next generation of Forest Planning may present an opportunity to include stronger direction to evaluate incidental land treatments. This would give rehabilitation monitoring a higher priority in annual budget allocations. Until that time, funds for monitoring must come from the several resource functions on a Forest that could use this information. This approach requires leadership to 'sell' the value of the ecological information to be gained to the silviculturists, wildlife biologists, fire management staff, range conservationists, soil scientists and hydrologists in the office (Reinig, pers comm., 1990). Another approach is to treat rehabilitation monitoring as a necessary administrative cost of revenue producing fire salvage logging projects (Ibid.).

Another potential source of monitoring and evaluation money would be the same FFF fund that enables the rehabilitation. This could be an additional fixed percentage of the rehabilitation request, or a capped amount, or an amount tailored to a monitoring proposal attached to the
request. While it seems logical that information feedback should be an integral part of the whole rehabilitation program, there is little optimism that funding authority will be expanded in this direction (Silverman, pers. comm., 1990).

Information via Research

There are two other major sources for fire effects and rehabilitation information. The land management branch of the Forest Service is supported by a research division. Scientists at the Forest and Range Experiment Stations respond to research needs identified by the National Forests and State and Private Forestry Cooperators. Research priorities are selected for broad applications. Standards for defensibility of data are high, more so than would be cost-effective for land managers to meet. Research results and management applications are widely available through publications, and research scientists often cooperate in educational field trips to train personnel in the management branch.

The Intermountain Research Station in Boise, Idaho is currently investigating the effects of postfire grass seeding on soil stability, forest succession, and tree regeneration (Geier-Hayes, pers. comm., 1990). This project will promote more efficient use of grass seeding by improving the knowledge base on native plant interactions and site conditions where native cover may be inadequate to prevent soil loss (Ibid.). Initiated in 1989, this study is the first major investigation of the topic in the Northern Rockies. The study is limited in ecological amplitude, however, and cannot satisfy the need for similar
information on other soil types, ecosystems, and seeding treatments. While this study will provide a defensible and much needed baseline for understanding grass seeding effects and interactions with the native flora, validation and/or extension of these relationships elsewhere is still needed.

The Soil Conservation Service's Plant Materials Centers also conduct research with broad applications. The portion of their work that is relevant to wildfire rehabilitation is limited, though it could be expanded in the future. Current studies on species for soil stabilization on forested sites in Montana focus on roadcuts and logging disturbance (Hunter, pers comm., 1990). Physical displacement and compaction of soil by heavy machinery has effects on soil and vegetation processes that are different from the effects of fire. The Plant Materials Center would be the appropriate facility to conduct agronomic trials involving competitive interactions of grass species in seed mixes, and different rates of application. From an agronomist's perspective, a thousand acres of forest is too big a plot to test a seed mix, yet this is routinely the case. Unfortunately, relationships apparent in lowland study plots may or may not hold true under field conditions in the forests. The assumptions of grass seed performance based on lab trials should be validated by testing the actual applications.

Information via Public Involvement

The National Environmental Policy Act (NEPA) provides for another source of information for land management decisions. The public. NEPA
intends "to help public officials make decisions that are based on understanding of environmental consequences, and take actions that protect, restore, and enhance the environment" (40 CFR 1500.1(b)).

Extensive procedural requirements for exchange of information with the public have been established to this end. The status of Emergency Burn Rehabilitation under NEPA is not entirely clear. There may be valid arguments for opening up the topic to public debate, but because this action appears discretionary, the question should turn on what may be gained by so doing.

The Emergency Burn Rehabilitation program meets several of the basic criteria that trigger NEPA review. It is a program of regular procedures affecting federal land, it involves irretrievable commitments of resources, and it has the potential for significantly affecting the human environment (40 CFR 1508.18(a), 1508.18(b)(3)). But meeting NEPA procedural requirements is not feasible for emergency actions and is exempted (36 CFR 1506.11). Public involvement procedures for environmental impact analysis include several minimum time periods for public notification, comment, and analysis and response to comments.

Postfire rehabilitation treatments have better chances for success if they are implemented before the area receives major precipitation. Direction in the Emergency Burn Rehabilitation Handbook is strictly limited to treatment of immediate needs so that this criterion is met. Similarities to NEPA procedures such as the preparation and analysis of alternatives are coincidental, this being a familiar approach to planning.

