8. Impact of Past, Present, and Future Fire Regimes on North American Mediterranean Shrublands

Jon E. Keeley and C.J. Fotheringham

Mediterranean shrublands occur in five regions of the world, under a climate of mild wet winters and hot summer–fall droughts lasting six months or more. In California they dominate landscapes below 2000 m in the central and southern coastal ranges and foothills of the Sierra Nevada. One consequence of this distribution is that these shrublands, more than any other vegetation type, interface with urban areas (Fig. 8.1). These shrublands are subject to periodic massive wildfires (Fig. 8.2) that account for 40% of all wildland acreage burned in the United States (Lillard 1961), creating a particularly hazardous urban–wildland interface. Contributing to this fire hazard are the moderate temperatures during the rainy winter and spring, which prolong the growing season and generate broad bands of dense contiguous fuels. The long drought makes these fuels readily ignitable and the autumn foëhn winds that come each year at the end of the dry season produce the worst fire climate conditions in the country (Schroeder et al. 1964).

This chapter examines the past, present, and future fire regimes in California shrublands, particularly chaparral and coastal sage scrub. Although shrublands are recorded from nearly all counties in the state (Callaham 1985), this review will focus on those in the central and southern coastal ranges with the largest expanses of contiguous shrubland (Fig. 8.3). Of particular concern are the extent to which humans have altered this regime in the past and the extent to which future global change will affect fire regimes and vegetation patterns.

Humans directly influence fire regimes in two ways: they ignite fires and they suppress fires. Evaluating the net effect of these impacts is not simple because

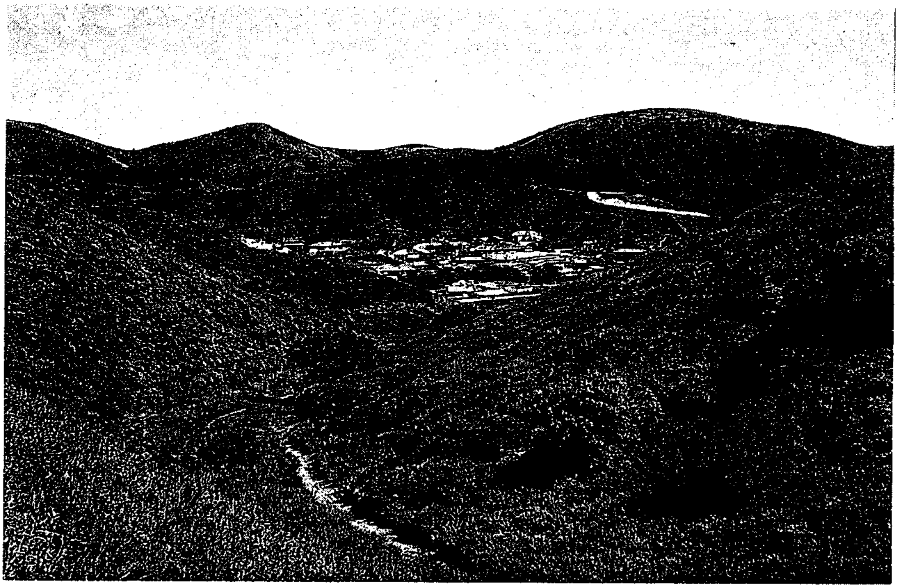

Figure 8.1. Interface between urban environments and evergreen chaparral (*right*) and semi-deciduous coastal sage scrub (*left*) in southern California (by J.E. Keeley).

their relative importance varies across the landscape. For example, in the montane coniferous forests of the Southwest, lightning-ignited fires are abundant and human ignitions are far less important than in lower-elevation shrublands of southern California where lightning is uncommon and humans cause the majority of fires (Fig. 8.4). Also fire suppression has been far more effective in western coniferous U.S. forests, often achieving nearly complete fire exclusion (Skinner and Chang 1996; Agee 1993), but this "fire-suppression = fire-exclusion" equation does not apply to shrublands of southern and central coastal California (Keeley and Fotheringham 2001b).

Determinants of Brushland Fire Regimes

Fire regimes are determined by the temporal and spatial pattern of ignitions, fuels, weather, and topography (Pyne, Andrews, and Laven 1996), and with regard to Californian shrublands there are two schools of thought on their relative importance. One is based on deductions from Rothermel's fire behavior model (Rothermel 1972) and argues that fire regime is a highly deterministic process driven by fuel load (Rothermel and Philpot 1973; Philpot 1974a,b, 1977). Under this model fire occurrence is unaffected by external drivers such as ignitions or weather, rather it is viewed as entirely dependent on community patterns of fuel accumulation (Minnich 1989, 1995,1998, 2001; Minnich and Cho 1997). The

Figure 8.2. Crown fire in chaparral (photo by USFS, Riverside Fire Lab).

alternative model argues that the fire regime is controlled by the coincidence of ignitions occurring under severe current and antecedent weather conditions that influence fuel flammability (Phillips 1971; Keeley et al. 1989; Davis and Michaelson 1995; Keeley and Fotheringham 2001a,b). Under this model any of these factors may be limiting, and the importance of each varies spatially and temporally with external drivers such as severe fire weather being of paramount importance in coastal California. These models have very different implications for fire management and affect our perception of anthropogenic impacts on fire regime and our ability to sort out future climatic signals.

Patterns of Ignition

In order to appreciate fully the role humans play in shrubland fire regimes, we need to first examine how ignitions, fuels, and weather interact to determine fire behavior. In California humans have been a source of ignitions for more than

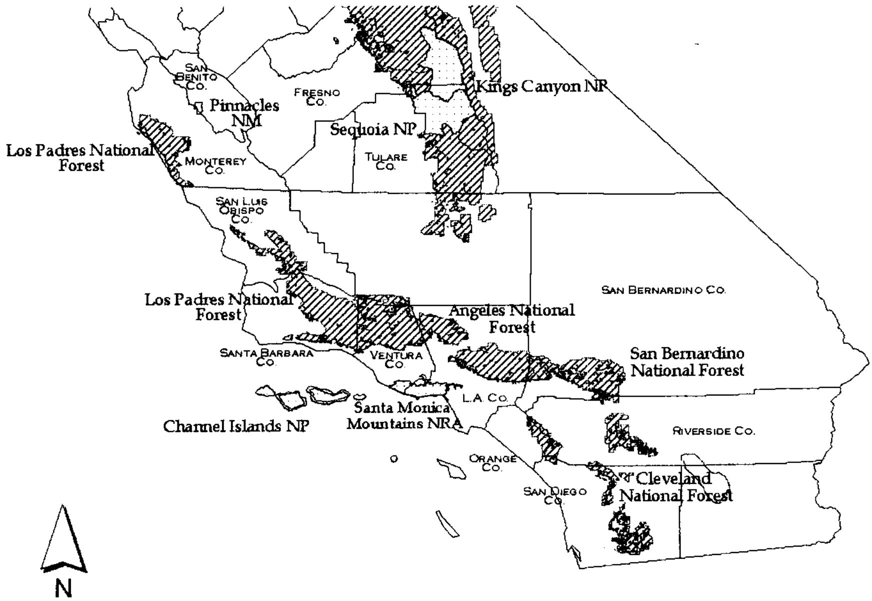

Figure 8.3. Central and southern California regions considered in this chapter. Central coastal California includes Monterey, San Luis Obispo, Santa Barbara, and Ventura counties, and southern California includes Los Angeles, San Bernardino, Riverside, Orange, and San Diego counties. Collectively these nine counties comprise nearly two million hectares of shrubland (Table 8.1).

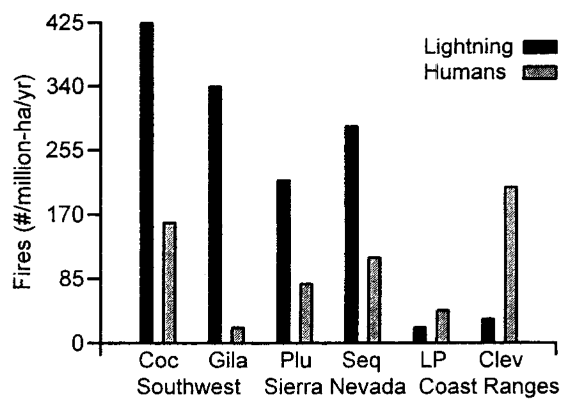

Figure 8.4. Regional comparison of lightning- and human-caused fires on USFS national forests. The Southwest includes the Coconino (Coc) in Arizona and Gilia in New Mexico. In California the Sierra Nevada forests are the Plumas (Plu) and Sequoia (Seq), and the California coastal ranges national forests are the Los Padres (LP) and Cleveland (Clev). Fire occurrence data from the published U.S. Forest Service, National Forest Fire Reports, 1970–1979, and forest area from (*http://www.fs.fed.us/land/*).

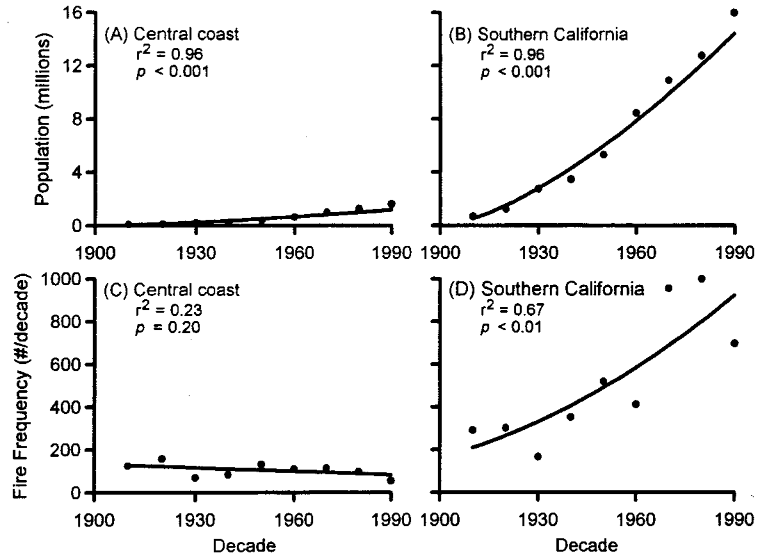

Figure 8.5. Decadal variation in population density (A–B) and fire frequency (C–D) for central coastal and southern California. Population data from the U.S. Department of Commerce, *http://www.census.gov/populations/cencounts/ca190090.txt*. (Fire data from the Statewide Fire History Data Base, California Department of Forestry, Fire and Resource Assessment Program (FRAP), Sacramento, CA, which includes historical fire records from the U.S. Forest Service national forests, California Division of Forestry ranger units and other protected areas, plus city and county records; minimum fire size recorded varied between 16 and 40 ha, depending on the agency).

10,000 years, but they likely have had a greater influence in the twentieth century due to the near exponential rise in population density and fire frequency in the southern part of the state (Fig. 8.5).

Under natural conditions lightning is a source of ignition but far less predictable than in other parts of the Southwest (Fig. 8.4). Within the state, lightning-ignited fires vary spatially because thunderstorms are rare near the coast and most frequent at higher elevations in the interior (Radtke, Atndt, and Wakimoto 1982; Keeley 1982; Greenlee and Moldenke 1982; Knipper 1998). Lightning is the dominant ignition source in the Sierra Nevada, but it is a far less common ignition source in the coastal ranges. Within the coastal ranges lightning varies with elevation; for example, in San Diego County lightning strikes are 10 times more abundant above 1800 m than below 500 m, and they vary temporally with 85% occurring between July and September (Wells and McKinsey 1994, 1995). Similar patterns are evident further south in Baja California (Minnich et al. 1993). The annual density of lightning discharges in this region is roughly 1 per 100 ha (Michael L. Wells, personal communication; Minnich et al. 1993). Based on the frequency of fires ignited by lightning in this region (Keeley 1982; Minnich et al. 1993), it would appear that only 2% to 5% of all lightning discharges ignite a wildfire. In other words, 95% of all lightning discharges strike inadequate fuels, or are extinguished by rain, before they reach a detectable size.

Lightning ignitions in coastal and southern California shrublands account for a highly variable amount of burning, ranging from less than 1% to more than 50% of the landscape per decade (Table 8.1). Both spatial and temporal factors are involved. Considering all of California, lightning ignitions account for an increasing fraction of burning from the coast to the interior and from south to north (Keeley 1982). Occasionally lightning may coincide with severe weather and fuel conditions and result in massive fires such as the Marble Cone Fire in 1977 on the Los Padres National Forest (Table 8.1). Longer-term data sets for the Los Padres show this to be an infrequent event (Davis and Michaelsen 1995), suggesting that lightning fires in these coastal ranges are capable of reaching extraordinary size but the temporal variance is high.

Lightning is more predictable in the higher interior Sierra Nevada Range (Fig. 8.4), and it varies inversely with elevation (van Wagtendonk 1992). In Sequoia National Park (located in the southern Sierra Nevada, Fig. 8.3) lightning-ignited fires reach a peak at elevations between 2000 and 3000 m and are considerably less frequent in the lower-elevation shrubland-dominated foothills (Parsons 1981; Vankat 1985). Within the park the lower-elevation shrublands experience fewer lightning-ignited fires than would be expected based on shrubland area ($p < 0.001$ with χ^2 test), and the opposite is true for higher-elevation mixed-coniferous forests. This pattern is repeated throughout the Sierra Nevada;

Table 8.1. Total number of fires and hectares burned and percentage due to lightning during the 1970s decade for lower-elevation foothills (California Division of Forestry Jurisdiction) and higher-elevation interior mountains (U.S. Forest Service national forests) in southern and central–coastal California

CDF Ranger Unit/USFS National Forest	Total fires (10^6 ha/ decade)	Total area burned (ha)	Fires due to lightning (%)	Area due to lightning (%)
Foothills (CDF)				
Monterey/San Benito	3,140	53,570	2	<1
San Luis Obispo	3,310	44,130	2	<1
San Bernardino	9,680	12,240	4	11
Riverside	17,620	332,950	1	5
Orange	42,900	120,830	<1	<1
San Diego	9,450	20,930	3	6
Mountains (USFS)				
Los Padres	2,340	49,720	9	56[a]
Angeles	4,980	214,460	15	4
San Bernardino	4,400	41,030	24	6
Cleveland	4,870	121,370	11	<1

Source: Keeley 1982.

Note: Sites are arranged from north to south, and national forest locations are shown in Figure 3. All of these ranger units or forests are dominated by chaparral, but they also include mixtures of grassland, sage scrub, woodlands, and forests.

[a]Much of this is due to a single lightning-ignited fire (Marble Cone Fire) in 1977.

foothill shrublands average about 10 lightning-ignited fires per year per million hectares and the higher-elevation montane forests experience 100 to 200 per year per million ha (Keeley 1982). Of course, making predictions about the elevational patterns of burning by lightning alone (i.e., in the absence of anthropogenic interference) is complicated by the likelihood that along this elevational gradient, conditions conducive to fire spread are inversely related to lightning fire frequency. Modeling is perhaps the only means of understanding the natural fire regimes in these ecosystems (e.g., Greenlee and Langenheim 1980; Davis and Burrows 1993; Davis and Michaelsen 1995; Zedler and Seiger 2000).

In general, rain or high humidity accompanies lightning fires, and there is often a time lag between ignition and changes in weather conducive to rapid fire spread. Thus, in forested ecosystems where lightning is the dominant ignition source (e.g., the Southwest, Fig. 8.4), fire suppression has been extraordinarily effective. Fire detection has become increasingly more reliable (Chandler 1960), and there is reason to believe that many suppressed lightning-ignited fires, in both forests and shrublands, would have burned out if never detected. This is supported by a greater number of reports of lightning-ignited wildfires in the latter half of the twentieth century (Keeley 1977, 1982; Greenlee and Moldenke 1982; Vankat 1985); however, it could reflect changes in fuel structure as well (Weatherspoon and Skinner 1996).

