

POSTFIRE ECOSYSTEM RECOVERY AND MANAGEMENT: THE OCTOBER 1993 LARGE FIRE EPISODE IN CALIFORNIA

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Introduction

Over a period of less than 10 days in autumn 1993, the southern California landscape exploded in massive wildfires that burned more than 80,000 ha (Keeley 1995). This ecological phenomenon presented an unparalleled opportunity for answering questions about the ecology and management of disturbance-prone ecosystems.

Opportunities presented by this unusual event included:

1. Fires occurred over an extensive range, > 150 km north/south and from the coast to 80 km inland. Thus, a wide range of environmental site characteristics were represented.
2. Fires burned more or less simultaneously, thus factoring out seasonal effects in comparisons of sites.
3. A wide range of age classes was represented. Due to the intense Santa Ana Winds, fires burned through young as well as old age stands (our study sites ranged from 3 to 85 years of age prior to the fire and thus comparisons could be made between sites which differed in their disturbance histories.
4. Fire intensities were apparently quite varied, as evidenced by the obvious differences in fire severity (measured by the diameter and height of skeletal remains). Thus, the role of disturbance intensity on the structuring of postfire communities could be examined.
5. Many different resource management agencies had jurisdiction over burned landscape and thus both active and passive postfire management techniques were available for comparison.

This paper summarizes findings of a study that monitored 90 sites, across 16 fires, for immediate postfire community recovery.

Study Sites and Methods

The primary criteria used in site selection were accessibility and lack of disturbances, other than the recent fires. Sites were selected to cover as great a range of environmental conditions as possible (Table 1). Of the 90 sites,

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Table 1. Summary of site characteristics for the 90 postfire monitoring sites established after the 1993 wildfires in southern California.

Elevation (m):	60 - 1225
Distance from the coast (km):	1 - 74
Aspect (°):	All aspects represented
Inclination (°):	0 - 42
Annual irradiance (Kcalories cm ⁻²):	146 - 318
Age (yrs since prior fire):	3 - 85 yrs
Fire severity:	low - high
Index 1	1.2 - 9.2
Index 2	2.0 - 9.5
Rock cover (% ground surface):	0 - 61
Soil:	
Rock (% by wt):	0 - 66
Organic Matter (%wt):	0 - 1
pH	5.9 - 7.1
% sand:	25 - 79
% clay:	8 - 39
Total Nitrogen (%):	0.06 - 0.49
Phosphorous (%):	0.01 - 1.20
Carbon:	0.81 - 8.10

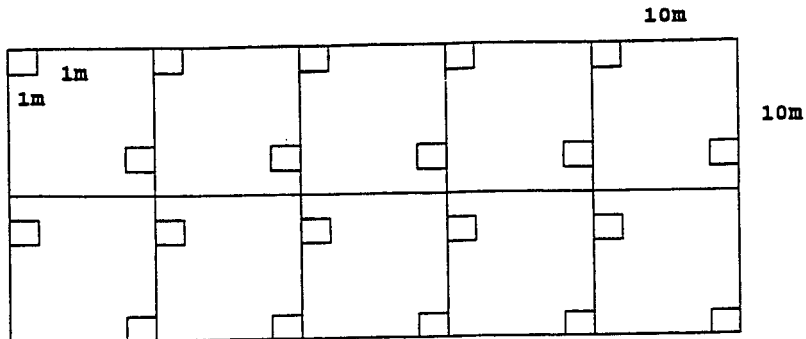


Fig. 1. Sampling plot design. Each site was 1000-m² (20-m x 50-m). Ten 10 m x 10 m plots were sampled for shrub recovery, prefire shrub density, shrub skeleton characteristics, and plant species diversity. Two 1 m x 1 m subplots in each plot were sampled for density, cover, and height of all plant species.

49 were in the lower stature, drought deciduous coastal sage scrub community and 41 in evergreen chaparral.

Each site was 1000-m², subdivided into 100-m² plots (in which prefire and postfire shrub composition and total species richness was determined) and further subdivided into 20 1-m² subplots (in which total herb and shrub density, coverage and height were recorded) (Fig. 1); a full description of the data collected with this nested quadrat technique is in Table 2.

Slope aspect was measured with a compass, inclination with a clinometer. For regression analysis, total annual solar insolation (Kcalories cm⁻²) was calculated from slope aspect, inclination and latitude (Frank and Lee 1966). Elevation and distance from the coast were taken from topographic maps.

Table 2. Summary of vegetation data collected for the 90 postfire monitoring sites established after the 1993 southern California wildfires.

Prefire (extrapolated from skeletal remains):

- Shrub density
- Dominant shrub species (relative density)
- Shrub size (basal diameter)

Postfire:

- Shrub sprouting success (percentage sprouting)
 - Shrub size comparison of dead vs resprouted plants
 - Shrub seedling recruitment
 - Density of all shrubs and herbs
 - Coverage of all shrubs and herbs
 - Height of all shrubs and herbs
 - Mode of regeneration of all species (vegetative sprouts or seedlings)
 - Fire severity (height and smallest twig diameter on shrub skeletons)
 - Diversity/species richness at scales of 1 m², 100 m², and 1000 m²
 - Survivorship through spring growing season
 - Survivorship through summer drought
-

Prefire stand age was determined from polished wood samples cut from shrub skeletons collected at each site (Keeley 1993).

Since fire intensity can have pronounced effects on vegetation recovery (Borchert and Odion 1995), two estimates of fire intensity were made from measurements of chaparral and coastal sage shrub skeletal remains. These were not direct measures of fire intensity and are best referred to as "fire severity indices." Index #1 was based on the diameter of the smallest twig remaining on the shrub skeleton in or nearest to each 1-m² subplot. The justification for using this as a surrogate measure for fire intensity is based on a study by Moreno and Oechel (1989), which related known fire intensity to diameter of the smallest twig remaining on shrub skeletons of *Adenostoma fasciculatum*. Since our study dealt with species in addition to *Adenostoma fasciculatum*, we assigned an index, between 1 and 10, to different diameter shrub skeleton twigs based on two assumptions: (1) smaller twigs reflected lower fire intensities and (2) for a given twig diameter, smaller-stature coastal sage shrubs would have generated a lower fire intensity than more robust chaparral shrubs (e.g., Fig. 2). Fire severity index #2 was based on the height above ground level of shrub skeletal remains within each of the 100-m² plots. Five skeletons of each dominant shrub or subshrub species were measured in each plot. As with index #1, indices from 1 to 10 were assigned to different heights based on two assumptions: (1) shorter skeletons reflected a higher fire intensity than taller skeletons, and (2) for a given skeleton height, coastal sage shrubs would have generated a lower fire intensity than chaparral shrubs. Although it is expected that stand age would affect this index, insufficient understanding of this relationship precluded using age in this index.

