Postfire Seeding for Erosion Control: Effectiveness and Impacts on Native Plant Communities

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Abstract: *Large, high-severity wildfires remove vegetation cover and expose mineral soil, ofen causing erosion and runoff during postfire rain events to increase dramatically. Land-management agencies in the United States are required to assess site conditions after wildfire and, where necessary, implement emergency watershed rehabilitation measures to help stabilize soil; control movement of water, sediment, and debris; prevent permanent impairment of ecosystem structure and function; and mitigate significant threats to human health, safety, life, property, or downstream values. One of the most common postfire treatments is broadcast seeding of grasses, usually from aircraft. Non-native annual or perennial grasses typically are used to provide quick, temporary ground cover to hold soil in place until native plants are reestablished. Critics argue that seeded grasses compete with native vegetation and do not effectively reduce erosion. Few data exist on the effectiveness of erosion control; less than half of the studies I reviewed showed reduced sediment movement with seeding. In all vegetation types, successful growth of seeded grasses—enough to affect erosion—appears to displace native or naturalized species, including shrub and tree seedlings. Due to the competitiveness of seeded grasses, they are used to attempt suppression of noxious weeds in some postfire seeding operations. In burned sagebrush range, postfire seeding is frequently used to replace non-native cheatgrass (*Bromus tectorum*) with native or introduced bunchgrasses, with at least short-term success. In recent years, native species and sterile cereal grains have increasingly been used for seeding. Use of aerially applied straw mulch has increased as well, with the risk of weed introduction from contaminated bales. More research on the effectiveness and ecosystem impacts of these alternatives is needed.*

Key Words: annual ryegrass, burned area rehabilitation, cereal grains, grasses, mulch

Siembra Post-incendio para Control de Erosión: Efectividad e Impactos sobre Comunidades de Plantas Nativas

Resumen: *Los incendios grandes, de alta severidad, remueven la vegetación y exponen el suelo mineral, con lo que la erosión y el escurrimiento incrementan dramáticamente durante eventos de lluvia post-incendio. Las agencias de gestión de tierras en los Estados Unidos deben evaluar las condiciones del sitio después del incendio y, donde sea necesario, implementar medidas de emergencia para la rehabilitación de cuencas de agua para ayudar a estabilizar el suelo; el control del movimiento de agua, sedimentos y detritos; la prevención de la degradación permanente de la estructura y función del ecosistema y la mitigación de amenazas significativas a la salud, seguridad, vida y propiedades humanas y a valores río abajo. Uno de los tratamientos post-incendio más comunes es la siembra de pastos, usualmente desde aviones. Típicamente se utilizan pastos anuales o perennes no nativos para proporcionar cobertura rápida, temporal para mantener al suelo en su lugar hasta que se puedan reestablecer plantas nativas. Los críticos argumentan que los pastos sembrados compiten con la vegetación nativa y no reducen la erosión efectivamente. Existen escasos datos sobre efectividad de control de erosión; menos de la mitad de los estudios que revisé mostraron reducción de movilidad de sedimentos con la siembra. En todos los tipos de vegetación, el crecimiento exitoso de pastos sembrados — suficiente para afectar a la erosión—parece desplazar a especies nativas o naturalizadas, incluyendo arbustos y plántulas de árboles. Debido a la competitividad de pastos sembrados, son utilizados para intentar la supresión de hierbas nocivas en algunas operaciones de siembra post-incendio. La siembra post-incendio en áreas de artemisa se utiliza frecuentemente, exitosamente por lo menos en el corto plazo, para reemplazar* Bromus tectorum *no*

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nativos con pastos nativos o introducidos. En años recientes se ha incrementado el uso de especies nativas y de granos estériles de cereal para la siembra. Asimismo, también ha incrementado el uso de mantillo de paja aplicado aéreamente, con el riesgo de la introducción de hierbas en pacas contaminadas. Se requiere más investigación de la efectividad del control de erosión y de los impactos en el ecosistema de estas alternativas.

Palabras Clave: ballico anual, cereales, mantillo, pastos, rehabilitación de área quemada

Introduction

Fire is a natural disturbance in most western North American ecosystems. The success of fire suppression during the twentieth century, however, has created fuel conditions in many plant communities that now result in fires of greater intensity and extent than would have occurred historically (Norris 1990; Agee 1993). In mountainous areas, severe wildfires can render the postburn landscape susceptible to massive soil erosion, flooding, and downstream sedimentation with the onset of summer thunderstorms or heavy winter rains.

