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Impacts of Postfire Grass Seeding on Chaparral Systems — What Do We Know and Where Do We Go from Here?

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Abstract. The most expensive effects of wildfires in urban interface areas are destruction of property during fires and the tremendous increases in erosion, sedimentation, and flooding that follow severe fires in the steep terrain that borders many of California's urban areas. Since the 1940's it has been standard practice to apply grass seed to denuded areas in an attempt to stabilize slopes and minimize downstream impacts of increased erosion. This practice is caught up in considerable controversy over its potential effects on the health and recovery of native ecosystems. The controversy has been exacerbated by lack of data concerning the effectiveness of seeding at reducing erosion and its long and short term influence on vegetation development. We are studying effects of seeding on erosion and vegetation development on four study sites in southern California chaparral. Data from this research support the idea that there are potential negative impacts of seeding on cover and composition of native herbaceous vegetation. However, we have seen little evidence of negative effects on shrub regeneration. Erosion data are inconclusive thus far, but suggest the possibility that seeding may cause some reduction in erosion, primarily during the second and third year after fire. Erosion on all burned sites, regardless of treatment, was much higher than prefire levels for 2 to 3 years. Variability in responses is high among sites, however, and we lack data from sites burned in years of normal or above average rainfall. A number of key questions must be addressed before we understand the effects of seeding on chaparral ecosystems. These include: impacts of seeding on mature vegetation structure, on the native post-fire flora, on genetic structure of plant populations, on habitat for key animal species, and on hillslope erosion. We also need to ask whether potential ecosystem damage is sufficient to suggest that the practice be abandoned, and what other options there are for minimizing and mitigating downslope damage following severe wildfires in urban interface areas.

Keywords: Chaparral; fire effects; *Lolium multiforum*; postfire erosion; postfire rehabilitation.

Introduction

Among the most damaging and costly after effects of fire in chaparral ecosystems in southern California are those resulting from the vast increases in erosion and sedimentation stimulated by the disruption of hillslope and channel systems during and following fire. Offsite effects such as flooding and debris flow deposits can impact developed areas downstream, often far removed from the fire area, and the costs of either preventing or cleaning up from these effects run into the millions of dollars.

During and immediately after a fire, dry erosion increases greatly as surface barriers to soil movement are removed. Dry ravel, as this material is called, moves downslope under gravity and fills stream channels. Early postfire rains can promote on-slope rill network development, enabling large amounts of water and soil to move rapidly off of steep burned slopes (Wells 1987, Wells *et al.*, unpublished manuscript). It is important to recognize that the relative importance of dry and wet erosion processes after a fire will be affected by site characteristics, such as the particle size of surface materials, and by weather patterns after the fire. Highest erosion would be expected where heavy rains occur early in the season; less erosion may result if early season rainfall is light and frequent, allowing vegetation to establish and stabilize loose hillslope materials. Some degree of downstream sedimentation will be inevitable, however, because of the channel loading by dry ravel processes immediately after a fire, and by sediment accumulation in stream channels during the interval between fires or extreme rainfall events. General patterns of post-fire erosion on chaparral sites have been documented by Wells (1987) and Campbell *et al. (1987)*. Erosion tends to be high for the first few years after a fire, and then gradually decreases with time, returning to prefire levels in 5 to 10 years (Rowe *et al.* 1954) as the increase in plant cover and root biomass over the first several postfire years helps to stabilize surface material.

Aerial Seeding

In response to the need to protect downstream structures and resources from these dramatic landscape responses to fire, managers began to explore ways of establishing rapid vegetation cover on burned hillslopes. Starting in the 1930's, Los Angeles County foresters first tried to seed native shrubs, then later experimented with herbaceous species such as mustards and grasses (Department of Forester and Fire Warden 1985). By the 1940's managers were routinely using annual ryegrass (*Lolium multiforum*) in an attempt to stabilize slopes after fires. Evaluation of seeding's effectiveness was based primarily on the level of grass cover established, with little attention given to any effects on native vegetation recovery; little or no quantification of the success of the practice at reducing erosion was attempted (California Division of Forestry 1957-1972). Nonetheless, results do provide some insights into the areas where ryegrass establishment is likely to be greatest and most reliable. In general, ryegrass establishment was more likely to be successful in northern California than in southern California (Fig. 1), and establishment tended to be poorer in dry inland areas than in coastal locations more prone to cool, foggy weather (Blanford and Gunter 1972).

Questions about the impact of seeding with annual grasses on natural vegetation recovery in chaparral have been raised for years. Schultz *et al.* (1955) and Gautier (1983) noted negative effects of ryegrass on native shrub seedling survival. Others have observed a negative relationship between ryegrass cover and native herb cover (Keeley *et al.* 1981, Gautier 1983, Taskey *et al.* 1989). Lower herbaceous species richness has been reported for ryegrass-seeded plots as well (Nadkarni and Odion 1986, Taskey *et al.* 1989). Additionally, there is evidence that under certain conditions seeding has the potential to increase erosion through localized increases in populations of burrowing rodents (Taskey *et al.* 1989). Still another concern is the effect of seeding on the likelihood of short-interval reburns, which have the potential to drastically alter species composition and stand structure (Zedler *et al.* 1983, Haidinger and Keeley 1993). There is also a need for additional information on effects of various seed mixes (including natives) on recovery processes and on how observed effects vary with differences in site, weather patterns, and fire characteristics (Barro and Conard 1987). These questions are increasingly important today as we try to manage in ways that maintain the health and resiliency of natural ecosystems and foster the maintenance of biodiversity, while still protecting human life, interests, and investments.

From a land management perspective, the bottom line, of course, is impact on erosion. If the practice of seeding does not achieve some measure of reduction in erosion or sediment output, then it loses its value. Unfortunately, despite the efforts that have gone into studying effects of post-fire seeding on vegetation, we have little information concerning effects on erosion. This is largely a function of the great difficulty in achieving sufficient replication and in evaluating treatment effects in a statistically sound manner when dealing with the steep slopes and diversity of chaparral systems. It is critical to remember that one of the basic characteristics of natural systems in southern California is their great spatial and temporal variability. So answers are not likely to be simple, and studies conducted on single sites or in single years will not yield results that can be widely generalized.