While each rehabilitation project is itself exempt from NEPA, the
overall program may not be. There are provisions in the regulations for public involvement in the formulation of Manual directives (36 CFR 216), and for Programmatic environmental impact statements (40 CFR 1508.18). The need for formal public review of proposed Manual directives is based on the level of public interest or controversy as determined by the responsible agency official (36 CFR 216.4). Evidence of any public review of the Service-wide Manual was not immediately available, and no supplements to the Manual have been prepared for Region One. The regulations do not provide any guidance for directives on topics that have become more controversial since they were originally issued, except that the public may make informal review and comment at any time (36 CFR 216.3(d)).

No programmatic Environmental Analysis document has been prepared for the Emergency Burn Rehabilitation program by the Forest Service (Putnam, pers. comm., 1990). The Soil Conservation Service (SCS) completed an Environmental Impact Statement (EIS) for their nationwide program of Emergency Assistance after natural disasters authorized by the Flood Control Act (Soil Conservation Service, 1975). In 1978, subsequent to Executive Order 12291, a formal regulatory review of the SCS Emergency Watershed Protection program and new rulemaking for implementation was initiated (46 FR 56574). Several public comments led to modification and clarification of the proposed rules (Ibid.). Among other things, the new rules provide for inclusion of a risk factor in assessment of cost-effectiveness of emergency measures (46 FR 56578).

It is not clear why a Programmatic Environmental Assessment on Emergency Burn Rehabilitation was not made at the time the program was
first authorized. The semantics of the regulations leave this question open:

EIS’s may be prepared, and are sometimes required, for broad federal actions such as the adoption of new agency programs or regulations (1508.18). Agencies shall prepare statements on broad actions so that they are relevant to policy and are timed to coincide with meaningful points in agency planning and decisionmaking (40 CFR 1502.4(b)).

Annual reauthorization of the PFF fund by Congress has apparently not been considered a "meaningful point" for review. The magnitude and expense of rehabilitation projects in the aftermath of the extremely busy fire seasons of 1987 and 1988 may yet trigger a new programmatic review.

What could a programmatic review accomplish? A formal and systematic review would have to acknowledge the many technical and administrative questions that have emerged over the life of the program. Despite the expense of going through the NEPA process, it would not necessarily provide answers to these questions. Public policy debates revolve around both facts and values. In the case of emergency burn rehabilitation, the vast majority of the existing factual information is already available to the agency. The substantial gaps and obstacles to improving the knowledge base would have to be acknowledged. The ways that the information is manipulated to calculate risks and estimate public values are less clear, and would be a focus of controversy. Such scrutiny might very well result in improved administrative procedures.

Public policy decisionmaking is often more a matter of process than of outcome (Wondolleck, 1988). This notion is supported by volumes of environmental case law, and also by research in the psychology of
risk assessment. Several anecdotal references have been made regarding potential liability of the Forest Service for failing to control flood damages from burned watersheds (MacDonald, 1989; Ruby 1989). Potential legal challenges would be less threatening if the agency had assurance of the soundness of its program, which could be achieved through a formal review. A sense of participation in the decision process by the public often engenders support for the outcome (Bloomfield, 1985). Continued public education efforts will be needed to increase the awareness of natural hazards in floodplain areas, so that people who build there may understand the voluntary nature of taking this risk, and that damages are only partially attributable to watershed land management. Public involvement in a programmatic review would present an opportunity for this educational outreach.

Despite these arguments, public involvement may not be the panacea desired. The special circumstances of emergency burn rehabilitation don't fit this model, as these actions are too infrequent and unpredictable for substantial participation of those that might eventually be personally involved. Author John McPhee (1989) chronicles an extreme of blind faith in technology in his recent work The Control of Nature. In the suburbs downstream from the fire-prone and dramatically erosive Transverse Range in California, laissez-faire attitudes prevail. In this case there is strong direct evidence of substantial risk, yet even the well-educated fail to take personal responsibility or involve themselves in the public dialogue. Public involvement is unlikely to solve the need for better technical information. It may, however, cause enough attention to be directed to
the issue to result in administrative changes that will indirectly produce the information.

