

Wildfire, Ryegrass Seeding, and Watershed Rehabilitation¹

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Abstract: Aerial seeding of Italian annual ryegrass (*Lolium multiflorum*) is a common, but controversial, emergency rehabilitation practice following wildfire in California. Replicated study plots, with and without ryegrass, established after a summertime chaparral wildfire on California's central coast revealed the following: 1. Ryegrass-seeded plots developed significantly greater total plant cover than unseeded plots in the first year. 2. Regeneration and growth of native species were significantly depressed in the presence of ryegrass. 3. Soil erosion was significantly greater on ryegrass-seeded plots than on unseeded plots. 4. Pocket gopher activity was greater on ryegrass-seeded plots than on unseeded plots. These results suggest that ryegrass seeding for emergency rehabilitation of burned areas can be ineffective, and even counterproductive, in certain cases.

THE WILDFIRE-GRASS SEEDING CONTROVERSY

The 1985 Las Pilitas fire burned 30,000 ha of predominantly chaparral watershed in California's central coastal region (fig. 1). Although fires such as the Las Pilitas are part of the natural order in chaparral, they can cause considerable watershed degradation, and predispose the land to greatly increased water runoff and soil erosion. The ensuing runoff water and erosional sediments may inflict further damage to property lower in the watershed.

In an effort to minimize post-fire damage and speed watershed recovery, land management and resource service agencies in California commonly seed severely burned brushlands with one or more plant species that exhibit early germination and rapid growth. Following commonly accepted practice, nearly two-thirds of the Las Pilitas burn was aeri ally seeded with either Italian annual ryegrass (*Lolium multiflorum*) or soft chess (*Bromus mollis*, also known commonly as Blando brome) (Calif. Dept. of For. 1985).

¹Presented at the Symposium on Fire and Watershed Management, October 26-28, 1988, Sacramento, Calif.

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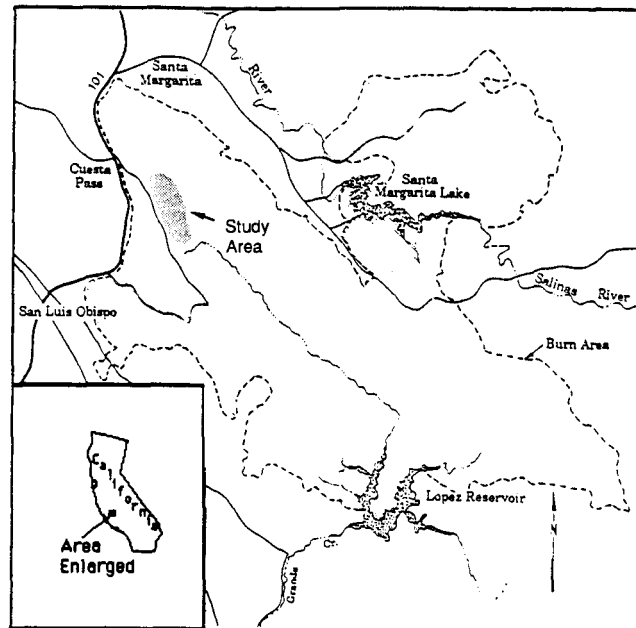


Figure 1--Study area location.

Since the 1940's, annual rye has been the most common grass seeded on burned chaparral lands of southern California. Its popularity in post-fire emergency rehabilitation work is due to its reliable germination, rapid early growth, short life span, effective ground cover and rooting characteristics, and broad site adaptability in mediterranean climates; moreover, the seed is inexpensive and readily available (Young and others 1975). Although seeding, especially with annual ryegrass, is a common post-fire rehabilitation practice, it is nonetheless highly controversial (Barro and Conard 1987, Gautier 1983).

Proponents of ryegrass seeding contend the following: The extreme surface runoff of rainwater from a denuded watershed erodes soil and threatens life and property by flooding and landsliding; therefore, plant cover must be quickly reestablished to mollify destructive forces. Although native species usually begin recolonization soon after a fire, their rate of recovery may be too slow to adequately protect the watershed during the first several years; therefore, artificial seeding is necessary.