Because of the high interest in effects of post-fire seeding on response of chaparral systems following fire, and to address some of the limitations of earlier research, in 1986 we started a study to investigate both the range of responses of chaparral vegetation development and hillslope erosion processes after fire and the impact of postfire seeding for emergency watershed protection on these natural processes. Because of the increasing interest in use of alternative seeding mixes, we also conducted a small-scale companion study to look at effects of different seed mixes on vegetation cover and on the development of natural postfire vegetation.

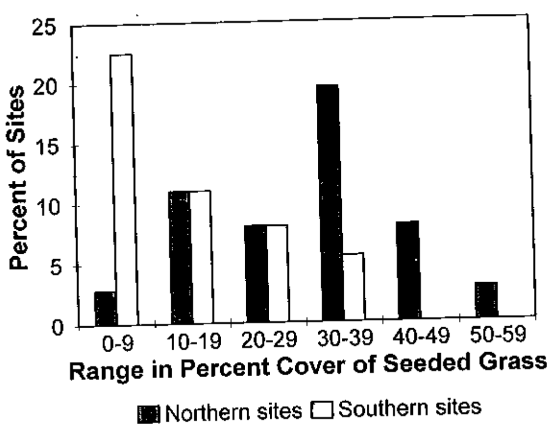

Figure 1. Range of ryegrass cover (percentage of ground surface) on burned sites the first year after emergency revegetation on northern and southern California sites. Data compiled from information in CDF emergency revegetation reports (California Division of Forestry 1957-1972).

Methods

Vegetation/erosion response studies

Results discussed here are for three study sites, one each in the Santa Monica, Santa Ana, and Santa Lucia Mountains. All study areas were burned in high-intensity prescribed fires following prefire vegetation and erosion plot installation (described briefly below). The Belmar site (Santa Monica Mountains; 34°3'N, 118°37'W) was burned in a prescribed fire in June, 1988. Before the fire, the vegetation was dominated by bigpod ceanothus (*Ceanothus megacarpus*), laurel sumac (*Malosma laurina*), hollyleaf cherry (*Prunus ilicifolia*), and chamise (*Adenostoma fasciculatum*) (nomenclature follows Hickman 1993). The site had last burned about 30 years earlier (Scott Franklin, personal communication). Rocks on the site are sedimentary and soils are gravelly loams in the Millsholm series. Slopes range from about 21 to 39 degrees in steepness and face SW to SE. The Bedford site (Santa Ana Mountains; 33°47'30"N, 117°32'30"W) was burned in late June and early July of 1990. Prefire vegetation consisted primarily of scrub oak (*Quercus berberidifolia*), toyon (*Heteromeles arbutifolia*), California ash (*Fraxinus dipetala*), hollyleaf redberry (*Rhamnus ilicifolia*), and hollyleaf cherry. This stand was at least 50 years old at the time of the fire (Gary Glotfelty, personal communication). Rocks are a mix of sedimentary and metamorphic, and soils are sandy loams in the Escondido series. Slopes vary in steepness, but range from 24 to 44 degrees and face NW to NE. The Vierra site (Santa Lucia Mountains; 30°29'N, 120°44'W) was burned in November, 1990. Prefire vegetation dominants were chamise, buckbrush (*Ceanothus cuneatus*), and black sage (*Salvia mellifera*). The site had not burned in approximately 40 years (Ben Parker, personal communication). Substrate is sedimentary and volcanic rocks, with soils in the Gaviota complex. Slope steepness ranges from 17 to 32 degrees, and slopes are predominately SW-facing.

Our basic study design called for installation of a series of erosion and vegetation monitoring plots in undisturbed mature mixed chaparral. Erosion was measured by collecting sediment in five adjacent 30 by 30-cm aperture sheet metal erosion troughs (treated as one unit for statistical purposes) at each of 60 unbordered erosion plots. At each location erosion was also measured on 10 control plots on nearby sites that were to remain unburned over the course of the experiment. Vegetation was measured on 40 2-m by 10-m permanent plots at each site with nested 1-m by 1-m subplots for herbaceous vegetation and for shrubs less than 0.5 m tall. Prefire baseline data were collected on hillslope erosion and on vegetation structure and composition. After burning, half of the completely-burned plots were randomly selected for seeding with annual ryegrass. Because of incomplete burns, postfire plot numbers were reduced to 20 at Belmar, 33 at Bedford, and 38 at Vierra. Seed was applied by hand with rotary fertilizer spreaders at a target rate of 430 seeds/m^2 (approximately 9 kg/ha [8 lb/ac, the most common density used for aerial seeding]). Following the burns, erosion was measured periodically throughout the year (at decreasing frequency as erosion rates returned toward prefire levels). Vegetation development on seeded and unseeded plots was measured each spring in April or May when herbaceous species were near peak biomass. Cover of all species present, shrub seedling density, and sprouting shrub canopy volume were measured. More detailed descriptions of methods can be found in Conard *et al.* (1991) and Beyers *et al.* (1994).

To assess impacts of seeding on vegetation at each study site, we analyzed for effects of seeding treatments on: seedling density of obligate-seeding *Ceanothus* species; seedling density of all other shrub species on a site (as a group); and percentage ground cover of seeded grasses, other herb species (as a group), shrub seedlings, and shrub resprouts (cover data for shrubs are not reported here). To look at dynamics of naturalized exotics and of native herbaceous species, we also compared cover of a group of four common naturalized species (*Hirschfeldia incana*, *Bromus madritensis*, *B. diandrus*, and *B. hordeaceous*) and of a group of native herbaceous annuals and perennials (see Table 2 for list) on seeded and unseeded plots. Differences in vegetation variables between seeding treatments were analyzed using a two-sample randomization test, because of the non-normal distribution of values; each p-value was estimated from 1,000 randomizations (Manly 1991).

Weather data were obtained from the nearest NOAA or local agency stations for each study site. Data presented in this paper are from the NOAA stations at Topanga (Belmar) and Santa Margarita (Vierra) (NOAA 1988-1993) and from the Corona Fire Station (Bedford) (Riverside County Flood Control and Water Conservation District records).

