Reexamining Fire Suppression Impacts on Brushland Fire Regimes

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California shrubland wildfires are increasingly destructive, and it is widely held that the problem has been intensified by fire suppression, leading to larger, more intense wildfires. However, analysis of the California Statewide Fire History Database shows that, since 1910, fire frequency and area burned have not declined, and fire size has not increased. Fire rotation intervals have declined, and fire season has not changed, implying that fire intensity has not increased. Fire frequency and population density were correlated, and it is suggested that fire suppression plays a critical role in offsetting potential impacts of increased ignitions. Large fires were not dependent on old age classes of fuels, and it is thus unlikely that age class manipulation of fuels can prevent large fires. Expansion of the urban-wildland interface is a key factor in wildland fire destruction.

California shrublands frequently fuel massive high-intensity wildfires that are of increasing concern to resource managers and the public. Despite increased expenditures on fire suppression, each new decade experiences increased loss of property and lives from brushland wildfires (*1*). By the middle of this century, it was suggested that the problem stemmed in large part from the burgeoning population and poor zoning regulations attendant with urban sprawl into the foothills (*2*).

Accepting expanded urbanization as the source of the wildfire problem has profound economic and political implications. An alternative view to emerge in the early 1970s was that the primary problem was tied to the overly successful state and federal fire suppression programs. As a consequence of eliminating fires from the wildland ecosystem, it has been widely held that we have exacerbated the situation by allowing unnatural fuel accumulation (*3*). Thus, when the inevitable fire does come, it is larger and more destructive. A computer model relating fire size to chaparral fuel loading predicted that the prevailing management strategy of fire suppression in California brushlands leads to fewer, but larger and more intense fires (*4*).

A 9-year Landsat imagery record that showed that fires between 5000 and 10,000 ha were slightly more abundant in southern California than in adjacent Baja California (*5*) has been widely cited as support for a link between fire suppression and fire size. On the basis of this study, it has been hypothesized that large wildfires in California shrublands are a modern artifact, due to fire suppression, and that they can be prevented by creation of a mosaic landscape of patches of different ages (*6*). The model is predicated on assertions that, because of fire suppression, (i) the number of fires has declined over time, (ii) fires are substantially larger today than in the past, (iii) contemporary fires burn with greater intensity than in the past, (iv) large fires result from extensive stands of very old age classes, and (v) there has been a decline in area burned, as suggested by some (*3*), but not all (*5*), studies. None of these assertions have been documented.

To investigate historical changes in fire regimes, we used the recently available California Statewide Fire History Database, which includes all records from the California Department of Forestry and U.S. Forest Service and other county records (*7*). We limited our analysis to counties dominated by shrublands with a stand-replacing fire regime: from north to south, Monterey, San Luis Obispo, Santa Barbara, Ventura, Los Angeles, San Bernardino, Riverside, Orange, and San Diego. Records date from the late 19th century for some counties and from at least 1910 for others (*8*).

Collectively, since 1910, there has been a highly significant increase ($r^2 = 0.61$, $P < 0.01$, $n = 9$) in the number of fires per decade. This increase is due largely to southern California counties, which also had significant increases in area burned (Fig. 1) (*9*). In no county was there a significant decline in number of fires or area burned. All counties exhibited significant interdecadal differences in area burned [$P < 0.01$, one-way analysis of variance (ANOVA)]. For most counties, the 1920s and 1970s were high and the 1930s and 1960s low. Collectively, area burned was significantly correlated ($r^2 = 0.71$, $P < 0.01$, $n = 9$) with number of fires, which was also correlated ($r^2 = 0.51$, $P < 0.05$, $n = 9$) with population density (*10*).

All counties reported very large fires from the beginning of record keeping; indeed, one of the largest fires in Los Angeles County was a 24,076-ha fire in 1878 (Fig. 2). During the 20th century, there has been no increase in mean fire size for any county, but four exhibited significant declines (Fig. 2). One contributor to this decline could be a purported inclination by agencies early in the century to not record very small fires (*8*). However, if fires less than 100 ha in size are removed from the data set, there is still a slight downward trend in fire size this century (all counties combined, $r^2 = 0.02$, $P < 0.001$, $n = 2766$). Another factor that could explain a trend toward smaller mean fire size is the increase in human-caused (*11*) ignitions (Fig. 1), coupled with the fact that many are ignited under moderate weather conditions and along roadways, factors contributing to their suppression at a small size (*12*). If we focus just on large fires, greater than 1000 ha, the trend toward smaller fires disappears, but still no county had a significant increase in fire size (ranges: $r^2 = 0.00$ to 0.02, $P > 0.10$ to 0.99, $n = 82$ to 159). The assertion that large wildfires are an artifact of modern fire suppression is not supported.