Monitoring rehabilitation treatments need not be highly sophisticated to produce useful data. Several recent papers have summarized the nature of the technical information needed (Barro and Conard, 1987; USDA, 1988). A review of the applicability of these questions could prioritize the situations and applications most in need of further evaluation in the Northern Region. Standardized methodology could be used for such projects throughout the region so that results would be comparable and quickly comprehensible by other personnel. A clearinghouse for information and methodology in the Regional Office would facilitate communication of results of ongoing monitoring studies to all rehabilitation team members. A regional coordinator would also be a logical contact person for assistance to District personnel in experimental design and sample stratification.

In summary, there are several refinements that could be made to the existing directives on emergency burn rehabilitation that would result in more successful and more efficient treatments. Emergency land treatments as a program must not escape the standard of review that is applied to more routine operations. Risk assessments must be made integral to the development of treatment alternatives, and later validated by monitoring data. The single most valuable change in policy would be to implement a systematic procedure by which each rehabilitation treatment would be considered as an opportunity to improve the base of information for future applications. Those projects
involving unusual or poorly understood treatments or sites types should be prioritized for study. There are several possible sources of funds to follow through with monitoring and evaluation, but stronger language in Forest Plans and/or Manual and Handbook directives will be needed to assure that such funds are secured. While the research division can supply extremely valuable baseline and conceptual studies, more site-specific validation monitoring will be required to further refine treatment applications. These studies do not demand a high degree of sophistication in methods, but rather conscientious attention to objectives in experimental design. Boyd, (1985) makes a case for operational research in a different forest management situation, silvicultural weed control. In this case also, there is so much variability in actual applications of the practice, and so many gaps in formal research results, that operational evaluation is essential. Finally, a formal review of the emergency rehabilitation program may be appropriate to force action on administrative shortcomings.
CHAPTER 4.

Summary and Recommendations

This technical review, case study, and policy analysis of the practice of seeding grass for emergency rehabilitation of wildfires in the Northern Rockies identifies many needs for improvement. Basic research is still needed to demonstrate the effectiveness of the technique in general, and to distribute high quality technical information to potential users. Site specific information is also needed to address the wide range of variability in natural conditions that affect rehabilitation success.

Operational monitoring of rehabilitation treatments could meet many of the information needs if it were conducted with appropriate study designs. Significant questions remain about the ability of seeded grass to establish on burns, and whether a substantial reduction in erosion or risk of erosion is actually gained by seeding grass. If grass does establish, which it often does, how does it compete with native plant species recovery, and how does it influence long term plant community composition? Can grass seeding directly or indirectly influence the productive capacity of forested landscapes? How do individual introduced grass species perform under field conditions? As yet, there are few answers to these questions in the Northern Region.

A case study of vegetation monitoring on the 1988 Canyon Creek Fire provides an overview of the potential range of vegetative response...
of both native and seeded species across a wide range of site conditions. Despite excellent growing conditions in the first postfire year, establishment and growth of seeded grasses was highly variable. Characterization of this variability will help in the design of future studies. Evaluation of paired treatment and control plots within habitat types or fire groups is recommended.

A critical review of Forest Service policy and procedures for implementation and administration of emergency fire rehabilitation suggests changes at several levels of authority. At the Washington office, Handbok directives need to be refined so that the risk of treatment failure is included in the cost-effectiveness analysis. Suggestions for monitoring and evaluation of rehabilitation projects must be strengthened to the point that they are taken seriously. The prohibition against using any Emergency Rehabilitation funds for administrative studies should be closely examined.

At the Regional level, coordination and assistance needs to be provided to the Forests to improve the level and usefulness of feedback from monitoring studies. Coordination and consistency in methodology and design would help distribute the burden of improving the knowledge base across the Forests. The Regional office should provide guidance and support to fill information gaps that are not covered formally by either Forest Plans or the Research branch of the Forest Service. While neither the specific sites nor timing of burn rehabilitation projects can be predicted, a programmatic review of both the policy and its implementation would be appropriate at the Regional level.
At the Forest level, monitoring of management practices needs to evolve beyond simple documentation of results to evaluation of effectiveness and testing and validation of underlying concepts. Monitoring projects that assess multiple resource values must be able to get financing, based on their merits, from each of the various functions (i.e. timber, range, wildlife, soil and water) that stand to benefit from this information. Amendments to Forest Plans might be necessary to make monitoring of land management practices a harder target.
Appendix A