Evaluation of seed mixes

In 1987 following the Silverado Fire in the Santa Ana Mountains, we established a small scale study to look at differences among seed mixes in their effects on plant cover and development of natural postfire vegetation. The four seeding treatments were: blando brome (*Bromus hordeaceous* 'Blando'), Zorro fescue

(*Vulpia myuros* 'Zorro'), annual ryegrass (*Lolium multiflorum*), and a mix of native postfire species (*Clarkia unguiculata, Gilia capitata, Lupinus sparsiflorus, Nemophila menziesii, Phacelia campanularia*, and *Phacelia parryi*). Grass treatments were selected based on the species most commonly being suggested at the time as possible replacements for *Lolium multiflorum* in postfire rehabilitation of chaparral sites. Seed was applied at both 9 and 18 kg/ha (8 and 16 lb/acre). Seeding treatments were compared to unseeded control plots. Treatments were laid out in a randomized complete block design, with two blocks each on north and south aspects. As we observed no differences between seeding levels, data for the two levels were combined, giving us a total of two samples per treatment within each block. Analysis of variance was used to test for differences among treatments in overall vegetation cover, total herbaceous cover, total shrub cover, shrub seedling cover, and cover of seeded species. Differences at $P<0.05$ were considered significant.

Results and Discussion

Vegetation/erosion response studies: vegetation

To date, we have five years of postfire data from the Belmar site and three years from Bedford and Vierra. Results are presented below by category of response.

Shrub seedling density — Two of our sites had significant prefire populations of obligate-seeding shrubs, species which are killed by fire and regenerate only as seedlings (*Ceanothus megacarpus* at Belmar and *C. cuneatus* at Vierra). At Belmar, data from the first post-fire year suggested the possibility of a lower density of *C. megacarpus* on seeded plots (1989, $P=0.07$), but in the second and subsequent years after fire there were no detectable differences in seedling density between treatments for obligate-seeding *Ceanothus* or other shrub species. No differences were found at Bedford or Vierra in any year. The first years after fire were very dry at Belmar: precipitation in the winters of 1988-89 and 1989-90 was only 51% of the long-term average at the nearest NOAA station (Table 1). High seedling mortality between the first and second growing seasons eliminated any difference in density that might have been attributable to ryegrass seeding. By the third year after fire, shrub seedling densities ranged from about 1 to 4 seedlings/m^2 on the three sites on both seeded and unseeded plots.

Cover — Ryegrass seeding did not significantly increase total herbaceous plant cover at Belmar or Bedford during the first or subsequent seasons after fire (Fig. 2). At Vierra, total herb cover was significantly greater on seeded plots in the first two years after fire ($p<0.05$), although even seeded plots had very little herbaceous cover during the first growing season. The winter of 1990-91 had nearly normal total precipitation (Table 1), but most of the rain fell during March, which may have affected herbaceous species germination and growth. Ryegrass cover was relatively low — less than 15% — during the first year after fire at all three sites (Fig. 2). Ryegrass all but disappeared from Belmar after the first year, but ryegrass cover increased dramatically the second year at both Bedford and Vierra, where the second winter after fire was comparatively wet. Ryegrass is known to require consistent moisture for successful establishment (Rice *et al.* 1965). Observations over many years by CDF (Fig. 1) found less than 20% cover of seeded ryegrass on over 70% of 46 sites observed in the first year after fire in southern California and on about 45% of 52 sites observed in northern California.

Table 1. Rainfall year (1 July to 30 June) precipitation at the weather station nearest to each research site for the winters after fire and seeding (station locations are given in the test). Percentage of the long-term average for each site are given in parentheses.

Rainfall Year	Belmar (mm)	Bedford (mm)	Vierra (mm)
1988-89	318 (51%)		
1989-90	323 (51%)		
1990-91	432 (69%)	428 (113%)	664 (85%)
1991-92	843 (134%)	460 (122%)	666 (85%)
1992-93	1,237 (196%)	886 (235%)	1,203 (153%)
Long-term average	630	377	785

Cover of herbaceous species other than ryegrass was significantly lower ($p<0.05$) on seeded plots during the first and third years after fire at Belmar, during the second year at Bedford, and during the second and third year at Vierra (Fig. 2). This suggests that ryegrass replaced natural regeneration, rather than adding to it, particularly at Belmar and Bedford. Cover of four common naturalized (non-native) species — *Hirschfeldia incana* (*Brassica geniculata*, Mediterranean mustard), *Bromus madritensis* (red brome), *B. diandrus* (ripgut brome), and *B. hordaceous* (soft chess) — increased dramatically in the third and fourth years after fire at Belmar, and by the fifth year they made up most of the herbaceous cover.

Natural postfire regeneration in chaparral, and to a lesser extent in coastal sage scrub, includes a group of native annual and perennial herbaceous species which reproduce from seed almost entirely in the first year or two after fire (Sampson 1944, Horton and Kraebel 1955, Keeley *et al.* 1981, O'Leary 1990). When we examined these species separately from the total her-