Some proponents contend that seeded ryegrass is most effective during the first year after the fire, when erosion is greatest. Others argue that ryegrass is nearly ineffective in the first winter, but it becomes increasingly effective in the succeeding two years. Nonetheless, most proponents agree that although ryegrass may interfere with native

species, it dies out within three to four years, and does not threaten the long-term integrity of the chaparral ecosystem (Conrad 1979, Corbett and Green 1965, Dodge 1979, Gautier 1982, Kay and others 1981, Krammes and Hill 1963, Leven 1985, Los Padres National Forest 1986, Partain 1985, Schultz and others 1955).

Proponents recognize that artificial seeding is a gamble: It does not guarantee significant control of post-fire runoff and erosion, but it reduces the risk, and perceived liability, of taking no action. No one, however, can reliably predict the amount of risk reduction. If early post-fire rains are gentle, and subsequent rains are moderate, ryegrass likely will become well established, and the seeding effort will be considered successful. Alternatively, if early rains are intense, the grass seed will be washed down the hillsides, and soils will erode. Given the uncertainties, the perceived risks, and the fear of litigation, proponents feel that the most prudent action is to seed.

Opponents of aerial ryegrass seeding contend that the practice is costly, ineffective and frequently detrimental. They make the following arguments: First, most erosion occurs during the first year after the fire, before seeded ryegrass becomes established (Boyle 1982, Blankenbaker and others 1985, Krammes 1960, Wells 1986).

Second, predictions of runoff and erosion are highly uncertain, largely because they are based on assumed, rather than known, values of post-fire vegetative cover. Moreover, the total effective cover established by seeding is assumed to be significantly greater than that which could be established by natural recovery. These uncertainties and assumptions may cause ryegrass effectiveness to be overestimated. As a result, benefit-cost analyses of proposed rehabilitation efforts err strongly in favor of seeding (Blankenbaker and others 1985, Gautier 1983, Griffin 1982, Sullivan and others 1987).

Third, the seeded ryegrass is a strong competitor for water, nutrients, light, and growing space; and it may compete allelopathically with native species. It may virtually eliminate fire-following annuals, deplete soil nitrogen, and out-compete nitrogen-fixing plants; moreover, ryegrass may interfere with development of deep rooting natives that are important for long-term watershed protection. These interferences inhibit ecosystem recovery and impede watershed

rehabilitation; thus, erosion may be greater than under natural recovery. Although the grass may be temporary in the ecosystem, its effects are not (Arndt 1979, Biswell 1974, Corbett and Green 1965, Corbett and Rice 1966, Gautier 1982, Griffin 1982, Hanes 1971, Keeley 1981, Krammes and Hill 1963, Nadkarni and Odion 1986, Rice and others 1965, Wakimoto 1979, Zedler and others 1983).

Fourth, ryegrass dries out during summer, producing a highly inflammable cover of thatch. A fire in this thatch could destroy the young regenerating chaparral plants, leaving the ground bare for the following winter rains, and effectively creating an unwanted vegetative type-conversion (Nadkarni and Odion 1986, Wakimoto 1979).

Finally, 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. Thus, success is based more on assumed effectiveness than on measured effectiveness.

OBJECTIVES

The study had two objectives: 1. evaluate the effectiveness of seeded ryegrass in controlling soil erosion on test plots in the Las Pilitas burn area, and 2. determine whether or not seeded ryegrass would influence natural reestablishment of chaparral species during the first year after the fire.

AREA

The study area is located in the coastal Santa Lucia Mountains, on East Cuesta Ridge, approximately 7 km northeast of San Luis Obispo, California, and 24 km east of the Pacific Ocean. The area is characterized by moderately sharp, windswept ridges, steep sideslopes, and deep, narrow canyons. The study sites lie at approximately 650 m elevation, on slopes ranging from 40 percent to 55 percent steepness, and on aspects ranging (clockwise) from north-northwest to south-southeast.

The area's mediterranean climate is characterized by cool, moist winters, and warm, dry summers. Between 1942 and 1987 annual precipitation at the Santa Margarita water-pumping station, near the study area, ranged from 322 mm to 1607 mm, and averaged 767 mm, with more than 80 percent falling between April and November (San Luis Obispo County 1988). We assume

that average annual precipitation in the overall study area is comparable to that at the pumping station, although the station may receive more rainfall due to orographic effects. Snow is rare, but rainfall is augmented by an unmeasured amount of summer and fall fog.