In the coniferous forests of the Sierra Nevada, lightning-ignited fires peak in the summer months of July and August and match closely the monthly distribution of human-ignited fires (Parsons 1981; Vankat 1985; van Wagtendonk 1992). Throughout the chaparral-dominated coastal ranges lightning-ignited fires are also concentrated in the summer months of July and August (Keeley 1982). However, humans are the dominant source of ignition (Fig. 8.4), and their impact on fire season varies from apparently very little effect in the central coast, as illustrated by a summer peak in burning to a much greater impact in the south, where anthropogenic fires result in a longer fire season and greater autumn burning (Fig. 8.6). Thus, in contrast to the situation in forests throughout the western United States where lightning is the dominant source of ignition and humans have successfully suppressed most fires, the vast majority of fires in chaparral and coastal sage scrub in the coastal ranges are ignited by humans (Keeley, Fotheringham, and Morais 1999; Keeley and Fotheringham 2001b). In short, fire suppression has not eliminated burning on this shrubland landscape. Human impact is most pronounced at lower elevations and in proximity to metropolitan areas. On shrubland landscapes under natural conditions, lightning is a predictable source of ignition but variably distributed in time and space.

Fuels and Weather

The spatial and temporal arrangement of fuels is a critical determinant of fire behavior, and fuel loading is determined largely by differences in site productivity and vegetation age. The extent to which fire will propagate across a landscape

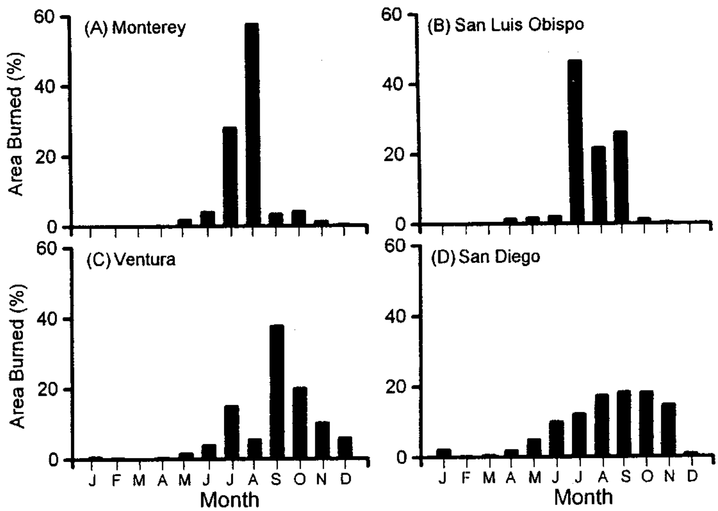

Figure 8.6. Seasonal distribution of burning reported for 1970 to 1999 for selected counties (data from the California Statewide Fire History Database; see Fig. 5).

is determined by the spatial arrangement of fuels and weather conditions prior to and during the fire. Fuel structure needs to be considered at different scales. In a stand of vegetation on a single slope face, the important fuel characteristics are the vertical and horizontal placement of fuels, fuel surface-area/volume ratio, and the moisture status of leaves and stems. At this scale shrubs are of uniform age and may be rather coarse grained in monotypic stands, becoming finer grained as the mixture of species increases. At the landscape level fuels are fine grained, and large expanses of homogeneous fuels are the exception. Barriers of reduced fuel loading, which could include rocks, rivers, alluvial fans, young age classes, or less flammable vegetation types, may inhibit fire spread. As seasonal drought progresses, different portions of the landscape are added as potential fuels, further contributing to inherent landscape heterogeneity of fuels. This interaction among landscape structure, fuels, and moisture limits the ability of models to predict fire spread, and the fine-grain nature of fuels leads to potentially large errors (Kessell and Cattelino 1978).

Fuel Structure and Fuel Moisture

In mature shrublands, surface fuels are insufficient to carry fire, and thus fires propagate through the canopy as crown fires. Recently burned sites have sufficient herbaceous growth to carry surface fires (Haidinger and Keeley 1993), and this may be exacerbated by artificial seeding of nonnative grasses (e.g., Zedler, Gautier, and McMaster 1983). However, sufficient herb biomass to carry surface fires is unlikely following dry winters or on highly infertile coarse-textured soils, such as occur in certain coastal sites (e.g., Lompoc, CA) or the interior ranges of Baja California (Franco-Vizcaino and J. Sosa-Ramirez 1997).

Normally, following a wet winter, high fuel moisture in chaparral shrubs makes them relatively resistant to fire in spring and early summer. However, as the amount of herbaceous matter in the stand increases, the seasonal window of burning increases. Dead herbaceous fuels dry rapidly and are capable of carrying fire within days of a rainfall event (Chandler 1963), and species composition plays a role as nonnative grasses typically die many weeks earlier than native herbs (Keeley, personal observations). As a result certain herbaceous fuels greatly extend the length of the fire season.

Shrublands that have been partially or fully type-converted to grasslands (e.g., Bentley 1967) have a greater probability of igniting but do not represent an extreme fire hazard as fire intensities are low and the fine herbaceous fuels fail to sustain embers or create the vortexes that carry the fire ahead of the moving front (Regelbrugge 2000). Even so, fires in a dense growth of non-native herbs, such as mustards (*Brassica nigra* and *Hirschfeldia incana*) on steep slopes, have been known to generate fire intensities sufficient to destroy homes (J. Keeley, personal observation).

For intact shrublands, two factors affect woody fuel moisture: the physiological activity (water potential) of live foliage and the quantity of dead fuels (Green 1981). Shrub species differ markedly in moisture status of foliage due in part to differences in rooting depth (Davis, Kolb, and Barton 1998)—shallow-rooted shrubs, such as chamise (*Adenostoma fasciculatum*) and *Ceanothus* spp., typically experience water potentials two to three times lower than more deeply rooted shrubs such as scrub oak (*Quercus berberidifolia*). Under prescription weather conditions fires may readily spread through *Adenostoma*-dominated chaparral but extinguish when they encounter patches of scrub oak (Chandler 1957; Green 1981). However, under extended drought, foliage moisture in scrub oak may drop to levels conducive to rapid-fire spread (Olsen 1960; Pirsko and Green 1967; Green 1981).

Dead fuels lack an internal water source and respond rapidly to changes in humidity; small diameter stems can dry completely within hours and larger fuels within days of experiencing low humidity (Chandler 1963; McCutchan 1977). Dead fuels not only combust readily, but as the proportion of dead/live material increases, there is an elevated potential for dead fuel combustion to cause drying of living foliage to a level sufficient for combustion. Because dead fuel carries fire and live fuel absorbs energy, the ratio of dead/live fuel is critical. This increase in combustibility of live fuels is enhanced by the common position of dead fuels beneath the living foliage. Topography plays a similar role. On steep terrain, head fires burning upslope enhance the combustion of fuels ahead of the front and may spread two to three times faster than on level ground—fire spread will roughly double for each 13 degree rise in slope (Green 1981).

Fuel Structure and Wind

At low wind speed, fuel structure and arrangement plays a critical role in fire spread. For example, fine-textured, low, compact fuels—particularly subshrubs with extremely high levels of volatiles, for example *Salvia* spp. (sage)—may readily combust and spread fire rapidly. However, under the same weather conditions, fire might naturally extinguish in a taller chaparral stand in which fuels are more widely scattered in the canopy, and there is little continuity with ground-level fuels (Green 1981). Under low to moderate wind conditions species-specific fuel characteristics in chaparral can promote fire spread. Many characteristics of *Adenostoma fasciculatum* (chamise) make it far more flammable than associated shrub species. About two-thirds of the plant is composed of twigs <25 mm diameter and thus has a stem surface area–volume ratio greater than that of other species (Conard and Regelbrugge 1994). Individual chamise leaves have a relatively low surface area/volume ratio, but they have an extremely high content of volatile compounds that vaporize and increase combustibility (Philpot 1969). On a whole plant basis, *Adenostoma* leaves have a very high surface area/volume ratio; they comprise 67% of surface area but only 16% of plant volume, reflecting the loose packing of foliage (Countryman and Philpot 1970; Barro and Conard 1991).

One of the key factors affecting flammability of *Adenostoma* is the fact that it does not self-prune dead twigs and branches; instead, they are held aloft in the canopy and increase canopy porosity (shrub canopy volume/leaf and stem volume), which often exceeds 99% (Rundel, Parsons, and Baker 1980). High canopy porosity increases flammability and extends the seasonal window of flammability. Also experimental studies demonstrate that this natural retention of dead branches substantially increases fire intensity over an artificial treatment of clipping and leaving as surface fuels (Schwilk 2000). Species with more densely packed fuels, and that self-prune dead branches and have thicker twigs and stems (e.g., scrub oak, *Quercus*, or chaparral holly *Heteromeles arbutifolia*), often will not burn under conditions suitable for fire spread in *Adenostoma*-dominated chaparral. It has been hypothesized that characteristics enhancing flammability have adaptive value (Mutch 1970) and shrubs with seedling recruitment restricted to postfire environments (*Adenostoma, Ceanothus, Arctostaphylos*) have significantly higher flammability than species that recruit independently of fire (*Prunus, Rhamnus, Quercus*) (Bond, unpublished data).

While high-canopy porosity increases flammability, it leads to lower bulk density (mass/volume) and fuel loading (mass/area), reducing the total energy available for combustion. Thus the *Adenostoma* fuel structure increases flammability under a wide range of conditions, whereas the *Quercus* fuel structure is limited in the range of conditions suitable for burning, but under the severest conditions *Quercus* fuels should be expected to generate the highest intensities.

Fuel structure appears to play a less deterministic role under windy conditions, but there is a complex interaction of wind, humidity, fuels, temperature, and topography. Cool moist marine air will extinguish fires (Coffin 1959), whereas warm dry air will lead to fire spread in fuels that otherwise would not burn. Wind accelerates oxygen supply and thus combustion (Green 1981) and is the primary mode of heat transfer. It carries heated air to adjacent fuels on the downwind side, raising the fuel temperature and driving off moisture. Wind also carries away water vapor as well as firebrands, which often occur when gusts are greater than 16 km/hr (Green 1981). Topographic features frequently cause unstable and erratic changes in velocity and direction as winds adapt to the topography. On coastal-facing slopes onshore winds are channeled up-canyon and produce eddies at ridgelines that may become turbulent and erratic. The typical pattern is for local daytime up-canyon wind and nighttime down-canyon winds, and on coastal slopes in the central coastal region extraordinarily strong down-canyon winds known as Sundowners are occasionally experienced (Ryan 1996).

Overriding synoptic-scale winds can upset these local wind patterns, e.g., foëhn winds known as "north winds" or "mono winds" in central California (Greenlee and Langenheim 1980) and Santa Ana winds further south (Lessard 1988) (Fig. 8.7). These winds are controlled by regional synoptic patterns that include a Great Basin high-pressure cell and Pacific Coast trough of low pressure, but their ultimate manifestation is a result of local topography (Schroeder and Buck 1970; Fosberg et al. 1966). For example, in the southern Sierra Nevada, the steep eastern escarpment and lack of low passes keeps these winds aloft (Mitchell 1969), and thus foëhn winds are not experienced on the lower western slopes. In Ventura and Los Angeles counties these winds are funneled through passes in the east west trending Transverse Ranges and thus are predominantly northern or northeastern winds (Weide 1968; Schroeder et al. 1964). In San Diego County they are strictly eastern due to the north–south orientation of the Peninsular Ranges winds (Campbell 1906; Sommers 1978). These ranges extend southward into Baja California where their sharp eastern escarpment, coupled with the Gulf of California to the east, limit the formation of foëhn winds on the west slopes of the Sierra San Pedro Mártir (Keeley and Fotheringham 2001a,b).

In southern California these hot, dry Santa Ana winds often have less than 10% relative humidity and may exceed 100 km per hour (Fosberg et al. 1966; Ryan 1969). Although referred to as "desert winds," the high temperatures and low humidity are the result of compression as air descends to form the "basin air mass" (Mitchell 1969), and on a local scale as it descends through coastal passes (Krick 1933). Santa Ana winds are most common in the autumn (Fig. 8.8). They have a mean life of about three days but may last two (Fosberg 1965) or three weeks (Campbell 1906), a critical factor since fire size is often determined by the duration of high wind conditions (McCutchan 1977). Under Santa Ana wind conditions fire spread is rapid. For example, the Kanan fire in the Santa Monica Mountains of southern California consumed 10,121 ha in 3 hours (Franklin 1987), and such fires may exceed 30,000 ha in a single day (Phillips 1971). Such fires are unimpeded by many potential barriers, since firebrands may be carried as much as 8 km beyond the front, igniting numerous new spot fires (Countryman

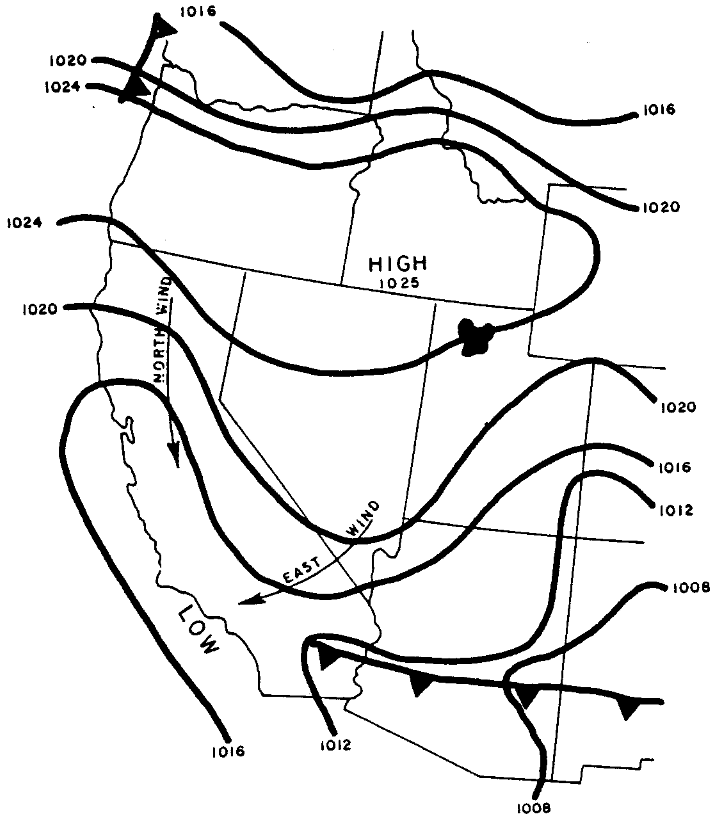

Figure 8.7. Surface weather map during the Great Basin high-pressure air mass that generates foëhn winds in central and southern California (from Phillips 1971).

1974). Under these conditions stands may burn regardless of stand age or species composition (Keeley, Fotheringham, and Morais 1999).

Fuel Mass and Stand Age

It has long been held that fuel mass increases with stand age (Philpot 1977), but this has been criticized as oversimplistic because it ignores tremendous species-specific variability in rates of biomass accumulation (Fig. 8.9). For example, some *Ceanothus* species may accumulate many times more biomass in less than 20 years than *Adenostoma fasciculatum* does in 60 years (Riggan et al. 1994; Regelbrugge 2000). Also at 10 years of age north-facing aspects may have greater biomass accumulation than drier south-facing slopes do at 80 years of age (Black 1987). This fact alone makes landscape-scale predictions of flammability based on stand age extremely difficult. Complicating the prediction of flammability with stand age is the increasing proportion of biomass in large diameter stems that

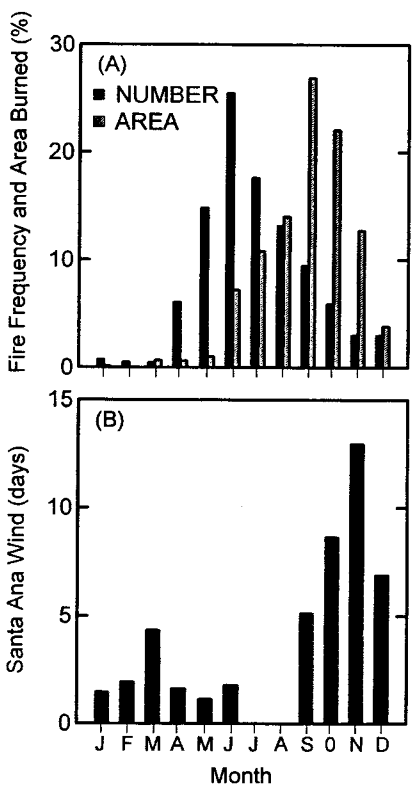

Figure 8.8. Seasonal distribution of fire occurrence and area burned during the twentieth century in Los Angeles County (data from Statewide Fire History Database; see Fig. 8.5) and seasonal distribution of Santa Ana winds (from Weide 1968).

combust only under the most extreme burning conditions. Also highly productive stands are often more mesic sites, and this, plus greater fuel density and higher fuel moisture, may reduce flammability. However, under extreme conditions, once ignited, productive sites sustain greater energy release than less productive stands (Riggan et al. 1988).