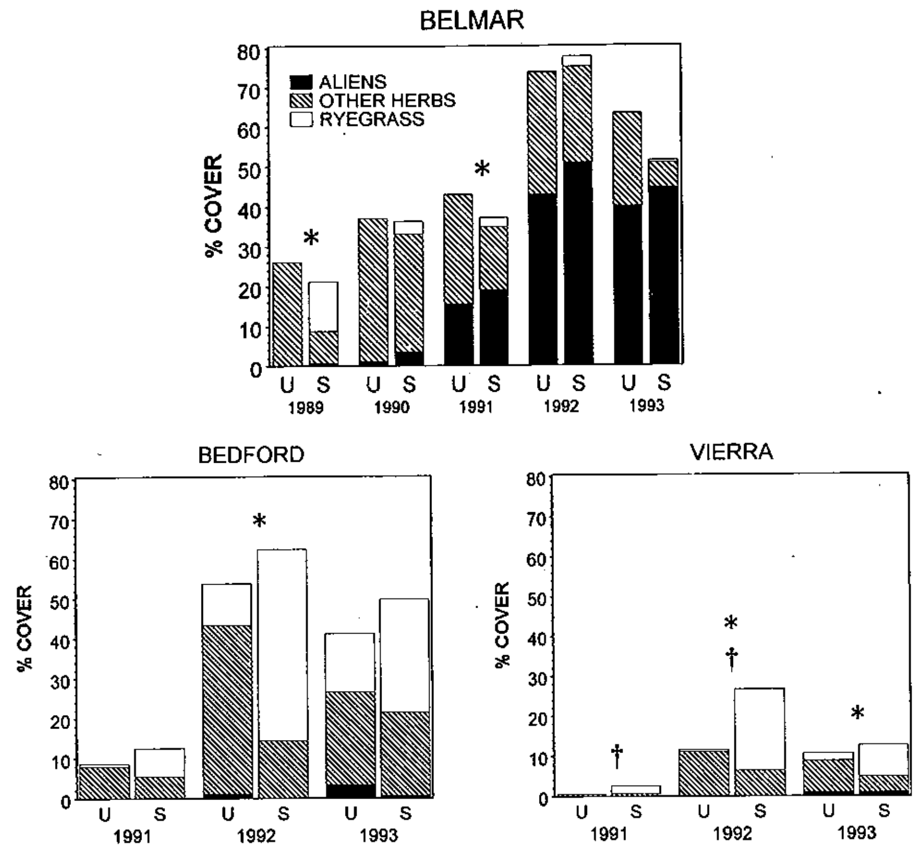

Figure 2. Cover (percentage of ground surface) of herbaceous species by category. ALIENS = *Hirschfeldia incana, Bromus madritensis, B. diandrus,* and *B. hordaceous*; OTHER HERBS = all other herbaceous species, native or introduced. U = unseeded plots; S = seeded plots. * = cover of OTHER HERBS different between seeding treatments at P<0.05; y = total herbaceous cover different between seeding treatments at P<0.05.

baceous flora, which includes many well-established non-native species, we found that in some years native species cover was dramatically lower on seeded plots (Table 2), even more so than overall herbaceous cover. Cover of postfire annuals was highest in the second year after fire at all three sites, which was also when ryegrass cover was greatest at Bedford and Vierra (Table 2, Fig. 2). The number of herbaceous species per plot — species richness, a measure of diversity — on seeded plots was 60 to 80% of that on unseeded plots (significantly different at P<0.05) during the second year after fire at all sites and during the third year at Belmar and Vierra (Beyers *et al.* 1994).

Vegetation/erosion response studies: erosion

On our study sites, hillslope sediment movement on burned plots seemed similar to that on unburned plots by about the second or third year after fire (Fig. 3), with some variation among sites (most likely a result of site characteristics combined with weather patterns). This

Table 2. Average percentage of ground surface covered by native prostfire annual and perennial herbs during the first three postfire years at three chaparral sites in southern California.[1]

Site (No. Plots)	Year	Cover (%ground surface) Seeded $\bar{x}$ (S.E.)	Unseeded $\bar{x}$ (S.E.)	P Value[2]
Belmar (n=10)	1989	7.6 (2.1)	22.6 (3.5)	0.004
	1990	26.4 (3.5)	30.1 (3.0)	0.432
	1991	11.8 (2.2)	19.8 (3.5)	0.068
Bedford (n=16,17)	1991	3.6 (1.2)	3.7 (0.9)	0.99
	1992	4.5 (1.3)	19.1 (5.4)	0.004
	1993	0.3 (0.1)	1.1 (0.6)	0.184
Vierra (n=19)	1991	0.4 (0.1)	0.2 (0.1)	0.572
	1992	5.7 (1.2)	10.3 (1.2)	0.012
	1993	2.9 (0.8)	6.4 (1.2)	0.022

[1]Annuals: *Antirrhinum kelloggii, Apiastrum angustifolium, Calandrinia breweri, Camissonia californica, C. intermedia, Caulanthus heterophyllus, Chaenactis artemesiifolia, Cryptantha species, Emmenanthe penduliflora, Eschscholzia californica, Eucrypta chrysanthemifolia, Lotus salsuginosus, Lupinus hirsutissimus, Malacothrix clevelandii, Papaver californicum, Phacelia species, Rafinesquia californica,* and *Silene multinerva*; perennials: *Calystegia macrostegia, Dichelostemma capitatum, Helianthemum scoparium,* and *Zigadenus fremontii.*

[2]For 1,000 randomizations.

