8. Perspectives on Fire Management in Mediterranean Ecosystems of Southern California

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San Dimas Canyon seems a wild place beyond the reach of civilization. It is home to black bears, gray foxes, Anna's hummingbirds, scrub jays, and in early summer, a multitude of biting insects. Along the steep, north-facing hillsides, the chaparral has the appearance of an ancient forest. From within the canyon it is difficult to remember that one is less than 7 km from metropolitan Los Angeles. It is also difficult to conceive of the landscape swept by flames 30- or 40-m high, or to visualize San Dimas Creek afterwards scoured by debris flows. Our difficulty in perceiving these catastrophic events makes it difficult to alter their course and consequences because to do so involves substantial cost and risks.

Catastrophic fires are undoubtedly part of the natural environment in southern California, given its Mediterranean-type climate and the flammable chaparral that results. This natural flammability, compounded by the common coincidence of human-caused ignitions with high winds and temperatures, has in recent decades produced a series of large and exceptionally severe fires. Since 1960, wildfires have consumed more than 350,000 ha in Los Angeles County (Los Angeles County Fire Department, records on file). There the larger fires dominate the landscape. Since 1919, the largest 10% – those greater than 2400 ha – have accounted for five-sevenths of the total area burned (Figure 8–1).

Fire prevention and suppression have undoubtedly reduced losses to a fraction of those that society and the environment would otherwise have sustained. But the immediate losses from recent wildfires have been so

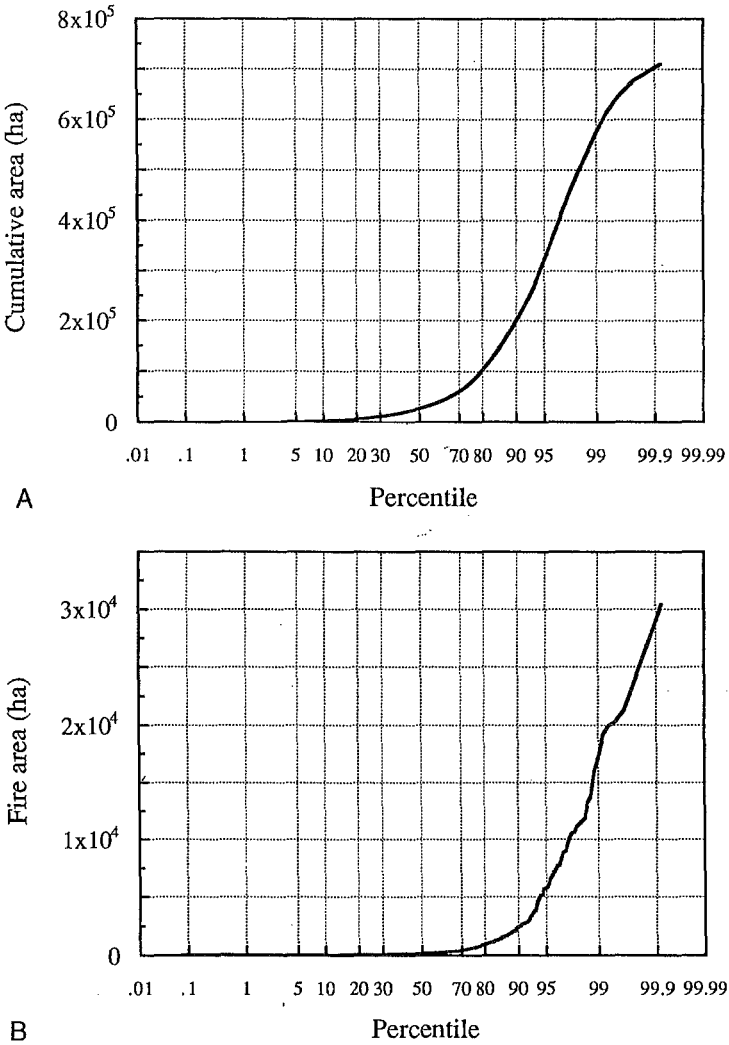

Figure 8–1. **(A)**, Cumulative area in Los Angeles County, CA, burned by all wildfires of area greater than 40 ha, 1919 to 1990, expressed as a percentile of fire number from smallest to largest (Los Angeles County Fire Department, unpublished records on file). **(B)**, Distribution of individual fire sizes as in *A*. These data show that larger fires have been dominant. Over the period, the largest 10 percent – those greater than 2400 ha – have accounted for five-sevenths of the total area burned.

high as to show that the current balance between ignitions and suppression is unacceptable. One needs only to cite the case of the Paint Fire, in Santa Barbara County, which was ignited late in the afternoon of June 26, 1990. It consumed 1800 ha of chaparral and planted landscaping and

destroyed or heavily damaged 648 residences and unattached structures, 221 apartment units, and 11 County buildings (Federal Emergency Management Agency 1990). Most of the destruction occurred during the first evening as local, adiabatically-heated Sundowner winds drove the fire. Estimated costs of the Paint Fire were $250 million in assessed valuation and $2 million for fire suppression (Federal Emergency Management Agency 1990). With continuing urban expansion into wild lands, the losses from catastrophic fires such as this will only accelerate.

There are other serious but less immediate impacts from catastrophic fires. Postfire floods continue to threaten life and property. Fire is a predominant force behind the 750,000 m^3 of soil and rock eroded annually on average in the watershed of the San Gabriel River (S. Kumar, Los Angeles County Department of Public Works, personal communication). This material must be trapped and dealt with before it reaches the urbanized flood plain. Fires accelerate the nitrate pollution of local waters in mountain regions that are subject to heavy air pollution (Riggan et al. 1994a). Frequent burning exacts a toll on native woodlands and has caused some coastal sage scrub communities (for instance, along the mouth of San Gabriel and San Dimas canyons in the San Gabriel Mountains) to yield to invasive European grasses.

Prescribed fire provides a way to mitigate the impacts of catastrophic wildfires by managing the mosaic of natural vegetation. This strategy has been adopted by Federal, state, and local agencies, yet its application is threatened by the primacy of short-term interests over long-term needs.

Ironies abound: In an era of budget cuts, fire suppression is essential, of course, but actions that might reduce the potential for severe fires are not. Fire suppression can be made more efficient by dispatching firefighters nationally, which reduces their availability for prescribed burning during critical months in southern California. Although wildfires emit tremendous quantities of air pollution, they are not conscious acts of society. Prescribed fires are and thus can be regulated and taxed to a point of ineffectiveness. Major wildfires in summer can destroy a multitude of small animals including endangered species, but inadvertent mortality of a smaller number during prescribed burning could be interpreted as a violation of migratory bird treaties and of the Endangered Species Act. When vegetation becomes most flammable from drought and dieback, and prescribed burning is most urgently needed, prescribed burning entails the greatest risk and is least likely to be used. The cumulative impact of legitimate questions about the use of fire could be to so restrict its application as to guarantee the reign of severe fire. Clearly, we need a broader perspective on how we deal – or fail to deal – with fire and ecosystem process in southern California.