Fire consumes the protective vegetation and organic litter cover from hillsides, which can destabilize surface soils on steep slopes. During and immediately after a fire, surface erosion increases by raveling, or gravity sliding, as organic barriers to sediment movement have been incinerated and soil structure disrupted (Krammes 1960). With the beginning of summer or fall rains, soil erosion again increases as the denuded hillsides are exposed to raindrop impact and surface runoff. In addition, the production of a fire-induced, near-surface, water-repellent soil layer in some soil types may restrict soil water infiltration, further increasing runoff potential (DeBano 1981; DeBano et al. 1998). These impacts vary with the severity of the fire: the more severe the effects of the fire—from greater fireline intensity or longer burning duration caused by heavy fuel loads—the greater the erosion and runoff potential. In an unburned watershed with greater than 75% of the ground covered by vegetation or litter, erosion is low or negligible and only 2% or less of rainfall may become surface runoff. After a severe fire, with <10% ground cover remaining, surface runoff can increase over 70% and erosion by three orders of magnitude (Bailey & Copeland 1961). Storm peak flow can increase by up to 9600% after fire (Anderson et al. 1976).

The potential for extensive damage and expensive clean-up costs from increased postfire runoff and erosion can be enormous, especially when large fires occur near the wildland-urban interface. Land-management agencies such as the U.S. Department of Agriculture Forest Service (USFS), the U.S. Bureau of Land Management, and the National Park Service are required to assess site conditions following wildfire and, where necessary, prescribe emergency watershed-rehabilitation measures to (1) help stabilize soil; (2) control water, sediment, and debris movement; (3) prevent permanent impairment of ecosystem structure and function; and (4) mitigate significant threats to human health, safety, life, property or downstream values (U.S. Department of Agriculture Forest Service 2000). Each year millions of dollars are spent on emergency postfire rehabilitation treatments (Robichaud et al. 2000). Rehabilitation treatments are short-term measures, however, designed to ameliorate the immediate emergency until natural vegetation regeneration stabilizes the burn area. These measures do not provide long-term ecosystem restoration. For the long term, land-management agencies take other actions, such as tree planting.

The most widely used postfire rehabilitation treatment is grass seeding, typically annuals or short-lived perennials applied from aircraft (Robichaud et al. 2000). Quick vegetation establishment is regarded as the most cost-effective method to promote rapid infiltration of water and to keep soil on hillsides, out of channels and downstream areas (Rice et al. 1965; Miles et al. 1989). Sediment production from burned or otherwise disturbed sites is inversely related to vegetative cover, with minimum erosion noted when plant cover was 60–70% (Noble 1965; Orr 1970), making vegetation enhancement a logical practice for reducing erosion at its source.

Brief History of Postfire Seeding in the Western United States

Foresters in southern California seeded burned-over chaparral slopes with native shrubs in the 1920s to try to reduce postfire erosion. Finding that shrub seeds germinated no earlier than natural regeneration, they experimented with faster-growing non-native herbaceous species, such as Mediterranean mustards (*Brassica* L. spp.; Gleason 1947; Department of Forester and Fire Warden 1985). Land-management agencies also tested various grasses, and by the 1950s they settled on annual ryegrass (*Lolium multiflorum* Lam., native to Europe and Asia) as the most effective and economical choice for burned-over California shrublands (Barro & Conard 1987). Ryegrass has also been widely applied on forestland burns in California (e.g., Griffin 1982), southern Oregon (e.g., Amaranthus et al. 1993), and elsewhere.

Seeding after fire for range improvement has been common since at least the 1930s, with the intent to gain useful products from land that would not yield timber for many years (Christ 1934; Friedrich 1947*a*; McClure 1956). Various species of predominantly non-native pasture grasses and forbs were tested for establishment, persistence, and forage production, in burned areas and on rangeland degraded by decades of overgrazing (e.g., Forsling 1931; Friedrich 1947*b*; Evanko 1955; Hull & Johnson 1955; Klock et al. 1975; Evans & Young 1978). Seed mixes were refined for particular areas as germination and establishment success were evaluated. Most mixes contain annual grasses to provide quick cover, perennial grasses to establish long-term protection, and often legumes to add nitrogen to the soil (Klock et al. 1975; Ratliff & McDonald 1987).