Successional changes in biomass (live and dead kg/ha) range from 1200 to 9000 in the first postfire year to 8000 to 13,000 after a decade, and 30,000 to 66,000

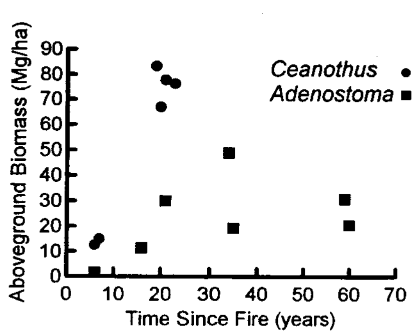

Figure 8.9. Live and dead aboveground biomass for chaparral shrubs at different times since fire based on several studies (data from Regelbrugge 2000).

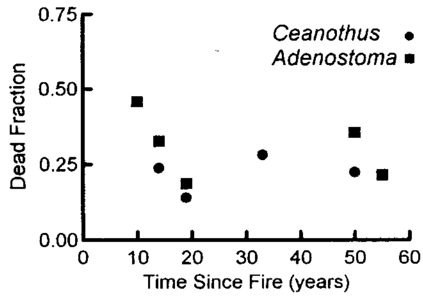

Figure 8.10. Fraction of total biomass comprising dead material for chaparral shrubs at different times since fire based on several studies (data from Paysen and Cohen 1990; Regelbrugge 2000).

(sometimes 100,000) in mature stands (Specht 1969, 1981; Green 1970; Keeley and Keeley 1984). Specht (1969) reported that the proportion of dead biomass exceeded 50% in mature chamise chaparral, and Green (1970) found 66% dead in mature *Cercocarpus betuloides*. These early reports led to the generalization of 1% dead for each year after canopy closure (Green 1981). However, more extensive studies (Fig. 8.10) report 30% dead/live ratios across the span from 20 to 60 years and no significant relationship with age (Paysen and Cohen 1990; Conard and Regelbrugge 1994; Regelbrugge 2000). It is apparent that dead/live ratios are a complicated function of many aspects of site composition and history. For example, unusually severe soil droughts may dramatically increase mortality, particularly of shallow-rooted *Ceanothus* shrubs, and this can occur in young or old stands (Keeley 2000; Davis et al. 2002). Also prior fire history may play a role; for example, chaparral stands burned by light fires leave large volumes of standing dead biomass that can produce very high dead/live ratios in young successional stands where high volume of dead fuels is not expected (e.g., Fig. 8.10).

In general, chaparral less than 25 years old has less than 20% dead, and this is insufficient to carry fire under "prescribed fire weather conditions" (Green 1981). Under severe weather conditions stand age (and total biomass and proportion dead) is less important in determining fire spread (Dunn 1989; Keeley, Fotheringham, and Morais 1999; Zedler and Seiger 2000).

The conclusion that older stands of chaparral generate fires of greater intensity needs to be viewed with caution. Fire intensity, which is often measured as fireline intensity or energy released per meter of fire front (Borchert and Odion 1995), can vary greatly depending on the interaction between weather and fuels. Sometimes intensity is equated with fire severity, which is defined as the ecological impact of the fire, and is often measured by mortality or the amount of plant biomass consumed, or alteration of nutrient cycles. However, a fast-moving fire that consumes little fuel and a slow-moving fire that consumes more fuel can achieve the same fireline intensity, and thus intensity and severity can not always be equated. In general, fire intensity is important to understanding options for fire suppression (Countryman 1974), whereas fire severity is most relevant to postfire ecosystem recovery (Keeley 1998b). Lastly, large fires often are equated with fires of high intensity, but they need not be. Large fires or mass fires are often described as catastrophic fires, but this latter term best refers to the impact of fire upon property and lives.

Past and Present Shrubland Fire Regimes

Understanding the extent of human impact on chaparral ecosystems requires that we reconstruct historical fire regimes. Stand-replacing crown fires typical of shrublands (Fig. 8.2) are not conducive to the formation of a tree-ring record of fires, as with surface fire regimes in montane coniferous forests. Thus reconstructing historical burning patterns for chaparral requires alternative approaches such as interpretation of sedimentary charcoal records.

Charcoal deposits in varved sediment cores from the Santa Barbara Channel have generated estimates of prehistoric fire frequency. Byrne, Michaelsen, and Soutar (1977) calibrated this procedure by comparison of annual varves from modern cores with U.S. Forest Service fire records. They found a significant correlation between large charcoal deposition events and incidence of large fires (>20,000 ha) in the adjacent mountain range less than 50 km from the core site. Using a core for the period from AD 730 to 1505, they were able to detect significant charcoal deposition but less than in the modern period, suggesting a lack of frequent small fires, unlike the contemporary pattern (Moritz 1997). They did, however, find two major peaks approximately 100 years apart with smaller peaks at 20- to 60-year intervals, and suggested this period had few fires, widely spaced, which became large conflagrations capable of generating large pulses of charcoal. Mensing, Michaelsen, and Byrne (1999) analyzed similar cores at a finer resolution and concluded that large fires were a feature of this region long before modern fire suppression.

Native American Impacts

Tree-ring records of fire scars from the coastal ranges and the Sierra Nevada have been interpreted to suggest that during the few hundred years prior to Euro-American colonization fire frequencies exceeded the level expected from lightning alone (Reynolds 1959; Greenlee and Langenheim 1990). From historical records and ethnographic accounts there can be no doubt that California Indians regularly utilized fire to manage their environment (e.g., Lewis 1973; Timbrook, Johnson, and Earle 1982; Wickstrom 1987; Anderson and Moratto 1996). The extent to which this management practice altered landscapes is a matter of debate. Due to the naturally high fire frequency of lightning fires in the coniferous forests of the Sierra Nevada, Vale (1998) has argued that the additional burning by Indians did not alter landscapes except in localized areas (but cf. Anderson, Barbour, and Whitworth 1998). On the other hand, it has been hypothesized that direct use of fire by Native Americans greatly altered landscape patterns in the lower elevation coastal range foothills, primarily through type conversion of shrublands and woodlands to grasslands and other herbaceous associations (Cooper 1922; Wells 1962; Huenneke 1989; Keeley 1990, 2002; Hamilton 1997). This hypothesis is supported by the low lightning activity, high Indian populations, shrub-dominated landscapes, limited resources for Native Americans in undisturbed shrublands, and weak resilience of shrublands to high fire frequency (Keeley, in review).

Euro-American Settlement Impacts

Euro-American settlers further increased fire frequency during the nineteenth century, primarily for the purpose of expanding rangeland into chaparral and coastal sage scrub dominated landscapes. The economy of the Spanish and later Mexican period was primarily based on pastoralism, and most historical sources indicate extensive grasslands at the time of colonization and limited need for immediate rangeland expansion (Keeley, 2002). Nonetheless, there are historical reports of these early pastoralists using fire to open up shrublands and increase forage (Kinney 1887), and this is reflected in increases in grass pollen from sediment cores (Russell 1983).

By the middle of the nineteenth century there was increasing pressure for rangeland expansion, and this was felt most severely in the coastal ranges south of San Francisco where 80% of livestock production was confined (Ewing et al. 1988). Following the Gold Rush of 1849, with an influx of American settlers, brush burning for the improvement of grazing became extensive throughout California. Ranchers in the foothill regions regularly burned large areas of brushland, and it became the practice of itinerant sheepherders, after leaving a grazing area, to set fires (Brown 1945; Bauer 1974; Nichols, Adams, and Menke 1984). Burcham (1957) contends that all rangelands in the state were fully occupied by 1880. A similar perspective is that of Brown and Show (1944) who stated, "It is generally conceded that what is known as the 'pastoral era' of California ended in 1870. In that year, good pasture land, which was also agricultural in character, rose to a price of from 75 cents to $6.00 per acre." In the succeeding decades there was extensive pressure to utilize fire for the purpose of opening up shrublands and increasing forage (Lee and Bonnicksen 1978). The burning by these stockmen in mountain watersheds of southern California were thought to be responsible for damaging floods on both the coastal and interior sides of the San Gabriel Mountains, leading to its designation as the first forest reserve in California (Lockmann 1981).

One factor contributing to the use of fire in the opening up of shrublands was apparently the homestead laws that allowed acquisition of 65-ha parcels from public domain land (Lee and Bonnicksen 1978). Such parcel sizes were generally sufficient to maintain a homestead based on stock production, but this plan did not work in the rugged hills of southern California, where homesteads were centered in small valleys known as *potreros*, surrounded by impenetrable chaparral. "Since the potreros were too small to support an economically sound cattle operation, stockmen supplemented meadow grazing with forage produced by periodically burning the adjacent chaparral" (Lee and Bonnicksen 1978). Since brush burning was an essential resource use practice for stockmen, they burned extensive areas of chaparral (Barrett 1935; Brown and Show 1944; Brown 1945). For instance, in 1887 it was reported, that in the southern portion of San Diego County that "at least one third of the land covered with brush, grass and oak timber has been burnt off by settlers in the past eighteen months" (Lee and Bonnicksen 1978). As a consequence of early settler burning, fire control laws were enacted soon after statehood in 1850 (Clar 1959). Not surprisingly, in the early part of the twentieth century, ranchers were often the primary opponents to fire exclusion policies, which in southern California was prompted by the need for watershed protection in the coastal plain (Lee and Bonnicksen 1978).

In summary, it is apparent that during this settlement period the primary alteration in fire regime was to increase the frequency of fires on shrubland landscapes. This was an era of very limited fire suppression, and yet fires were much as they are today in that large crown fires covering tens of thousands of hectares were not uncommon (Kinney 1900; Barrett 1935; Brown and Show 1944; Brown 1945; Minnich 1987). For example, one of the largest fires in Los Angeles County (24,000 ha) occurred in 1878 (Keeley, Fotheringham, and Morais 1999), and the largest fire in Orange County's history was over a quarter million hectares and occurred in 1889 (Lee and Bonnicksen 1978).

Twentieth-Century Patterns of Burning

Burning patterns during the twentieth century are shown for the nine counties in central and southern coastal California (Fig. 8.11). Most counties exhibited little or no change in area burned except for Los Angeles and Riverside counties in southern California, which exhibited highly significant increases in area burned during the twentieth century. In contrast to the situation in western U.S. coniferous forests, fire suppression clearly has not excluded fire from these shrubland landscapes. Collectively the 1920s, 1940s, and 1970s were high decades, and the 1930s and 1960s were low. Possible explanations for these patterns are that they result from (1) decadal-scale variation in climate, (2) natural cycles resulting from fuel buildup, and/or (3) human demographic patterns.

Role of Climate/Weather

There are numerous suggestions in the literature of extended droughts contributing to extraordinarily severe fire seasons, but with a few notable exceptions, most lack statistical rigor. Minnich (1983) reported that there was a significant positive relationship between precipitation and area burned in coastal sage scrub of southern California and adjacent Baja California, but he presented no statistics to support this contention. He also inspected patterns of chaparral burning over this time period and concluded no such relationship existed with chaparral. However, others have reported a relationship between precipitation and burning

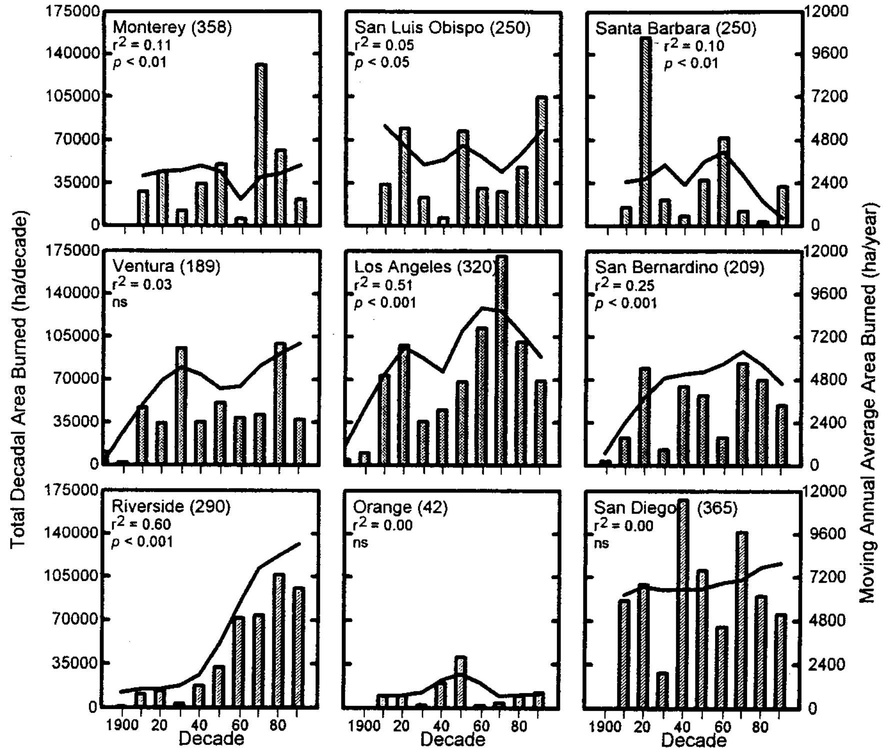

Figure 8.11. Area burned per decade and 10-year running annual average during the twentieth century for nine counties in central and southern California (data from the Statewide Fire History Database; see Fig. 8.5). Shrubland area in thousands of hectares shown in parentheses following the county name (from Callaham 1985).

in chaparral. One line of evidence is the spatial relationship between average precipitation and fire occurrence within the chaparral zone of San Diego County (Krausmann 1981). Another line of evidence is the demonstration that chaparral burning varies temporally with changes in precipitation; little area is burned following rainfall years where spring precipitation is >200 mm (Davis and Michaelsen 1995). These observations have been interpreted to mean that more rain translates into more biomass and thus greater fuels for burning in the subsequent fire season.

Using the FRAP data set (Fig. 8.11), we found few statistically significant correlations between patterns of rainfall and burning for chaparral and coastal sage shrublands combined. For each county separately, or all counties collectively, there was no significant relationship between total acreage burned per year and the nearest station with long-term records for:

1. total annual (January–December) precipitation,
2. growing season (November–June) precipitation,

3. spring (January–May) precipitation,
4. summer (June–August) precipitation, or
5. previous growing season's precipitation.

There was, however, a weak, but significant negative correlation ($p < 0.05$, $r^2 = 0.05$–0.06, $n \geq 88$) between October precipitation and area burned in each of the southern California counties, indicating that early autumn rains cut short the fire season at its peak.