Our purpose in this chapter is to examine some of the consequences of managing fire with fire in southern California. These depend on the structure of biomass and the rate it accumulates, the fire behavior that

results from that fuel, the relative environmental impacts of moderate-intensity or severe fires, and the potential for intervention.

We argue as follows:

1. Chaparral structure and flammability change year by year during stand development with as much as a threefold increase in energy release during burning between ages 7 and 22 years in some *Ceanothus* communities. Flammability in the *Ceanothus* has also been altered, perhaps dramatically, by climate extremes during the past decade.
2. Postfire flood peaks, sedimentation, and water quality are related to fire severity and should be manageable through the use of prescribed fire to create a mosaic of vegetation that differs in flammability, thereby reducing the likelihood, potential size, or severity of catastrophic fires. Yet, since we cannot readily predict when fire can spread through young chaparral, we cannot specify the level of intervention necessary to achieve substantial reduction in wildfire losses and costs.
3. The concentrations of several radiation-absorbing gases have been rising relentlessly in the atmosphere and raising expectations that global climate will be altered. If climate does change, one of the most important responses to track in southern California will be the potential for catastrophic fires, but this is most affected by extreme conditions that will be difficult to predict.

How Do Chaparral Fuels Develop?

Chaparral fires are fueled by foliage, live stems of small diameter, and deadwood; the spatial arrangement and biomass of these change greatly through the first few decades of postfire community development. *Ceanothus*-dominated chaparral, the development of which has been extensively studied, accumulates above-ground biomass at an exponential rate through at least 2 decades after stand establishment (Riggan et al. 1988). Stand basal area rises exponentially through at least 3 decades (USDA Forest Service records, Riverside, CA). The biomass accumulation depends markedly on site productivity. Biomass of closed-canopy chaparral can range at least from 2 to 12 kg/m^2 at 20 years of age (Figure 8–2). Biomass accumulation rates at older ages are not well established because estimates have been made from only a few chronosequences that confound site factors with age (Marion and Black 1988; Specht 1969).

Foliage biomass of closed-canopy stands of *Ceanothus crassifolius* (hoaryleaf ceanothus) approaches a maximum after 15 to 20 years, although it is likely that there is considerable annual variation about this trend. For instance, we noted that two age classes of foliage were retained through the dry season in 1983, which followed several years of plentiful rainfall (Riggan et al. 1988), yet only one age class was retained through

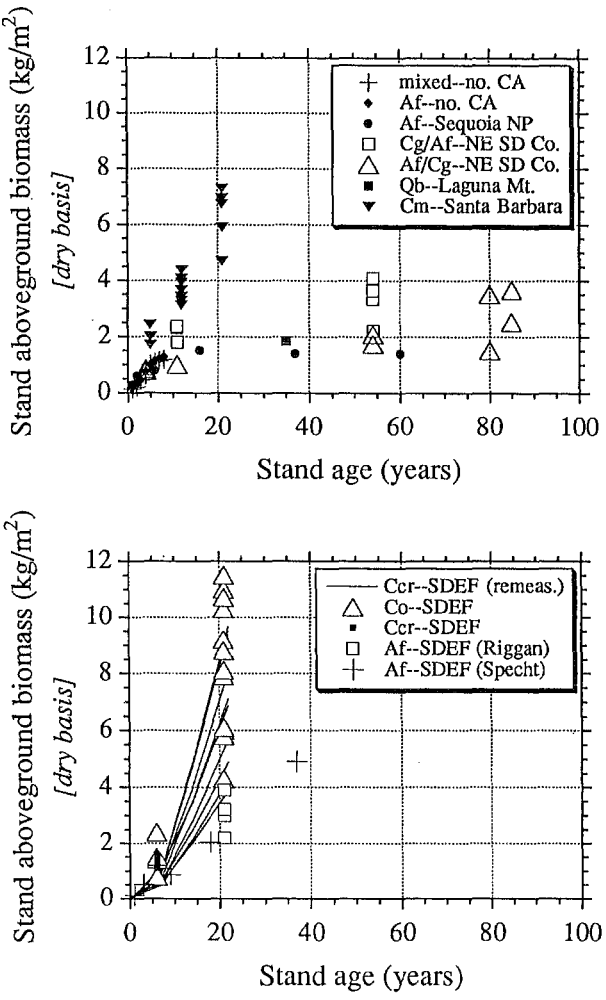

Figure 8–2. Trends in accumulation of above-ground biomass in chaparral. Connected symbols are from reconstructed or remeasured growth. Unconnected symbols represent measurements from stands of different ages within a given region. Production varies markedly across sites, so age does not account for a high proportion of the variance. Stands older than 20 years of age, which are largely from northeastern San Diego County, are not well represented in the data. References are as follows: Cm-Santa Barbara, *Ceanothus megacarpus* (Schlesinger and Gill 1980); mixed and Af-no. CA, mixed chaparral and *Adenostoma fasciculatum* from Mendocino and Shasta counties, CA (Sampson 1944); Af-SDEF, *A. fasciculatum* at San Dimas Experimental Forest (prior to the Johnstone Fire in 1960) (Specht 1969); Ccr, Co, and Af-SDEF, *C. crassifolius*, *C. oliganthus*, and *A. fasciculatum* at San Dimas Experimental Forest (subsequent to the Johnstone Fire in 1960); Qb-Laguna Mt., *Quercus berberidifolia* in southern San Diego County (Riggan et al. 1988); Cg/Af and Af/Cg-NE SD Co. (with species order denoting dominance), *C. greggii*, *A. fasciculatum* (with elements of *A. sparsifolium* (redshanks) and *Quercus berberidifolia*) in northeast San Diego County (Marion and Black 1988); Af-Sequoia NP, *A. fasciculatum* at Sequoia National Park (Rundel and Parsons 1979).

the years of low rainfall from 1984 through 1991. Foliage biomass probably approaches a maximum when the developing leaf area of dominant shrubs hastens the onset of seasonal water stress and the effects of this extended physiological drought are no longer balanced by enhanced carbon uptake during the early growing season or more efficient competition for water with smaller plants (Riggan et al. 1988). Foliage biomass of mature chaparral may also vary substantially across sites; measured values in 21-year-old *Ceanothus crassifolius* stands ranged almost threefold from 0.5 to 1.3 kg/m^2 (Riggan et al. 1988).

As chaparral ages, deadwood slowly accumulates through progressive shading of lower branches, drought- or pathogen-induced declines or dieback, intraspecific competition, and random mortality such as that from snow or freezing. Accumulation rates vary according to the life histories of the predominant species (nomenclature follows that of the Jepson Manual [Hickman 1993]):

Quercus berberidifolia (scrub oak) accumulates little standing deadwood, which constitutes only one-tenth of the above-ground biomass in mature stands 21 to 37 years of age (Riggan et al. 1988).

Adenostoma fasciculatum (chamise) apparently accumulates little deadwood in low productivity environments (0.07 of the total) and a substantial fraction (0.45 to 0.48) in high-productivity stands, especially where subject to competition from tall *Ceanothus*, such as along the coastal slope of the San Gabriel Mountains (Riggan et al. 1988). Median fraction deadwood across 15 southern California locations has been reported to be 0.23 (Paysen and Cohen 1990). Shoot mortality is probably high when a sequence of years with relatively high rainfall are followed by prolonged drought (T. Paysen, personal communication).