At each site, rock cover (as a percentage of ground surface covered) was visually estimated, using the average of three estimates for each of the 20

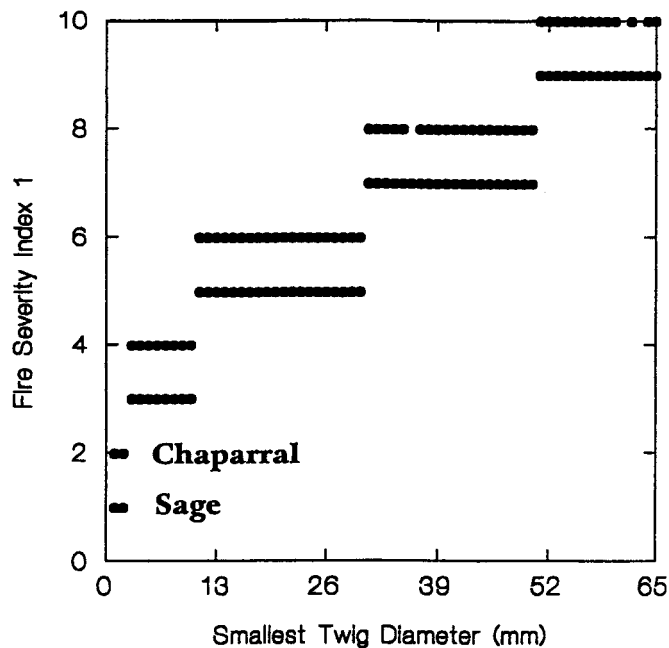


Fig. 2. Fire severity index 1 values assigned to different diameters of the smallest twig on the shrub skeleton nearest to each subplot. For a given twig size, the lower index was assigned to coastal sage shrubs, thus, odd numbers are for coastal sage and even numbers for chaparral.

1-m² subplots. Soil texture was analyzed according to Cox (1985) and nutrient analyses were performed by the soil testing lab in the USDA Forest Service, Riverside Fire Lab on three samples collected in early spring from each site.

Pairwise comparisons were made with either a one-tailed or two-tailed t-test, as appropriate. Correlations were calculated with the Pearson product moment correlation coefficient.

Results

Vegetative Recovery

Peak vegetative recovery was reached in early spring at interior sites and in late spring near the coast. Across all 90 sites percentage ground surface cover ranged from 5% to > 200% (nearly one-third of the sites had > 100% cover, which occurs when plant canopies overlap). Average cover across all 90 sites was 72% (S.E.=5). On most sites annuals dominated the postfire cover (\bar{X} = 50% of total cover, S.E.=3, n=90), but there were exceptions, e.g., on a few sites herbaceous perennials dominated and at other sites suffrutescents (species woody only near base) dominated (Fig. 3).

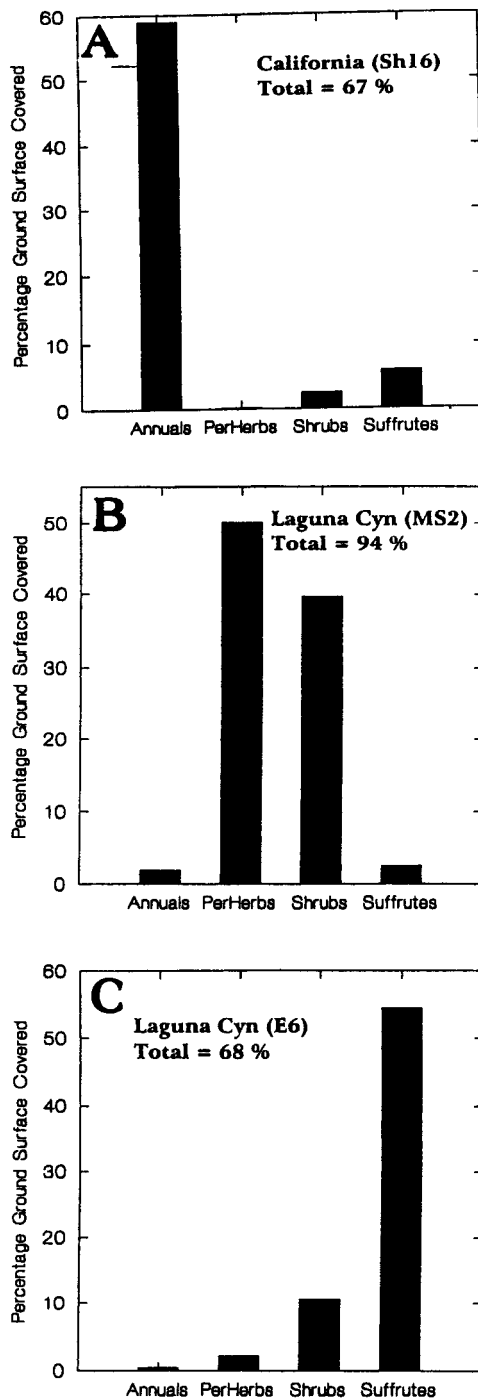


Fig. 3. Cover of annuals, herbaceous perennials, shrubs and suffrutescents at three representative sites.

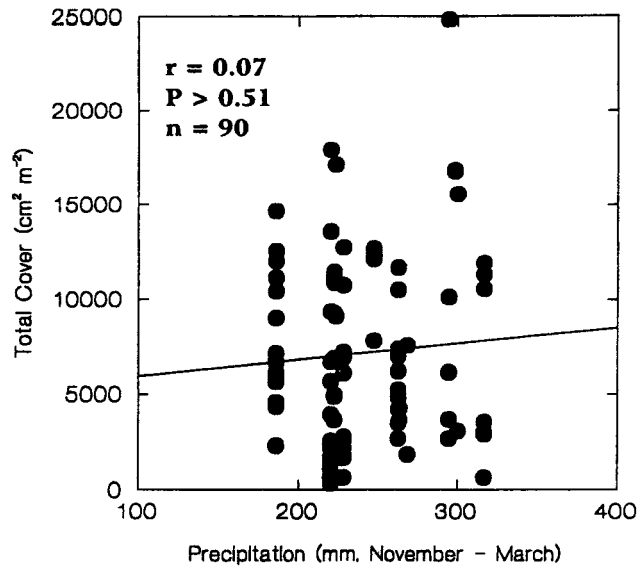


Fig. 4. Correlation between 1993 winter precipitation and total cover of the postfire flora for all 90 sites.

The fact that these fires burned nearly simultaneously, and the observation that regional differences in precipitation can not account for patterns of recovery (Fig. 4), results in an ideal situation for examining how differences in site characteristics and fire parameters affect recovery. For example, annuals increased significantly with distance from the coast (Fig. 5), and parallel to this, herbaceous perennials declined with distance from the coast. This pattern is apparently widespread as postfire data from Ammirati (1967) show annuals more common on interior sites and herbaceous perennials more common near the coast in northern California (700 km north of our sites). In our study weak correlations were evident between growth form and soil texture. Annuals were positively correlated with the percentage of sand in the soil ($r=0.23$, $P<0.05$, $n=90$) and herbaceous perennials were positively correlated with the percentage of clay in the soil ($r=0.26$, $P<0.05$, $n=90$). Thus, annuals appear to be favored on better-drained, hotter interior sites and herbaceous perennials on milder coastal sites with better water-holding capacity.