Descriptions of Landtypes Sampled on the Canyon Creek Fire

Landtypes are designed to describe soils, potential climax vegetation, and landforms found on the Forest and their suitability for commonly applied land management practices (Holdorf, 1981). Landtype maps are developed primarily from aerial photo interpretation of landforms, with ground truthing to correlate data from soil and vegetation sampling. Of necessity, mapping units are broad, and contain inclusions of other landtypes that may have differing management implications. Common inclusions in mapping units are small areas of dissimilar parent rock, or microclimatic differences in plant habitat. The resolution of mapping is intended to be appropriate for many typical Forest planning functions, with the understanding that site specific projects will require field verification for accuracy. Wilderness lands are subject to only a few management practices (mainly fire and watershed planning), so a coarser scale in mapping and description is adequate (Holdorf et al, 1980). The landtype recorded for each fire monitoring plot was determined simply by reading it off the map. As no key to the landtypes is available, no attempt was made to verify the accuracy of the landtype maps. The main constituent landforms, Habitat Types, Soil classification, slopes, and geology are given below for the eight landtypes sampled in the front country, and the five landtype associations sampled in Wilderness plots.
In addition, some of the interpretations of soil stability, vegetative productivity, and suitability reforestation are given for each landtype. The Soil Resource Inventory (Holdorf, 1981) lists several interpretations of onsite erosion hazards and offsite sediment pollution hazards for each landtype, of which two are listed below.

Sediment delivery efficiency is a rating of the probability of eroded material becoming stream sediment, and is derived from slope steepness and drainage density. Sediment hazard after fire is derived from several factors: time required for establishment of native vegetation, probability of heat induced water repellency in topsoil, probability of slope mass failure due to loss of plant roots and reduced evapotranspiration, and probability of accelerated dry soil creep due to removal of canopy shade. Forage productivity in clearcuts and burns is an estimate in pounds per acre per year of forage palatable to livestock, elk, and deer. No clipping studies were available to support these estimates. Suitability for reforestation was rated as limitations to full stocking on a clearcut or burn within five years, based in plant moisture stress, short growing seasons, or competition from understory vegetation. This does not include possible rodent or livestock damage, or lack of a tree seed source. The descriptions below are paraphrased from Holdorf, 1981, and Holdorf et al, 1980.

Landtype 18. Steep west-facing slopes, Scree Habitat Types, Lithic cryorthent soils, 40-60% slopes on limestone. Sediment delivery efficiency is low, postfire sediment yield is low, productivity is 400 lbs browse, 100 lbs herbaceous; slight limitations to reforestation by plant moisture stress.
Landtype 21. Glacial drift deposits, Subalpine fir / twinflower Habitat Types, Andic Cryochrept soils, 10-25% slopes on undifferentiated parent materials. Sediment delivery efficiency is low, postfire sediment yield is low, productivity is 300 lbs browse, 75 lbs herbaceous; No limitations to reforestation.

Landtype 21A. Steep, drift plastered glacial trough walls, Subalpine fir / twinflower Habitat Types, Andic Cryochrept soils, 25-40% slopes on sandstone or shale. Sediment delivery efficiency is low, postfire sediment yield is low, productivity is 300 lbs browse, 75 lbs herbaceous; No limitations to reforestation.

Landtype 23A. Steep, drift plastered glacial trough walls, Douglas fir / snowberry and Douglas fir / pinegrass Habitat Types, Typic Cryoboralf soils, 25-40% slopes on sandstone or shale. Sediment delivery efficiency is low, postfire sediment yield is low, productivity is 300 lbs browse, 300 lbs herbaceous; Slight limitations on reforestation due to plant moisture stress.

Landtype 71A. Steep valley sideslopes, Douglas fir / pinegrass and Douglas fir / twinflower Habitat Types, Typic and Andic Cryochrept soils, 25-60% slopes on sandstone and shale. Sediment delivery efficiency is low, postfire sediment yield is low, productivity is 150 lbs browse, 450 lbs herbaceous; no limitations to reforestation.

Landtype 161. Low relief ridges and slopes, Rough fescue / Idaho fescue and Limber pine / Idaho fescue Habitat Types, Argic cryoborolls to Typic cryochrept soils, 10-40% slopes on sandstone and shale. Sediment delivery efficiency is low, postfire sediment yield is low, productivity is 225 lbs browse, 600 lbs herbaceous; reforestation limits not applicable.