Contrasting fires after 1950, when fire suppression impacts would be greatest (*13*), with those in and before 1950, we see no significant change in pattern of burning (Fig. 3A); a small percentage of fires account for the bulk of area burned, now and in the past [10% of the fires accounted for 75% (in and before 1950) to 79% (after 1950) of the area burned]. The primary change has been in the proliferation of fires between 10 and 100 ha (Fig. 3B), reflecting both increased ignitions under moderate conditions—that favor suppression—and increased reporting of small fires. In these brushland ecosystems, the frequency of small to medium size fires cannot be used to quantify the risk of large fires (*14*).

Contrasting fire regimes between the first and second halves of this century, we found that fire frequency increased in all but one county (Table 1). The majority of counties exhibited no significant change in mean or median fire size; however, three southern California counties had highly significant declines in mean fire size. Fire rotation intervals, the time required to burn the equivalent

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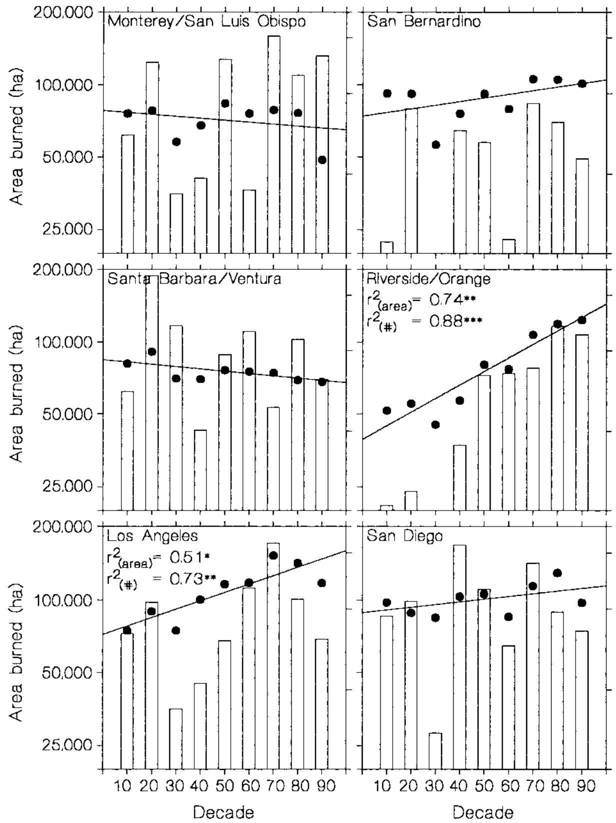

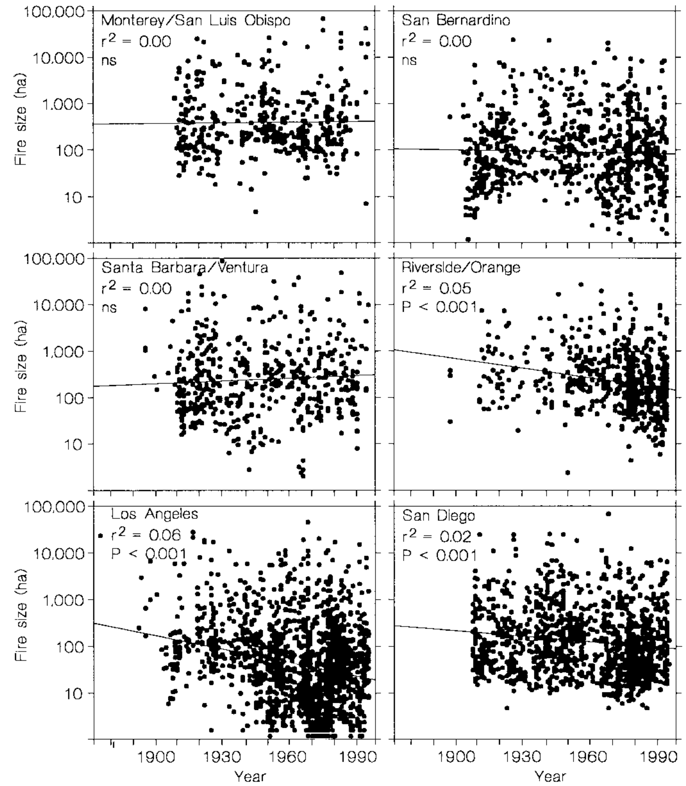

Fig. 1 (left). Area burned (bars) and fire frequency (circles) by decade (1910–1990) for brush-dominated counties in central-coastal and southern California. r^2 is included only when significant: *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$. **Fig. 2 (right).** Magnitude of individual fire size for all records for brush-dominated counties.

Table 1. Brush-covered area as of 1985 and fire statistics for 1910–1950 and 1951–1997 with estimated fire rotation interval (area of brush (*22*)/average area burned) for California counties. Trends with medians are the same for each county.