In recent years, federal land-management agencies have been directed to use native species, when practical, for revegetation projects, including postfire rehabilitation (Richards et al. 1998). However, the difficulty of acquiring enough native seed for large fires, combined with concern about the ecological impacts of using nonlocal genotypes of native species, has limited the amount of postfire seeding actually done with native plants (Richards et al. 1998; Goodrich & Rooks 1999; Robichaud et al. 2000). On some burn areas, native species are seeded only on firelines (bulldozer scrapes or handlines made in the effort to control and contain the fire), where the ecological damage is most severe and the need to establish persistent cover is great but the total acreage in need of seeding is low. Cereal grains or pasture grasses are then used on the more extensive hillslope areas in need of protection (various USFS reports on burned areas).

Seeding Controversy: Effectiveness versus Ecological Impacts

The practice of postfire seeding has had its critics. Postfire mustard seeding was discontinued in the 1950s in southern California, partly because the mustard species proved to be troublesome weeds in downslope orchards and farm fields (Barro & Conard 1987). In the late 1970s and 1980s, the impacts and effectiveness of ryegrass seeding in burned-over California chaparral, in particular, came into question (Conrad 1979; Gautier 1983). Concern focused on chaparral ecosystems because a specialized annual flora there takes advantage of the light, space, and soil nutrients available immediately after fire (Sweeney 1956; Keeley et al. 1981). In addition, some dominant chaparral shrub species regenerate after fire only from seed (Sampson 1944; Keeley 1991), so competition from seeded grasses could have long-term impacts on plant community composition (Barro & Conard 1987). Silviculturists have long been aware, as well, that successful grass establishment can interfere with the survival of tree seedlings (e.g., Friedrich 1947*a*; Elliott & White 1987; Ratliff & McDonald 1987; Van de Water 1998).

If grass seeding is effective at controlling erosion, some degree of negative ecosystem impact may be tolerable as a trade-off for retention of long-term site productivity and protection of downstream values. How well does postfire seeding reduce erosion and sedimentation? How seriously does seeded grass compete with native plants?

Evaluation of Erosion Control

In California, chaparral brush fields prone to fire occur at the wildland-urban interface, where the societal impacts of accelerated postfire erosion are enormous, as are the pressures to treat burned hillslopes with grass seed to protect life and property (Gibbons 1995). Consequently, southern California chaparral has been a focus of research on the effectiveness of erosion control. During the first year after a wildfire near San Diego, Gautier (1983) recorded 31% less sediment from plots seeded with annual ryegrass than from unseeded plots. The seeded plots had significantly greater plant cover as well—approximately double on average—than the unseeded plots in a year with above-average rainfall. In contrast, Taskey et al. (1989) found no significant difference in hillslope erosion between seeded and unseeded plots near San Luis Obispo in a year with average rainfall, despite higher average cover on the seeded plots (71% vs. 58% measured in late summer). Interestingly, 4.5 times greater dry erosion occurred during the summer on the seeded plots, which was attributed to the actions of pocket gophers attracted to seeded areas by the abundant ryegrass (Taskey et al. 1989).

Wohlgemuth et al. (1998) assessed hillslope sediment movement after four hot prescribed fires and one wind-driven wildfire on mixed chaparral sites in coastal southern California. Plots seeded with annual ryegrass had significantly greater total plant cover at only one site—about 5% compared with 3% cover (Beyers et al. 1998)—and there was no significant difference in measured erosion at any site during the first year after fire. At three sites, there was less sediment movement on seeded plots than on unseeded plots during later years after fire, but by that time erosion rates had decreased to prefire levels or lower (Wohlgemuth et al. 1998). This study has been criticized for using prescribed fires rather than wildfires to provide a baseline record of prefire erosion and vegetation because fire severity is intentionally lower in a prescribed burn and thus erosion would be less extreme (USFS rehabilitation team personnel, personal communication). The one wildfire site measured produced over 10 times as much sediment the first winter after fire than had a prescribed-fire site in the same area (Wohlgemuth et al. 1999). However, vegetation response was similar to the prescribed fires—no significant increase in cover due to seeding (Beyers et al. 1998)—and erosion was not significantly different between seeded and unseeded plots (Wohlgemuth et al. 1998).

Less research has documented the impacts of grass seeding on erosion in conifer forest ecosystems. In plots on a southern Oregon study site, 75–90% of the first winter sediment movement occurred by December, before the seeded annual ryegrass was well-established, resulting in no significant difference between seeded and unseeded plots, even though seeded plots had roughly twice the plant cover of unseeded plots by late spring (Amaranthus 1989). In the northern Sierra Nevada of California, no difference in total plant cover or erosion was detected between seeded and unseeded watersheds 2 years after fire (Roby 1989). The seeded watershed was treated with a mix of orchard grass (*Dactylis glomerata* L.), tall fescue (*Festuca arundinacea* Schreber), timothy (*Phleum pratense* L.), and slender wheatgrass (*Elymus trachycaulus* [Link] Shinn. spp. *trachycaulus*). In both studies, fertilizer was included with the grass seed, a common practice at the time. Although not measuring erosion, Geier-Hayes (1997) found that total plant cover did not differ between seeded and unseeded plots for 5 years after a fire in Idaho.