Ceanothus crassifolius and *C. megacarpus* (big-pod ceanothus) are little affected by shading and competition. Deadwood in the lower canopy of live *C. crassifolius* at the San Dimas Experimental Forest constitutes less than 0.01 of above-ground biomass at age 6, and 0.12 at age 21. Mortality of small shrubs, which occurs primarily between the ages of 5 to 10 years, can raise the fraction of deadwood to 0.19 of the community's above-ground biomass by age 21 (Riggan et al. 1988). Accumulated deadwood in *C. megacarpus* constitutes 0.23 of standing biomass at this age (Schlesinger and Gill 1980). Several of the Ceanothus species are also subject to high, pathogen-induced mortality rates as we discuss later.

Salvia mellifera (black sage) is a semiwoody shrub of low stature which, in competition with tall *C. crassifolius*, accumulates a substantial biomass of deadwood, constituting as much as one-third of above-ground biomass at age 21 years (Riggan et al. 1988). *Salvia mellifera* can be widely established when postfire regeneration of competing taller shrubs such as *Ceanothus* is poor, thereby increasing the probability of burning at young ages or under moderate weather (Riggan et al. 1988).

Snow or frost is occasionally an important cause of mortality. The 40,000-ha Marble Cone Fire of 1977 was an especially severe chaparral

fire fueled by stems broken by a wet, heavy snowfall (T. Plumb, personal communication). Frost also occasionally kills stems and foliage of *Malosma laurina* (laurel sumac), thereby producing scattered dead shrubs as was widely observed in the winter of 1990–91 (D. Neff, California Department of Forestry and Fire Protection, personal communication). This mortality is most likely to affect fire behavior during the limited time that dead foliage is retained on the plant. Exceptionally low temperatures in the absence of an insulating snow cover in the winter of 1990–91 also caused widespread mortality of *Arctostaphylos* spp. in stands regenerating after the 1987 wildfires in northeastern California (California Department of Forestry and Fire Protection, personal communication).

Chaparral Dieback in Southern California

Extensive chaparral dieback may have caused an abrupt increase in fire potential in southern California beginning in 1984 and 1985 (Figure 8–3). This mortality appears to have been most pronounced on coastal slopes in the San Gabriel and the Santa Ynez mountains, in the Santa Monica Mountains, and in southern San Diego County. Accelerated mortality has been most notable in *Ceanothus crassifolius* and *C. megacarpus*, although several chaparral species have been affected.

Climatic Stress

Severe drought is most likely the primary stress factor that predisposed the affected chaparral to infection by a fungal pathogen. We first observed dieback to be extensive in the San Gabriel Mountains after the hydrologic year beginning October 1, 1983, during which precipitation was in the eleventh percentile of a 107-year record (as measured at Tanbark Flats, San Dimas Experimental Forest [USDA Forest Service, Riverside, CA; unpublished records on file]). More important, nine-tenths of the autumn and winter rains that year fell before the end of December; the largest storm during the succeeding 11 months occurred during August and brought only 1.3 cm of rain. The 1983–84 drought followed a hydrologic year with twice the average annual rainfall, during which an abundance of foliage developed. With an extensive leaf area, shrubs probably exhausted available soil water and suffered substantial water stress even during the growing season. The dieback progressed annually through 1992 with a second peak in activity late in 1986 after the fourth lowest annual rainfall on record.

A similar pattern of abundant rainfall, followed by a tenth-percentile drought occurred during hydrologic years 1979–80 and 1980–81; however, no extensive dieback occurred. This lack of response may have been because rain during the low-precipitation year of 1980–81 fell largely from late January through early April, coincident with the growing season.

Figure 8–3. Dieback in *Ceanothus crassifolius* in Ham Canyon in the San Dimas Experimental Forest, March 1985.

The initial pattern of dieback may also have been linked with topographic gradients in water stress. We observed in Bell Canyon on the San Dimas Experimental Forest that the dieback was most common on the steepest and driest slopes where *Ceanothus crassifolius* is mixed with *Adenostoma fasciculatum*. Ridge lines dominated by the *Ceanothus* were least affected.

Another unusual weather pattern during August 1984 may have affected the dieback. A subtropical air mass over southern California caused extended high temperatures and humidity with occasional light rain. This rain could have facilitated dispersal of fungal spores as the high temperatures and moisture promoted subsequent secondary infections.

Botryosphaeria dothidea *and Its Mode of Action*

An opportunistic pathogen, *Botryospheaeria dothidea* (or *B. ribis*), was identified in association with dieback of stems of *Ceanothus crassifolius*, *C. oliganthus*, *C. megacarpus*, *C. spinosus*, *C. leucodermis*, *Heteromeles arbutifolia*, *Arctostaphylos glauca*, *A. glandulosa*, and *Quercus agrifolia* (J. Pronos, USDA Forest Service, San Francisco, CA; personal communication). Subsequent inoculation trials have established that *Botryosphaeria dothidea* can reproduce the dieback symptoms (Brooks and Ferrin 1991).

Botryosphaeria dothidea is a common canker fungus (Hodges 1983) that is probably ubiquitous in the chaparral environment. It can attack a variety of hosts, including *Cornus stolonifera* (red osier dogwood) (Schoeneweiss 1979), *Sequoiadendron giganteum* (giant sequoia), *Sequoia sempervirens* (coast redwood) (Worrall et al. 1986), and *Pinus taeda* (Loblolly pine) (Hodges 1983). Chaparral dieback in 1984 and 1985 was coincident with widespread mortality in urban vegetation, including plantings of *Ceanothus* and *Arctostaphylos* at the Rancho Santa Ana Botanic Garden in Claremont, CA. Dieback in chaparral also was apparently widespread circa 1948 to 1951 (Kittredge 1955) and 1958 to 1961 (Pirsko and Green 1967) during periods of record-low rainfall.

Botryosphaeria dothidea is considered a facultative parasite and occurs naturally as a saprophyte on dead and dying plant tissue. Infection proceeds through wounded plant tissue (Schreiber 1964; Worrall et al. 1986) or natural openings such as lenticels (Shearer et al. 1987) or leaf scars (Spiers 1977). Tissue entry does not assure spread of the pathogen. The host is generally attacked when subject to a predisposing stress, which can occur after initial entry. Possible stresses to the host in this instance could be abnormally low water potential (Crist and Schoeneweiss 1975; Schoeneweiss 1979, 1981), prolonged duration of water stress, defoliation (Schoeneweiss 1981), or possibly impact of air pollution (Hinrichsen 1987; Schoeneweiss 1975). Plants do mount defenses to the spread of the pathogen within a stem, as shown by the canker, but this defense may be

inadequate under prolonged stress conditions. Fungal hyphae progressing through vessel elements in a stem restrict water movement to the distal leaves. Once the hyphae infect a stem below the point at which live leaves are attached, the entire stem is killed.