Soil parent materials originate from well-consolidated, thinly bedded siliceous shales of the Monterey Formation (Hart 1976). The well-consolidated bedrock often lies within a meter of the ground surface. Small fissures and minor synclines are filled with ancient alluvial and colluvial deposits, which may be several meters thick. Soils are gravelly sandy loams to gravelly clay loams, which range from shallow over residuum to deep over colluvium and alluvium. Fragments of cherty shale cover 15 to 70 percent (mean = 25 percent) of the ground surface in study plots. Soils are mapped as Santa Lucia-Lopez-rock outcrop complex (O'Hare and others 1986).

The study area is a burned chamise chaparral community. Prefire vegetation consisted of dense stands of mature shrubs dominated by chamise (*Adenostoma fasciculatum*). On moister sites, manzanita (*Arctostaphylos glandulosa* var. *cushingiana*, and *A. luciana*) was a codominant, and toyon (*Heteromeles arbutifolia*), and scrub oak (*Quercus dumosa*) were associated species. The area previously had burned in 1929.

METHODS

Field sites were selected to meet the following criteria: 1. burned chamise chaparral; 2. unseeded by emergency rehabilitation efforts; 3. readily accessible throughout the year; 4. uniform geology and, as closely as possible, soil parent material; 5. uniform topography of smooth, upper portions of backslopes; 6. little chance of disturbance by people or cattle, and unaffected by runoff from roads or unusual features.

Soil Erosion Study

Eleven field sites, spread over 4.5 km, were established in November 1985. Each site supported two similar adjacent plots approximately 3 to 6 meters apart, and each measuring 6 m by 15 m, parallel and perpendicular, respectively, to the slope contour. Ten erosion troughs were installed along the bottom of each plot, for a total of 220 troughs. The troughs are welded sheet metal boxes 30 cm long,

10 cm wide, and 13 cm deep, with a 13 cm long apron on the uphill side (Ryan 1982, Wells and Wohlgemuth 1987).

One randomly selected plot in each pair was left untreated, and the other was seeded with Italian annual ryegrass (*Lolium multiflorum* Lam.) at the rate of 17.5 kg/ha, to give an application of approximately 400 seeds/m². This rate corresponds to approximately 15.5 lb/ac, or 37 seeds/ft². California Department of Forestry and US Forest Service recommendation for Las Pilitas burned area emergency rehabilitation was 8 lb/ac, based on approximately 40 seeds/ft² at 200,000 seeds/lb (California Department of Forestry 1985, US Forest Service 1978). The seed used in this study measured 104,000 seeds/lb; therefore, the weight per unit land area was increased accordingly.

Sediment trapped in each trough was collected, dried and weighed periodically from April 1986 to May 1988.

Vegetative cover was determined in September 1986, by estimating the percentage of ground covered within a one square meter sampling frame placed in five random locations in each plot. Sample locations for individual plots were chosen by coordinates selected from a random number table (Wonnacott and Wonnacott 1972). Each set of five values, which were averaged, gave a 5.5 percent sampling intensity.

Precipitation was measured by two weighing-bucket recording rain gauges and two nonrecording rain gauges, distributed throughout the study area.

Analysis of variance was performed on data using a completely randomized block study design, arranged to test differences between seeded and unseeded treatments, differences among site locations, and interaction between treatment and site location. The number of troughs (10) in each plot constituted the sample size. The test statistics $F = MST/MSE$ and $F = MSB/MSE$ were applied to treatment main effects and location (block) main effects, respectively; $F = MSTB/MSE$ was applied to interaction. MST is the mean square of seeding treatment; MSB is the mean square of site location; MSTB is the mean square of treatment x location; and MSE is the mean square error (Little and Hills 1978). Although statistical calculations considered each trough as an observation, the histograms present mean values per plot to allow simplicity and clarity of presentation.

Plant Interaction Study

The plant interaction study included field and laboratory components. Field plots were established in November 1985 on seven sites, each adjacent to an erosion site. Each site contained six plots--three seeded treatment plots and three unseeded control plots, for a total of 42 plots. Plot size was 2 m by 2 m. The treatment plots were seeded with Italian annual ryegrass (*Lolium multiflorum* Lam.) at the rate of 17.5 kg/ha, a rate equal to that applied in the erosion study.

Native and ryegrass cover, and species composition, abundance, and richness were evaluated on each plot in May 1986, using the Braun-Blanquet method (Westhoff and van der Maarel 1978).