The effectiveness of a wide range of postfire emergency rehabilitation measures was reviewed by Robichaud et al. (2000). They collected USFS postfire rehabilitation needs assessments, internal monitoring reports, and published literature. In addition, they interviewed burned-area rehabilitation team leaders. Grass seeding was the most extensively studied of the treatments reviewed, but even for seeding Robichaud et al. (2000) found relatively few quantitative studies—published or unpublished—of the effectiveness of erosion control conducted in wildland burn situations. Measuring erosion is time-consuming and labor-intensive, requiring the installation of some kind of hillslope or small-channel collection structures, and few investigators have done it, particularly with replication. The internal monitoring reports reviewed did not include statistical comparison of erosion from seeded versus unseeded plots (Robichaud et al. 2000).

Among the USFS monitoring reports examined, only three attempted to quantify erosion from chaparral sites; one of those concluded that seeding reduced erosion (Robichaud et al. 2000), although statistical analysis was not done. Four of seven reports that quantified vegetation cover found greater cover on seeded sites, a measure often equated with seeding effectiveness based on known relationships between plant cover and sediment movement (e.g., Noble 1965). On conifer sites, two of four reports that measured erosion found less sediment movement on seeded plots the first year after fire, though again no statistical analysis was done. Three of four reports comparing measured plant cover found greater cover on seeded than unseeded sites. A number of the monitoring reports noted that reduction in erosion with seeding is often not apparent until the second year after fire, when dead grass from the first growing season provides protective mulch, but only three monitoring studies reported second-year erosion measurements (Robichaud et al. 2000). One of these noted less erosion on seeded plots, though no statistical tests were conducted on the results.

Table 1. Effectiveness ratings for aerial grass seeding provided by burned-area rehabilitation specialists of the U.S. Department of Agriculture Forest Service (USFS), based on particular projects, by USFS region.*

Region	*No. of replies*	*Excellent (%)*	*Good (%)*	*Fair (%)*	*Poor (%)*
1	8	62.5	12.5	12.5	12.5
2	6	33.3	33.3	0	33.3
3	16	6.3	18.7	37.5	37.5
4	11	63.6	18.2	0	18.2
5	32	3.0	34.4	43.8	18.8
6	10	40.0	40.0	20.0	0

**Percentages of replies in each rating class are shown. Regions: 1, northern (northern Idaho, Montana, North Dakota); 2, Rocky Mountain (Wyoming, South Dakota, Colorado, Nebraska); 3, southwestern (New Mexico, Arizona); 4, intermountain (southern Idaho, Nevada, Utah); 5, Pacific Southwest (California); 6, Pacific Northwest (Oregon, Washington). (Modified from Robichaud et al. 2000.)*

In interviews, rehabilitation team members ranked seeding highest among hillslope treatments for overall effectiveness, yet seeding was also the treatment most often cited as overused (applied but probably unnecessary) (Robichaud et al. 2000). Regional differences in opinion on the effectiveness of grass seeding were revealed. Interviewees from California, Arizona, and New Mexico rated seeding as "fair" or "poor" in effectiveness, based on past projects, far more often than interviewees in the Rocky Mountain states, Intermountain West, and Pacific Northwest, who tended to rate seeding "good" or "excellent" (Table 1). Robichaud et al. (2000) speculated that unpredictable summer monsoon (Southwest) or early winter (California) rains could result in less-reliable grass growth in these areas than in the other areas. An additional possibility is that the most damaging rains, against which seeding is supposed to protect, tend to occur soon after fire in California and the Southwest, before seeded grass has much chance to become established. In the other areas, seeding is done in the fall or early spring to protect hillslopes from high-intensity summer thunderstorms, which occur after the grass has had more time to grow and thus to produce cover that can stabilize the soil. Seeding is also done to protect against winter rains in the Pacific Northwest (e.g., Amaranthus 1989), but greater rainfall and more reliable production of cover by seeded grasses there may have resulted in better evaluations of success. Grass seeding was often considered successful in agency evaluations if the seeded species simply produced cover (Robichaud et al. 2000).