Levels of Mortality

We estimated the accumulation of deadwood in *Ceanothus crassifolius* chaparral after the initial dieback at Lodi and Ham canyons at the San Dimas Experimental Forest (Basal area was measured 50 cm from the base of each stem found in 33 plots, each 120 m^2 in area, in which *Ceanothus crassifolius* accounted for more than one-half of the basal area. The fraction of deadwood within live stems was measured for a sample of 183 stems destructively sampled in 1986 and 1988 at Lodi and Ham Canyons; variance of the ratio estimate was 4.8×10^{-4}. Woody tissue at any point was classified as live if any live foliage remained distal to that point along a branch.) Deadwood constituted 0.12 of the woody biomass within live stems. Dead stems accounted for an average of one-quarter of the total basal area. Thus, cumulative mortality affected three-eighths of the woody biomass.

Relation to Stand Age

Chaparral stands older than 30 to 40 years have been described as senescent, a condition marked by mortality of individual shrubs and loss of canopy continuity (Specht 1969; Hanes 1971). For example, Hanes (1971) observed in the San Gabriel and the San Bernardino mountains in 1966 and 1967 that over one-half of the population of *Ceanothus crassifolius* and *C. greggii* var. *vestitus* was dead in stands older than 40 years. Dead shrubs constituted 70% of the total *Ceanothus crassifolius* biomass in 37-year-old mixed-composition stands measured in 1956 at the San Dimas Experimental Forest (Specht 1969). Yet, high rates of mortality are not a universal characteristic of older Ceanothus stands. Schlesinger and Gill (1980) observed 54-year-old *Ceanothus megacarpus* that had no greater mortality than stands less than one-half that age. We surmise that mortality previously attributed to senescence or to "intraspecific competition" may actually be driven by *Botryosphaeria dothidea* or other pathogens. Furthermore, the dieback does not appear to be merely a result of chaparral shrubs exceeding their natural longevity because we have observed it in 6-year-old *C. megacarpus* in the Santa Monica Mountains.

How Is Fire Behavior Affected by Biomass Accumulation and Dieback?

Several components of fire behavior are important to managing fire effects. These include likelihood that fire will spread, total energy release,

forward velocity or rate of spread of the fire front, and fire intensity. Rate of spread affects the ultimate size of a fire given concerted effort at suppression and a finite duration of severe weather. Unfortunately, there are no direct, quantitative observations of how fire behavior changes with either stand development or dieback. However, we can show some qualitative effects of stand age from our observations of the July 1985 Sherwood Fire in western Los Angeles County, and deduce some properties from fuel distributions and, by analogy, model fires in porous fuel beds.

Observations of Fire Spread in Two Age Classes

The Sherwood Fire burned concurrently in 50- and 8-year-old chaparral under the influence of a subtropical high aloft. (Afternoon temperatures were 38 to 40°C, relative humidity was below 20%, and winds were from the northwest at approximately 15 km/h.) Before burning, the older age class had had a several-fold greater biomass than the young, as judged from vegetation at the fire perimeter. Efforts at fire suppression were minimal in the young age class until the late stages of the fire.

Fire in the young chaparral produced a mosaic of burned and unburned vegetation, spreading in part through the *Salvia mellifera* and grass fuel embedded in this *Adenostoma fasciculatum* and *Ceanothus megacarpus* community. There fire consumed foliage and only small live twigs, incompletely consumed the leaf litter, and left a black ash surface. Radiometric temperatures of ash under solar heating 3 weeks after the fire were as high as 72°C (Figure 8–4*A*).

In stark contrast, the older chaparral burned with exceptionally high intensity as all foliage, fine wood, leaf litter, and many live stems 5- to 10-cm in diameter were incinerated. Virtually no unburned vegetation remained within the fire perimeter in this age class, and the ground surface was uniformly covered with a fine white ash. This ash was approximately 10°C cooler under solar heating than that of the young age class, and it was more reflective at intermediate wavelengths of infrared light (Figure 8–4*B*). We interpret the temperature and reflectance differences as showing greater carbon loss from soils and leaf litter under the older vegetation. This illustrates that both the propensity of chaparral to propagate fire and a fire's severity are related to changes in biomass associated with stand aging.

The vegetation mosaic created in young chaparral by the Sherwood Fire shows, for wildfire conditions, a general observation from prescribed burning: Chaparral fires spread only when the rate of energy transfer to unburned fuel exceeds a relatively substantial threshold. Weather that allowed a severe fire in older chaparral was only marginally sufficient to spread fire in the young chaparral. Such marginal conditions during prescribed burning, as judged by an inability to sustain combustion on shallow or shaded slopes or at any but the lowest live-fuel moisture, have

produced large flames of 6 to 10 m in 35-year-old *Quercus berberidifolia* (live fuel moisture, 0.8 g water g^{-1} dry mass; wind run, 4.8 km/h with gusts to 24; relative humidity, 21 to 27%) (Dougherty and Riggan 1982) and 30 m in 20- to 25-year-old *Ceanothus crassifolius* (Live-fuel moisture, 0.7 to 0.8 g water g^{-1} dry mass; wind gusts to 15 km/h) (Riggan et al. 1994a). In the absence of substantial deadwood biomass, fire is readily spread only with wind and low live-fuel moisture.

Total Energy Release During Burning

As we have noted, chaparral fires typically consume a fraction of live wood, most leaf litter, and all foliage and deadwood. For a given stand, the fraction of live wood consumed apparently varies little across the range of weather and fuel moisture that allows fire propagation (Figure 8–5). In 22-year-old *Ceanothus crassifolius*, 0.09 of the live wood was lost, an amount that corresponded to six-tenths of the wood less than 0.5 cm in diameter. If we assume that fires in younger stands consume the

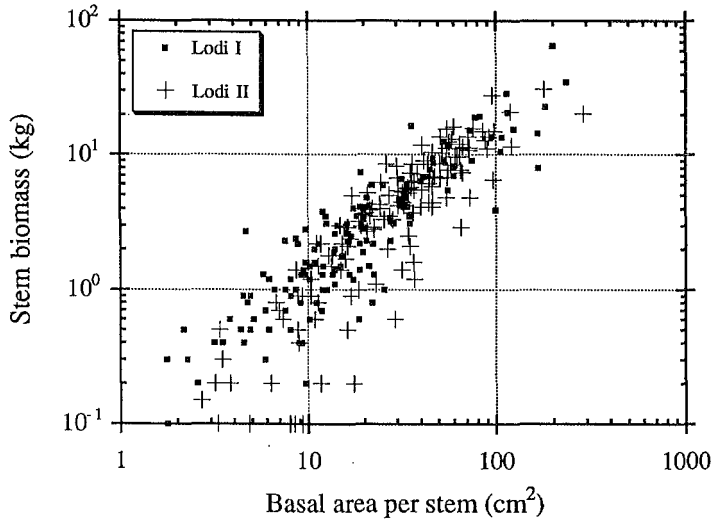

Figure 8–5. Relation of unconsumed biomass to stem basal area after two prescribed fires in *Ceanothus crassifolius* chaparral during December 1986 and June 1987 at Lodi Canyon in the San Dimas Experimental Forest. Basal area of live stems was not reduced by burning, and there was no discernible difference between the fires in the rate of biomass consumption per stem. The winter fire (Lodi I) was ignited with difficulty and only spread slowly on steep slopes and southerly aspects; the summer fire (Lodi II) spread readily on similar slopes, yet both fires consumed approximately 0.09 of the live-wood biomass and equivalent masses of foliage and deadwood. Thus, despite large differences in rate of spread and flame length, the fires did not differ in total energy release.