For the laboratory portion of the study, 20 wooden boxes, measuring 0.5 m by 0.5 m, were filled with surface-soil collected from the burn area, and placed on a rooftop at California Polytechnic State University. Ten of the boxes were seeded in early February 1986 with Italian annual ryegrass at the same rate as the field plots, and ten boxes were left unseeded. No native seed was added to that which was naturally in the collected soil. Species composition, abundance, and richness were assessed in each box periodically for 21 weeks after emergence.

Statistical analyses of field data were similar to those used in the erosion portion of the study. Planter box data were analyzed by t-test for a completely randomized design (Little and Hills 1978).

RESULTS AND DISCUSSION

In the year after the fire, plant cover varied significantly ($\bullet = 0.05$) with site location; nonetheless, it was greater with ryegrass seeding than with natural recovery. Moreover, ryegrass was the dominant species on all seeded plots. In May 1986, 10 months after the fire and 6 months after ryegrass seeding, plant cover with seeding significantly exceeded ($\bullet = 0.05$) that without seeding by 14 percent (mean) in the plant-interaction field plots. At the same time, native cover was depressed 23 percent (mean) in the presence of ryegrass ($\bullet = 0.001$):

	Percent Cover ¹	
Vegetation:	Seeded	Unseeded
Ryegrass	37.1 $\pm$ 24.0	--
Native	34.3 $\pm$ 21.0	57.7 $\pm$ 30.9
Total	71.4 $\pm$ 19.6	57.7 $\pm$ 30.9

¹Mean $\pm$ 1 std. dev.

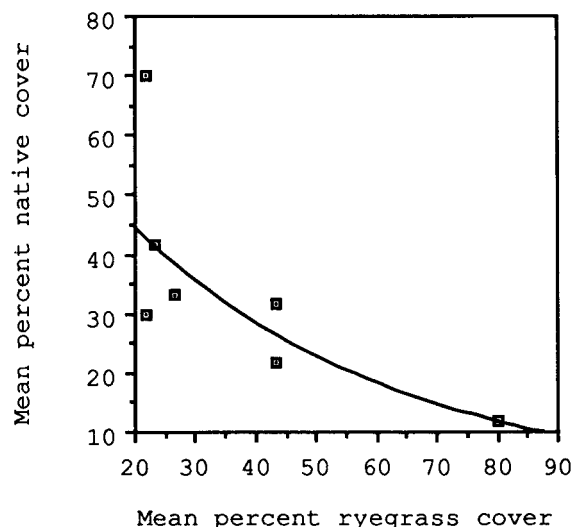


Figure 2--Native cover decreased as ryegrass cover increased on ryegrass-seeded field plots 6 months after seeding.

Native plant cover decreased exponentially as ryegrass cover increased (fig. 2). The high variability due to site location ($\bullet = 0.001$) is reflected in the large differences in native cover with low ryegrass cover. Note that as ryegrass increased, native cover variability decreased, perhaps because the ryegrass treatment effect over-rode the site location effect.

Native species richness was significantly less ($\bullet = 0.05$) on ryegrass-seeded plots than on unseeded plots: each seeded plot averaged 4.2 ± 2.1 native species, whereas each unseeded plot averaged 5.2 ± 1.6 native species.

Plant cover in the soil erosion plots showed a similar significant ($\bullet = 0.05$) trend in differences (12 percent), but mean values were considerably less: 39.0 ± 18.0 percent with ryegrass, compared to 27.1 ± 12.1 percent without ryegrass. Two factors might explain the lower cover on erosion plots compared to plant-interaction plots: One, these data were collected in September 1986, after many plants had desiccated in the summer dry season; two, the measurements were made by a different researcher.

Ryegrass seedlings outnumbered native seedlings by 19 to 1 six weeks after planting ryegrass in half the planter boxes. Native seedlings without ryegrass outnumbered those with ryegrass by 2.5 times ($\bullet = 0.001$); this ratio increased to