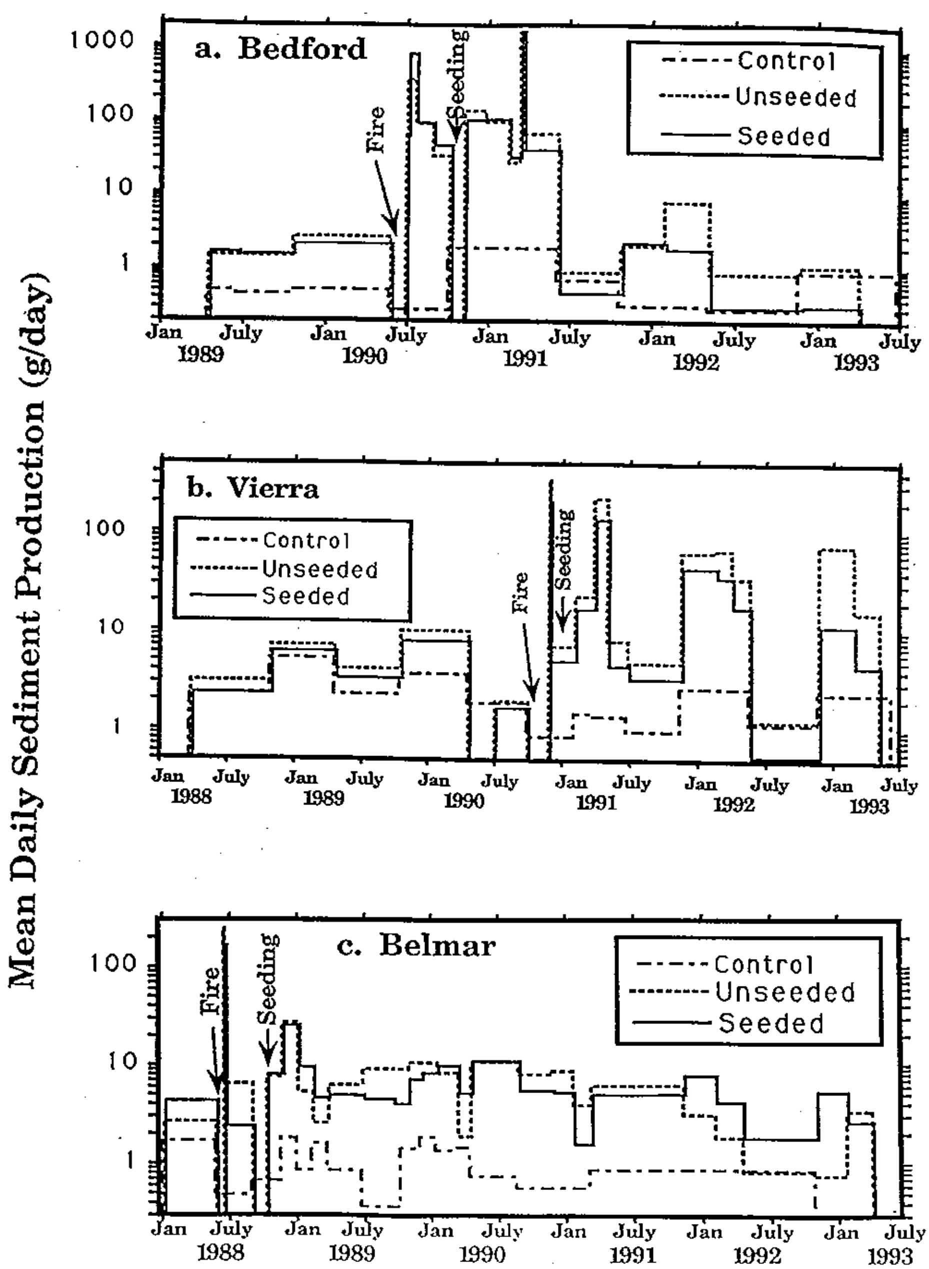

Figure 3. Average daily sediment production (based on grams of sediment collected per trough during each sampling interval) on the three study sites: a. Bedford, b. Vierra, c. Belmar. Note log scale and differences in vertical and horizontal scales among graphs.

suggests that in many areas hillslope erosion may return to near preburn levels more rapidly than is indicated in much of the literature (Rowe *et al.* 1954). Thus, at least on our sites, any effect from seeding on erosion rates after the second postfire year — even if statistically significant — can be expected to have a minimal influence on overall sediment output from the site. Although we have not completed statistical analysis, it appears that some effect of seeding on erosion may have occurred during one or two collection periods during the second and third year after fire at Bedford (Fig. 3a) and at Vierra (Fig. 3b). Nonetheless, even on seeded plots, erosion was many times higher in the first season following fire than in the unburned controls (Fig. 3). Herbaceous cover at Belmar was substantially higher in the first year postfire than at the other two sites (Fig. 2), and there is no evidence of differences in sediment output from seeded and unseeded plots (Fig. 3c). Postfire erosion rates in the first postfire year were highest at Bedford, probably due to the extremely steep slopes on parts of this site, combined with high rainfall intensities. In the second and third years after fire, erosion rates were typically highest at Vierra, intermediate at Belmar, and lowest at Bedford, where establishment of plant cover was most rapid. Whether results on any of our sites might have been different in wetter years, or years with better initial establishment of seeded grasses, is still open to question. Our data (Fig. 3) point up the episodic nature of erosion responses after fire. This, coupled with the tremendous spatial and inter-annual variability in hillslope sediment movement, unfortunately makes statistical analysis and detection of treatment differences difficult. We are currently developing new statistical approaches to deal with these problems, and, hopefully, allow us to draw conclusions with greater confidence.

Evaluation of seed mixes

In our study of the effects of different seed mixes, we observed no statistically significant effects of seed mix treatment on total percentage ground cover or on cover of different vegetation components on

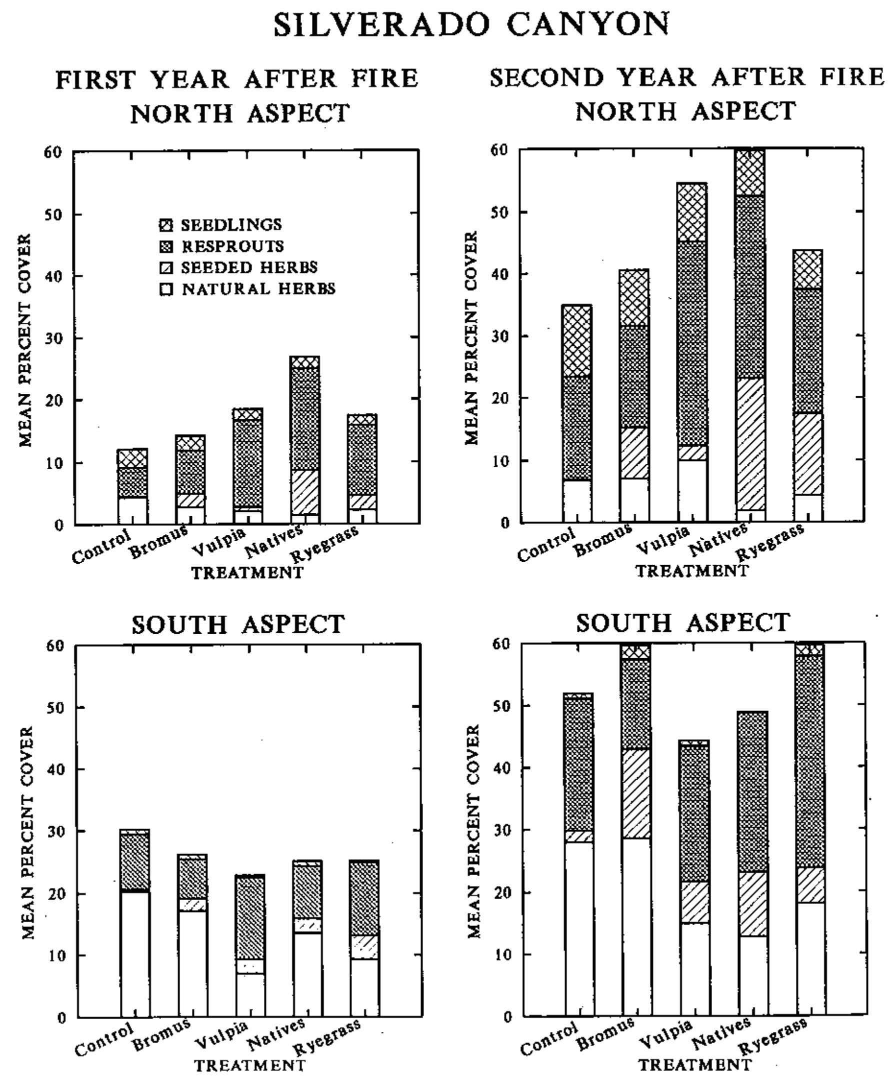

Figure 4. Average vegetation cover (percent of ground surface) on north and south aspect plots the first and second year after fire for the Silverado study sites.