Weather conditions affecting autumn foëhn wind-driven fires are most critical in determining area burned. Santa Ana wind conditions are largely responsible for fires becoming large and is reflected by the strong correlation between fire size and high temperatures. On the Los Padres National Forest fires generally ignite on days when the temperature is 3 to 5°C greater than the monthly average, and large fires never originate on days where temperatures are <25°C at the Santa Barbara airport (Davis and Michaelsen 1995; Moritz 1997). Moritz (1999) examined this relationship between severe fire weather (defined as days with maximum temperatures at the Santa Barbara airport ≥32°C) and extreme fire events in the central portion of the Los Padres National Forest. He found that large fires (>4000 ha) were strongly associated with severe fire weather. In this part of California severe fire weather is often, but not always, associated with foëhn winds (Schroeder et al. 1964; Dunn and Piierto 1987; Ryan 1996). However, farther south, for example, in the Santa Monica Mountains, all large fires appear to be driven by Santa Ana winds (NPS, Santa Monica Mountains National Recreation Area, unpublished data). In general, the largest wildfires in the central and southern coastal region are during severe fire weather conditions that include high temperatures, coupled with low humidity and high winds (Coffin 1959; Pirsko 1960; Schroeder et al. 1964; Weide 1968; Countryman, McCutchan, and Ryan 1969; Phillips 1971; Countryman 1974; Dunn and Piierto 1987; Gomes et al. 1993; Davis and Michaelson 1995; Minnich and Chou 1997).

Role of Fuel Cycles

Fuel accumulation was implicated in burning patterns in California shrublands by modeling studies published in the 1970s (Rothermel and Philpot 1973; Philpot 1974a,b). Based on untested assumptions about rates of fuel accumulation and effectiveness of fire suppression, it was concluded that large fires were increasingly more common because of an accumulation of older age classes of vegetation (Fig. 8.12). In support of this idea are many anecdotal references that fire fighters and fire researchers often relate about the tendency of fires to stop upon encountering young age classes of fuels (e.g., Philpot 1974a,b; Minnich 1998). An example of how these anecdotes are often used is the story about the 1970 Laguna Fire (one of the largest in California's history), in which it is claimed the fire died out when it encountered young age classes of vegetation (Rich Minnich, public communication, National Public Radio's "All Things Considered" radio broadcast, June 10, 1999). While that observation may be true, the deduction that

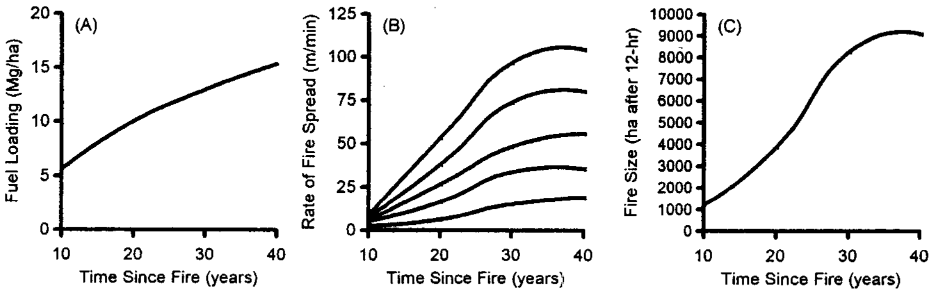

Figure 8.12. Modeling studies by Philpot (1974a, 1974b). (A) Assumed successional changes in fuel loads, (B) predicted rate of fire spread at increasing wind speeds from 10 to 50 kph, and (C) predicted fire size after 12 hours burning under sustained 50 kph wind speed. From these models it was concluded that as chaparral stands increase in age due to fire exclusion, there is a resultant increase in fuels, fire spread rate, and fire size. Following suggestions by Countryman (1974), these models were interpreted to support a fire management policy that relied heavily on prescription burning to produce a landscape comprising a mosaic of age classes.

there is a causal relationship is doubtful because the Laguna Fire burned over 10,000 ha of young vegetation (5–20 years) prior to its being extinguished (Dunn 1989), and the fire was contained only after a week of very severe Santa Ana winds subsided (Keeley, personal observations). In this fire, as well as other catastrophic fires, changes in fire behavior leading to containment often have had more to do with temporal changes in weather than spatial changes in fuels (Dunn and Piirto 1987). Although one can point to various fire perimeters that suggest fuel age is a barrier to fire spread (e.g., Philpot 1974), there are others that indicate it is not, such as half of the 5900 ha Romero Fire that burned above Santa Barbara in 1971 consumed seven-year old fuels from the 1964 Coyote Fire (Gomes et al. 1993). In short, there is no statistical evidence to support the notion that southern California landscapes supporting young vegetation are effective barriers to the spread of catastrophic fires. This of course is not meant to suggest that stand age has no effect on fire spread, only that its effectiveness is strongly controlled by weather (see the section below on Future Fire Management Strategies).

Fire history data also have been used to support the idea of fuel-driven fire behavior. Radtke, Arndt, and Wakimoto (1982) observed that peak decades of burning were followed by decades of very little burning in the Santa Monica Mountains of Ventura and Los Angeles counties. It was suggested that these decadal variations in burning represented a cyclical pattern driven by fuel loading. Their confidence in this model is illustrated by their future prediction that for the Santa Monica Mountains, the 1980s decade would be a peak and would be followed by a decline in burning during the 1990s. In retrospect we now know that, although the 1980s were high, the 1990s were even higher (Santa Monica Mountains Recreation Area, unpublished data). The primary weakness in explaining

Table 8.2. Shrubland area,[a] population density,[b] and estimated fire rotation intervals[c] for the shrub-dominated counties in California, arranged north to south

County	Brush (10^3 ha)	People/ 10^6 ha brush	Fire rotation interval (yr) pre-1951	Fire rotation interval (yr) post-1950
Monterey	358	0.99	115	64
San Luis Obispo	250	0.87	60	48
Santa Barbara	250	1.48	47	81
Ventura	189	3.54	121	34
Los Angeles	320	27.69	44	30
San Bernardino	209	6.79	46	37
Riverside	290	4.04	225	38
Orange	42	57.39	36	29
San Diego	365	6.84	35	41

[a] Area as of 1985, from Callaham 1985.
[b] Population density for 1990, from *http://www.census.gov/population/cencounts/ca190090.txt*.
[c] From Keeley et al. 1999.

decadal variations in burning by changes in fuel loads is the fact that the total burning during a decade comprises only a fraction of the fuels on the landscape and substantial fuel loads are available for burning every decade. For example, fire rotation intervals (Table 8.2) indicate that in most counties only 20% to 30% of the landscape burned in any given decade; thus decades of peak burning should not automatically be assumed to alter the future course of burning by leaving the landscape with limited fuels.

For southern and central coastal California, fire history data refute the contention made by Minnich (1989, 1998, 2001; Minnich and Cho 1997) that chaparral fire occurrence is constrained by the rate of fuel accumulation. Fire hazard estimates are either independent of age (Moritz 1999) or only weakly dependent up to 20 years of age (Schoenberg et al. 2001; Peng and Schoenberg 2001). In addition stand-age classes burned in the eight largest wildfires in the Santa Monica Mountains illustrate that these extreme events are not dependent on accumulations of older fuels (Keeley, Fotheringham, and Morais 1999). Indeed, in this range the greatest proportion of burned vegetation is in the younger aged stands, for both coastal sage scrub and chaparral (Fig. 8.13). Also vegetation type, which has a profound influence on fuel distribution (e.g., Fig. 8.9), has been shown to have little influence on fire history in the Los Padres National Forest (Moritz 1999).

Alterations in the landscape distribution of fuels have also been implicated in changes in fire size. It has been proposed that due to fire suppression, there has been an increase in the age and homogeneity of fuel distribution leading to larger and higher-intensity fires (e.g., Minnich 1989, 1995, 1998; Minnich and Cho 1997). The only data in support of this model are the high frequencies of small fires south of the U.S. border, which are interpreted as solely the result of natural burning cycles in the absence of fire suppression. However, north of the border

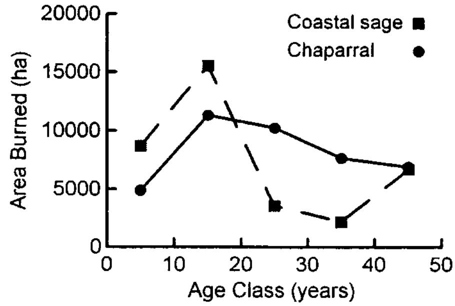

Figure 8.13. Age classes of chaparral and coastal sage scrub stands burned by all fires over 5000ha from 1967 to 1996 in the Santa Monica Mountains (data from the U.S. National Park Service, Santa Monica Mountains National Recreation Area, Thousand Oaks, CA). Greater burning of young age classes of coastal sage scrub is likely due to more flammable fuels, longer fire season, and the concentration of this vegetation adjacent to urban centers, which are major sources of ignition.

fire suppression activities have not resulted in fire exclusion (Moritz 1997; Conard and Weise 1998; Keeley, Fotheringham, and Morais 1999; Weise et al., in press). Thus the patterns north and south of the border, while interesting, cannot be held up as an example of what happens to landscapes subjected to a fire suppression policy. Such fire management policies can not be held responsible for large destructive wildfires (Keeley and Fotheringham 2001a,b) as large fires have been a common feature of the southern California landscape throughout the ninteenth and twentieth centuries (Keeley, Fotheringham, and Morais 1999; Keeley and Fotheringham 2001a). Additionally sediment cores show the frequency of large fires has not changed during the past 450 years (Mensing, Michaelsen, and Byrne 1999), and colorful, but less authoritative, is the Digueño Indian legend of a large catastrophic fire sufficient to lead to migrations of tribes in San Diego County at about the time of Columbus (Odens 1971, p. 8). All of these observations suggest large fires are not a modern artifact of fire suppression as proposed elsewhere (Minnich 1989, 1995, 1998; Minnich and Dezzani 1991; Minnich and Chou 1997).

We now know that although the models developed by Philpot and others may be sound, their conclusions were flawed by incorrect assumptions. The assumption of a steady increase in fuels was inaccurate (Figs. 8.9 and 8.10), and the assumption that fire suppression was effectively excluding fire from shrubland landscapes was wrong (Fig. 8.11). In short, patterns of burning on shrubland landscapes cannot be explained solely by changes in accumulation of fuels (Moritz 1997, 1999; Conard and Weise 1998; Keeley, Fotheringham, and Morais 1999; Peng and Schoenberg 2001). Indeed, modeling studies that consider landscape patterns of fire spread conclude that stand age alone cannot constrain fire size (Zedler and Seiger 2000). If that were true, then just a single large Santa Ana wind-driven fire would reset the landscape to the same age class, which would forever be doomed to burn as a single large unit. Zedler and Seiger's model shows that even in the absence of Santa Ana fires, if stand age were the only controlling factor, over time, burn units would coalesce and become larger and larger with each fire cycle.

Role of Human Demography

Clearly, humans have perturbed shrubland fire regimes, but unlike the situation in many western U.S. forests, the primary impact has been through increased fire frequency (Table 8.1, Figs. 8.4 and 8.5c–d) and not through fire exclusion (Fig. 8.11). Collectively, across all counties considered in Figure 11, there was a significant correlation between fire frequency and population density and between fire frequency ($r^2 = 0.51$, $p < 0.05$, $n = 9$) and area burned ($r^2 = 0.71$, $p < 0.01$, $n = 9$). Southern California (defined in Fig. 8.3 legend), with the highest rate of population growth (Fig. 8.5b), also has had the greatest increase in wildfire ignitions (Fig. 8.5d). In contrast, the central coastal region has far fewer human ignitions (Figs. 8.4 and 8.5c), which is in line with the much lower population (Fig. 8.5a).

Indirectly the public infrastructure of roads contributes to patterns of burning. The central coastal region has substantial portions of its landscape lacking public roads, which is in stark contrast to the vast highway network connecting most parts of southern California. Fully one-third of all human-caused fires on Forest Service and CDF protected lands in southern California occur along roads (Gee 1974; Conard and Weise 1998). On shrubland landscapes near metropolitan areas, such as the Santa Monica Mountains, the vast majority of fires originate along roadways (Los Angeles County Fire Department, unpublished data).

In light of these considerations it seems probable that some portion of the decadal variation in burning during the twentieth century (Fig. 8.11) may have a human dimension. In the early part of the twentieth century populations in many parts of California were increasing rapidly (Fig. 8.5a–b). With this influx of people, came increased anthropogenic impact on the natural fire regime, driven largely by the increased mobility the automobile afforded; car registrations in California rose from 191,000 in 1915 to 1,500,000 in 1925 (Davis 1967). During the period 1908 to 1920, every county in southern California voted large bonds for road building (Davis 1967). Roads provided increased access to wildland areas. For example, a doubling in wildland use between 1916 and 1920 (Show and Kotok 1923) coincided with a marked increase in wildfire incidence in southern California (Fig. 8.5d). This rapid population growth and increased mobility strained the ability of fire protection in California during the early part of the twentieth century (Clar 1959). With the expanding population came an expansion of development at the urban–wildland interface, which then increased public susceptibility to wildfire impacts. As a consequence the decade of the 1920s witnessed some particularly destructive wildfires that increased public pressure for fire protection and prevention (Clar 1959).

In response, during the 1930s fire management agencies stepped up their attack on wildland fires by the introduction of lookout towers and aircraft for better reconnaissance, which decreased the size of some fires due to early detection (Clar 1969; Pyne 1982). During this period various innovations were introduced to suppress fires, although effective suppression was elusive due to the inaccessibility of remote wilderness areas (Brown and Show 1944; Pyne 1982). A system of fuel breaks was one early answer to this problem, and creation of 200 CCC (Civilian Conservation Corp) camps throughout the state during the Depression contributed significantly to this network. Increased fire suppression activities due to an excess of man power from federal relief programs (Clar 1969 described it as a "forced feed" of the California Division of Forestry), coupled with reduced "motor touring" (e.g., AAA memberships dropped 40% in the five years following 1929; Davis 1967) perhaps contributed to the drop in area burned during the Great Depression in many counties.

Diversion of resources to the "war effort" during the first half of the 1940s contributed to diminished fire suppression capacity (Brown 1945; Clar 1969) and may account for the peak burning that occurred in some counties during that decade (Fig. 8.11). San Diego County stands out because its worst decade for wildfires was the 1940s. Zahn (1944) suggests the extraordinary fires of this era were the result of the aircraft industry, which had concentrated a great deal of the war effort in San Diego County. He described the situation at the time as follows: "Bootleg fuel, high payrolls and a yen for the open spaces have resulted in hundreds of aircraft workers motoring to the hills—night or day—between work shifts. Most of these workers are newcomers to California, unfamiliar with the tinder-box potentialities of local brush."

The modern era of effective fire suppression was introduced in the 1950s with the development of air tankers for fire fighting (Pyne 1982), and this impact was evident in a 10-fold drop in burning across the country (Dombeck 2001). However, in California these techniques have not proved very successful in halting fires during extreme Santa Ana wind conditions (Countryman 1974).

In short, despite innovations, fire suppression has not diminished the wildland fire problem in California. Indeed, since the 1950s, there has been an increase in the allocation of funds to the California brushfire problem (Bonnicksen and Lee 1979; Kinney 1984), and an increase in the loss of property and lives (Rogers 1982; Martin and Sapsis 1995). Additionally, due to television, there has been increased public awareness of large-scale wildfires. Over this period there have been a number of workshops, conferences, and proceedings volumes published on this wildland fire problem—roughly one every 5 to 10 years since 1950—and these offer a diversity of opinions on the role of fire in the California landscape. Although not a popular view, it has been frequently suggested that the problem stemmed in large part to the burgeoning population and poor zoning regulations attendant with urban sprawl into the foothills. The problem was evident 50 years ago. For example, Zivnuska, Arnold, and Arment (1950) warned of this "potentially explosive situation," and noted "it is known that one of the significant trends in recent population changes has been the increase in number of residences in the flash-fuel types adjacent to primary watersheds." Under these conditions catastrophic fires are not necessarily the largest fires, as witnessed by the rather small

Oakland Hills Tunnel Fire in October 1991 (725 ha) that burned nearly 3000 structures and killed 25 people (Booker, Dietrich, and Collins 1995).