One parameter of particular interest is the role of fire intensity on postfire recovery. Our two surrogate measures of fire intensity were based on the diameter of the smallest twig on shrub skeletons (index 1) and the height of shrub skeletons (index 2). Mean fire severity index 1 was significantly correlated with mean fire severity index 2 (Fig. 6), which suggests they are measuring the same thing, viz. fire intensity.

Fire intensity, as measured by our fire severity index 1, was negatively correlated with total plant cover in the first postfire season (Fig. 7). Three possibilities present themselves as to how fire intensity could impact postfire

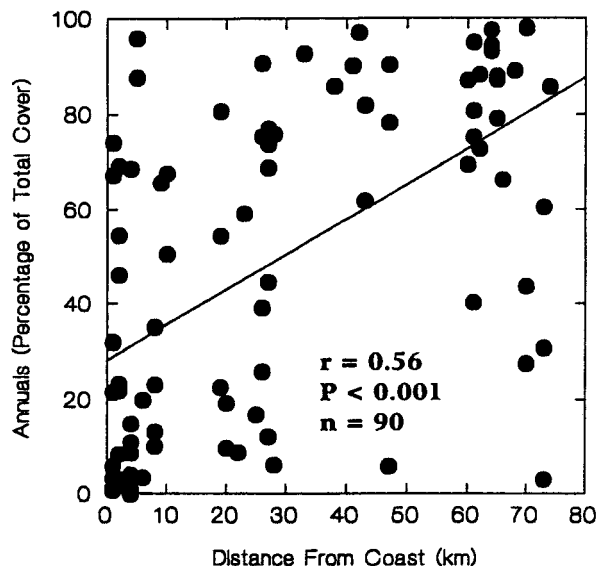


Fig. 5. Correlation between distance from the coast and proportion of total cover contributed by annuals in the postfire flora.

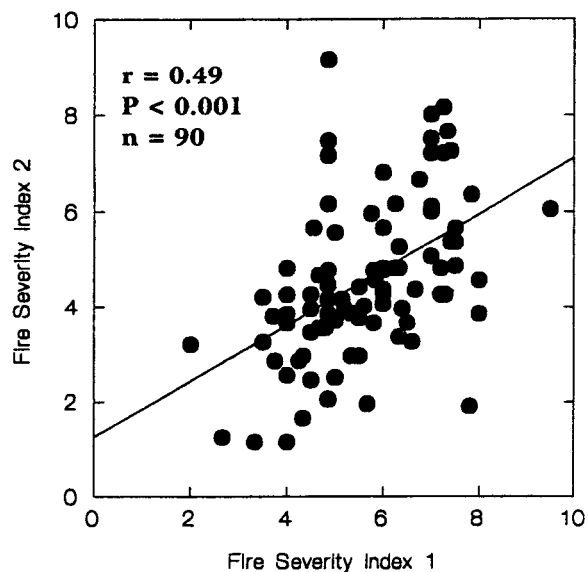


Fig. 6. Correlation between the fire severity index 1, based on diameter of smallest twig on shrub skeletons and fire severity index 2, based on height of shrub skeletons.

recovery: (1) it could alter soil properties, e.g., nutrient status, which could reduce plant production, (2) heat could diminish soil seed banks, or (3) heat could increase mortality of resprouting organs. There is some evidence that all three of these may be involved. Step-wise multiple regression showed

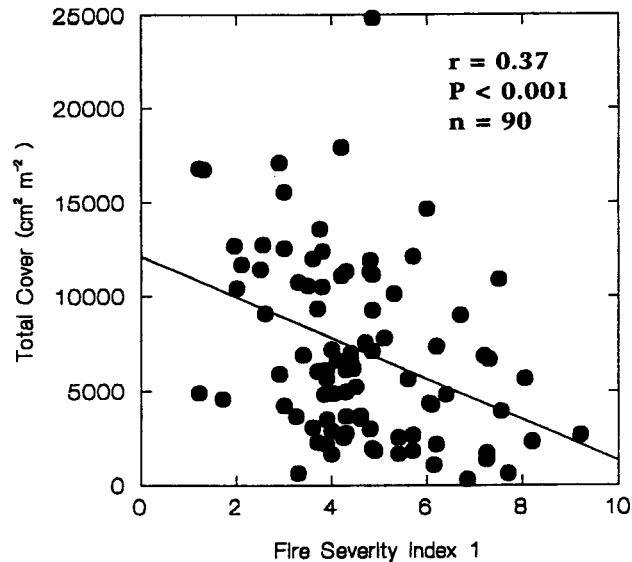


Fig. 7. Correlation between fire severity index 1 and total postfire cover.

that soil phosphorous was negatively correlated with fire severity index 1 and soil nitrogen was negatively correlated with fire severity index 2 (Table 3). However, in both cases these soil parameters were minor components and other factors were of far greater importance, although it is noteworthy that many of the same factors were correlated with both indices.

Evidence that fire intensity might diminish soil seed banks is attested to by the fact that fire severity was negatively correlated with seedling density (Fig. 8). The fact that cover of resprouting herbs and shrubs was negatively correlated with fire severity (Fig. 9) suggests that high fire intensity may affect mortality of resprouting species.

Although many factors affect fire intensity, fuel load is of primary importance (Borchert and Odion 1995). Since fuel load increases with stand age (Philpot 1974), it is not surprising that prefire stand age was significantly correlated with fire severity (Fig. 10). Interestingly, mean fire severity was negatively correlated with the coefficient of variation in fire severity (Fig. 11), suggesting that as fire intensity increases, the within site heterogeneity in fire intensity declines.

Regeneration Modes

In light of the importance of annuals, it is not surprising that the total cover contributed by resprouts was, on average, far less than cover contributed by seedlings; resprouts = 29% S.E.=3 and seedlings = 71% S.E.=3. Distribution of regeneration mode for the 433 species encountered in this study is shown in Table 4. In the perennial postfire flora obligate seeding species are far less

Table 3. Multiple regression of fire severity indices and site factors.