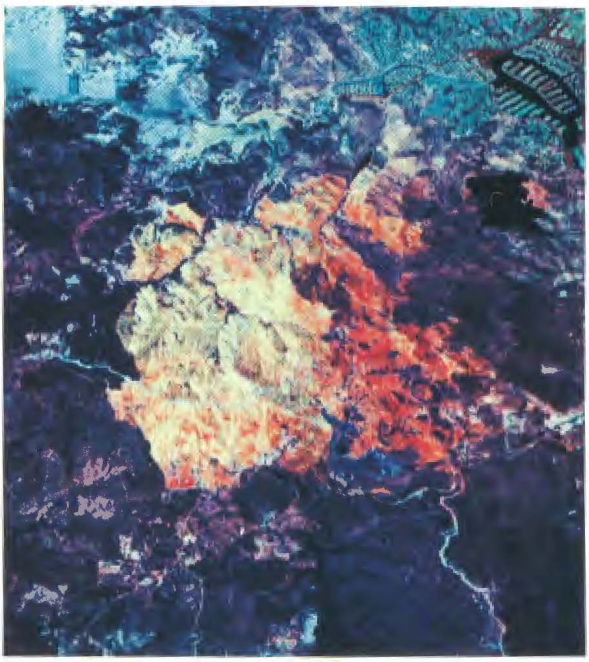

Figure 8–4. *A*, Ash from the 1985 Sherwood Fire in the western Santa Monica Mountains, Los Angeles and Ventura counties, CA. In this false-color composite image, reflected light at 1.55 to 1.75 μm is mapped in blue and that at 2.08 to 2.35 μm is mapped in green; emitted long-wave infrared light at 10.4 to 12.5 μm is mapped in red. Radiometric temperatures of the surface ash, estimated from the 10.4 to 12.5 μm radiance, were as high as 72°C in the younger age class and 10°C cooler in the older age class. The eastern one-third of the fire area (tones in orange at the right) is the burned 8-year-old age class. The yellow tones in the northwest quadrant of the fire area are from a north-facing aspect where 50-year-old chaparral was burned. Slopes at the southwest (below the fuel break that appears as an east-west trending white line) were south-facing and of intermediate age. Data were collected 23 July 1985 by an ADDS 1268 Thematic Mapper Simulator (Daedalus Corporation) aboard a NASA high-altitude aircraft.

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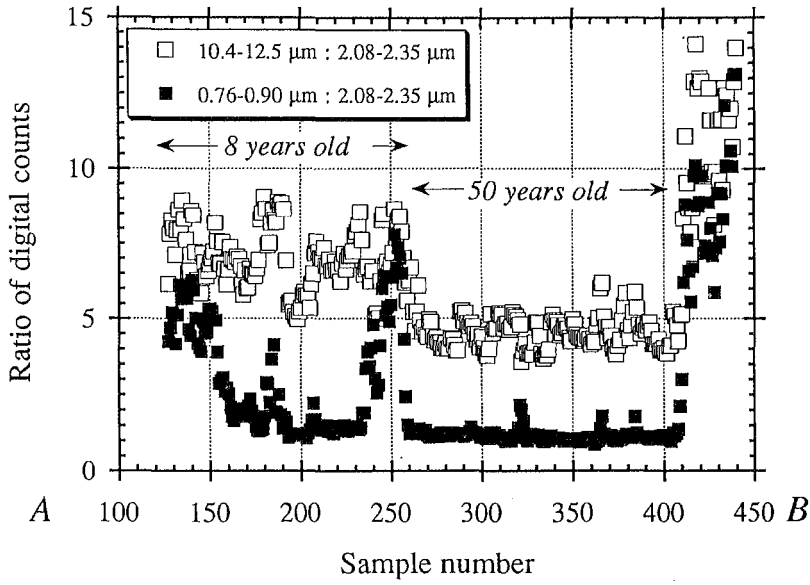

Figure 8–4 (*Continued*). *B*, Graphical representation of the reflected and emitted light along a transect from east to west through the fire area depicted in *A*. Fire in the 8-year-old chaparral (sample numbers 120 to 260) left a mosaic of ash and unconsumed vegetation, which is marked by high values of the ratio of near-infrared (0.76 to 0.90 μm) to intermediate-infrared radiances (2.08 to 2.35 μm). Ash there was warmer and less reflective at intermediate infrared wavelengths (depicted by high values of the ratio of radiances at 10.4 to 12.5 μm and 2.08 to 2.35 μm) than was the ash from the older age class.

same fraction of live wood in this size class, then fire spreading in 7-year-old stands of this species, necessarily burning with low live-fuel moisture under considerable winds, would yield from standing biomass approximately one-tenth of the energy of a fire burning in a healthy stand at age 22 years (Figure 8–6). In each case, the heat of vaporization would be small in relation to the total energy released.

Foliage, soil O horizon (organic material at the soil surface), and consumed wood at age 22 years are roughly equivalent sources of carbon for combustion (Table 8–1). Thus, the O horizon adds energy to the fire equivalent to one-half of that from the standing vegetation, although it will not propagate fire by itself because it is too compact.

Dieback of one-half (a high rate) of the live stems in a 22-year-old stand of *Ceanothus crassifolius* might be expected to increase energy release by one-half compared with the same stand in a healthy condition (see Table 8–1).

Changes in Rate of Spread During Stand Development

It is difficult to translate biomass consumption or energy release from these cases into predicted rates of fire spread because to do so would

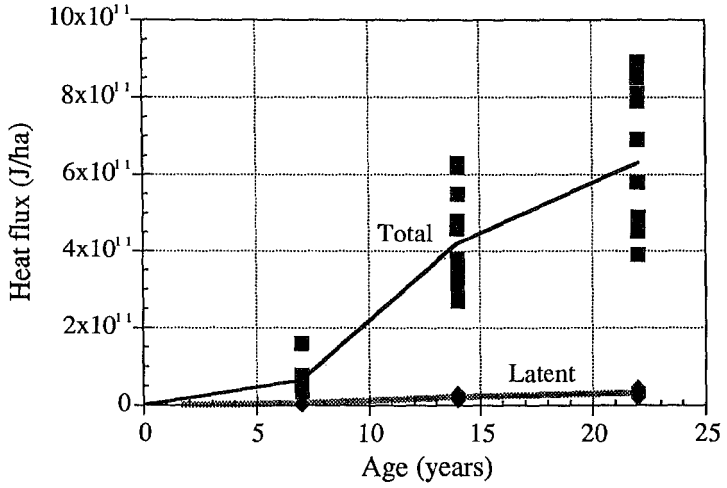

Figure 8–6. Estimated total and latent heat fluxes during burning of standing biomass in 7-, 14-, and 22-year-old *Ceanothus crassifolius*. Estimates were based on reconstructed biomass trends (Riggan et al. 1988) and the following assumptions: live wood consumed (0.09 of the total) is a constant fraction (0.6) of wood with diameter <0.5 cm; by analogy to *Quercus berberidifolia*, wood with diameter <0.5 cm is one-quarter of the total at age 7 and one-fifth of that at age 14; moisture contents are 0.7 g water g^{-1} dry biomass for foliage and live wood and 0.07 g water g^{-1} dry biomass for deadwood; specific heat of vaporization is 2.26×10^6 J/kg; specific heat of combustion is 2×10^7 J/kg (Albini 1980).