In summary, severe fire weather occurring each autumn coupled with human demographic patterns would seem to explain patterns of burning (Fig. 8.11) far better than changes in available fuels. During the twentieth century any changes in the fire regime have been dwarfed by the changes in land development patterns, which have increasingly placed more people at risk to the natural forces long present on the landscape (Davis 1965; Bradshaw 1987). This pattern continues—for example, in the 25 years prior to 1980, 2408 homes and other structures were destroyed by wildfires in California but in the subsequent 14 years the number tripled (*http://www.prefire.ucfpl.ucop.edu/wildfire.htm*). Preference for a rural lifestyle and the skyrocketing cost of suburban housing in large metropolitan areas has progressively increased the urban–wildland interface. Of particular concern is the prediction that rural population will exceed urban growth in the foreseeable future (Bradshaw 1987). For both economic and political reasons the notion that urban sprawl is responsible for natural wildfires becoming catastrophic fires is unpopular, in part, because it seems to defy the inherent belief that it is possible to engineer solutions to all environmental problems.

The Contemporary Versus Natural Fire Regime

There is reason to believe that the contemporary fire regime in these shrublands mirrors the natural crown fire regime far more than is generally accepted (cf. Bonnicksen and Lee 1979; Minnich 1983; Pyne 1982). Today in southern California, fire incidence peaks in the summer, but most area burned is from autumn fires (Fig. 8.8a). Likewise the natural fire regime was probably characterized by many small summer lightning-ignited fires and a few large autumn fires driven by Santa Ana or Mono winds that burned large areas (Keeley and Fotheringham 2001a). This model would seem to be contradicted by the fact that Santa Ana or Mono winds are northeast winds, whereas summer thunderstorms are associated with south winds, and the two do not commonly coincide (Coffin 1959). Consequently today it is rare for Santa Ana wind-driven fires to be other than anthropogenic in origin. However, under natural conditions the fact that lightning fires burned for months (Minnich 1987), coupled with the relatively close temporal juxtaposition of the July–August lightning fire season (Keeley 1982) with the September–November Santa Ana winds (Fig. 8.8b), makes it inevitable that lightning ignitions would occasionally have been spread by these foëhn winds (Keeley and Fotheringham 2001a). While such events could not have been frequent, we know from historical documents that summer lightning-ignited fires can burn for more than a month and consume on the order of 10^3 ha (Minnich 1987). This pales in comparison to the 10^4 ha that are often covered in a single day by a Santa Ana wind-driven fire (Phillips 1971).

Davis and Burrows (1993, 1994) modeled the long-term fire regime in chaparral by linking physical models based on fire spread equations to fuel models of stand senescence. Their simulations predicted a prehistoric fire regime of variable sized fires that produced a landscape mosaic of different age classes. With one ignition every 10 years (typical lightning-ignited fire frequency for coastal California; Keeley 1982) their model predicted that most fires would be large and over 80% of the landscape would burn at ages greater than 95 years. These conclusions are supported by other evidence that points to a natural fire regime of large fires and long fire return intervals for these coastal range landscapes (Greenlee and Langenheim 1990; Byrne, Michaelsen, and Soutar 1977; Mensing, Michaelsen, and Byrne 1999; Keeley and Fotheringham 2001a).

Alternatively, it has been argued that prior to the current fire suppression policy, landscapes were immune to Santa Ana wind-driven fires because lightning fires kept the shrublands in a fine-scale mosaic of young age classes (Minnich 1989, 1995, 1998). It is presumed that this mosaic was quickly erased by highly effective fire suppression during the first couple decades of the twentieth century (Minnich 1990). However, historical records do not support this notion. For example, 90% of the 214,000 ha of shrublands on the San Jacinto Forest Reserve were estimated to be 30 years or older when surveyed at the end of the nineteenth century, which represent far older age classes then present today (Keeley and Fotheringham 2001a). Clearly, this landscape was not a fine-scale mosaic immune to large fires. In addition the early history of forest protection does not support the idea that highly effective fire suppression was present in the opening decades of the twentieth century (Clar 1959; Lockmann 1981). Minnich and Chou's (1997) suggestion that fire suppression activities "culminated in extensive fire outbreaks as early as 1919" is contradicted by historical documentation that reports large fires in the region long before this date, and before any fire suppression activities (e.g., Kinney 1900; Barrett 1935; Brown and Show 1944; Brown 1945; Lee and Bonnicksen 1978; Radtke, Arndt, and Wakimoto 1982). Nationwide there is no evidence of substantive reductions in area burned due to fire suppression until midway through the twentieth century (Dombeck 2001).

Historically fire intensity was variable, and there is no credible evidence that it has increased during the era of fire suppression (Keeley, Fotheringham, and Morais 1999). The primary changes in the fire regime are that humans have replaced lightning as the primary source of ignition and fire frequency has increased, particularly in areas of high population density such as southern California (Figs. 8.4 and 8.5). Because fire prevention has been ineffective at eliminating human fires, presently and for the foreseeable future, fire suppression is required just to maintain some semblance of the natural fire regime.

Impacts on Vegetation

In contrast to the paradigm suggested for many western U.S. forests, ecosystem health of shrublands is threatened not by a lack of fire but by high fire frequencies that exceed the resilience of many species. Examples of high fire frequency induced extirpations are numerous (e.g., Gause 1966; Zedler et al. 1983;

Haidinger and Keeley 1993; Zedler 1995; Keeley 2000). Generally, the threat of high fire frequency is lessened on very low nutrient soils where postfire annual biomass is limited and less likely to carry a repeat fire. Where fires occur more than once in a decade, nonsprouting chaparral shrubs are entirely lost from the system. Commonly exotic grasses and forbs will take their place, and as these increase in importance, they appear to competitively displace the native annuals. A similar course is evident in coastal sage scrub under higher fire frequency. In both of these crown fire ecosystems, high fire frequency favors annuals over woody plants, and this advantage increases with increasing soil aridity (Wells 1962).

As fire frequency increases, fuel structure changes and subsequent fire behavior changes. With increasing exotic herbaceous cover, the seasonal window of flammability increases (Radtke, Arndt, and Wakimoto 1982), and fire behavior becomes a mixture of crown and surface fires. This has two very important consequences. Surface fires connect the woody fuels where otherwise they might be too widely spaced to carry a crown fire, and thus exotic herbs shorten the fire return interval. Another important consequence is that fire intensity is lower where surface fires occur and this contributes to increased survivorship of exotic annual seed banks (Fig. 8.14). With continued disturbance these nonnative invasives may replace the entire ecosystem (Keeley, 2001), and type conversions of shrublands to exotic grasslands are well documented (e.g., Cooper 1922; Bentley 1967; CDF 1978; Biswell 1989; Minnich and Dezzani 1998). As a consequence exotic grasslands tend to replace shrublands in the proximity to urban environments, where the higher ignition sources in the company of flashy fuels have the potential for even greater fire frequency. Evidence of this is seen in the substantially shorter fire return interval in grassland vegetation at the urban–wildland interface than observed for shrublands at the interface (J. Spero, California Division of Forestry, personal communication, 1999).

The extent of such type conversion is unknown because of past disturbances, which includes Indian burning throughout the Holocene and burning coupled with intensive livestock grazing in the past 200 years. In the coastal counties from Monterey southward (Fig. 8.3) exotic annual grasslands cover nearly two million hectares or 25% of the wildland landscape, and less than 1% has significant patches of native perennial bunchgrass (Huenneke 1989). Although it is often taken as a matter of faith that these landscapes have always been grassland (Heady 1977), there is evidence that many exotic grasslands were formerly dominated by woody associations (Cooper 1922; Wells 1962; Oberbauer 1978; Huenneke 1989; Keeley 1990, Hamilton 1997). Today these landscapes comprise a mosaic of vegetation patterns (Fig. 8.15) that appear to be disturbance induced (Wells 1962). Grasslands on this modern landscape comprise a new quasi-equilibrium of nonnative annuals that are somewhat resistant to recolonization by native shrubs. It is a dynamic process whereby as disturbances increase or wane, vegetation physiognomy shifts between exotic grassland and shrubland/woodland (Hobbs 1983; Freudenberger, Fish, and Keeley 1987; Callaway and Davis 1993).

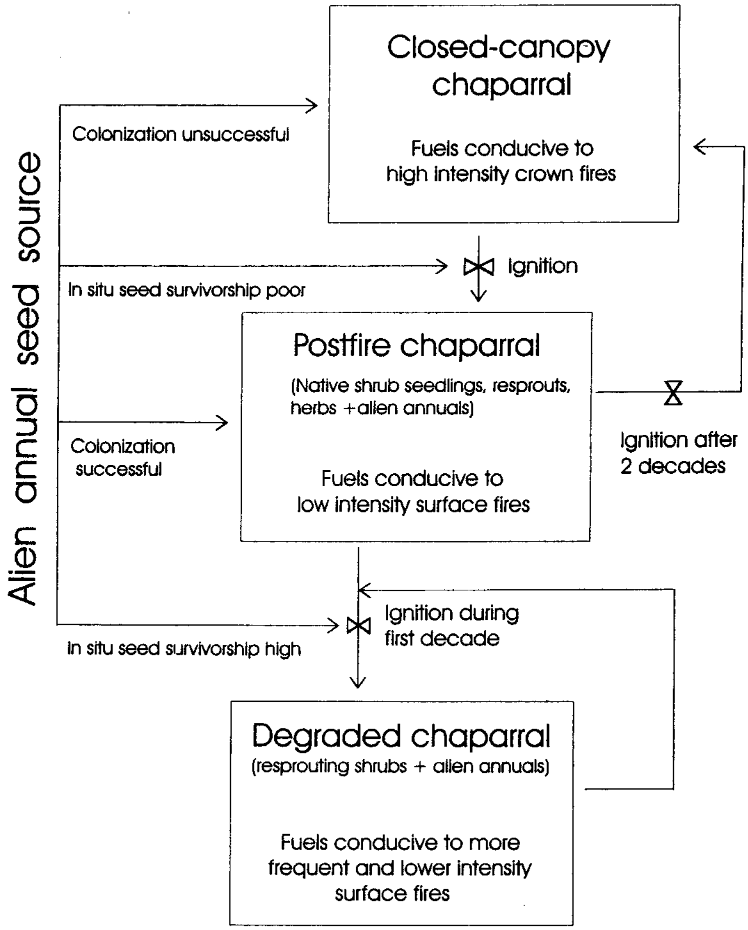

Figure 8.14. Schematic diagram of how rate of fire ignitions in chaparral affects alien plant invasion and how alien invasions affect fuel loads, which in turn alter fire frequencies, making sites more conducive to further invasion (from Keeley, 2001).

Future Fire Management Strategies

It has been suggested that "after nearly a century of suppression" there is a need for a reintroduction of fire into chaparral through prescribed burning (Minnich and Dezzani 1991; Minnich and Franco-Vizcaíno 1999). However, fire history data do not support this management strategy. On most shrubland landscapes there is an abundance of fire, and 60- to 70-year-old stands, considered to be the normal age for burning (Minnich 1989), are rare at the present time (Keeley 1992). Indeed, the current fire rotation interval of 30 to 40 years is shorter than

Figure 8.15. Vegetation mosaic of nonnative annual grassland and shrublands in the central coastal ranges of California (photo by J. Keeley).

that calculated for the early part of the twentieth century (Keeley, Fotheringham, and Morais 1999). In light of the expected trends in population growth in California, and the close association between population density and fire incidence (Fig. 8.5), increased fire prevention is far more important to protecting natural resources than prescription burning or other methods of "fire restoration."

Consequently there is a need to reevaluate prescribed burning strategies for California shrubland landscapes. There are two common motivations for prescription burning: (1) for the benefit of natural resources and (2) as a fuel manipulation technique, primarily to reduce fire hazard but also to reduce the threat of soil erosion or air quality hazards, which may be worse under wildfire conditions. In many western U.S. forests, prescription burning provides both resource benefits and a reduction in fire hazard However, the reality for some ecosystems is that prescriptions reducing fire hazard, may not always enhance resource values and sometimes may detract (Johnson and Miyanishi 1995; Keeley and Fotheringham 2001b).

Prescription Burning for Resource Benefit

There may be little justification for using fire for resource benefit, since vast portions of shrubland landscape currently experience a higher than normal fire frequency. Lack of fire does not appear to pose a risk because postfire studies demonstrate that both chaparral and coastal sage scrub regeneration are highly resilient to even the most extreme fire events occurring after a long hiatus of burning (Keeley 1998, 2000).

One proposed benefit of prescribed burns is that they are done under more moderate weather conditions than are typical for wildfires, leading to less intense fires and less severe impacts on plant and soil resources (Green 1981; Moreno and Oechel 1991; Riggan et al. 1994; Wohlgemuth, Beyers, and Conard 1999). However, some of the experimental work demonstrating fire intensity effects on seed banks and soils have been done on piles of cut fuels, which do not accurately represent the fuel structure under natural conditions and are likely to generate unnaturally high soil temperatures. More importantly, however, even the most extreme fire wildfire events today probably do not fall outside the natural range of variation for these ecosystems.

Other resource benefits from prescription burning include invasive plant control, but the primary invasive problems involve herbaceous species, which invade shrublands when fire frequency increases (Fig. 8.14). It is not likely that prescription burning would displace these invasive species, unless the target is vulnerable to a particular seasonal window of burning. In shrublands there are no such windows of opportunity that are not equally damaging to some native species. Nonnative legume shrubs known as brooms (*Cytisus scoparius* and *Genista monspesulanus*) are sometimes targeted for removal with prescription burning, but these are inevitably replaced by exotic grasses (D'Antonio 2000). However, prescription burning for restoration of shrubland communities may be useful if accompanied by vigorous revegetation with native shrubs and herbs.

In general, there are few places where fire-dependent shrublands are threatened by the lack of fire and few instances where prescription burning is needed for natural resource benefits. The primary justification for prescription burning is for fire hazard reduction. However, in these ecosystems any additional fire carries with it the potential for negative impacts on resources. Negative impacts may arise not just from burning but can be associated with other fuel manipulations. For example, fuel breaks are possible corridors for bringing nonnative invasive species into wildland areas (Keeley, 2001).

Prescription Burning for Fire Hazard Reduction

Prescription burning carries with it a risk of fires escaping, and escaped fires are quite hazardous in crown fire ecosystems, most particularly chaparral landscapes with a complex urban–wildland interface. In order to ensure successful containment of a prescribed burn, there are strict limitations on the acceptable wind speed, air temperature, relative humidity, and fuel moisture—typically wind speeds below 17 kph (10 mph), relative humidities above 30%, air temperature below 32°C (95°F), and fuel moisture above 75% (Fenner, Arnold, and Buck 1955; Green 1981). This, of course, varies with the fuel load and landscape, and various combinations will produce acceptable prescriptions (Paysen, Narog, and Cohen 1998). One approach to reducing the risk of escaped fires is to burn in the spring, assisted by pretreatment of mechanical crushing and drying. Thus the target fuels are surrounded by less flammable living vegetation (Wolfram 1962). This procedure is expensive, and it has the potential for producing resource damage. For example, unseasonable application of fire inhibits postfire vegetation recovery (Florence 1985; Rundel, Parsons, and Baker 1987; Parker 1990) and is correlated with increased soil erosion (Turner and Lampinen 1983).

Because prescriptions are designed for safety, they are often marginal for burning. Under prescription weather conditions, fire spread is markedly influenced by fuel structure, and fire spread is often inhibited in stands less than 20 years of age (Green 1981; Paysen and Cohen 1990; Conard and Regelbrugge 1994). This is largely due to the lack of sufficient dead fuels required to spread fire to live foliage, and to the lack of fuel continuity between the ground and the shrub canopy and between adjacent canopies, factors that are extremely critical to fire spread under low wind and high humidity. Controlled burning of younger stands requires either prescriptions with risky weather conditions or pretreatment with biodegradable herbicides (which increase the dead fuels) coupled with seeding of exotic grasses that increase flashy (readily ignitable) fuels and increase surface fire spread.

Evaluating the effectiveness of prescribed burning at reducing fire hazard is complicated by the fact that such fuel management practices are never going to be fully effective against all fires. Wildfires are often more readily contained when they encounter young stands of vegetation, largely because lower fire intensities allow for safer access by fire suppression forces (Countryman 1974). However, landscape age mosaics created by rotational burning will not pose a barrier to wildfires ignited under severe fire weather, since the high winds readily push fires through young age classes (e.g., Fig. 8.13). Under these conditions young vegetation is of minimal value in halting the forward spread, and also firebrands are capable of spreading the fire kilometers beyond the front. Containment of shrubland fires burning under severe weather conditions usually requires a change to more favorable weather (Rogers 1982; Dunn and Piirto 1987; Gomes et al. 1993).