south aspect plots (Fig. 4). On north aspects, there was very little natural herbaceous regeneration in any treatment (maximum cover of 10 percent). The only significant effect ($P<0.05$) was an increase in total herbaceous cover (seeded herbs plus natural herbs) with the native mix over the control, suggesting some possibility that there was inadequate onsite seed after the fire. During the first postfire year, shrub seedling regeneration was not significantly affected by the specific seeding treatment, but did vary inversely with total herbaceous cover and varied with aspect (best regeneration on north aspects, where herbaceous cover was generally lower; Fig. 5). These effects persisted into the second year after treatment, suggesting that they may be long-term. Our results strongly suggest that it is not the species composition of herbaceous regeneration after a fire that is most critical in determining effects on natural shrub regeneration processes, but the amount of herbaceous vegetation that becomes established and the rainfall patterns in the first critical postfire years.

To expand our data base to include a greater range of site and weather conditions, we have installed two additional sites in the past year (one burned in the Malibu fire in November, 1993, and the other in the Santa Ynez Mountains, burned in a March, 1994, prescribed fire). We hope to burn two additional sites in the Santa Lucia Mountains in Fall 1994. We also need to complete analysis on erosion data to determine significance of observed differences and to identify effects of variables (such as physical site characteristics, vegetation cover, and rainfall patterns) that can influence erosion responses.

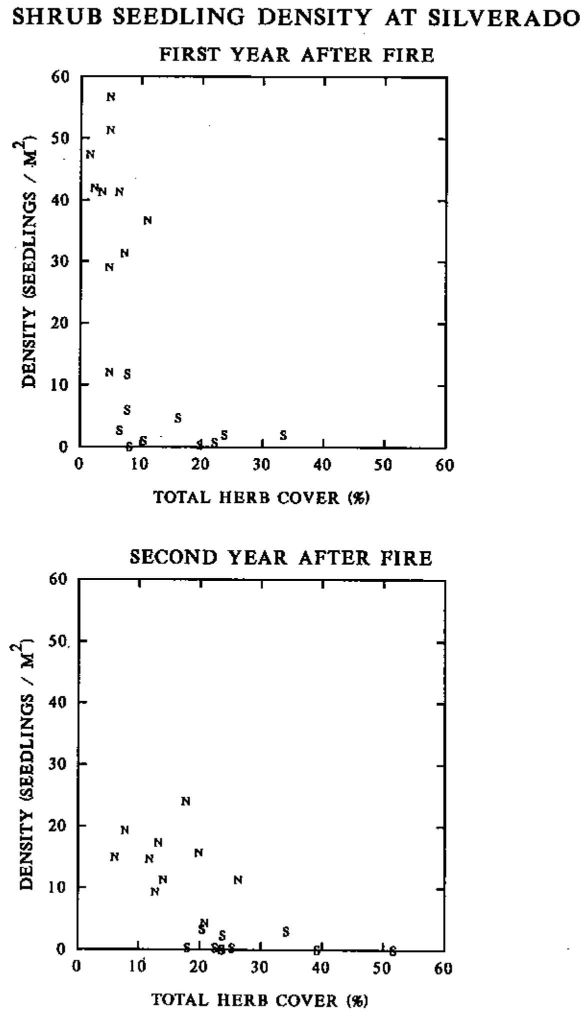

Figure 5. Shrub seedling densities on north and south aspects in the first two years after fire for the Silverado study sites.

Synthesis

Weather data from nearby stations illustrate the tremendous year to year and site to site variability in rainfall patterns typical of southern California chaparral systems (Table 1). It is important to recognize that this is only one component of the variable nature of chaparral sites, with their wide range in soils, climate, topography (e.g. slope steepness), fire history, fuel development, and vegetation composition. Chaparral systems have a complex set of interactive ecosystem processes relating to erosion, hydrology, chemical cycling, soil development, vegetation development, and animal population dynamics. These are all affected by site characteristics, site history, weather patterns, and fire characteristics. The way a given chaparral site responds to fire, and to postfire management treatments, will depend on all of these processes and the ways in which they interact. Therefore, while the data in the literature, and that which we present here, can shed considerable light on important issues, there are still some major missing pieces to the puzzle.

Our vegetation response data represent results from four sites, burned in three different years. Erosion data are from three sites, burned in two years. All of our data thus far are from sites burned the summer or fall before either drought years (1987-88; 1988-89) or a year with very anomalous timing of rainfall (1990-91; the year of the "March miracle," when a dry winter was followed by a very wet March). Our results, along with information available in the literature, suggest a general preliminary model of herbaceous species response to fire and to the effects of postfire ryegrass seeding (Fig. 6). Under unfavorable environmental conditions in the first winter following a fire, we expect low herb cover on both seeded and unseeded sites. While there