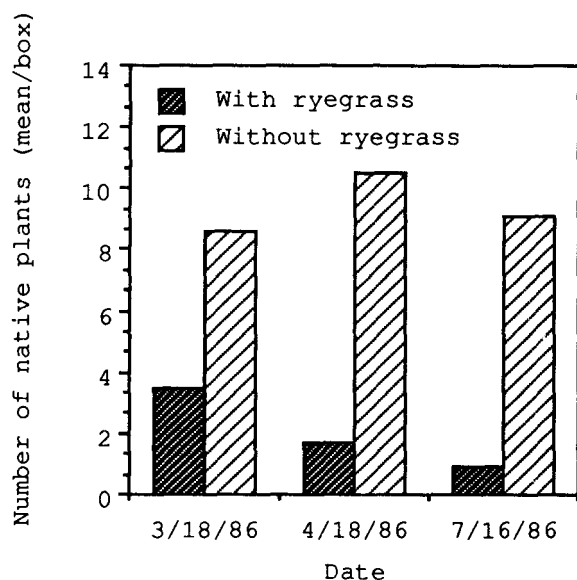


Figure 3-- Mean number of native plants per planter box on 3 dates, 6, 10 and 21 weeks after planting ryegrass.

6:1 after 10 weeks, and to 10:1 after 21 weeks (fig. 3).

Although fire-following annuals were the plants most restricted in the presence of ryegrass, shrubs also were affected. At 21 weeks, chamise seedlings grew in nine of ten boxes without ryegrass, but in only four of ten boxes with ryegrass. Average seedling height was 10 cm without ryegrass, and 1 cm with ryegrass. Manzanita growth showed similar trends, but the manzanita population was less than that of chamise.

Precipitation in the study area after the fire was near or below the assumed average. Rainfall collected from Nov. 10, 1985, to Apr. 18, 1986, ranged from 487 mm to 726 mm, and averaged 636 mm for the four rain gauges distributed over the study area. Rainfall at the Santa Margarita pumping station from Nov. 1, 1985, to Apr. 30, 1986, was considerably higher, at 1026 mm. From Sept. 1986, to Apr. 1987, the study area average value was approximately 336 mm, whereas the pumping station precipitation was 476 mm. The limited precipitation, consisting of light to moderate rains and fog, kept soil erosion to considerably less than the amount anticipated.

Ryegrass seeding appeared ineffective in controlling erosion during the first rainy season after the fire, from November 1985 to April 1986. Comparing sediment collected from seeded and unseeded plots

at the eleven sites, we found that four sites had less erosion with ryegrass, four sites had more erosion with ryegrass, and three sites showed almost no difference between treatment and control (fig. 4). The net result was no significant difference, at the $\alpha = 0.1$ level, in erosion between seeded and unseeded plots, although the seeded plots yielded 16 percent more sediment. Erosion did vary among site locations ($\alpha = 0.001$). Sheetting was the primary overall erosional process on the plots. Rilling was secondary; nonetheless, it contributed substantially to the sediment collected on plot numbers 3-seeded, 9-seeded, and 9-unseeded. Rilling tended to cut no deeper than to the depth of a clearly observable water-repellant layer.

During the dry season, from April to November 1986, soil erosion was greater on seven of eleven ryegrass-seeded plots than on the companion unseeded plots. Overall erosion on the eleven sites was 4.5 times greater with ryegrass seeding than without ryegrass seeding (fig.5). For the year, from November 1985 to November 1986, erosion was greater on nine of eleven ryegrass-seeded plots. Overall for the eleven sites, erosion with seeding exceeded that without seeding by 2.2 times (fig. 6). Erosion continued to differ with high significance among site locations. These data are statistically very highly significant ($\alpha = 0.001$).

Annual ryegrass seed is applied to control soil erosion. Why, then, did we

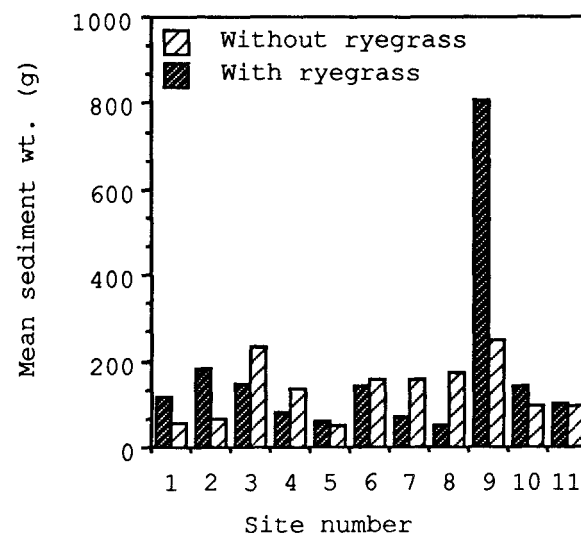


Figure 4--Sediment weights for the first rainy season, November 1985 to April 1986 (mean of 10 erosion-trough measurements per site).