Table 8–1. Estimated Biomass Consumption During Burning for 22-Year-old Stands of *Ceanothus crassifolius* Both With and Without An Additional 50% Mortality of Stems

Component	Biomass Consumption (Mg/ha)		Carbon Loss (Mg/ha)	
	No Dieback	50% Dieback	No Dieback	50% Dieback
Live biomass				
Foliage	14.3	7.2	7.2	3.6
Live Wood	4.0	2.6	2.0	1.3
Deadwood				
Within live shrubs	8.0	30.3	4.0	15.2
Dead shrubs	4.2	4.2	2.1	2.1
Readily consumed (<0.5 cm)	4.6	7.9	2.3	4.0
O horizon	45.5	52.7	6.4	10.0
Total consumed			21.7	32.2
Readily consumed			17.9	18.9

Note. Biomass is reported by Riggan et al. 1988. Carbon loss from the O horizon was estimated from measured carbon fraction there (mean = 0.24, s = 0.081, n = 72) and in ash (mean = 0.096, s = 0.038, n = 61) after prescribed fires in standing chaparral at San Dimas in 1984; the case with dieback assumes consumption of abscised foliage before appreciable decomposition could begin. Readily consumed deadwood biomass is assumed to be that with a diameter less than 0.5 cm; larger material may also be consumed within the flaming front.

require a questionable extrapolation from semiempirical descriptions of model fires in a laboratory. Some inferences may be possible, however, from two models: One described by Rothermel (1972) and elaborated by Wilson (1990), and one of Fendell et al. (1990).

Rothermel (1972) gives a semiempirical model of fire spread that was parameterized first for small fires burning with no wind in dry excelsior or in wood elements with a nominal width of 1/4 or 1/2 in. The fuel array was approximately 90-cm wide and 11- to 15-cm deep. Rate of spread with wind was then described by $V(U) = f(V_0\ (1 + f_w))$ where V is the forward rate of spread at wind speed U, V_0 is the spread rate with no wind, and f_w is a function of wind speed, relative surface area to volume of the fuel, and the fuel packing, b, which can be interpreted as the proportion of the fuel bed volume that is actually occupied by fuel elements. The wind speed function, f_w, was estimated from observations of grass fires in Australia, in which no fuel properties were measured, and from limited trials with fine fuels in the laboratory.

This model was not explicitly developed to describe fire spread in sparse live fuels such as those that propagate chaparral fires. The form of the model (which relates spread with wind to a finite rate with no wind) is

also inappropriate for chaparral, where there is commonly a threshold of wind required for fire propagation (i.e., the no-wind rate of spread is zero).

Fendell et al. (1990) solved a differential equation for fire spread driven by convective heating. The resulting model, $V(U) = a(U/m)^{1/2}$ (where a is an experimentally determined constant), is simple in form and explicitly includes interaction of wind and mass, m, for fuel of a given height and ratio of surface area to volume. The model parameter, a, was estimated from burning trials conducted with ranges of wind speed (up to 4.6 m/s) and mass loading of toothpicks. The ratio of surface area to volume for toothpicks is similar to that of chaparral foliage, but the packing used experimentally was much denser than that of chaparral. The experiments used combustion tables of 55- or 100-cm width and incorporated variations in fuel height below 22 cm. For most experiments, which were conducted with a constant height of fuel, the changes in mass were equivalent to a change in packing.

These models are limited by the experimental situations from which they were developed. Spread rate of the model fires was dependent on width of the fuel bed and the magnitude of edge effects on forward energy transfer, roughly doubling as the width was increased from 55 to 100 cm (Fendell et al. 1990). Fendell et al. (1990) also generally observed reduced rates of spread when b and U are held constant, but the mass is increased by using taller fuel elements. This reduction presumably resulted from dispersal of convective energy through a larger volume of air within the fuel bed. Thus, velocity of fire spread was dependent on the geometry of the experimental situation, and there is no way to know a priori how to scale the results from model fires to those of chaparral fires where flame lengths are typically one to two orders of magnitude greater; large live woody stems do not burn but reduce energy transfer to unburned fuel; and there is substantial oxygen depletion, fire-caused acceleration of ground-level winds, and turbulence.

Both sets of experimental results show that rate of spread in fine fuels reaches a maximum at low values of packing. Furthermore, the experimental results of Fendell et al. (1990) show that the packing that maximizes spread obtains lower values at higher wind speeds. Given appreciable wind, fire spread was very sensitive to the packing at levels below the optimum (which can act as a threshold for spread), and zero spread was observed at finite values of fuel loading. (The model describes only conditions with packing greater than the optimum.)

We assume by analogy to the laboratory-based fires that chaparral with low deadwood biomass, which may only marginally sustain fire spread at lower wind speeds, is essentially at an optimum packing for those conditions. Furthermore, we assume that an increase in foliage mass as a stand matures (e.g., between age 10 and 20 years) increases packing above this optimum because of greater overlap between the canopy of

adjacent shrubs and slow concurrent height growth. Therefore, the rate of spread would be expected to decline accordingly. Furthermore, if the theoretical model form proposed by Fendell et al. (1990), $V = f(U/m)^{1/2}$, remains valid at the low packing and requisite wind speeds of chaparral fires, then the accumulation of foliage, fine live wood, and fine deadwood (as given in Table 8–1 and Figure 8–6) between ages 14 and 22 years would reduce the forward rate of spread by about one-fifth.

The impact of the dieback on rate of spread is more problematical because we cannot readily predict, for the combined live and dead fuels, the interactions of wind, packing, and moisture or the effect of fuel stratification. In general, we expect that the primary effects will be to increase fire residence time and soil heating, because of the greater total energy release and larger-diameter fuels that become involved, and to allow fire spread at lower wind speeds and higher fuel moistures. The latter effect could extend, perhaps by several weeks in early summer or late autumn, the conditions under which large fires may occur or could increase fire size during more severe weather – for example, by accelerating fire spread at night.

How Are Fire Impacts Related to Fire Severity?

Flooding and Sedimentation

Destructive postfire flooding, such as that which caused 30 deaths and destroyed 484 homes in the towns of La Cresenta and Montrose in 1933, has historically been an impetus for fire and flood control in southern California (Eaton 1935). An extensive system of engineered debris catchments and flood channels has been constructed to contain such flows, but these provide limited protection even today (S. Kumar, Los Angeles County Department of Public Works, personal communication). Debris flows from recently burned chaparral watersheds can mobilize a large volume of material from sand to boulders of 1- to 2-m in breadth. When the capacity of a debris basin has been exceeded, this high-velocity mix cuts its own unpredictable and destructive path. Even when the flows are contained, the expense of emergency sediment removal is high.