Landtype 183. Very steep peaks and upper slopes, Scree and Subalpine
fir / Whitebark pine Habitat Types. Rockland to Typic cryochrept soils, 60% and steeper slopes on non-carbonate rock. Sediment delivery efficiency is low, postfire sediment yield is moderate, productivity is 50 lbs browse, 20 lbs herbaceous; Slight limitations to reforestation due to high elevation, short growing season, and slow plant succession.

Landtype 202. Steep rocklands. Scree Habitat Types, no soil development, 60% and steeper slopes on limestone. Sediment delivery efficiency is low, postfire sediment yield is low, productivity is 50 lbs browse, 10 lbs herbaceous; reforestation limitations are not applicable.

Landtype Association III. Glacial moraine in valley floors and low relief rolling hills, forested with Spruce / dwarf huckleberry and Subalpine fir / dwarf huckleberry Habitat Types, with Subalpine fir / twinfower H.T. in frost pockets and Douglas fir / pinegrass H.T. on drier south aspect slopes, loamy soils developing in glacial drift or sandy or gravelly glacial outwash. Erosion rates are low and postfire vegetative recovery is rapid.

Landtype Association IIIA. Steep, forested lower valley sideslopes with a dense pattern of parallel, low order drainages, dominant Habitat Types are Subalpine fir / beargrass and Subalpine fir / rusty menziesia, andic cryochrept soil predominate, developing in slowly permeable glacial drift. Water erosion hazard is low, but susceptible to rotational slumping when undercut by a stream or when denuded by fire.

Landtype Association VB. Steep, smooth, forested residual slopes, dominant Habitat Types are Subalpine fir/ beargrass and Subalpine fir/ rusty menziesia, with Douglas fir H.T.'s on lower south slopes, soils are predominantly andic cryochrepts developing in stony loamy weathered bedrock. Erosion hazards are low and postfire vegetative recovery is moderate.
Landtype Association VE. Steep, warm aspect residual slopes with mixed forest and grassland, with Subalpine fir / beargrass and Subalpine fir / blue huckleberry Habitat Types supporting seral lodgepole pine, and Douglas fir / pinegrass, Douglas fir/ Idaho fescue, and Rough fescue/ Idaho fescue H.T.'s in open areas. Typic and andic cryocrepts support forested sites, and typic and lithic cryoborolls support open Douglas fir forests and grasslands. Erosion hazards are low and postfire vegetative recovery is slow.

Landtype Association VIII. Very steep, forested, warm aspect breaklands with frequent rock outcrops, with Subalpine fir series Habitat Types supporting seral lodgepole pine and open forests on Douglas fir/ Idaho fescue, Douglas fir/ pinegrass, or Douglas fir/ snowberry H.T. sites. Typic ustochrept and typic cryochrept soils develop in very gravely or stony colluvium, and are highly permeable. Water erosion hazards are low, but dry creep erosion hazard is moderate. Droughty soils limit postfire vegetative recovery.
APPENDIX B

Fire Group Descriptions

The Forest Habitat Types of Montana (Pfister et al., 1977) classifies sites with common associations of trees and understory plants according successional patterns and environmental gradients. Fischer and Clayton (1983) examined the fire ecology of forest Habitat Types occurring east of the Continental Divide in Montana and grouped them by similarity of fire response. Postfire plant succession varies with the responses of major tree species to fire, as well as the commonly associated understory species and the typical frequency and intensity of fires on those sites. The brief descriptions below are paraphrased from Fire Ecology of Montana Forest Habitat Types East of the Continental Divide (Fischer and Clayton, 1983). Seven of the twelve Fire Groups were sampled in the Canyon Creek Fire.

Fire Group 0. Miscellaneous special habitats such as Scree, forested rockland, aspen or alder groves, wet meadows and grassy balds. These types generally act as firebreaks during summer burning season.

Fire Group 1. Dry Limber Pine Habitat Types. Fuel loads are light and discontinuous, fire frequencies are low, but wind-driven stand-replacement fires can revert the sites back to grassland for a long time. Spring burning can stimulate range productivity and check conifer encroachment.
Fire Group 2. Warm, dry ponderosa pine Habitat Types. Existing as fire-maintained grasslands, open pine stands, or mixed-age ponderosa without Douglas fir encroachment. Natural fire frequencies are high, and act to maintain grasslands, open up pine stands, or create seedbeds for pine regeneration.