Effects on Native Vegetation

The ecological impacts of grass seeding have been investigated more thoroughly than have its effects on erosion. In southern California chaparral, significantly less cover or biomass of native and naturalized herbaceous species was found on plots with high annual ryegrass cover (>10%) in both aerially seeded areas and hand-seeded experimental sites, whether or not seeded plots had higher total plant cover than unseeded plots (Keeley et al. 1981; Gautier 1983; Nadkarni & Odion 1986; Taskey et al. 1989; Beyers et al. 1998). Conard et al. (1995) observed that ryegrass seeding appeared to reduce cover of native fire-followers more dramatically than overall herbaceous cover (which included many naturalized non-native species) in years when ryegrass cover was high on their southern California sites. Average species richness of native plants was also lower on ryegrass-seeded plots (Nadkarni & Odion 1986; Taskey et al. 1989; Beyers et al. 1994). Reduced abundance of fire-following herbaceous species on seeded sites in chaparral could result in a reduced seed bank available for growth after the next fire (Conard et al. 1995). Beyers et al. (1998) examined this possibility and found no difference in cover of the 10 most abundant native fire-followers between formerly seeded and unseeded plots after a short-interval reburn of one experimental site, but data were insufficient to determine the residual effects of seeding on less common species.

Reduced cover of native herbaceous species in seeded areas has also been observed in conifer ecosystems. Although seeded perennial grasses provided greater vegetation cover than natural regeneration during the first 3 years after a fire in Oregon, the growth of native annual forbs appeared to be suppressed during the second and third year (Anderson & Brooks 1975). Conard et al. (1991) noted a negative relationship between annual ryegrass cover and native herbaceous cover in the central Sierra Nevada two growing seasons after the Stanislaus Complex fires. Geier-Hayes (1997) measured significantly lower cover of native annual species on grass-seeded plots than on unseeded plots during the third growing season after fire, and herbaceous cover (annual and perennial) trended lower in other postfire years and on several conifer sites in central Idaho. Orchard grass made up most of the seeded cover in that study (over 80% of non-native species cover on most sites in most years; it was 66% of the seed mix). Plots in a seeded stand of grand fir (*Abies grandis* [Douglas] Lindley) had no more total plant cover than unseeded plots in the eastern Cascades (Schoennagel & Waller 1999); mean seeded cover totaled 16.1% and consisted of soft white winter wheat (*Triticum aestivum* L.) and slender wheatgrass. Native plant cover and species richness were significantly lower on the seeded plots, particularly for on-site and wind-blown colonizing species (Schoennagel & Waller 1999).

Of importance to long-term chaparral stand composition, significantly lower shrub seedling density occurred on seeded compared to unseeded plots in studies where seeding increased total plant cover or biomass (Gautier 1983; Nadkarni & Odion 1986; Taskey et al. 1989). Where seeded plots did not have greater cover than unseeded plots, shrub seedling density did not differ significantly (Beyers et al. 1998). Grass seeding after fire has been used deliberately to reduce shrub density for grazing improvement in central California rangelands. Ryegrass foliar density (essentially cover) of 30% during the first year after fire increased shrub seedling mortality, and 55% ryegrass cover was sufficient to reduce shrub seedling density to zero by the end of the summer (Schultz et al. 1955). Conard et al. (1995) speculated that total herbaceous competition, rather than the presence of seeded grass per se, determines the success of shrub seedlings in the postfire chaparral environment. Soil moisture is depleted more rapidly on sites with greater herbaceous plant biomass, reducing water available to shrub seedlings earlier in the growing season (Schultz et al. 1955). In chaparral stands where the dominant prefire shrub species were those that survive fire only as seeds, any reduction in shrub seedling density due to grass seeding could have long-term ecosystem repercussions. Stand dominance could change from shrub species that reproduce from seed to those that resprout after fire. No published studies have documented such an alteration, however.

The negative effects of grass competition on conifer seedling growth are well documented in the forestry literature (e.g., Pearson 1942; Larson & Schubert 1969; Elliott & White 1987). Results of field studies on aerially seeded sites in California showed low densities of pine (*Pinus* L.) seedlings on most plots with annual ryegrass cover of >40% (Griffin 1982; Conard et al. 1991). In the southern Cascades, high mortality of planted sugar pine (*Pinus lambertiana* Douglas) seedlings occurred during the first year after fire on plots with high annual ryegrass cover (49% when seedlings were planted, 85% by midsummer) (Amaranthus et al. 1993). However, a second batch of planted seedlings had greater survival during the second year after fire, when seeded plots contained only mulch from dead ryegrass but pine seedlings in unseeded plots faced competition from native shrubs (which were much less abundant in the seeded plots) (Amaranthus et al. 1993). Once conifer seedlings are well established, grass competition may be less detrimental to their growth than shrub competition (McDonald & Oliver 1984; McDonald 1986).