Fire Severity Index 1 (n = 90; r = 0.73; r ² = 0.53; P < 0.001)		Fire Severity Index 2 (n = 90; r = 0.53; r ² = 0.28; P < 0.001)	
Variable	Coefficient	Variable	Coefficient
Constant	0.00	Constant	0.00
Soil clay	-0.67	Soil clay	-0.51
Soil sand	-0.43	Soil sand	-0.41
Distance from coast	-0.35	Prefire age	0.31
Soil rock	-0.29	Solar insolation	0.19
Prefire age	0.28	Soil nitrogen	-0.18
Prefire shrub density	-0.25		
Soil phosphorous	-0.19		
Soil organic matter	-0.19		
Solar insolation	0.14		

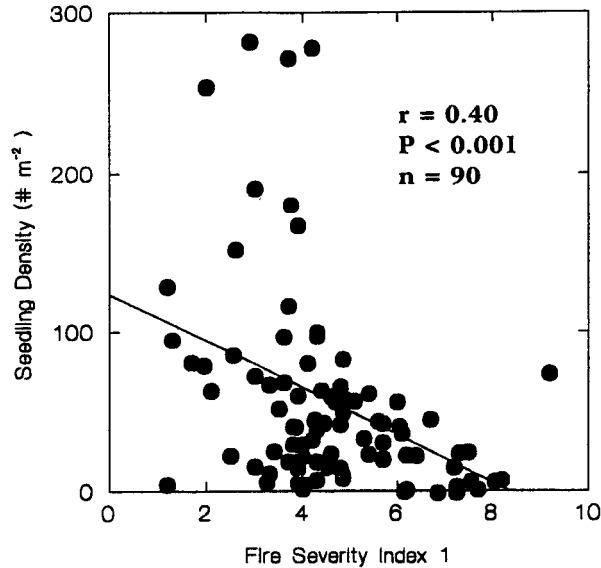


Fig. 8. Correlation between fire severity index 1 and seedling density in the postfire flora.

common in chaparral and coastal sage scrub than obligate resprouting species (Table 4).

It is also apparent from Table 4 that different growth forms are clearly more compatible with one mode of recovery after fire than with others. All but one of the 57 herbaceous perennial species is an obligate resprouter after fire. Postfire resprouting in these taxa is not surprising in light of their life cycle which involves winter resprouting from underground organs that persist through the summer drought. Further thought, however, needs to be given to why the herbaceous perennial growth form has not been selected

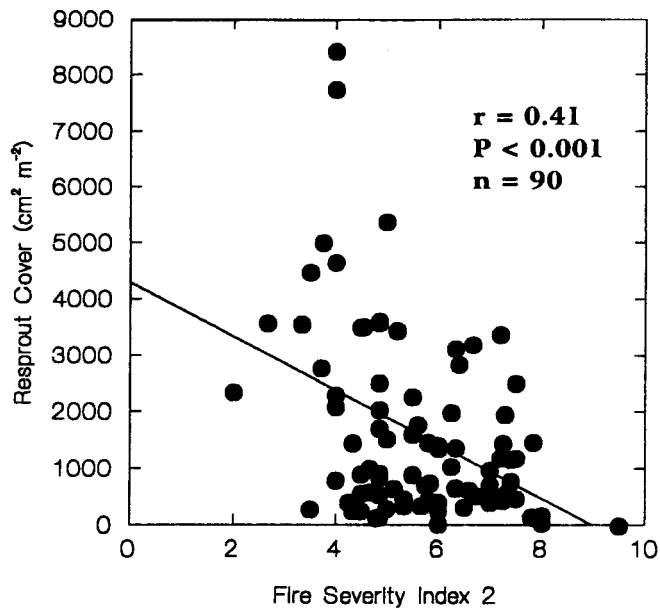


Fig. 9. Correlation between fire severity index 2 and cover by herbaceous and woody resprouts.

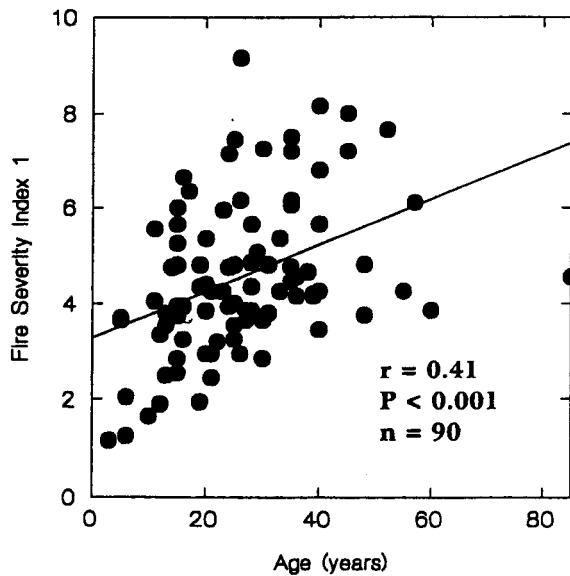


Fig. 10. Correlation between prefire stand age and fire severity index 1.

to maintain dormant soil seed banks that could germinate after fire.

Suffrutescent species die back each year but maintain a woody base aboveground. In many of these, e.g., *Lotus scoparius* and *Helianthemum scoparium*, resprouting after fire is essentially non-existent, however, these

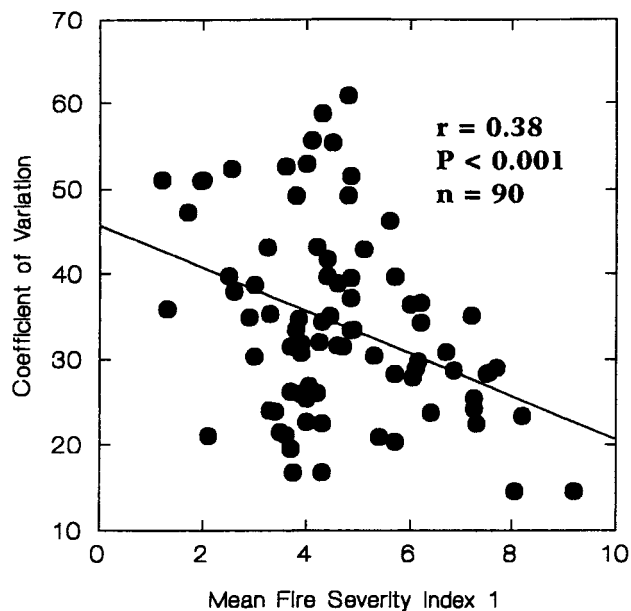


Fig. 11. Correlation between mean fire severity index 1 and the coefficient of variation in fire severity index 1 for subplots within a site.

Table 4. Distribution of postfire obligate seeders, facultative seeders, and obligate resprouters by growth form.

	Number of species in this study		
	Obligate seeders	Facultative seeders	Obligate resprouters
Annuals	274	-	-
Herbaceous perennials	0	1	56
Suffrutescents	11	5	28
Subshrubs	0	3	18
Shrubs	8	8	21

taxa maintain large dormant soil seed banks and seedlings are very common after fire. Coastal sage species with the subshrub growth form are largely obligate resprouters, and two of them, *Eriogonum cinerum* and *Salvia apiana* produce distinct basal burls. Three of these subshrubs, *Artemisia californica*, *Eriogonum fasciculatum*, and *Salvia mellifera* commonly establish seedlings. In the shrub growth form only certain species of *Arctostaphylos* and *Ceanothus* are obligate seeders, although a number of species are facultative seeders.