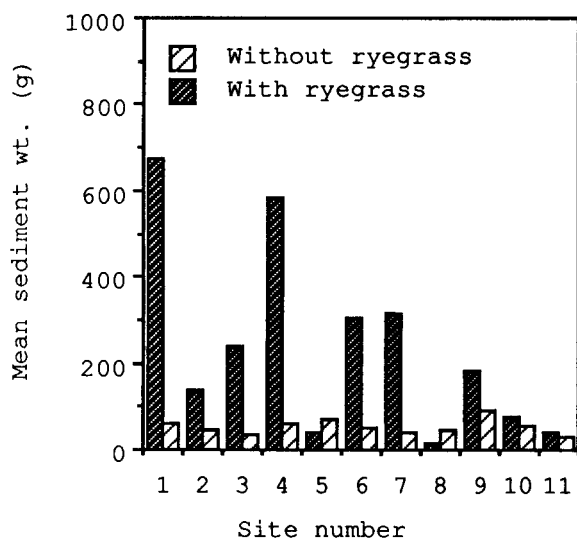


Figure 5--Sediment weights for dry season the year after the fire, April to November 1986 (mean of 10 erosion-trough measurements per site).

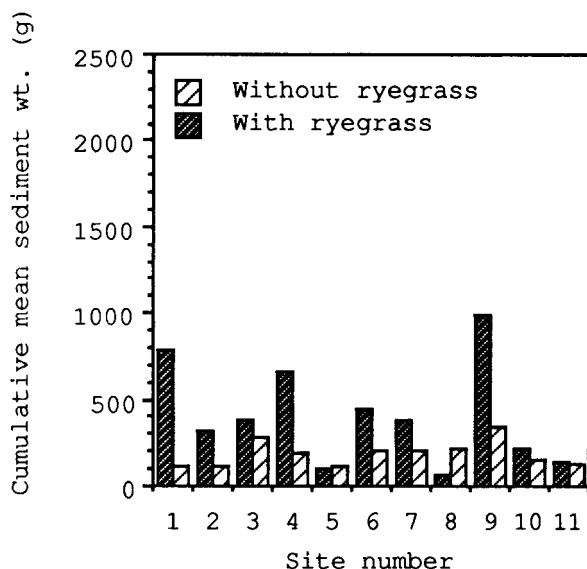


Figure 6--Cumulative sediment weights for 1 year of collection, November 1985 to November 1986 (mean of 10 erosion-trough measurements per site).

find greater soil erosion on ryegrass-seeded plots than on adjacent unseeded plots, especially when the seeded plots had greater plant cover? The answer appears to be gopher activity. The number of mounds made by pocket gophers (*Thomomys bottae*) was far greater on ryegrass-seeded plots than on unseeded plots. In September 1986, we counted 204 mounds on

ryegrass-seeded plots, and 31 mounds on unseeded plots. As the number of gopher mounds increased, the amount of soil trapped by the sediment troughs tended to increase; further study is needed to adequately quantify this relationship.

The gophers contributed to erosion by piling soil loosely on the surface, from where it was easily moved by sheeting and rilling, and by casting soil downslope during excavations. Occasionally the excavated soil was deposited directly into an erosion trough.

Additional correlations needing further quantitative study were noted between erosion and site aspect and soil depth. (Perhaps some of these could help explain the high statistical significance ($p = 0.01$) between amount of sediment collected and site location, and interaction of treatment and site location.) Site aspects were concentrated equally in the northeast and southeast compass quadrants, except for one site in the northwest quadrant. Soil erosion from ryegrass-seeded plots appeared to increase generally with aspect progression from northeast to southeast. Gopher activity followed a similar progression, with greatest activity occurring in the southeast quadrant. In contrast, soil erosion from unseeded plots did not vary appreciably among aspects. Gopher activity and soil erosion also tended to increase with increasing soil depth; few or no gopher mounds were noted on sites having soil less than 40 cm deep to bedrock.

We questioned whether or not the plot sizes were so small as to cause crowding of gophers, and if larger plots would allow the animals to disperse, thereby decreasing the concentration of mounds. To answer this, gopher mounds were counted on three sites, outside the study area, which had been aerially seeded with annual ryegrass as part of the burned area emergency rehabilitation efforts. Site conditions and plot sizes were similar to those of the study area. Gopher mounds on these plots ranged from 28 to 72, a density comparable to that in the study plots which ranged from 0 to 73. These densities are also similar to those reported in the literature. Although the size of our study plots is somewhat smaller than the average territory of an adult male pocket gopher, the plot size is well within the range of reported territorial sizes (Bryant 1973, Chase and others 1982, Pollock 1984).