County	Brush (10^3 ha)	Number of fires		Mean fire size (ha)			Fire rotation interval (years)	
		Before 1951	After 1950	Before 1951	After 1950	P	Before 1951	After 1950
Monterey	358	102	129	1220	1998	>0.32	115	64
San Luis Obispo	250	93	119	1760	2068	>0.68	60	48
Santa Barbara	250	125	61	1622	2341	>0.45	47	81
Ventura	189	143	172	1568	1508	>0.93	121	34
Los Angeles	320	357	1392	827	360	<0.001	44	30
San Bernardino	209	311	544	609	480	>0.33	46	37
Riverside	290	57	613	871	565	<0.01	225	38
Orange	42	25	48	1721	1317	>0.68	36	29
San Diego	365	456	770	939	544	<0.001	35	41

of the total brush area in the county (Table 1), declined in all but two counties (*15*).

These fire rotation intervals do not support the assertion that large fires derive from ancient stands of brush. To investigate the true fire return interval, we used digitized fire maps for the Santa Monica Mountains in Los Angeles and Ventura counties (*16*). Fires in this brush-dominated range have included numerous large catastrophic and costly fires, such as the 1961 Bel Aire Fire or the recent 1993 Green Meadow Fire. Age classes of fuels consumed by all fires exceeding 5000 ha in the past 30 years demonstrate that large fires are not dependent on old classes (Fig. 4). Collectively, there was a significant ($P < 0.05$ with one-way ANOVA, n = 8) difference across age classes, with fuels 11 to 20 years old representing 38%, which was more than double the consumption of older age class fuels. Because of the proximity of this range to urban centers, the age classes consumed may not be representative of more remote sites; however, these data demonstrate that large catastrophic wildfires are not dependent on ancient stands of brush and contradict the assertion that young stands less than 20 years of age prevent fire spread (*5*, *6*).

Inferences that fires today are of greater intensity are based on the assertions that fire rotation intervals have increased and there has been a seasonal shift toward autumn burning (*6*). However, rotation intervals have generally declined (Table 1) and September has remained the peak month of burning throughout this century (Fig. 5).

Humans directly affect fire regimes in two ways: They ignite fires and they suppress fires. In brush-covered landscapes of southern and central-coastal California, there is no evidence that fire suppression has altered the natural stand-replacing fire regime in the manner suggested by others (*3*, *5*). This is

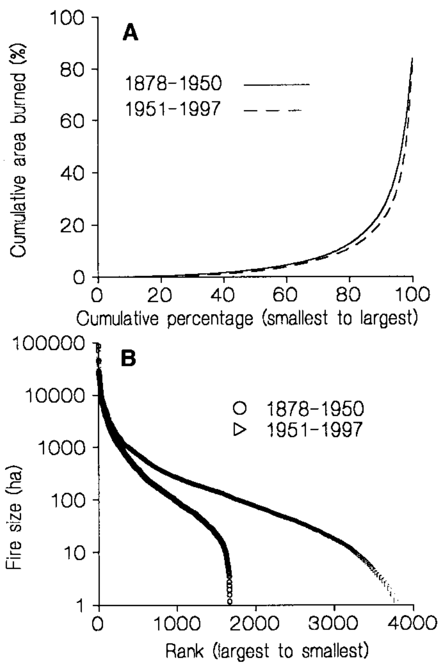

Fig. 3. (**A**) Cumulative area burned and (**B**) fire size distribution, for all counties before and including 1950 and after 1950.

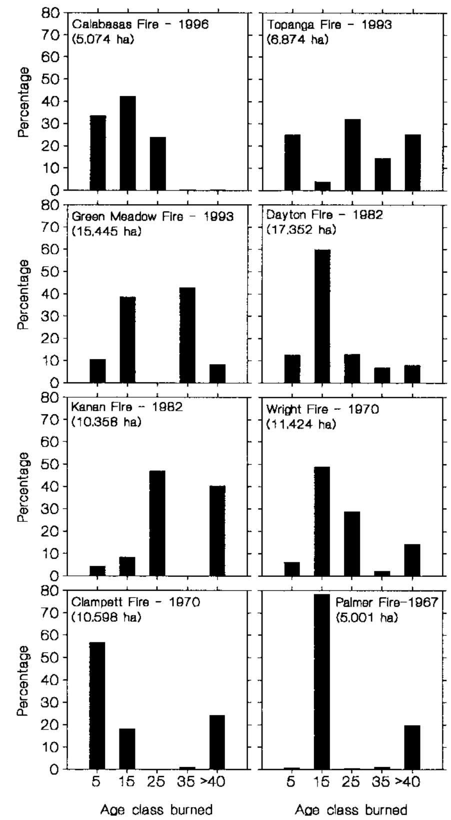

Fig. 4. Age classes burned by all fires over 5000 ha from 1967 to 1996 in the Santa Monica Mountains. Indicated on the abscissa are midpoints of age classes 1 to 10, 11 to 20, 21 to 30, 31 to 40, and over 40 years.