Ratios relating seedling recruitment to the size of the prefire shrub population for the dominant subshrubs in coastal sage and the dominant shrubs in chaparral are illustrated in Figure 12. In both communities, this ratio

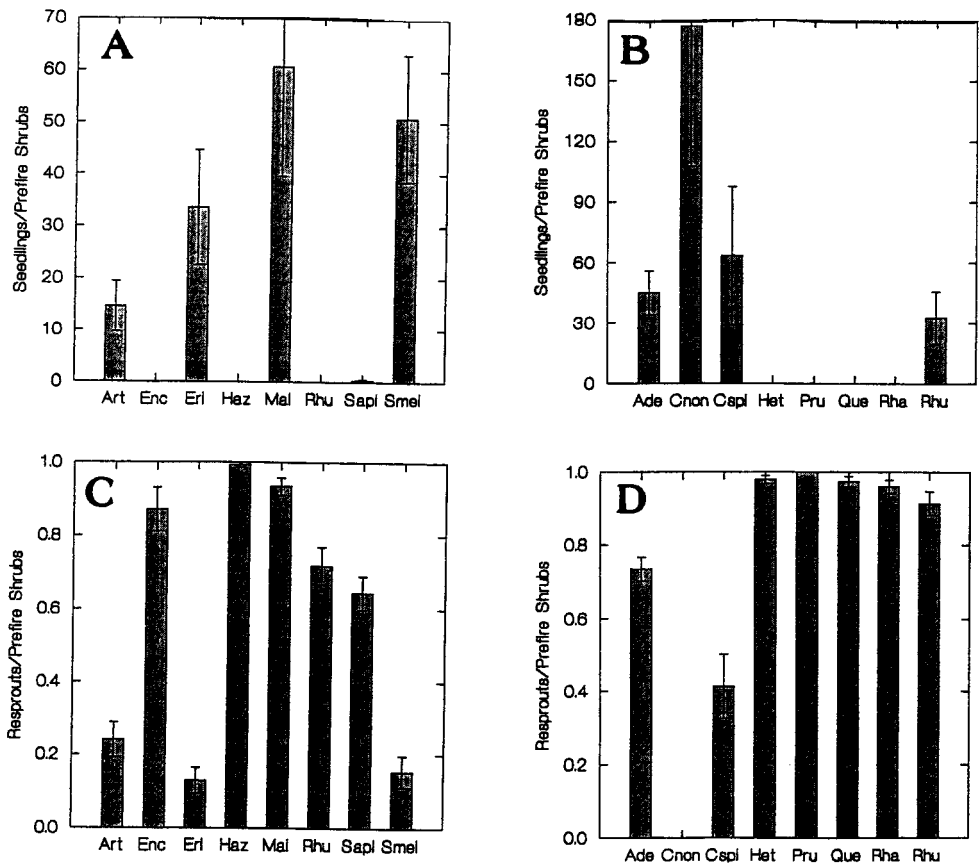


Fig. 12. Ratios of seedlings/prefire shrubs for dominant species in coastal sage scrub (A) and chaparral (B). Ratios of resprouts/prefire shrubs for dominant species in coastal sage scrub (C) and chaparral (D). Species codes and sample size (number of sites) are (A/C): Art=*Artemisia californica* (35), Enc=*Encelia californica* (11), Eri=*Eriogonum fasciculatum* (44), Haz=*Hazardia squarrosa* (35), Mal=*Malosma laurina* (42), Rhu=*Rhus integrifolia* (19), Sapi=*Salvia apiana* (18), Smel=*Salvia mellifera* (40). (B/D): Ade=*Adenostoma fasciculatum* (44), Cnon=non-sprouting species of *Ceanothus* (27), Cspi=*Ceanothus spinosus* (7), Het=*Heteromeles arbutifolia* (20), Pru=*Prunus ilicifolia* (8), Que=*Quercus berberifolia* (21), Rha=*Rhamnus crocea* (50), Rhu=*Rhus ovata* (17).

ranged from 0 to ≥ 60 , and half of the dominant shrub species did not establish seedlings after fire. There is some evidence that fire intensity plays a role in generating these ratios. For example, non-sprouting *Ceanothus* species exhibited a negative relationship between fire severity and the ratio of seedling density/prefire shrub density (Fig. 13).

Species failing to establish seedlings on a site would be extirpated from the site if they did not regenerate by resprouts. The ratio of resprouts/prefire shrubs is also presented in Figure 12. Although coastal sage scrub lacks obligate seeding species, some such as *Artemisia californica*, *Eriogonum fasciculatum*, and *Salvia mellifera* experienced complete mortality on certain

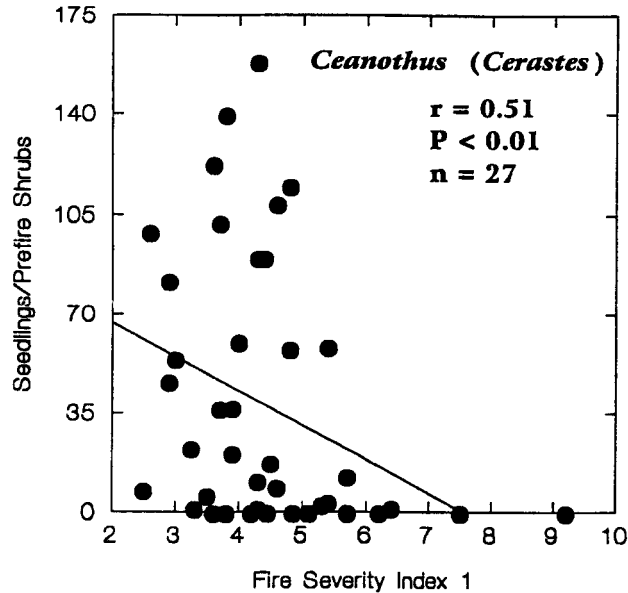


Fig. 13. Correlation between fire severity index 1 and seedling/prefire shrub ratio for non-sprouting species of *Ceanothus*.

sites and throughout their range they averaged $\leq 25\%$ resprouting (Fig. 12). In chaparral, *Adenostoma fasciculatum* varied from 0% of the population resprouting at some sites to 100% at other sites.

Reasons for variable resprouting success in some taxa but not others have not previously been offered in the literature. Two factors are potentially involved; plant size and fire intensity. To correlate resprouting with these two parameters we did extensive sampling of shrub skeletons of all dominant subshrubs and shrubs, measuring the height and basal diameter of skeletons on dead individuals and on resprouting individuals (Table 5). In shrubs such as *Adenostoma fasciculatum*, *Ceanothus spinosus*, and *Rhus integrifolia*, resprouting individuals had skeletons that were 65-88% taller than skeletons that failed to resprout, suggesting that dead shrubs had been burned much more intensely. Prefire shrub stem diameters, however, were relatively similar between resprouting and non-resprouting individuals suggesting little difference in age between dead and resprouting individuals. These data support the hypothesis that resprouting is more likely from individuals burned less intensely and this is consistent with the observation that *Adenostoma fasciculatum* resprouting success declined with increases in our surrogate measure of fire intensity based on twig diameter (Fig. 14).