Thus prescription burning presents a catch-22 situation. It can only be done safely under weather conditions that require mature chaparral, 20 years of age or more, but stands of vegetation this age and younger will not form effective barriers to fire spread under severe weather conditions. Modeling studies indicate that to be effective even under moderate weather conditions requires a substantial portion of the landscape be treated (Mark Finney, public communication, 2001). Thus, while landscapes managed by rotational burning may contribute to easier containment of fires burning under moderate weather conditions, they are of limited value during severe weather. However, these latter fires are the ones that become truly catastrophic and are responsible for the greatest losses. Consequently National Forest Service policy of landscape-scale rotational burning to produce a mosaic of age classes needs to be reconsidered (Conard and Weise

1998). This type of fuel management is extremely expensive, unlikely to prevent catastrophic wildfires, and has little resource benefit.

Future fire management policy needs to steer away from extensive landscape-scale prescription burning and focus on intensive and strategic use of fire hazard reduction techniques, both to minimize negative impacts of high fire frequency on natural resources and to maximize fire hazard reduction. The marked differences observed between the central coastal ranges and southern California (Fig. 8.5) suggests that regions may require different fire management strategies. Greater focus needs to be given to transportation corridors as roadways are primary sites of ignitions, and since roadways are required to connect developments, as the urban/wildland interface increases, these fire hazards increase. Roads could also play a role in minimizing the negative impacts of fire hazard reduction programs, since many of the negative impacts of fuel reduction techniques (e.g., aesthetic impacts, promoting invasive plants and animals) are also shared by roadways. Thus greater attention needs to be given to co-locating roads and fuel manipulations such as fuel breaks.

Considering the psychology of many who inhabit the urban–wildland interface, it is questionable whether or not education can play a substantive role in reducing future losses from wildfires (Gardiner, Cortner, and Widaman 1987). Regulations requiring fire "safe" construction have been implicated in reducing property losses in the past and will possibly reduce the degree of future losses (Cohen 2000). It seems inevitable that fire management policy will increasingly require involvement of city and county planners in order to solve the primary fire hazard problem of how to constrain the ever-expanding urban–wildland interface. Fire managers can play a key role in providing accurate analytical models of causal factors driving extreme fire events and educating planners on the limitations to fire hazard reduction (e.g., Sapsis 2001).

Global Change Impacts on Future Fire Regimes

Fire regime is an emergent property of landscapes arising from the interaction of vegetation, weather, topography, and land management (Davis and Michaelsen 1995). Fire regime is influenced directly by vegetation through flammability characteristics and the structural distribution of fuels. Weather affects fire regimes through timing of ignitions, and through frequency and severity of burning conditions as well as direct effects on vegetation distribution. Topography affects rates of natural lightning ignitions and wind patterns that ultimately control fire behavior. Land management affects the distribution of vegetation types and thus landscape patterns of fuels. Land management, in the broad sense, also controls the extent and pattern of the urban–wildland interface, which acts as a porous boundary where fires diffuse across in both directions. Wildland fires may diffuse out from the urban–wildland interface, but the most catastrophic fires result from wildfires burning into the urban environment. Global changes, including direct effects of increased atmospheric CO_2 levels, climate changes, and changing land use, all have the potential for changing fire regimes by altering vegetation, weather, and land management.

Future increases in atmospheric CO_2 levels may directly affect plant growth and potentially alter patterns of fuel distribution. In chaparral the effects are predicted to be variable and strongly dependent on levels of other resources (Oechel et al. 1995). Along gradients of increasing soil fertility we might expect increased biomass production, but the increased leaf area may place greater demands on the limited soil water resources in this semi-arid region, dampening potential increases in primary production. Further exacerbating this dampening effect is the expected increase in summer temperature. However, this could be offset by increased water use efficiency expected with elevated CO_2.

Climate change in California shrubland landscapes over the next half-century is predicted to increase winter and summer temperatures by 3°C and 1°C, respectively, and to increase winter precipitation by 25% (Field et al. 1999). Warmer and much wetter winter conditions will almost certainly contribute to higher primary production, although the magnitude is likely to decline with decreasing soil nutrients. It is assumed that this increased production will lead to higher fuel accumulation and more intense fires. However, these climate changes may also accelerate decomposition of dead fuels, which are critical to fire spread, and the importance of this dampening effect on fuel accumulation has not been evaluated. Expected increases in C:N ratios of dead fuels imply variations in rates of decomposition along soil fertility gradients, paralleling expected increases in primary production. Thus sites with the greatest increases in fuels may also experience the greatest increases in decomposition. Even if the net effect is an increase in rate of fuel accumulation, this should not automatically be assumed to lead to major changes in the fire regime. This is based on the fact that currently rates of fuel accumulation do not play a highly deterministic role in shrubland fire regimes (Moritz 1999; Schoenberg et al. 2001; Peng and Schoenberg 2001).

Expected changes in climate will affect vegetation structure through changes in energy balance as well as nutrient cycling, but this involves such complexity that presently one can only speculate what the future holds (Oechel et al. 1995). Attempts to understand how changes in precipitation and temperature will affect vegetation composition include documentation of contemporary climatic responses (Westman 1991) and growth simulations (Malanson and Westman 1991a, b; Westman and Malanson 1992; Malanson and O'Leary 1995). Realistic parameterization of these models is one limitation to their current usefulness, and thus the primary conclusion one can draw at this point is that changes in the relative abundance of species are to be expected. Another possibility is that changes in fire intensity due to greater fuel loads may affect changes in postfire recovery, although shrublands currently exhibit extraordinary resilience to a wide range of fire intensities (Keeley 1998). Ecotones are expected to be sites of greatest sensitivity to climate change (Peteet 2000), and the complex vegetation mosaic of California landscapes (e.g., Fig. 8.15) presents many opportunities for shifts in vegetation distribution. Considering the large role played by human interference, it seems likely that the greatest alteration in fire regimes will occur at the urban–wildland ecotone.

In general, GCM predictions for twenty-first-century climates in California are of limited value in understanding future fire regimes. Patterns of burning are driven by extreme events (Moritz 1997), and these are not well modeled (Field et al. 1999). One of the primary determinants of area burned is the coincidence of ignition with severe weather, and future changes in patterns of ignition might be expected to play a determining role in fire regimes. Climate-based models predict the California region will have a few percent increase in lightning fires (Price and Rind 1994), but this may not affect these shrubland landscapes where humans are the primary ignition source (Table 8.1, Figs. 8.4 and 8.5).

Future changes in land use are likely to have a more profound impact on shrubland fire regimes than other types of global change. Land use may also be the primary driver behind losses in biodiversity in California as well as in other Mediterranean-climate regions (Sala et al. 2000). Diversity loss is expected to result from increased population growth contributing to habitat loss, habitat fragmentation, and loss of corridors. Some of these factors will affect fire regimes, but, we expect that increased fire ignitions predicted from increased population growth will have a far more profound impact on these landscapes. As the fire return interval shortens, the native shrublands are degraded to mixtures of exotic grasses and forbs, and these invasives contribute to further decreases in fire return interval and loss of native plant diversity (Fig. 14). However, dampening this potential impact of shortened fire return intervals is the stepped-up rate of postfire shrub recovery expected from predicted increases in winter temperature and precipitation. The impact of land-use changes on these landscapes makes it likely that it will far outweigh other global change impacts on fire regimes.

Conclusion

Throughout much of the shrubland landscape humans play a dominant role in promoting fires beyond what was likely the natural fire cycle. Future climate change is expected to have a minor role in altering fire regimes relative to other global changes such as population growth and habitat fragmentation. Future fire management needs to take a strategic approach to fuel manipulations and move beyond evaluating effectiveness strictly in terms of area treated. Fire management should consider designing strategies tailored to different regions as there are marked differences between the central coastal region and southern California in source of ignition (e.g., Table 8.1, Fig. 8.4), season of burning (Fig. 8.6), and historical patterns of population growth (Fig. 8.5a–b) and burning (Figs. 8.5c–d and 8.11). Presently we know relatively little about fire regimes in shrublands in the foothills of the Sierra Nevada and interior foothills of the northern coastal ranges, and thus it would be prudent to not transfer the conclusions drawn here too broadly until we have a clearer understanding of the extent of regional variation in shrubland fire regimes. One of the primary threats that all regions share is the increasing number of people being placed at risk to the natural wildfire threat because of the rapidly expanding urban–wildland interface. Fire management will need to play an increasingly active role in the planning process through critical analysis of causal factors driving fire regimes and the limitations to hazard reduction.

Acknowledgments. We thank Jim Agee, Max Moritz, Carl Skinner, Nate Stephenson, and Paul Zedler for helpful comments on an earlier version of this ms. CJF acknowledges funding from EPA S.T.A.R. Graduate Fellowship #U-915606. We thank Karen Folger and Denise Krieger for assistance with data acquisition.

References

Agee, J.K. 1993. *Fire Ecology of Pacific Northwest Forests.* Covelo, CA: Island Press.

Anderson, M.K., Barbour, M.G., and Whitworth, V. 1998. A world of balance and plenty. In *Contested Eden. California before the Gold Rush,* eds. R.A., Gutierrez and R.J., Orsi, pp. 12–47. Los Angeles: University of California Press.

Anderson, M.K., and Moratto, M.J. 1996. Native American land-use practices and ecological impacts. In *Sierra Nevada Ecosystem Project: Final Report to Congress: Status of the Sierra Nevada,* vol. 2, eds. SNEP Team, pp. 187–206. Davis: Centers for Water and Wildland Resources, University of California.

Barrett, L.A. 1935. *A Record of Forest and Field Fires in California from the Days of the Early Explorers to the Creation of the Forest Reserves.* San Francisco: USDA Forest Service.

Barro, S.C., and Conard, S.G. 1991. Fire effects on California chaparral systems: An overview. *Environ. Int.* 17:135–149.

Bauer, D.R. 1974. A history of forest-fire control in southern California. In *Symposium on Living with the Chaparral, Proceedings,* ed. M. Rosenthal, pp. 121–129. San Francisco: Sierra Club.

Bentley, J.R. 1967. *Conversion of Chaparral to Grassland: Techniques Used in California.* Washington, DC: USDA Forest Service, Agriculture Handbook 328.

Biswell, H.H. 1989. *Prescribed Burning in California Wildlands Vegetation Management.* Los Angeles: University of California Press.

Black, C.H. 1987. Biomass, nitrogen and phosphorus accumulation over a southern California fire cycle chronosequence. *In Plant Response to Stress: Functional Analysis in Mediterranean Ecosystems,* eds. J.D. Tenhunen, F.M. Catarino, O.L. Lange, and W.C. Oechel, pp. 445–458. Berlin: Springer.

Bonnicksen, T.M., and Lee, R.G. 1979. Persistence of a fire exclusion policy in southern California: A biosocial interpretation. *J. Environ. Manag.* 8:277–293.

Booker, F.A., Dietrich, W.M., and Collins, L.M. 1995. The Oakland Hills fire of October 20, 1991, an evaluation of post-fire response. In *Brushfires in California Wildlands: Ecology and Resource Management,* eds. J.E. Keeley, and T. Scott, pp. 163–170. Fairfield, WA: International Association of Wildland Fire.

Borchert, M.I., and Odion, D.C. 1995. Fire intensity and vegetation recovery in chaparral: A review. In *Brushfires in California Wildlands: Ecology and Resource Management,* eds. J.E. Keeley, and T. Scott, pp. 91–100. Fairfield, WA: International Association of Wildland Fire.

Bradshaw, T.D. 1987. The intrusion of human population into forest and range lands of California. In *Proceedings of the Symposium on Wildland Fire 2000,* April 27–30, South Lake Tahoe, CA, eds. J.B. Davis, and R.E. Martin, pp. 15–21. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Gen. Tech. Rep. PSW-101.

Brown, W.S. 1945. History of Los Padres National Forest. Goleta, CA. USDA Forest Service, Unpublished rep. on file.

Brown, W.S., and Show, S.B. 1944. *California Rural Land Use and Management: A History of the Use and Occupancy of Rural Lands in California*. Berkeley: USDA Forest Service, California Region.

Burcham, L.T. 1957. *California Range Land: an Historic-Ecological Study of the Range Resources of California*. Sacramento: State of California, Department of Natural Resources, Division of Forestry.

Byrne, R., Michaelsen, J., and Soutar, S. 1977. Fossil charcoal as a measure of wildfire frequency in southern California: A preliminary analysis. In *Proceedings of the Symposium on Environmental Consequences of Fire and Fuel Management in Mediterranean Ecosystems*, eds. H. A. Mooney, and C. E. Conrad, pp. 361–367. Washington, DC: USDA Forest Service, Gen. Tech. Rep. WO-3.

Callaham, R.Z. 1985. *California's Shrublands: A Vast Area in Transition and Need.* Berkeley: University of California, Wildland Resources Center.

Callaway, R.M., and Davis, F.W. 1993. Vegetation dynamics, fire, and the physical environment in coastal central California. *Ecol.* 74:1567–1578.

Campbell, A. 1906. Sonora storms and Sonora clouds of California. *Mon. Wea. Re.* 34: 464–465.

CDF. 1978. *Brushland Range Improvement*. Annual report 1974–1977 inclusive. Sacramento: California Department of Forestry.

Chandler, C.C. 1957. *"Light burning" in Southern California fuels*. Berkeley: USDA Forest Service, California Forest and Range Experiment Station, Forest Res. Notes 119.

Chandler, C.C. 1960. How good are statistics on fire causes? *J. For.* 58:515–517.

Chandler, C.C. 1963. *A Study of Mass Fires and Conflagrations*. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Res. Note PSW-22.

Clar, C.R. 1959. *California Government and Forestry from Spanish Days until the Creation of the Department of Natural Resources in 1927.* Sacramento: State of California, Department of Natural Resources, Division of Forestry.

Clar, C.R. 1969. *California Government and Forestry—II. During the Young and Rolph Administrations*. Sacramento: State of California, Department of Natural Resources, Division of Forestry.

Coffin, H. 1959. Effect of marine air on the fireclimate in the mountains of southern California. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Tech. Pap. 39.

Cohen, J.D. 2000. Preventing disaster: Home ignitability in the wildland–urban interface. *J. For.* 98:15–21.

Conard, S.G., and Regelbrugge, J.C. 1994. On estimating fuel characteristics in California chaparral. In *12th Conference on Fire and Forest Meteorology*, pp. 120–129. Boston: Society of American Foresters.

Conard, S.G., and Weise, D.R. 1998. Management of fire regime, fuels, and fire effects in southern California chaparral: Lessons from the past and thoughts for the future. *Tall Timbers Ecol. Conf. Proc.* 20:342–350.

Cooper, W.S. 1922. *The Broad-Sclerophyll Vegetation of California: An Ecological Study of the Chaparral and Its Related Communities.* Washington, DC: Carnegie Institution of Washington, Pub. 319.

Countryman, C.M. 1974. Can southern California wildland conflagrations be stopped? Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Gen. Tech. Note PSW-7.

Countryman, C.M., McCutchan, M.H., and Ryan, B.C. 1969. Fire weather and fire behavior at the 1968 Canyon Fire. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Res. Pap. PSW-55.

Countryman, C.M., and Philpot, C.W. 1970. Physical characteristics of chamise as wildland fuel. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Res. Pap. PSW-66.

D'Antonio, C.M. 2000. Fire, plant invasions, and global changes. In *Invasive Species in a Changing World*, eds. H.A. Mooney, and R.J. Hobbs, pp. 65–93. Covelo, CA: Island Press.

Davis, F.W., and Burrows, D.A. 1993. Modeling fire regime in Mediterranean landscapes. In *Patch Dynamics*, eds. S.A. Levin, T.M. Powell, and J.H. Steele, pp. 247–259. New York: Springer-Verlag.