Fire Group 3. Warm, moist ponderosa Habitat Types. These stands are often dense, stagnant thickets of ponderosa saplings. High natural fire frequencies maintained open, parklike ponderosa stands, but fire in older, stagnant, multistoried stands often crowns out and replaces the stand.

Fire Group 4. Warm, dry, Douglas fir Habitat Types. Cover is often Ponderosa pine stands maintained by frequent low intensity fires, but which will be replaced by the more shade-tolerant Douglas fir in the absence of fire. Ladder fuels in multistoried mixed stands often lead to stand replacing crown fires. Above the cold limits of ponderosa, succession moves from shrubs and herbs to open Douglas fir forests, to closed, multistoried Douglas fir.

Fire Group 5. Cool, dry Douglas fir Habitat Types. Stands are typically open Douglas fir, but dense Douglas fir understories may develop in the absence of fire. Fuel loads are light and undergrowth is sparse. Regeneration is vulnerable to competition on droughty sites.

Fire Group 6. Moist Douglas fir Habitat Types. These sites will support substantial amounts of Douglas fir even with periodic fire. Stands tend to be overstocked and fuel loading can be heavy, including ladder fuels, with the exception of lodgepole stands. Natural fire frequencies either thinned or replaced stands. Periodic light fire in open stands maintains excellent wildlife habitat.
Fire Group 7. Cool Habitat Types usually dominated by lodgepole pine, where lodgepole may be a persistent dominant species or where seral lodgepole stands are renewed by fire. Holocausitic fires can be expected as fuels build up from suppression mortality or pine beetle epidemics. Postfire dominance of understory plants is short-lived unless reburn halts the vigorous and well-stocked lodgepole regeneration.

Fire Group 8. Dry, lower subalpine Habitat Types. Often in Spruce or Subalpine fir series, but occupied mainly by mixed stands including Douglas fir and lodgepole pine. Severe fires lead to lodgepole dominance over large areas, while moderate fires tend to favor Douglas fir. Fire in the relatively deep duff under dry conditions can cause substantial mortality to understory plants.

Fire Group 9. Moist or wet lower subalpine Habitat Types. Spruce is often a major component of seral stands. Fires are infrequent but with severe and long-lasting effects. Undergrowth is abundant, and duff layers are often deep. Under dry conditions, even surface fires can cause substantial tree mortality from cambial heating.

Fire Group 10. Cold, moist upper subalpine and timberline Habitat Types. Undergrowth is generally sparse and discontinuous. Fuel loads of large-diameter wood may be heavy, but short fire seasons and lack of continuous fuels generally lead to infrequent, small fires. Fire effects are severe and prolonged.

Fire Group 11. Moist grand fir, western red cedar, and western hemlock Habitat Types. These forest types are not found east of the Continental Divide.
### APPENDIX C

**MATRIX OF FLORISTIC SIMILARITY BETWEEN PLOTS**

<table>
<thead>
<tr>
<th>Fire Group #4</th>
<th>F.G. #5</th>
<th>F.G. #6</th>
<th>F.G. #7</th>
<th>F.G. #8</th>
<th>F.G. #9</th>
</tr>
</thead>
<tbody>
<tr>
<td>plot # 06</td>
<td>16 33</td>
<td>62 64</td>
<td>67 69</td>
<td>14 46</td>
<td>52 86</td>
</tr>
<tr>
<td>plot # 04</td>
<td>68 58</td>
<td>59 80</td>
<td>85 88</td>
<td>66 75</td>
<td></td>
</tr>
<tr>
<td>plot # 12</td>
<td>71 66</td>
<td>78 69</td>
<td>75 66</td>
<td>84 67</td>
<td></td>
</tr>
<tr>
<td>plot # 19</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Figures in this matrix represent the similarity of the native floristic composition of plot pairs, with low numbers indicating higher similarity. Jaccard’s similarity index is the ratio of the number of single occurrences of a plant species in a pair of plots to the total number of species in the two plots together. Similarity was calculated within Fire Groups of the 30 high intensity burn plots. The most similar plot pairs are in boldface type and are discussed in the text.
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