This pattern with shrubs is in marked contrast to the pattern observed in subshrubs (Table 5), in particular *Artemisia californica* and *Eriogonum fasciculatum*. In these subshrubs, resprouting individuals had only slightly taller skeletons than dead individuals. Surprisingly, however, the skeleton diameter on resprouting individuals was approximately 60% smaller than on dead

Table 5. Basal diameter and height of postfire shrub skeletons for non-resprouting (dead) individuals and resprouting (alive) individuals of the subshrubs and shrubs in coastal sage scrub and chaparral exhibiting variable resprouting success.

	Basal diameter (mm) [$\bar{X} \pm SE$ (n)]	Height (cm) [$\bar{X} \pm SE$ (n)]
Shrubs		
<i>Adenostoma fasciculatum</i>		
Dead	31 \pm 1 (851)	37 \pm 2 (851)
Alive	32 \pm 1 (1430)ns	101 \pm 2 (1430)***
<i>Ceanothus spinosus</i>		
Dead	48 \pm 3 (131)	53 \pm 8 (131)
Alive	58 \pm 3 (169)*	201 \pm 11 (169)***
<i>Rhus integrifolia</i>		
Dead	50 \pm 2 (163)	13 \pm 2 (163)
Alive	69 \pm 2 (364)***	106 \pm 6 (364)***
Subshrubs		
<i>Artemisia californica</i>		
Dead	32 \pm 1 (952)	19 \pm 1 (951)
Alive	11 \pm 1 (376)***	32 \pm 2 (376)***
<i>Eriogonum fasciculatum</i>		
Dead	24 \pm 1 (1454)	7 \pm 1 (1454)
Alive	10 \pm 1 (256)***	10 \pm 1 (256)***
<i>Salvia apiana</i>		
Dead	22 \pm 1 (200)	13 \pm 1 (200)
Alive	22 \pm 1 (326)ns	24 \pm 1 (326)***
<i>Salvia leucophylla</i>		
Dead	33 \pm 1 (116)	42 \pm 4 (116)
Alive	19 \pm 1 (247)***	49 \pm 3 (247)ns
<i>Salvia mellifera</i>		
Dead	34 \pm 1 (973)	18 \pm 1 (973)
Alive	21 \pm 1 (183)***	52 \pm 3 (183)***

Results of statistical tests: ns= not significant ($P>0.05$); *= $0.05 \geq P > 0.01$; **= $0.01 \geq P > 0.001$; ***= $P \leq 0.001$

individuals. Indeed, in both species, 95% of the resprouting individuals had skeleton diameters ≤ 25 mm, whereas more than half of the skeletons of non-resprouting individuals were ≥ 25 mm and many were twice that size. Generally stem diameter is strongly correlated with age (Keeley 1992), thus it appears that resprouting individuals were the younger individuals and this pattern is also evident in two of the other three subshrub species (Table 5).

In summary, it is apparent that in shrubs such as *Adenostoma fasciculatum*, resprouting individuals are either not different in age or older than non-sprouting individuals. This is in marked contrast to subshrubs, where the younger individuals are the ones more likely to resprout. I suggest that the latter pattern is the result of resprouting being a juvenile trait in these subshrubs, a trait that is lost with age. In the genera *Artemisia*, *Eriogonum* and *Salvia*, subshrubs are uncommon and the majority of taxa are herbaceous perennials, which, as noted above (Table 4) are typically vigorous resprouters. Cladistic analysis would support the evolution of these coastal sage scrub

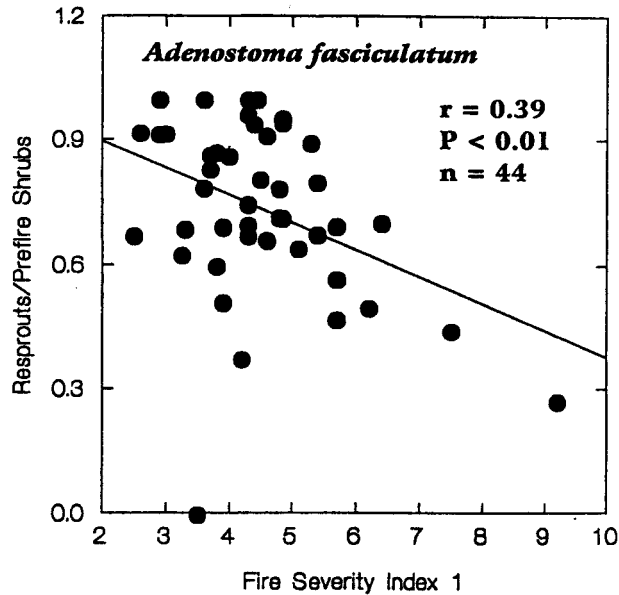


Fig. 14. Correlation between fire severity index 1 and resprouts/prefire shrub ratio for *Adenostoma fasciculatum*.

subshrubs from herbaceous perennial ancestors. Thus, resprouting in juveniles is not surprising, and it appears that the propensity for resprouting declines as the woody habit increases with age in these taxa. This conclusion has important implications for prescribed burning programs in coastal sage scrub.

Diversity and Species Richness Patterns

An example of the pattern of changes in species richness with scale is illustrated in Figure 15. Although the number of species varied from site to site, the pattern observed for the sites in Figure 15 was similar for all 90 sites. This is fascinating because it is in marked contrast to the pattern observed for other Mediterranean-type ecosystems, although those data are for communities of varying ages after fire (Fig. 16). Relative to California chaparral and coastal sage, these other Mediterranean-type communities show a greater rate of species addition over the scale from 1 to 100 m² and a lower rate of species addition over the scale from 100 to 1000 m². This pattern suggests that the processes underlying maintenance of species richness in these communities may differ from the processes controlling species richness in Californian ecosystems.