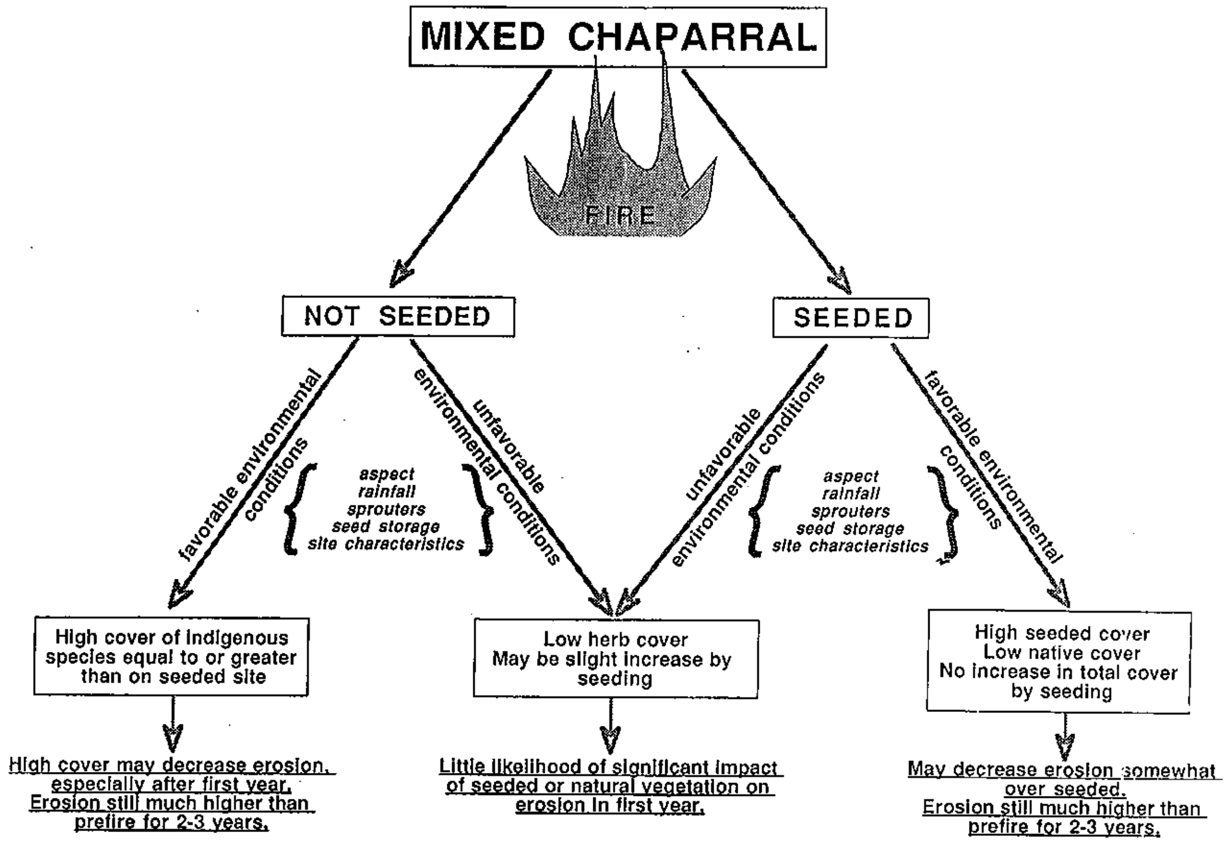

Figure 6. Preliminary model of some of the potential effects of seeding on herbaceous vegetation development and erosion in chaparral systems. Variable site conditions are shown in brackets.

is a possibility of some increase in cover from seeding, in many cases it is likely to be insufficient to affect erosion. Where environmental conditions are favorable for plant establishment and growth, we expect that high cover can be achieved on both seeded and unseeded sites (as long as there is sufficient seed remaining in the soil). However, on seeded sites, data suggest competition between seeded species and the native postfire herbaceous flora. The net results are likely to be minimal effects of seeding on total cover or on erosion. Before developing comprehensive models, however, data are clearly needed from a wider range of sites and weather conditions.

There are, nonetheless, some important conclusions that can be drawn at this point. First, although there may be conditions under which seeding can effect a significant reduction in erosion on chaparral sites in the first two years after fire, any reduction is unlikely to be large enough to eliminate the potentially catastrophic increases in sedimentation that are so characteristic of steep chaparral sites following wildfires. The most likely scenario for maximum effectiveness of seeding at reducing erosion would be one where rainfall is of low intensity and regularly spaced in the fall and early winter, allowing good grass cover to establish before heavy rains. However, this does not appear to be a reliable or frequently-occurring scenario on southern California chaparral sites. This raises the question of whether it is wise to rely solely on seeding for erosion control.

Second, in years of even moderately favorable weather conditions, seeded grasses appear to compete with the natural postfire herbaceous flora rather than enhancing total cover. This competition decreases both species richness and cover of the native postfire flora. We do not know yet whether these decreases are sufficient to cause long-lasting effects on the soil seed bank, but we can speculate that this might be the case.

Third, we have no evidence from our southern California chaparral sites that seeding, in and of itself, is affecting shrub seedling establishment (once the pulse of first year postfire mortality has occurred) on chaparral sites. However, high herbaceous cover of any type may inhibit seedling regeneration, and maximum cover of seeded annual grasses in our studies have rarely exceeded 20 percent. Although Nadkarni and Odion (1986) noted significant negative impacts of ryegrass on chaparral shrub seedling establishment in the first year after fire, their results are difficult to interpret as no mention is made of the level of seeded grass cover or biomass on their study site, and results were limited to the first postfire year. On *Ceanothus cuneatus/C. leucodermis* chaparral sites in the Sierra Nevada, Schultz *et al.* (1955) attributed increased shrub seedling mortality to seeded grasses where grass cover exceeded 30 percent. Shrub seedling regeneration was also reduced where seeded ryegrass cover was over 40 percent in ponderosa pine forests in the Sierra Nevada (Conard *et al.* 1991). However, rainfall and site conditions in the Sierra Nevada are quite different from those in southern California chaparral, where cover of seeded ryegrass of over 30 percent is rare, occurring on only about 5 percent of sites evaluated by CDF (Fig. 1). Bond (1987), in a study of effects of ephemeral herbs on shrub seedling survival after fire in the San Gabriel and Santa Monica Mountains, reported that herbs enhanced first-year shrub seedling survival on south aspects and decreased seedling survival on north and east aspects, and suggested this as a possible influence on species composition on different aspects in chaparral. In our study in the Santa Ana Mountains, however, we observed significantly higher shrub seedling densities on north aspects than on south aspects, with higher cover of seeded and unseeded herbaceous species on south aspects. Clearly, the impacts of seeding or of postfire herbs on shrub regeneration are not simple.

Key Issues for the Future

What are the impacts of seeding on vegetation composition and structure?