Davis, F.W., and Burrows, D.A. 1994. Spatial simulation of fire regime in Mediterranean-climate landscapes. In *The Role of Fire in Mediterranean-Type Ecosystems*, eds. J.M. Moreno, and W.C. Oechel, pp. 117–139. New York: Springer-Verlag.

Davis, F.W., and Michaelsen, J. 1995. Sensitivity of fire regime in chaparral ecosystems to climate change. In *Global Change and Mediterranean-Type Ecosystems*, eds. J.M. Moreno, and W.C. Oechel, pp. 435–456. New York: Springer-Verlag.

Davis, J.A. 1967. *The Friend to All Motorists: The Story of the Automobile Club of Southern California through 65 Years, 1900–1965*. Los Angeles: Automobile Club of Southern California.

Davis, L.S. 1965. *The Economics of Wildfire Protection with Emphasis on Fuel Break Systems*. Sacramento: State of California, Resources Agency, Division of Forestry.

Davis, S.D., Ewers, F.W., Sperry, J.S., Portwood, K.A., Crocker, M.C., and Adams, G.C. 2002. Shoot dieback during prolonged drought in *Ceanothus* (Rhamnaceae) chaparra: a possible case of hydraulic failure. *Amer. J. Bot.* 89:820–828.

Davis, S.D., Kolb, K.J., and Barton, K.P. 1998. Ecophysiological processes and demographic patterns in the structuring of California chaparral. In *Landscape Diversity and Biodiversity in Mediterranean-Type Ecosystems*, eds. P.W. Rundel, G. Montenegro, and F.M. Jaksic, pp. 297–310. New York: Springer-Verlag.

Dombeck, M. 2001. How can we reduce the fire danger in the interior West? *Fire Management Today* 61(1):5–13.

Dunn, A.T. 1989. The effects of prescribed burning on fire hazard in the chaparral: Toward a new conceptual synthesis. In *Proceedings of the Symposium on Fire and Watershed Management*, ed. N.H. Berg, pp. 23–29. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Gen. Tech. Rep. PSW-109.

Dunn, A.T., and Piirto, D. 1987. The Wheeler fire in retrospect: factors affecting fire spread and perimeter formation. Riverside: USDA Forest Service, Pacific Southwest Research Station, unpublished report on file.

Ewing, R.A., Tosta, N., Tuaszon, R., Huntsinger, L., Marose, R., Nielson, K., Motroni, R., and Turan, S. 1988. *California's Forests and Rangelands: Growing Conflict Over Changing Uses*. Sacramento: State of California, Department of Forestry and Fire Protection.

Fenner, R.L., Arnold, R.K., and Buck, C.C. 1955. Area ignition for brush burning. Berkeley: USDA Forest Service, California Forest and Range Experiment Station, Tech. Pap. 10.

Field, C.B., Daily, G.C., Davis, F.W., Gaines, S., Matson, P.A., Melack, J., and Miller, N.L. 1999. *Confronting Climate Change in California. Ecological Impacts on the Golden State*. Cambridge, MA, and Washington, DC: Union of Concerned Scientists and Ecological Society of America.

Florence, M.A. 1985. Successional trends in plant species composition following fall, winter and spring prescribed burns of chamise chaparral in the central coast range of California. M.S. thesis: California State University, Sacramento.

Fosberg, M.A. 1965. A case study of the Santa Ana winds in the San Gabriel Mountains. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Res. Note PSW-78.

Fosberg, M.A., O'Dell, C.A., and Schroeder, M.J. 1966. Some characteristics of the three-dimensional structure of Santa Ana winds. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Res. Pap. PSW-30.

Franco-Vizcaíno, E., and. Sosa-Ramirez, J., 1997. Soil properties and nutrient relations in burned and unburned mediterranean-climate shrublands of Baja California, Mexico. *Acta Oecol.* 18:503–517.

Franklin, S.E. 1987. Urban-wildland fire defense strategy, precision prescribed fire: The Los Angeles County approach. In *Proceedings of the Symposium on Wildland Fire 2000*, April 27–30, 1987, South Lake Tahoe, CA, eds. J.B. Davis, and R.E. Martin, pp. 22–25. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Gen. Tech. Rep. PSW-101.

FRAP. 1999. Fire management for California ecosystems. Sacramento: State of California, Resources Agency, California Department of Forestry, Fire and Resource Assessment Program, *http://frap.cdf.ca.gov/projects/fire_mgmt/ftp_main.html.*

Freudenberger, D.O., Fish, B.E., and Keeley, J.E. 1987. Distribution and stability of grasslands in the Los Angeles Basin. *Bull. Southern California Acad. Sci.* 86:13–26.

Gardner, P.D., Cortner, H.J., and Widaman, K. 1987. The risk perceptions and policy response toward wildland fire hazards by urban home-owners. *Landscape Urban Plan.* 14:163–172.

Gause, G.W. 1966. Silvical characteristics of bigcone Douglas-fir. Berkeley: USDA Forest Service, PSW-39.

Gee, P.J. 1974. Roadside fire hazard in California. M.S., thesis. University of California, Berkeley.

Gomes, D., Graham, O.L., Jr., Marshall, E.H., and Schmidt, A.J. 1993. Sifting through the ashes: Lessons learned from the Painted Cave Fire. Graduate Program for Public Historical Studies, University of California, Santa Barbara.

Green, L.R. 1970. An experimental prescribed burn to reduce fuel hazard in chaparral. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Res. Note PSW-216.

Green, L.R. 1981. Burning by prescription in chaparral. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Gen. Tech. Rep. PSW-51.

Greenlee, J.M., and Langenheim, J.H. 1980. The history of wildfires in the region of Monterey Bay. Sacramento: California Department of Parks and Recreation, unpublished rep.

Greenlee, J.M., and Langenheim, J.H. 1990. Historic fire regimes and their relation to vegetation patterns in the Monterey Bay area of California. *Am. Midland Natural.* 124:239–253.

Greenlee, J.M., and Moldenke, A. 1982. History of wildland fires in the Gabilan Mountains region of central coastal California. San Francisco: USDI National Park Service, Unpublished rep.

Haidinger, T.L., and Keeley, J.E. 1993. Role of high fire frequency in destruction of mixed chaparral. *Madroño* 40:141–147.

Hamilton, J.G. 1997. Changing perceptions of pre-European grasslands in California. *Madroño* 44:311–333.

Heady, H.F. 1977. Valley grasslands. In *Terrestrial Vegetation of North America*, eds. M.G. Barbour, and J. Major, pp. 491–514. New York: Wiley.

Hobbs, E.R. 1983. Factors controlling the form and location of the boundary between coastal sage scrub and grassland in southern California. Ph.D. dissertation. University of California, Los Angeles.

Huenneke, L.F. 1989. Distribution and regional patterns of Californian grasslands. In *Grassland Structure and Function: California Annual Grassland*, eds. L.F. Huenneke, and H.A. Mooney, pp. 1–12. Dordrecht: Kluwer Academic.

Johnson, E.A., and K., Miyanishi. 1995. The need for consideration of fire behavior and effects in prescribed burning. *Restor. Ecol.* 3:271–278.

Keeley, J.E. 1977. Fire dependent reproductive strategies in *Arctostaphylos* and *Ceanothus*. In *Proceedings of The Symposium on Environmental Consequences of Fire and Fuel Management in Mediterranean Ecosystems*, eds. H.A. Mooney, and C.E. Conrad, pp. 371–376. Washington, DC: USDA Forest Service, Gen. Tech. Rep. WO-3.

Keeley, J.E. 1982. Distribution of lightning and man-caused wildfires in California. In *Proceedings of the Symposium on Dynamics and Management of Mediterranean-Type Ecosystems*, eds. C.E. Conrad, and W.C. Oechel, pp. 431–437. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Gen. Tech. Rep. PSW-58.

Keeley, J.E. 1990. The California valley grassland. In *Endangered Plant Communities of Southern California*, ed. A.A. Schoenherr, pp. 2–23. Fullerton: Southern California Botanists, Special Publication 3.

Keeley, J.E. 1992. Demographic structure of California chaparral in the long-term absence of fire. *J. Veg. Sci.* 3:79–90.

Keeley, J.E. 1998a. Coupling demography, physiology and evolution in chaparral shrubs. In *Landscape Diversity and Biodiversity in Mediterranean-Type Ecosystems*, eds. P.W. Rundel, G. Montenegro, and F.M. Jaksic, pp. 257–264. New York: Springer-Verlag.

Keeley, J.E. 1998b. Postfire ecosystem recovery and management: the October 1993 large fire episode in California. In *Large Forest Fires*, ed. J.M. Moreno, pp. 69–90. Leiden, The Netherlands: Backhuys.

Keeley, J.E. 2000. Chaparral. In *North American Terrestrial Vegetation*, eds. M.G. Barbour, and W.D. Billings, pp. 201–251. Cambridge: Cambridge University Press.

Keeley, J.E. (in press). Fire and invasives in Mediterranean-climate ecosystems of California. Tall Timbers Research Station Miscellaneous Publication 11:81–94.

Keeley, J.E. 2002. Native American impacts on fire regimes of the California coastal ranges. *J. Biogeogr.* 29:303–320.

Keeley, J.E., and Fotheringham, C.J. 2001a. Historic fire regime in California shrublands. *Conserv. Biol.* 15:1534–1548.

Keeley, J.E., and Fotheringham, C.J. 2001b. History and management of crown-fire ecosystems: A summary and response. *Conserv. Biol.* 15:1561–1567

Keeley, J.E., Fotheringham, C.J., and Morais, M. 1999. Reexamining fire suppression impacts on brushland fire regimes. *Science* 284:1829–1832.

Keeley, J.E., and Keeley, S.C. 1984. Postfire recovery of California coastal sage scrub. *Am. Midland Natural.* 111:105–117.

Keeley, J.E., Zedler, P.H., Zammit, C.A., and Stohlgren, T.J. 1989. Fire and demography. In *The California Chaparral: Paradigms Reexamined*, ed. S.C. Keeley, pp. 151–153. Los Angeles: Natural History Museum of Los Angeles County, Science Series 34.

Kessell, S.R., and Cattelino, P.J. 1978. Evaluation of a fire behaviour information integration system for southern California chaparral wildlands. *Environ. Manag.* 2:135–159.

Kinney, A., 1887. Report on the forests of the counties of Los Angeles, San Bernardino, and San Diego, California. Sacramento: First Biennial Report, California State Board of Forestry.

Kinney, A. 1900. *Forest and Water*. Los Angeles: Post.

Kinney, W. 1984. Economics and policy of shrubland management. In *Proceedings of the Chaparral Ecosystems Research Conference*, ed. J.J. DeVries, pp. 129–136. Davis: University of California, Water Resources Center, Rep. 62.

Knipper, C. 1998. Fire: The rejuvenating force. *Explorer* 5(8):8.

Krausman, W.J. 1981. An analysis of several variables affecting fire occurrence and size in San Diego County, California. M.A., thesis. San Diego State University.

Krick, I.P. 1933. Foehn winds of southern California. *Beitr. Geophys.* 39:399–407.

Lee, R.G., and Bonnicksen, T.M. 1978. Brushland watershed fire management policy in southern California: biosocial considerations. Davis. University of California, California Water Resources Center, Contribution 172.

Lessard, A.G. 1988. The Santa Ana wind of southern California. *Weatherwise* 41:100–104.

Lewis, H.T. 1973. *Patterns of Indian Burning in California: Ecology and Ethnohistory.* Ramona, CA: Ballena Press.

Lillard, R.G. 1961. Black horizons. *Westways* 62(10):17–19, 64–65.

Lockmann, R.F. 1981. *Guarding the Forest of Southern California.* Glendale, CA: Clark.

Malanson, G.P. 1985. Fire management in coastal sage-scrub, southern California, USA. *Biolog. Conserv.* 12:141–146.

Malanson, G.P., and O'Leary, J.F. 1995. The coastal sage scrub—Chaparral boundary and response to global climatic change. In *Global Climate Change in Mediterranean-Type Ecosystems*, eds. J.M. Moreno, and W.C. Oechel, pp. 203–224. Berlin: Springer-Verlag.

Malanson, G.P., and Westman, W.E. 1991a. Climatic change and the modeling of fire effects in coastal sage scrub and chaparral. In *Fire and the Environment: Ecological and Cultural Perspectives, Proceedings of an International Symposium*, eds. S.C. Nodvin, and T.A. Waldrop, pp. 91–96. USDA Forest Service Station, Southeastern Forest and Experiment Station, Gen. Tech. Rep. SE-69.

Malanson, G.P., and Westman, W.E. 1991b. Modeling interactive effects of climate change, air pollution, and fire on a California shrubland. *Clim. Change* 18:363–376.

Martin, R.E., and Sapsis, D.B. 1995. A synopsis of large or disastrous wildland fires. In *The Biswell Symposium: Fire Issues and Solutions in Urban Interface and Wildland Ecosystems*, eds. D.R. Weise, and R.E. Martin, pp. 35–38. Berkeley: USDA Forest Service, Gen. Tech. Rep. PSW-GTR-158.

McCutchan, M.H. 1977. Climatic features as a fire determinant. In *Proceedings of the Symposium on Environmental Consequences of Fire and Fuel Management in Mediterranean Ecosystems*, eds. H.A. Mooney, and C.E. Conrad, pp. 1–11. Washington, DC: USDA Forest Service, Gen. Tech. Rep. WO-3.

Mensing, S.A., Michaelsen, J., and Byrne, R. 1999. A 560-year record of Santa Ana fires reconstructed from charcoal deposited in the Santa Barbara Basin, California. *Quat. Res.* 51:295–305.

Minnich, R.A. 1983. Fire mosaics in southern California and northern Baja California. *Science* 219:1287–1294.

Minnich, R.A. 1987. Fire behavior in southern California chaparral before fire control: the Mount Wilson burns at the turn of the century. *Ann. Assoc. Am. Geogr.* 77:599–618.

Minnich, R.A. 1989. Chaparral fire history in San Diego County and adjacent northern Baja California: An evaluation of natural fire regimes and the effects of suppression management. In *The California Chaparral: Paradigms Reexamined*, ed. S.C. Keeley, pp. 37–47. Los Angeles: Natural History Museum of Los Angeles County, Science Series 34.

Minnich, R.A. 1990. Fire suppression in chaparral: what the United States can learn from Mexico. In *Environmental Hazards and Bioresource Management in the United States-Mexico Borderlands*, eds. P. Ganster, and H. Walter, pp. 329–342. Los Angeles: UCLA Latin American Center Publications, University of California.

Minnich, R.A. 1995. Fuel-driven fire regimes of the California chaparral. In *Brushfires in California: Ecology and Resource Management.* eds. J.E. Keeley, and T. Scott, pp. 21–27. Fairfield, WA: International Association of Wildland Fire.

Minnich, R.A. 1998. Landscapes, land-use and fire policy: where do large fires come from? In *Large Forest Fires*, ed. J.M. Moreno, pp. 133–158. Leiden, The Netherlands: Backhuys.

Minnich, R.A. 2001. An integrated model of two fire regimes. *Conservation Biology* 15:1549–1553.

Minnich, R.A., and Chou, Y. H. 1997. Wildland fire patch dynamics in the chaparral of southern California and northern Baja California. *Int. J. Wild. Fire* 7:221–248.

Minnich, R.A., and Dezzani, R.J. 1991. Suppression, fire behavior, and fire magnitudes in Californian chaparral at the urban/wildland interface. In *California Watersheds at the Urban Interface, Proceedings of the Third Biennial Watershed Conference*, ed. J. J. DeVries, pp. 67–83. Davis: University of California, Water Resources Center, Report 75.

Minnich, R.A., and Dezzani, R.J. 1998. Historical decline of coastal sage scrub in the Riverside-Perris Plain, California. *Western Birds* 29:366–391.