It is apparent that one factor affecting species richness in our communities is fire intensity (Fig. 17). Also species richness increased significantly with distance from the coast ($r=0.61$, $P<0.001$, $n=90$), and with cover by annuals ($r=0.49$, $P<0.001$, $n=90$). Within site heterogeneity in species richness (measured by the coefficient of variation between plots) was negatively correlated

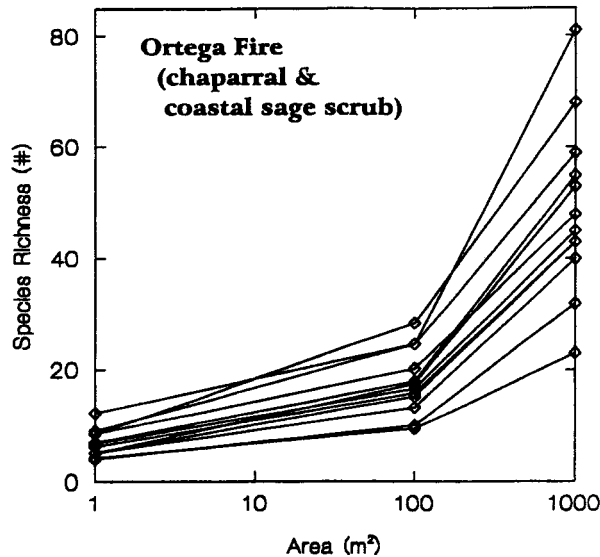


Fig. 15. Species richness at 1 m², 100 m², and 1000 m² for representative sites in chaparral and coastal sage scrub in this study.

with species richness, reflecting that as species richness increases, number of species per plot are more evenly distributed across the site (Fig. 18).

Deterministic models of species diversity (e.g., Tilman 1982) are based on the idea that species diversity is determined by niche diversity. A corollary to this is that, if resource competition is driving diversity patterns, then species richness will be correlated with growth form diversity (Cody 1991). In Californian chaparral and coastal sage there exist a diverse array of growth forms and different functional types. My attempt at recognizing those encountered in this study (Table 6) distinguished 33 functional/growth forms for 433 species. Considering all 1800 subplots sampled in this study there was a highly significant correlation between growth form richness and species richness (Fig. 19), supporting the hypothesis that competition leads to increased diversity in our ecosystems.

Postfire Management

Due to the close juxtaposition of flammable wildlands and urban development in southern California, management of burned brushlands is of major concern to resource agencies. For over 50 years, both state and federal agencies have used aerial seeding programs for the purpose of enhancing postfire plant cover and thus reducing erosion. However, increasingly this form of management is attacked for a variety of reasons (Keeley and Scott 1995). One criticism lodged against this form of management is that the species traditionally used (*Brassica* spp. originally, but in recent decades *Lolium multiflorum*, *Vulpia myuros* and *Trifolium hirtum*) are aggressive non-

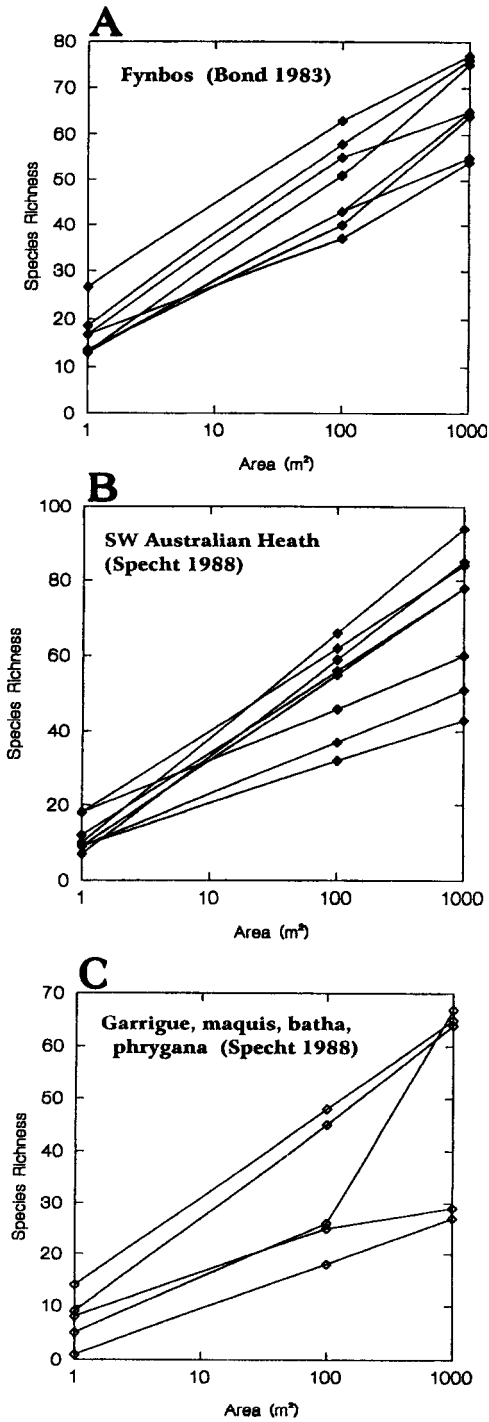


Fig. 16. Species richness at 1 m², 100 m², and 1000 m² for representative sites in Australian heath (A), South African fynbos (B) and Mediterranean Basin garrigue, maquis, and batha (C).

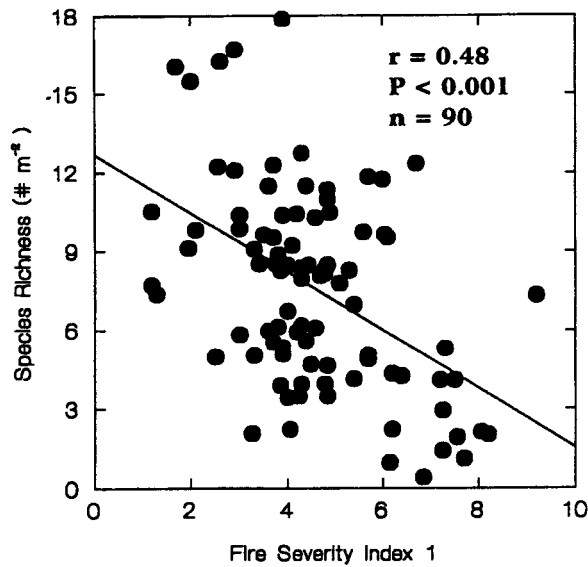


Figure 17. Correlation between fire severity index 1 and species richness at 1 m².

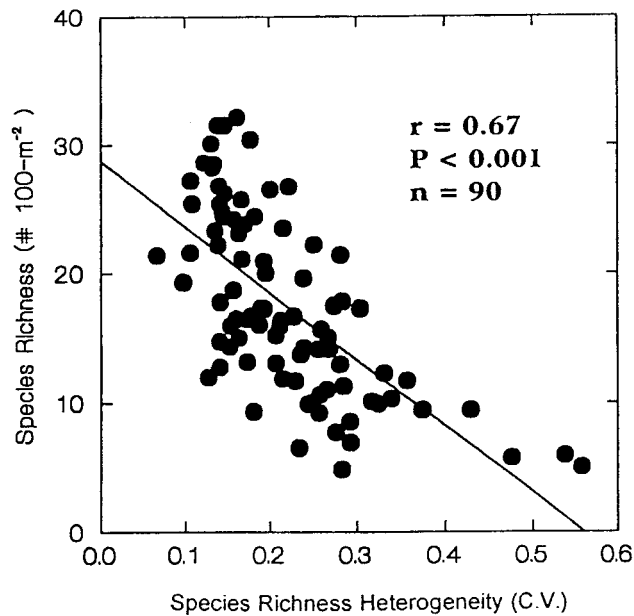


Fig. 18. Correlation between species richness heterogeneity (measured by the coefficient of variation within a site) and mean species richness for the site.

native species that displace the native flora and have detrimental impacts on ecosystem recovery (Conard *et al.* 1995). To counter this criticism, some agencies have utilized, on a very limited scale, native species in the seed mix applied to recent burns.