The severity of a chaparral fire can affect the magnitude of subsequent flooding and sediment production. This was demonstrated in 1984 and 1985 in a replicated watershed experiment at the San Dimas Experimental Forest (Riggan et al. 1994a, 1994b), where two canyons burned by moderate-intensity prescribed fires in standing chaparral yielded three-eighths as much sediment and water as did two that had been burned by severe fires in felled *Ceanothus*. The severe fires consumed all of the above-ground biomass, thereby releasing approximately three times the energy of the more moderate fires. The postfire sedimentation was

dominated by debris flows that were generated at a rainfall intensity of 5 mm in 5 minutes and comprised 59% sediment by weight (Weirich 1988; Riggan et al. 1994b). Associated peak discharge from the severely burned canyons exceeded $330\,L\,s^{-1}\,ha^{-1}$; concurrent flows peaked in unburned watersheds at less than $1\,L\,s^{-1}\,ha^{-1}$.

Massive soil erosion after intense chaparral fires probably follows from the loss of vegetative cover, organic matter destruction, and development of water repellency in soils. Loss of organic matter on and beneath the soil surface probably parallels the intensity and depth of soil heating and contributes to sediment production by exposing mineral soil and reducing its cohesiveness. Water repellency forms in soils when pyrolytic organic compounds condense below the soil surface and block subsequent infiltration of water (DeBano 1981). The depth of this condensation, as well as that of initial rill and surface erosion, is probably greatest under prolonged or intense soil heating (Wells 1981).

Nitrogen Flux from Burned Watersheds

Fire also affects the accumulation and cycling of mineral nitrogen in soils and the flux of nitrate in stream water. In the San Dimas experiments (Riggan et al. 1994a), burning mobilized nitrogen that had accumulated in part from the air pollution of metropolitan Los Angeles and caused stream water to be polluted with nitrate (NO_3^-) at concentrations sometimes exceeding the Federal water quality standard (0.7 meq/L). Streamwater NO_3^- concentrations were elevated during peak flows, reaching 1.12 meq/L during a debris flow.

The postfire NO_3^- concentrations and flux in stream water reflected the severity of burning. Severe fires produced daily-mean volume-weighted NO_3^- concentrations that were 1.7 times those after the moderate-intensity fires; moderate fires produced concentrations that were 3.0 times those of unburned controls. Annual NO_3^- loss from severely burned watersheds, averaging 1.2 keq/ha, was 40 times greater than that from areas that remained unburned. Fires of moderate intensity produced less than one-seventh the NO_3^- loss observed after severe burning.

The response of stream-water NO_3^- to burning was due in part to changes in the net rates of soil mineralization and nitrification, both of which were positively associated with fire severity (Riggan et al. 1994a). Severe fires undoubtedly subjected soils to the greatest heating below the immediate soil surface and caused the greatest rates of nitrogen volatilization. As a result, they elevated the NH_4^+ concentration in surface soils, but to a level below that of the moderate fires. NH_4^+ concentrations were subsequently highest in soils that had been subject to severe heating, because of either long-term inhibition of NH_4^+ immobilization or accelerated mineralization. Nitrate concentration after storms in November and December was greatest in soils subjected to severe fire (1.20 meq/kg

of soil), intermediate in those from the moderate-intensity fire (0.79 meq/kg), and least in the unburned soils (0.49 meq/kg).

The results from the San Dimas fire severity experiments show that flooding, sedimentation, and nitrate water pollution after a moderate-intensity prescribed fire are likely to be substantially less than would be produced from an equivalent area burned by a severe, catastrophic wildfire. The experiment was designed to produce a range in severity between fires, and although a fuel comprised entirely of deadwood would rarely be found in nature, the difference in energy release rate between the fires is comparable to that we estimate between fires burning in 10- and 22-year-old *Ceanothus crassifolius*.

Fire Severity and Community Composition

Fire severity can also affect the subsequent species composition of chaparral with potentially long-lived effects. This was apparently the case after prescribed fires we examined at Stone Canyon in the Santa Monica Mountains where *Ceanothus megacarpus* seedling density 1 year after burning was 5 times higher where the chaparral had been crushed and burned than where standing vegetation had been burned. (Seedling density was estimated in 46 plots each 10 m^2 in area; average density was 22.5 m^{-2} after crushing and burning and 4 m^{-2} after burning in standing vegetation. The null hypothesis that densities were not different was tested with a Mann–Whitney test and rejected with $P = 0.01$.) As at San Dimas, crushing allowed virtually all of the above-ground biomass to be burned with correspondingly greater energy release.

Species abundance in mature stands at the San Dimas Experimental Forest also shifted dramatically after successive fires of widely differing severity in 1919 and 1960 (Jacks 1984). The cover of *Ceanothus crassifolius* in 15-year-old stands after the 1960 Johnstone Fire was six times that which had been estimated at a comparable age in the same area after the 1919 San Gabriel Fire (Horton 1941), whereas cover of *Adenostoma fasciculatum* was one-third as great and the amount of bare ground was reduced. Changes in cover were primarily due to an increase in the Ceanothus population density from 0.03 to 0.14 m^{-2}; the mature *Adenostoma fasciculatum* population concurrently declined from 0.06 to 0.04 m^{-2}. The Johnstone Fire must have had relatively high energy yield because it burned 41-year-old chaparral that had been subjected to substantial mortality during an interval of record drought (Kittredge 1955); the San Gabriel Fire burned in 24-year-old stands. Drought after the Johnstone Fire may also have enhanced *Ceanothus crassifolius* seedling survival relative to that of *Adenostoma fasciculatum* (Jacks 1984).

Fuel structure and biomass accumulation depend to a large extent on species composition, so a change in composition could alter the nature of the next fire. An increase in *Ceanothus* abundance, such as that which

occurred at San Dimas after 1960, could yield a structure more conducive to the propagation of infrequent, severe fires (Riggan et al. 1988).

How Might Prescribed Burning Alter the Pattern of Catastrophic Fires?

Now, let us consider some of the values and environmental liabilities that might be derived from an active program of prescribed burning. The strategy is to maintain a shifting mosaic of age classes both to alter the potential for catastrophic fire and to reduce overall environmental impacts of an unmanaged wildfire regime. Our analysis is somewhat speculative because the ability of any chaparral community to propagate fire at a given age or fuel structure is probably the least known property of fire behavior.

Our approach is to ask what might result when a wildfire ignites and spreads in mature chaparral, 30 or 40 years of age, and encounters a young age class. Let us assume that it is autumn with low fuel moisture, humidity near 20%, and moderate winds (under 20 km/h).

The Wildfire Front is Large Compared to the Size of the Younger Age Class. In this case, the ultimate *perimeter* and rate of progress of the wildfire will not be materially reduced, but there are several alternative outcomes within the younger chaparral.