Many of the concerns raised by environmental groups and professional ecologists over the practice of postfire seeding center around the potential effects of this practice on the structure, composition, and stability of native vegetation. Because vegetation is a key component of animal habitat, concerns also center on how vegetation changes induced by seeding might affect habitat and, therefore, population levels of key animal species. Some of the specific issues involved are addressed in more detail below.

1. *How does artificial seeding of grasses (or other species) after fires affect mature vegetation structure?* This will depend on whether shrub regeneration is reduced sufficiently to prevent stands from achieving full cover by 5-10 years after the fire or whether species composition of the shrub community is significantly altered by seeding. We know that there is a certain amount of natural variability in stand density and composition following chaparral fires (e.g. Riggan *et al.* 1988). The important question may be whether structure is altered enough to cause long-term degradation and loss of resiliency in the vegetation. Another issue relating to vegetation resiliency is whether seeding enhances the probability of an early reburn of the site, as has been suggested by Zedler *et al.* (1983). Unfortunately, we are just beginning to get the data needed to address this question.

2. *How does seeding affect native postfire flora?* As discussed earlier, there are indications that seeded grasses can decrease diversity and population levels of native postfire herbaceous species. Many of these species are dependent on the seed bank established in the first few years after fire to repopulate a site in the next fire cycle. A decreased population level might lead to decreased seed bank of native herbs, perhaps reflected in diminished ability to recolonize after the next fire. Low population levels of native species then might leave more openings for the invasion of various widespread non-native grasses and forbs, as is suggested by our data and those of Haidinger and Keeley (1993). Other concerns include possible competition of seeded species with native flora, including endangered species, and effects of changed species composition on potential flammability of coastal sage and chaparral sites (see, for example, Zedler *et al.* 1983). There are few data available on this question to date.

3. *What is the impact on genetic and population structure of native species of using on-site versus off-site seed of native shrubs and herbs for revegetation projects?* There are increasing efforts to reseed with native (or naturalized) species after fires in chaparral and coastal sage scrub communities. We have no information on the effects of these practices on the vigor and fitness of native populations. Introduction of certain easily naturalized non-native species, such as Zorro fescue, may have greater impacts on native seed banks and population and community structure than would less well-adapted species, such as ryegrass, that will drop out after a few years. Their introduction may add to the problem of naturalized aliens that is so pervasive in postfire chaparral and throughout succession in coastal sage scrub. Introduction of off-site seed of native species, on the other hand, may cause decreased fitness of native populations through interbreeding with less well-adapted genotypes. We have little information on the genetic structure or gene flow characteristics of populations of native herbaceous or shrub species common to chaparral or coastal sage

scrub sites, and therefore, can only speculate about possible effects. Studies to address these issues are critically needed.

4. *How does seeding affect habitat for key animal species?* This is an increasingly important issue, especially on coastal sage scrub sites, which support a number of animal species that are threatened or candidates for endangered species listing. Seeding with either introduced or native species has unknown effects on recovery of key habitat features for these species. Alterations in stand structure, or any slowing of the recovery of the shrub component of chaparral or coastal sage scrub types following fire, has the potential to extend the recovery period for populations of animal species displaced by wildfire. One result might be the need for larger habitat areas on a regional level to accommodate slowed recovery following fires and to incorporate a mosaic of fire history patterns and stand ages (since fire) across the landscape. Research and monitoring started after the 1993 fires should help to answer some of these questions.

How do we best address the question of postfire erosion?

The purpose of emergency seeding after wildfires is primarily to protect downstream values (structures, water supplies, etc.) from damage due to accelerated postfire erosion by reducing the amount of material eroded from burned slopes. The goal of keeping soil on the slopes, especially in southern California, may not be a completely realistic one. Our mountains here are among the fastest growing in the world, with net uplift rates (uplift minus erosion) of about 0.53 meters per century (Scott and Williams 1978; see also Spittler 1995). Slopes in many areas are exceedingly steep, making them highly susceptible to erosion once the vegetation is removed.

1. *Is seeding to control erosion a practice that should be abandoned?* Seeding is obviously not a panacea, even if it does have some effect on postfire erosion. It will most likely continue to be used in particularly high hazard situations (and perhaps should be) until we have more conclusive data on its effects and effectiveness. Seeding costs on recent chaparral fires in California have ranged from around \$20/ha (\$8/acre) for seeding with ryegrass to over \$250/ha (\$100/acre) for seeding with native species (Dale Wierman, personal communication). The low cost of seeding with annual ryegrass makes it an attractive option for agencies when they are required to mitigate potential erosion and sedimentation damage following fires. Seeding with other species, however, can be quite costly, so it becomes even more important to have a clear idea of the costs and the benefits that will be derived. Because seeding has been the accepted postfire treatment for many years, agencies may be exposed to potential liability if they abandon the practice without adequate supporting data.

2. *Are there viable alternatives to seeding for controlling hillslope erosion after fires?* A number of other methods for controlling on-slope erosion were tried after the Oakland fire in 1991 (Booker *et al.* 1993) and many of these are also being tried in the wake of the fall 1993 fires in southern California. These include various types of hydromulch, silt fences, and straw bale check dams, techniques that have been tested and recommended for small-scale erosion control on road cuts and construction sites. These methods generally have the disadvantage of considerably higher cost and greater logistical difficulty than seeding, and many of them may require regular maintenance to work as intended (Goldman *et al.* 1986). Furthermore, due to the intrinsically high postfire erosion rates on many chaparral sites, the effectiveness of any method at keeping sediment on the hillslopes over the long-term is likely to be minimal.

3. *What about other options to protect downstream values?* It would also be worthwhile to consider alternatives for managing sediment once it gets into the channel system rather than relying on hillslope stabilization efforts for protection. Natural barriers in channels, such as downed trees, may serve to meter out sediment over a longer period (Barro *et al.* 1989) and minimize some of the catastrophic effects of fires. Leaving channels uncleared, however, does increase the risk of destructive flood bores and organic and sediment debris in reservoirs if natural barriers are suddenly breached. Human-built structures (small and large in channel structures and debris basins) may be costly, but they have long been shown to be effective at containing and retaining downstream transport of sediment. Where possible, it is important to consider zoning restrictions on development in areas at risk for postfire sedimentation and flooding. Unfortunately, in many developed areas this is no longer an option.

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