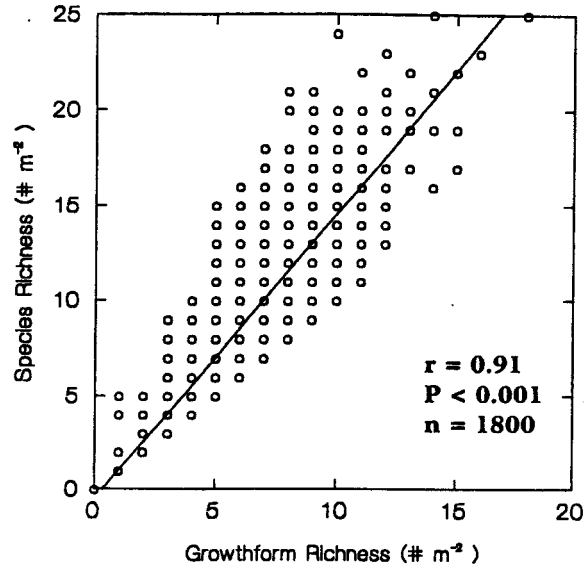


Fig. 19. Correlation between growthform richness and species richness in 1 m² all 1800 subplots distributed across 90 sites.

Out of our 90 sites, 24, distributed on three different burns, were aerially seeded by either state or federal agencies within 1-2 weeks following fire. Our data showed that by late winter, after the threat of soil erosion had passed, cover contributed by seeded species was less than 15% ground surface cover on 21 of the 24 sites (Fig. 20). In general seeding on the Kinneloa Fire had the greatest success in enhancing cover and the least success on the Old Topanga Fire. *Vulpia myuros* was the dominant seeded species on all three sites, although there were marked differences in the seeding density (Table 7) and it seems likely that seeding density may have contributed to the greater success of seeding on the Kinneloa Fire. On all three burns, California native species were a part of the seed mixture, however, on nearly all sites, these species failed to establish or represented a minor portion of the cover. The fact that these species comprised a minor portion of the seed mix is one factor accounting for their poor establishment. Other reasons, however, include the poor choice of species, since four of the six native species were herbaceous perennials, a growth form that almost never establishes from seed in recent burns (Table 4).

In conclusion, relative to the natural regeneration, cover contributed by seeded species was relatively insignificant (Fig. 20) and could not possibly justify the nearly one million dollars spent on these projects.

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Table 6. Functional growthform categories recognized for the 433 species encountered in this study.

Trees	Herbaceous perennials
Evergreen	Geophyte
Winter deciduous	Geophyte (summer flowering)
Shrubs	Vine
Evergreen	Cespitose
Evergreen (summer flowering)	Caulесcent
Summer deciduous	Monopodial
Summer deciduous (summer flowering)	Prostrate
Winter deciduous	Grass
Cactus	Fern
Yucca	Annuals
Monopodial	Cespitose
Subshrubs	Caulесcent
Evergreen	Caulесcent (summer flowering)
Summer deciduous	Monopodial
Summer deciduous (summer flowering)	Prostrate
Suffrutescents	Vine
Caulесcent	Grass
Monopodial	
Prostrate	
Lianas	

Table 7. Seeding prescriptions used on the three fires actively managed with aerial seeding. (*) Indicates non-native.

Fire	Seeding density (kg/ha)	Cost	Species	% by count
Kinneloa	15.4	\$158,600 (\$112/ha)	* <i>Vulpia myuros</i>	59
			* <i>Lolium multiflorum</i>	15
			* <i>Trifolium hirtum</i>	10
			<i>Lotus scoparius</i>	10
			<i>Eschscholzia californica</i>	6
Laguna Cyn	9.1	\$463,000 (\$378/ha)	* <i>Vulpia myuros</i>	95
			<i>Nasella pulchra</i>	3
			<i>N. cernua</i>	1
			<i>N. lepida</i>	1
Old Topanga	5.4	\$438,600 (\$87/ha)	* <i>Vulpia myuros</i>	56
			* <i>Trifolium hirtum</i>	22
			* <i>Bromus hordeaceus</i>	19
			<i>B. carinatus</i>	3

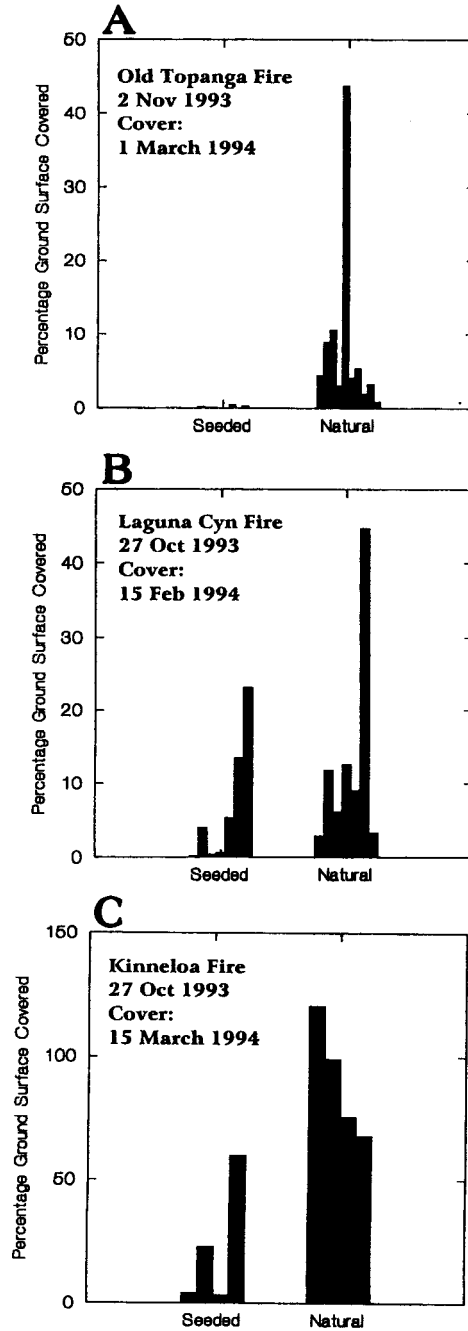


Fig. 20. Percentage ground surface cover by seeded and natural regeneration at the end of winter 1994 at the three sites managed by aerial seeding.

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