Minnich, R.A., and Franco-Vizcaíno, E. 1999. Prescribed mosaic burning in California chaparral. In *Proceedings of the Symposium on Fire Economics, Planning, and Policy: Bottom Lines*, eds. A. González-Cabán, and P.N. Omi, pp. 243–246. Berkeley: USDA Forest Service, Pacific Southwest Research Station, Gen. Tech. Rep. PSW-GTR-173.

Minnich, R.A., Franco-Vizcaíno, E., Sosa-Ramirez, J., and Chou, Y., 1993. Lightning detection rates and wildland fire in the mountains of northern Baja California, Mexico. *Atmósfera* 6:235–253.

Mitchell, V.L. 1969. The regionalization of climate in montane areas. Ph.D. dissertation. University of Wisconsin, Madison.

Moreno, J.M., and Oechel, W.C. 1991. Fire intensity effects on germination of shrubs and herbs in southern California chaparral. *Ecology* 72:1993–2004.

Moritz, M.A. 1997. Analyzing extreme disturbance events: fire in the Los Padres National Forest. *Ecol. Appl.* 7:1252–1262.

Moritz, M.A. 1999. Controls on disturbance regime dynamics: fire in Los Padres National Forest. Ph.D. dissertation. University of California, Santa Barbara.

Mutch, R.W. 1970. Wildland fires and ecosystems: a hypothesis. *Ecology* 51:1046–1051.

Nichols, R., Adams, T., and Menke, J. 1984. Shrubland management for livestock forage. In *Shrublands in California: Literature Review and Research Needed for Management*, ed. J.J. DeVries, pp. 104–121. Davis: University of California, Water Resources Center, Contribution 191.

Oberbauer, A.T. 1978. Distribution dynamics of San Diego County grasslands. M.S. thesis. San Diego State University.

Odens, P. 1971. *The Indians and I. Visits with Dieguenos, Quechans, Fort Mojaves, Zumis, Hopis, Navajos and Piutes*. El Centro, CA: Imperial Printers.

Oechel, W.C., Hastings, S.J., Vourlitis, G.L., Jenkins, M.A., and Hinkson, C.L. 1995. Direct effects of elevated CO_2 in chaparral and Mediterranean-type ecosystems. In *Global Change and Mediterranean-Type Ecosystems*, eds. J.M. Moreno, and W.C. Oechel, pp. 58–75. New York: Springer-Verlag.

Olsen, J.M. 1960. 1959 green-fuel moisture and soil moisture trends in southern California. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Res. Note 161.

Parker, V.T. 1990. Problems encountered while mimicking nature in vegetation management: An example from a fire-prone vegetation. In *Ecosystem Management: Rare Species and Significant Habitats. Proceedings of the 15th Annual Natural Areas Conference*, eds. R.S. Mitchell, C.J. Sheviak, and D.J. Leopold, pp. 231–234. Albany New York State Museum, Bulletin 471.

Parsons, D.J. 1981. The historical role of fire in the foothill communities of Sequoia National Park. *Madroño* 28:111–120.

Payson, T.E., and Cohen, J.D. 1990. Chamise chaparral dead fuel fraction is not reliably predicted by age. *Western J. For.* 5:127–131.

Paysen, T.E., Narog, M.G., and Cohen, J.D. 1998. The science of prescribed fire: to enable a different kind of control. *Tall Timbers Ecol. Conf. Proc.* 20:31–36.

Peng, R., and Schoenberg, F. 2001. Estimation of wildfire hazard using spatial-temporal fire history data. *J. Am. Stat. Assoc.*, in press.

Peteet, D. 2000. Sensitivity and rapidity of vegetational response to abrupt climate change. *Proc. Nat. Acad. Sci.* 97:1359–1361.

Phillips, C.B. 1971. *California Aflame! September 22–October 4, 1970.* Sacramento: State of California, Department of Conservation, Division of Forestry.

Philpot, C.W. 1969. Seasonal changes in heat content and ether extractive content of chamise. Berkeley: USDA Forest Service, Intermountain Forest and Range Experiment Station, Res. Pap. INT-61.

Philpot, C.W. 1974a. The changing role of fire on chaparral lands. In *Symposium on Living with the chaparral, Proceedings*, ed. M. Rosenthal, pp. 131–150. San Francisco: Sierra Club.

Philpot, C.W. 1974b. New fire control strategy developed for chaparral. *Fire Manag.* 37: 3–7.

Philpot, C.W. 1977. Vegetative features as determinants of fire frequency and intensity. In *Proceedings of the Symposium on Environmental Consequences of Fire and Fuel Management in Mediterranean Ecosystems*, eds. H.A. Mooney, and C.E. Conrad, pp. 12–16. Washington, DC: USDA Forest Service, Gen. Tech. Rep. WO-3.

Pirsko, A.R. 1960. 1960 fire weather severity in California. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Miscellaneous Pap. 54.

Pirsko, A.R., and Green, L.R. 1967. Record low fuel moisture follows drought in southern California. *J. For.* 65:642–643.

Price, C., and Rind, D. 1994. Lightning fires in a 2 × CO_2 world. In *12th Conference on Fire and Forest Meteorology*, October 26–28, Jekyll Island, GA, pp. 77–84. Washington, DC: Society of American Foresters.

Pyne, S.J. 1982. *Fire In America: A Cultural History of Wildland and Rural Fire.* Princeton, NY: Princeton University Press.

Pyne, S.J., Andrews, P.L., and Laven, R.D. 1996. *Introduction to Wildland Fire.* New York: Wiley.

Radtke, K.W.H., Arndt, A.M., and Wakimoto, R.H. 1982. Fire history of the Santa Monica Mountains. In *Proceedings of the Symposium on Dynamics and Management of Mediterranean-Type Ecosystems*, eds. C.E. Conrad, and W.C. Oechel, pp. 438–443. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Gen. Tech. Rep. PSW-58.

Regelbrugge, J.C. 2000. Role of prescribed burning in the management of chaparral ecosystems in southern California. In *2nd Interface between Ecology and Land Development in California*, eds. J.E. Keeley, M.B. Keeley, and C.J. Fotheringham, pp. 19–26. Sacramento: U.S. Geological Survey Open-File Rep. 00–62.

Reynolds, R.D. 1959. Effect of natural fires and aboriginal burning upon the forest of the central Sierra Nevada. M.A., thesis. University of California, Berkeley

Riggan, P.J., Franklin, S.E., Brass, J.A., and Brooks, F.E. 1994. Perspectives on fire management in Mediterranean ecosystems of southern California. In *The Role of Fire in Mediterranean-Type Ecosystems*, eds. J.M. Moreno, and W.C. Oechel, pp. 140–162. New York: Springer-Verlag.

Riggan, P.J., Goode, S., Jacks, P.M., and Lockwood, R.W. 1988. Interaction of fire and community development in chaparral of southern California. *Ecol. Monogr.* 58: 155–175.

Rogers, M.J. 1982. Fire management in southern California. In *Proceedings of the Symposium on Dynamics and Management of Mediterranean-Type Ecosystems*, eds. C.E. Conrad and W.C. Oechel, pp. 496–497. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Gen. Tech. Rep. PSW-58.

Rothermel, R.C. 1972. *A Mathematical Model for Predicting Fire Spread in Wildland Fuels.* Ogden, UT: USDA Forest Service, INT-115.

Rothermel, R.C., and Philpot, C.W. 1973. Predicting changes in chaparral flammability. *J. For.* 71:640–643.

Rundel, P.W., Baker, G.A., Parsons, D.J., and Stohlgren, T.J. 1987. Postfire demography of resprouting and seedling establishment by *Adenostoma fasciculatum* in the California chaparral. In *Plant Response to Stress: Functional Analysis in Mediterranean Ecosystems*, eds. J.D. Tenhunen, F.M. Catarino, O.L. Lange, and W.C. Oechel, pp. 575–596. Berlin: Springer-Verlag.

Rundel, P.W., Parsons, D.J., and Baker, G.A. 1980. The role of shrub structure and chemistry in the flammability of chaparral shrubs. In Fire Ecology: *Proceedings of the Second Conference on Scientific Research in National Parks*, vol. 10, pp. 248–260. Washington, DC: USDI National Park Service.

Russell, E.W.B. 1983. Pollen analysis of past vegetation at Point Reyes National Seashore, California. *Madroño* 30:1–11.

Ryan, B.C. 1969. A vertical perspective of Santa Ana winds in a canyon. Berkeley: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Res. Pap. PSW-52.

Ryan, G. 1996. Downslope winds of Santa Barbara, California. Washington, DC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Weather Service, NOAA Tech. Memo. NWS WR-240.

Sala, O.E., et al. 2000. Global biodiversity scenarios for the year 2100. *Science* 287:1770–1774.

Sampson, A.W. 1944. Plant succession and burned chaparral lands in northern California. Berkeley: University of California, Agricultural Experiment Station, Bull. 685.

Sapsis, D. 2001. Development patterns and fire suppression. Sacramento: State of California, Resources Agency, California Department of Forestry, Fire and Resource Assessment Program, *http://frap.cdf.ca.gov/publications/development_patterns/toc.html.*

Schoenberg, F., Peng, R., Huang, Z., and Rundel, P. 2001. Exploratory analysis of wildfire data in Los Angeles County, California. *http://www.stat.ucla.edu/~frederic/papers/fire1.pdf.*

Schroeder, M.J., et al. 1964. Synoptic weather types associated with critical fire weather. Washington, DC: U.S. Department of Commerce, National Bureau of Standards, Institute for Applied Technology, AD 449–630.

Schroeder, M.J., and Buck, C.C. 1970. *Fire Weather . . . A Guide for Application of Meteorological Information to Forest Fire Control Operations.* Washington, DC: USDA Forest Service, Agricultural Handbook 360.

Schwilk, D.W. 2000. Flammability as niche construction: Canopy architecture's effect on the flammability of a chaparral species. In *Mediterranean-Type Ecosystems: Past, Present and Future*, pp. 68–69. Stellenbosch, South Africa: MEDECOS 2000, Stellenbosh University.

Show, S.B., and Kotok, E.I. 1923. *Forest Fires in California 1911–1920: An Analytical Study.* Washington, D.C.: U.S. Department of Agriculture, Circular 243.

Skinner, C.N., and Chang, C.-R. 1996. Fire regimes, past and present. In *Sierra Nevada Ecosystem Project: Final Report to Congress. Status of the Sierra Nevada*, eds. SNEP Team, pp. 1041–1069. Davis: Centers for Water and Wildland Resources, University of California.

Sommers, W.T. 1978. LFM forecast variables related to Santa Ana wind occurrences. Mon. *Wea. Rev.* 106:1307–1316.

Specht, R.L. 1969. A comparison of the sclerophyllous vegetation characteristics of Mediterranean type climate in France, California and Southern Australia. I. Structure, morphology, and succession. *Austral. J. Bot.* 17:277–292.

Specht, R.L. 1981. Primary production in Mediterranean-climate ecosystems regenerating after fire. In *Ecosystems of the World: Mediterranean-Type Shrublands*, vol. 2, eds. F. di Castri, D.W. Goodall, and R.L. Specht, pp. 257–268. New York: Elsevier Scientific.

Timbrook, J., Johnson, J.R., and Earle, D.D. 1982. Vegetation burning by the Chumash. *J. Cal. Great Basin Anthropol.* 4:163–186.

Turner, K.M., and Lampinen, B.D. 1983. Prescribed burning of chaparral: some effects on soil movement. Sacramento: State of California, Resources Agency, Department of Water Resources.

Vale, T.T. 1998. The myth of the humanized landscape: an example from Yosemite National Park. *Natural Areas J.* 18:231–236.

van Wagtendonk, J.W. 1992. Spatial analysis of lightning strikes in Yosemite National Park. In *Proceedings of the 11th Conference on Fire and Forest Meteorology*, eds. P.L. Andrews, and D.F. Potts, pp, 605–611. Bethesda, MD: Society of American Foresters.

Vankat, J.L. 1985. General patterns of lightning ignitions in Sequoia National Park, California. *Proceedings—Symposium and Workshop on Wilderness Fire*, eds. J.E. Lotan, B.M. Kilgore, W.C. Fischer, and R.W. Mutch, pp. 408–411. Fort Collins, CO: USDA Forest Service, Intermountain Forest and Range Experiment Station, Gen. Tech. Rep. INT-182.

Weatherspoon, C.P., and C.N., Skinner. 1996. Landscape-level strategies for forest fuel management. In *Sierra Nevada Ecosystem Project: Final report to Congress. Status of the Sierra Nevada*, eds. SNEP Team, pp. 1471–1492. Davis: Centers for Water and Wildland Resources, University of California.

Weide, D.L. 1968. The geography of fire in the Santa Monica Mountains. M.S. thesis. California State University, Los Angeles.

Weise, D.R., Regelbrugge, J.C., Paysen, T.E., and Conard, S.G., (in press). Fire occurrence on southern Californian national forests—Has it changed recently? In *Proceedings of Fire in California Ecosystems: Integrating Ecology, Prevention, and Management*, eds. N.G. Sugihara, and M.I. Borchert. Davis: University of California.

Wells, M.L., and McKinsey, D.E. 1994. The spatial and temporal distribution of lightning strikes in San Diego County, California. *GIS/LIS Proc.* 2:768–777.

Wells, M.L., and McKinsey, D.E. 1995. Lightning strikes and natural fire regimes in San Diego County, California. In *Biswell Symposium: Fire Issues and Solutions in Urban Interface and Wildland Ecosystems*, eds. D.R. Weise, and R.E. Martin, pp. 193–194. Berkeley: USDA Forest Service, Gen. Tech. Rep. PSW-GTR-158.

Wells, P.V. 1962. Vegetation in relation to geological substratum and fire in the San Luis Obispo quadrangle, California. *Ecol. Monogr.* 32:79–103.

Westman, W.E. 1991. Measuring realized niche spaces: Climatic response of chaparral and coastal sage scrub. *Ecology* 72:1678–1684.

Westman, W.E., and Malanson, G.P. 1992. Effects of climate change on Mediterranean-type ecosystems in California and Baja California. In *Global Warming and Biological Diversity*, eds. R.L. Peters, and T.E. Lovejoy, pp. 258–276. New Haven: Yale University Press.

Wickstrom, C.K.R. 1987. Issues concerning Native American use of fire: a literature review. Yosemite National Park, CA: Yosemite Research Center, Publ. Anthropol. 6.

Wohlgemuth, P.M., Beyers, J.L., and Conard, S.G. 1999. Postfire hillslope erosion in southern California chaparral: A case study of prescribed fire as a sediment management tool. In *Proceedings of the Symposium on Fire Economics, Planning, and Policy: Bottom Lines*, eds. A. Gonzâlez-Cabân, and P.N. Omi, pp. 269–276. Berkeley: USDA Forest Service, Pacific Southwest Research Station, Gen. Tech. Rep. PSW-GTR-173.

Wolfram, H. 1962. Brush can be burned in the early spring. Sacramento: State of California, Department of Natural Resources, California Division of Forestry, Range Improvement Studies 6.

Zahn, C. 1944. The San Diego fires . . . an inquest. *Am. For.* 50:161–164.

Zedler, P.H. 1995. Fire frequency in southern California shrublands: Biological effects and mana-gement options. In *Brushfires in California: Ecology and Resource Management.* eds. J.E. Keeley, and T. Scott, pp. 101–112. Fairfield, WA: International Association of Wildland Fire.

Zedler, P.H., Gautier, C.R., and McMaster, G.S. 1983. Vegetation change in response to extreme events: The effect of a short interval between fires in California chaparral and coastal scrub. *Ecology* 64:809–818.

Zedler, P.H., and Seiger, L.A. 2000. Age mosaics and fire size in chaparral: A simulation study. In *2nd Interface between Ecology and Land Development in California*, eds. J.E. Keeley, M.B. Keeley, and C.J. Fotheringham, pp. 9–18. Sacramento: U.S. Geological Survey Open-File Rep. 00–62.

Zivnuska, J.A., Arnold, K., and Arment, C. 1950. Wildfire damage and cost far-reaching. *Cal. Agric.* 4(9):8–10.