Conifer seedlings were half as abundant on seeded than on unseeded plots in the eastern Cascades. As noted above, there was no difference in total cover between the two treatments, but winter wheat and slender wheatgrass comprised 16% of the cover on seeded plots (Schoennagel & Waller 1999). This suggests that the seeded species were more vigorous competitors of tree seedlings than were native species during the first 2 years after fire. The stature and growth pattern of seeded grasses, relative to that of native species, may account for this impact. Elliott and White (1987) found that non-native orchard grass and crested wheatgrass (*Agrypyron desertorum* [Fischer] Schultes) significantly reduced the height and diameter growth of ponderosa pine (*Pinus ponderosa* Laws.) seedlings more than did blue grama (*Bouteloua* *gracilis* [Kunth] Griffiths) and squirreltail (*Elymus elymoides* [Raf.] Swezey), both native species, because the non-native grasses grew more vigorously early in the growing season and depleted soil moisture to a greater depth than the native grass species. Pine-seedling growth with the two native species was not significantly different from growth in denuded plots (Elliott & White 1987). Winter wheat would also be expected to grow early and vigorously (though Schoennagel and Waller [1999] did not evaluate this). This suggests that the grass species most likely to be effective for erosion control are also those most likely to reduce the establishment of conifer seedlings.

Recent Trends

Federal agencies are now encouraged to use native species for postfire rehabilitation seeding to the extent practical, but the cost and availability of seed limits their use (Richards et al. 1998; Goodrich & Rooks 1999; Robichaud et al. 2000). Native grasses are well known for their ability to suppress the growth of conifer seedlings, although some species may be less inhibitory than others (e.g., Pearson 1942; Larson & Schubert 1969). One USFS postfire rehabilitation team decided against the use of a well-adapted native grass for postfire seeding after taking into account the cost of using herbicides to suppress the grass when replanting tree seedlings (Griffith 1998). Seeding sterile cereal grains and cereal-grass hybrids that will grow for only one season has also increased in recent years (Robichaud et al. 2000). Sterile cereal grains would be expected to suppress some native regeneration during the first year after fire if they become established successfully, like other grasses. However, many studies of "traditional" seeded species, such as annual ryegrass and orchard grass, have found the greatest reductions in native species cover, relative to unseeded plots, during the second or even later years after fire (e.g., Anderson & Brooks 1976; Conard et al. 1991; Geier-Hayes 1997; Beyers et al. 1998). Will sterile cereals have fewer impacts on recovering ecosystems than species that grow for several years? Little research has been conducted on these species so far. Winter wheat contributed 11.9% average cover 2 years after fire and seeding on plots in the northeastern Cascades, Washington (Schoennagel & Waller 1999), implying that it successfully reseeded itself. Successful regeneration of cereal grains was reported by Robichaud et al. (2000) in areas where stands were disturbed by cattle grazing or salvage logging. Regreen, a sterile wheat-wheatgrass (*Elytrigia elongata* [Host] Nevski) hybrid, produced little cover (<1%) on a burned area in the southern Sierra Nevada (Beyers 2004). Most of the few plants that grew (0.5 plants/m^2 the first year) persisted into the second growing season after fire (0.4 plants/m^2), suggesting that Regreen could have extended ecosystem impacts if its initial establishment had been high. Results from additional research and monitoring conducted on recently burned areas seeded with sterile cereal grains are needed to better answer questions about their effectiveness and impacts.

Seeding to Prevent Undesirable Species or Noxious Weeds

The recognized ability of seeded grass to compete with other vegetation is used intentionally to displace undesirable species or noxious weeds after fires. As noted above, annual grasses have been used to suppress shrub seedlings after fire in order to increase grazing acreage in the foothills of the Sierra Nevada and Coast ranges in California (Schultz et al. 1955). Seeded annual grasses not only competitively exclude shrub seedlings, they will also carry fire after curing and can be burned to further reduce shrub regeneration. The risk of an early reburn in seeded areas also exists where seeding has been done for erosion control, with the potential to severely reduce regeneration of woody species (Zedler et al. 1983).