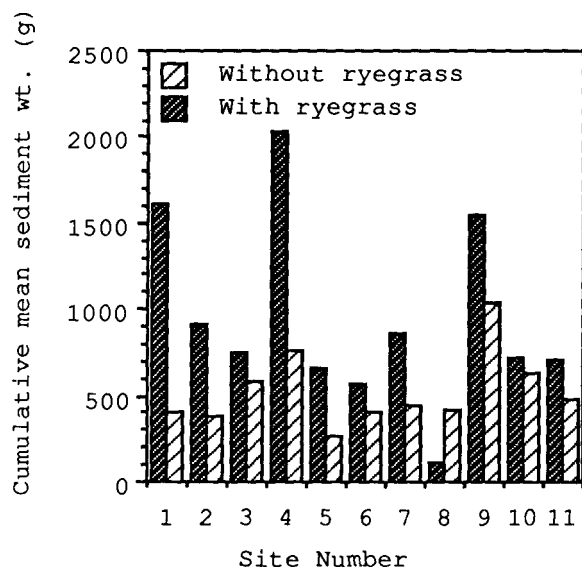


Figure 7--Cumulative sediment weights for 2-1/2 years of collection, November 1985 to May 1988 (mean of 10 erosion-trough measurements per site).

The erosion trends noted during the first year of the study continued in the following two years (fig. 7). The ryegrass-seeded plots continued to produce more sediment, and in May 1988, 2-1/2 years after seeding with ryegrass, overall erosion was 1.8 times greater with ryegrass than without it; moreover, erosion was greater with ryegrass seeding on ten of the eleven sites. Differences between treatment and nontreatment, and among site locations continued to have high statistical significance ($\bullet = 0.01$).

Additional important observations were made during the latter part of the study, but have not been quantified:

1. After going to seed in 1986, ryegrass spread to outside of the experimental plots. The spreading continued in 1987 and, to a lesser extent, 1988.
2. Gopher activity followed the spreading ryegrass, and soil erosion increased accordingly.
3. Ryegrass declined greatly on the southerly aspects in 1988, 2-1/2 years after seeding, but continued to increase on the northerly aspects.
4. Gopher activity declined as ryegrass disappeared from southerly aspects, but gopher activity

increased as ryegrass increased on northerly aspects. Unexpectedly, gopher activity increased in shallow soils, which previously had very few or no gopher mounds, as ryegrass persisted in those soils.

5. Ryegrass continued to interfere with recovery of native species, most notably those reproducing from seed, including lupine, lotus, and chamise. Lupine, for example, was dramatically excluded from two ryegrass-seeded plots on a slope which was purple with lupine outside the seeded plots.
6. In the third year after seeding, total cover appeared greater on unseeded plots than on seeded plots. As the ryegrass died out on the seeded plots, uncovered spots were left where ryegrass cover was heaviest.

CONCLUSIONS

Italian annual ryegrass seeded on the burn area increased total vegetative cover in the first year after the fire, but it failed to fulfill the ultimate goal of post-fire emergency rehabilitation--namely, to control soil erosion and enhance post-fire watershed recovery. Although seeding increased plant cover during the first year after the fire, it had four negative impacts: (1) The seeded ryegrass clearly interfered with recovery of native species, which are important for long-term stability of the ecosystem. (2) It failed to significantly control soil erosion any more than did natural recovery. (3) It stimulated an unwanted environmental factor, in this case, pocket gophers. (4) The gophers, in turn, moved large amounts of soil which otherwise would not have been disturbed.

In burned area emergency rehabilitation, we must be concerned not only with vegetative cover, but, more importantly, with the effectiveness of that cover in meeting our goals. Seeding an introduced species can prove counterproductive if that species interferes with natural recovery, or if it stimulates an unwanted factor in the ecosystem.

ACKNOWLEDGMENTS

This study was funded by a cooperative agreement with Pacific Southwest Forest and Range Experiment

Station, USDA Forest Service, and by an Agricultural Education Grant from the School of Agriculture, California Polytechnic State University.

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