If the young chaparral is 2 to 5 years old (and not dominated by introduced grasses), then it is likely that it will not burn and the overall area of the wildfire will be reduced accordingly. Such was the case where the 1985 Wheeler Fire encountered the 1983 Matilija Fire (Figure 8–7). Considering the wildfire and prescribed fire together, the total area burned will not be affected, but there will be a reduction in watershed impacts within the area of the prescribed burn, and a partitioning of impacts between the years after the two fires. Thus, within the uncertainty of ensuing storm intensity, there should be reduced flood peaks and less sediment and debris to deal with in an emergency.

If the young age class is 10 to 15 years old, there is a greater, but still small, chance that it will propagate the wildfire. If it does burn, biomass consumption and total energy release within the young stand would probably be one-quarter to one-half that which would have resulted without management, and the relative reduction in subsequent flood peaks and sedimentation from the area might be roughly comparable to that observed between moderate-intensity and severe fires in the San Dimas experiments. The result of the two fires together would be an increase in fire frequency within the perimeter of the prescribed fire, but a small reduction in sedimentation (since a moderate-intensity fire yields less than one-half as much sediment as would a severe fire).

If the young age class is 20 to 25 years, it would most likely be burned by the wildfire, there would be only a relatively small change in the wildfire effects, and there could be a net increase in sediment production relative to a single wildfire. (We assume that the supply of sediment is not quickly exhausted by a few fires.)

The Wildfire Front is of Comparable Size or is Small in Relation to the Younger Age Class. In these cases, there could be a substantial reduction in perimeter and area of the wildfire for those cases in which the young age class will not propagate fire or burning there is readily suppressed. Such was probably the case where the 1985 Wheeler Fire encountered under moderating weather the area of the 1971 Romero Fire (Salazar and González-Cabán 1987). The area so protected continues to age, and by continued mortality, to accumulate dead fuel. Depending on rates of fuel accumulation at later ages, which are not well established, the impact of a later wildfire there could be worsened if another prescribed fire does not intervene.

Use of prescribed fire is still limited in southern California, and although it has aided suppression of a few subsequent wildfires (Dougherty and Riggan 1982), it probably has not substantially altered the general impact of wildfires over the past decade. As we have discussed here, prescribed burning has potential for reducing some societal and environmental losses from wildfire, especially those associated with flooding, sedimentation, and water quality. But public policy for fire management remains driven by crises in funding rather than by the potential for averting catastrophic losses. The potential for fire management will be realized only when we are capable of monitoring and predicting long-term change in wildfire risks and articulating the benefits of an appropriate level of intervention. The most limiting factors in this analysis at present are the uncertainties regarding the propensity of young age classes to carry fire, the regional distribution of consumable biomass, and the rate of fuel accumulation in stands older than 3 decades.

How Predictable Will Be the Response to Climate Change?

Current predictions are that "business as usual" on earth will cause global mean air temperature to warm by as much as 2 to 5°C over the next century as a result of rising concentrations of radiation-absorbing trace gases in the atmosphere (Houghton et al. 1990). Regional predictions of potential climate change are highly uncertain, especially with regard to climate extremes such as those that most affect fires in southern California: Incidence of Santa Ana winds in autumn; heat waves associated with subtropical high pressure in midsummer; frequency of high rainfall years followed by drought; length and depth of multiannual

Figure 8–7. False-color composite image of the 1983 Matilija Fire (orange tones in center) within the perimeter of the 1985 Wheeler Fire. The 2-year-old-age class remained unburned despite extreme fire weather. The image was constructed from radiances measured at 0.91 to 1.05 (red), 0.63 to 0.69 (green), and 0.52 to 0.60 μm wavelength (blue) by an ADDS 1268 spectrophotometer (Daedalus Corporation). Data were collected 16 October 1985 aboard a NASA high-altitude aircraft.

droughts; and possibly, episodes of high temperatures, high humidity, and intermittent rain in midsummer.

Several of these conditions are now predicted with considerable uncertainty even in short-range forecasts and may remain unpredictable from long-range global climate models. One indicator of the adiabatically-heated and high-velocity Santa Ana winds, for example, is the development of a 4-mbar or greater offshore pressure gradient between Los Angeles and Tonopah, NV. Error in predicting the strength or relative location of the Great Basin high by a few hundred kilometers, and thereby the strength of this gradient and the Santa Ana winds, is fairly common within a 5-day forecast (N. Dean, National Weather Service, Fire Weather Forecasting Office, Riverside, CA, personal communication).

Annual precipitation and summer drought stress can be dominated by one or a few storm systems that may produce locally heavy or prolonged rainfall. Closely spaced storms during 9 successive days in February 1980 raised the total precipitation in the San Gabriel Mountains that season from average to twice the annual average and increased the area-specific stream flow to 55 cm from the expected average of 7.5 (Riggan et al. 1985). This recharged deeper soils and probably limited water stress during the succeeding summer. Rainfall of this magnitude in 1983 may have allowed an accumulation of leaf area that aggravated water stress condition in 1983–84 and caused the most severe dieback in decades. The 5-day forecasts are generally poor in predicting storm locations within 100 to 200 km or storm residence at one location, yet such large storms are probably one of the more important events in the ecosystem for a span of several years or even decades.

Even if global models could reliably predict average conditions for a season, we would have difficulty predicting extreme events in winds or rainfall, and these can dominate the ecosystem for years. Thus, by analogy, we find ourselves riding a rollercoaster in fog and not knowing if the amplitude of change is widening or subsiding. At present, the dips are catching us by surprise.

Conclusion

Without a doubt, southern California is an extremely challenging setting for natural resource management: Its environment is physically, biologically, and culturally dynamic. Management can affect the lives of millions of people, and these people exert a strong pressure on the environment. Catastrophic fires here will always be a threat to people and the environment. Climate extremes and impending climate change could drastically alter the situation in ways that would be very difficult to predict. But the impact of a fire regime can be altered through use of fire

and a reduction of values at risk. To fail to act before the threat is immediate is to be continually responding to wildfires and paying their escalating price.

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Cover illustration. Map of documented fires occurring between 1938 and 1985 in Burton Mesa, California. Adapted from Figure 7-4, page 128.

Library of Congress Cataloging-in-Publication Data
The role of fire in Mediterranean-type ecosystems / [edited by] José M. Moreno, Walter C. Oechel.
p. cm. – (Ecological studies; v. 107)
Includes bibliographical references and index.
ISBN 0-387-94215-7 (New York). – ISBN 3-540-94215-7 (Berlin):
1. Fire ecology. 2. Mediterranean climate. 3. Plants, Effect of fires on. I. Moreno, José Manuel. II. Oechel, W. C. III. Series.
QH545.F5R65 1994
574.5' 222 – dc20 93-42622.

Printed on acid-free paper.

Production coordinated by Chernow Editorial Services, Inc., and managed by Ellen Seham; manufacturing supervised by Genieve Shaw.
Typeset by Best-set Typesetter Ltd., Hong Kong.
Printed and bound by Braun-Brumfield, Inc., Ann Arbor, MI.
Printed in the United States of America.

9 8 7 6 5 4 3 2 1

ISBN 0-387-94215-7 Springer-Verlag New York Berlin Heidelberg
ISBN 3-540-94215-7 Springer-Verlag Berlin Heidelberg New York