In the intermountain western United States, sagebrush rangelands are often seeded with perennial grasses, such as non-native crested wheatgrass (*A. cristatum* [L.] Gaertn., *A. desertorum*) or native wheatgrasses, and alfalfa (*Medicago sativa* L.) or other legumes are planted in an attempt to suppress non-native cheatgrass (*Bromus tectorum* L.) and/or medusahead (*Taeniatherum caput-medusae* [L.] Nevski) and provide more forage for livestock and wildlife (Evans & Young 1978; Goodrich & Rooks 1999; Pyke & McArthur 2002). Seeding must be done immediately after fire to effectively reduce the abundance of cheatgrass, and the seeded perennials must be protected from grazing if they are to establish themselves successfully (Evans & Young 1978). Goodrich and Rooks (1999) measured a higher frequency of the native perennial grass squirreltail in a burned and seeded pinyon-juniper site than in an unseeded site, as well as less cheatgrass and musk thistle (*Carduus nutans* L., a noxious weed) on the seeded site. On the other hand, Ratzlaff and Anderson (1995) found no difference in the amount of cheatgrass on seeded compared with unseeded plots in an Idaho sagebrush rangeland, where cheatgrass was not particularly abundant before fire. Establishment of seeded species was poor on their site because of low precipitation the first year after fire, and unseeded plots had higher cover than seeded plots, an effect they attributed to impacts of the rangeland drill used for seeding (Ratzlaff & Anderson 1995). As with chaparral and forested sites, it appears that rangeland seeding has had mixed success.

Impacts of Mulch

Another postfire rehabilitation treatment with the potential to affect natural ecosystems is straw mulch, which can introduce the seed of non-native plants. Straw is applied to far fewer acres of burned wildlands annually than is grass seeding, and its use has been largely confined to areas accessible by road (Robichaud et al. 2000), where invasive non-native plants may already be part of the vegetation. Mulch treatments are used to protect assets of particularly high value—such as roads, streams and reservoirs—because the technique is highly effective at reducing erosion (Bautista et al. 1996; Miles et al. 1989) but is generally labor-intensive and thus expensive to apply (Robichaud et al. 2000). Seed of the straw species or field weeds may be introduced with mulch. In the past this was not considered a problem, because the pasture grasses or cereals grown from the straw would just add to soil-holding ground cover (Robichaud et al. 2000).

Now, however, as part of programs to prevent noxious weeds, agency personnel are increasingly required to use "weed-free" straw (and seed) on federal lands. In the western United States, rice (*Oryza sativa* L.) straw has become the mulch material of choice, because it is assumed to contain only seeds of wetland species that will not germinate or grow in upland settings. Whether this assumption is justified is still being tested. White et al. (1995), examining a burned coastal sage scrub stand, found one weedy species that they attributed to the rice straw applied to protect an adjacent municipal reservoir. They reported on only one season of postfire monitoring, however, so we do not know whether the weed (*Echinochloa* sp., a non-native annual grass) successfully set seed or spread in the summer-dry environment of southern California. Robichaud et al. (2000) recorded anecdotal accounts of weeds growing from "certified" weed-free straw, other than rice straw, in their interviews with personnel of USFS postfire rehabilitation teams. Some straw bales used for rehabilitation treatments on one fire in Colorado were contaminated with cheatgrass (Chong et al. 2002) (the type of straw used was not specified in the report). However, weed seeds in "certified" straw can come from ground contact in the bale staging area, not the straw bales themselves, if the staging area is not chosen carefully (Faust 2004). Because of its immediate effectiveness for erosion control and because of questions about the usefulness of seeding, mulch is increasingly being applied, via helicopter, to remote, high-value areas (aerial mulching; Robichaud et al. 2002; Faust 2004). This will increase the possibility of spreading noxious weeds to currently weed-free areas if adequate precautions to prevent contamination of the straw with weed seed are not implemented.

Discussion

The published literature on the erosion-control effectiveness of postfire grass seeding does not make a compelling case for the practice: few studies demonstrated statistically significant decreases in sediment movement. Little rigorous erosion research or monitoring has been conducted, however, presumably because of the expense and effort needed to get quantitative data. Erosion data are highly variable, and some of the studies I examined showed a trend toward lower sediment production on seeded sites that was not statistically significant (e.g., Amaranthus 1989; Wohlgemuth et al. 1998). On the other hand, there is clear evidence that successful establishment of seeded grass displaces native herbaceous vegetation, particularly annuals, and can reduce the survival rates of shrubs and tree seedlings. Shrubs and perennial herbaceous species that resprout after fire are generally not reported to be adversely affected by seeding (Beyers et al. 1998; Schoennagel & Waller 1999).

Table 2. Percentage of study sites in publications and monitoring reports reviewed by Robichaud et al. (2000) that had at least 30% and 60% cover by the end of the first and second growing seasons after fire.*

	Sites with >30% cover (%)		*Sites with >60% cover (%)*	
Study	*seeded*	*unseeded*	*seeded*	*unseeded*
1 year after fire				
19 publications	42	26	26	10.5
21 reports	74	38	35	8
2 years after fire				
18 publications	78	67	56	17
4 reports	75	75	25	50

**All published studies contained data from both seeded and unseeded plots. Monitoring reports did not always contain both treatments. Multiple sites within one publication or report are tabulated separately (modified from Robichaud et al. 2000).*

To estimate potential erosion-control effectiveness from studies that only measured vegetation cover, Robichaud et al. (2000) tallied the number of reported study sites that recorded at least 30% cover (considered partially effective erosion control) and at least 60% cover (considered effective erosion control) (Noble 1965; Orr 1970) on seeded and unseeded sites (Table 2). These cover thresholds are remarkably similar to the grass-cover percentages that Schultz et al. (1955) found would increase shrub seedling mortality (30%) or eliminate shrub seedlings altogether (55%); tree seedlings probably respond similarly. The dilemma for the land manager is thus obvious: if seeding produces enough cover to effectively control erosion, it will also effectively suppress or eliminate woody plant seedlings in the seeded area.

The good news for natural vegetation recovery is that seeding seldom produces effective cover the first year after fire (Table 2). Only 26% of seeded sites in published studies and 35% of seeded sites in USFS monitoring reports reviewed by Robichaud et al. (2000) had >60% cover. A higher proportion of seeded sites had partially effective (at least 30%) cover and, presumably, partially suppressed woody regeneration and native annual plants. In the second year after fire, 78% of seeded sites reported in published studies had at least 30% cover, compared with 67% of unseeded sites, and 56% had at least 60% cover, compared with only 17% of unseeded sites (Table 2).

For the land manager concerned primarily with erosion, seeding may be a reasonable gamble for trying to increase plant cover during the first year after fire (Table 2). Seedling is likely to stabilize a site more quickly than natural regeneration. Where control of erosion for protection of life, property, or infrastructure is essential, however, seeding would not be a good choice; effective control of sediment movement is likely to be achieved by seeding at the end of the first year only one-third of the time. More expensive but effective treatments such as straw mulch should be considered, especially where protection is needed from the first storms that occur after the fire.

Table 2 also illustrates the risks to conifer and other woody seedling regeneration faced by the manager considering seeding. In 26–35% of the cases examined, sites would probably have experienced nearly complete failure of tree or shrub seedling establishment in the first year after fire, compared with 10% of sites regenerating naturally. Partial control of erosion with only partial suppression of woody regeneration would occur more often. A land manager must weigh the potential erosion-control benefit against the economic cost of seeding, the ecological cost of native plant suppression, and the possible economic cost of replanting timber species. Seeding of grass species at rates likely to produce high levels of first-year cover should probably be reserved for very high-value timberland that will be replanted and intensively managed.

Along with suppressing native plant regeneration, seeded grasses may suppress noxious weeds. In addition to the work done in rangelands, Schoennagel and Waller (1999) found less cover of a non-native annual species in seeded plots than in unseeded plots in a burned grand fir forest. Invasive plant prevention is sometimes used as justification for postfire seeding, particularly where sources of infestation are known to be close to the burn area. Seeding for weed prevention is most likely to be successful against annual, rather than established perennial, species. When seeding for weed control is being considered, the probable impacts to native herbaceous and woody species must be weighed against the benefit of reducing the potential weed infestation.

Future Research Needs

More work is critically needed on the effectiveness of seeding with native species, the impacts of seeded cereal grains on natural regeneration, the effectiveness of seeding for prevention of non-native species establishment, and the likelihood of weed introduction from "certified" weed-free seed or straw. Monitoring the effectiveness of agency rehabilitation treatments should yield some of this information, but monitoring efforts, if they occur at all, often do not include untreated (control) sites on which to compare seeding with natural regeneration (Robichaud et al. 2000). Research scientists should seek opportunities to become involved in postfire rehabilitation projects, to help design studies that include controls, and to test methods, such as seeding with native species, that are not regarded as "proven" and thus may not be approved for widespread application in emergency stabilization projects (Robichaud et al. 2000, 2002). High-profile wildfires at the wildland-urban interface will continue to generate public demand for postfire rehabilitation, increasing the need to understand the effectiveness and ecological impacts of these treatments.

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