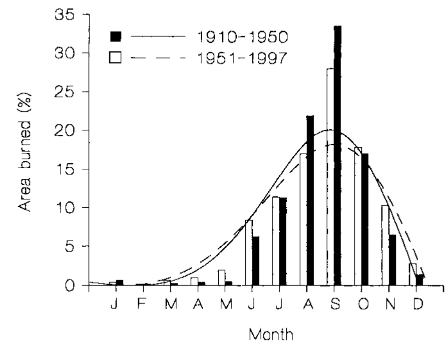

Fig. 5. Area burned by month for 1910–1950 and 1951–1997, for all counties except Riverside and San Bernardino, which were excluded because of incomplete data.

in striking contrast to coniferous forests throughout much of the western United States, where the stand-thinning fire regime has proven amenable to near total fire exclusion, resulting in demonstrably hazardous fuel accumulation and increased potential for catastrophic wildfires (*17*). The primary hazard in brushland ecosystems is the marked increase in fire frequency during the latter half of this century that often results in type conversion to nonnative exotic grasslands (*18*), and fire suppression plays a crucial role in offsetting this impact.

Large catastrophic wildfires in brush-covered regions of California are often driven by high winds, and under these conditions even modern fire suppression techniques are ineffective (*19*). Today, people ignite most of these fires; however, in their absence, lightning storms that typically occur just weeks before the autumn foëhn winds (*11*) would have provided a natural source of ignition. Although fuel structure is an important determining factor in fire behavior, the role of structure diminishes markedly under foëhn winds that can blow at speeds exceeding 100 km/hour and are responsible for the majority of area burned in California brushlands (*19*). Under these conditions, fires readily burn through all age classes of fuels (Fig. 4), and thus, rotational burning programs that attempt to modify vast stretches of chaparral landscape through age class modification are not likely to be effective in stopping these catastrophic fires.

This may come as welcome news to resource managers because the combination of legal restrictions and financial constraints makes large-scale prescribed burning of brushland landscapes unobtainable. Our results support the conclusion that the most effective strategy (*20*) for reducing catastrophic losses from wildfires is to minimize the management effort spent on the bulk of the chaparral landscape and focus on strategic locations. The worst fires predictably follow landscape features, and these patterns can be used to select buffer zones at the urban-wildland interface for more intensive fuel management. However, the urban-wildland interface is so extensive now that even strategically focused intensive management could have enormous ecological impacts. Preference for a rural life-style and the skyrocketing cost of suburban housing in large metropolitan areas continue to expand the urban-wildland interface, and of particular concern is the prediction that rural population will soon exceed urban growth (*21*).

References and Notes

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2. J. A. Zivnuska and K. Arnold [*Calif. Agric.* **4**, 8 (1950)] warned "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."
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7. California Department of Forestry, Fire and Resource Assessment Program (FRAP), Sacramento, CA. Small fires are not recorded; for example, the U.S. Forest Service records only fires over 16 ha. However, the threshold limit varies with the agency.
8. Fires recorded here burned predominantly in chaparral, which sometimes forms a mosaic with coastal sage scrub, grassland, oak woodland, and coniferous forests. Early in the century, there may be a deficit of small fires because of incomplete reporting, but total area burned is not likely to be affected because small fires are a minor portion and large fires are less likely to have been missed.
9. Adjacent counties were combined for presentation purposes; statistical tests discussed in the text were performed on counties both separate and combined. Data for the 1990 decade were standardized by dividing the average for the first 8 years by 0.8.
10. For population density statistics, see www.census.gov/population/cencounts/ca190090.txt
11. For these counties, natural lightning-ignited fires typically make up less than 5% of all fires [J. E. Keeley, *U.S. Forest Serv. Gen. Tech. Rep. PSW-58* (1982), pp. 431–437].
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Wildland Fire (Wiley, New York, ed. 2, 1996)]. Additionally, because of low rates of decomposition in these ecosystems, if fire suppression were to result in fuel accumulation, the magnitude of this impact would be cumulative with time and be greatest in the latter half of the century.
14. Compare B. D. Malamud, G. Morein, D. L. Turcotte, *Science* **281**, 1840 (1998).
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Positive Feedbacks in the Fire Dynamic of Closed Canopy Tropical Forests

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The incidence and importance of fire in the Amazon have increased substantially during the past decade, but the effects of this disturbance force are still poorly understood. The forest fire dynamics in two regions of the eastern Amazon were studied. Accidental fires have affected nearly 50 percent of the remaining forests and have caused more deforestation than has intentional clearing in recent years. Forest fires create positive feedbacks in future fire susceptibility, fuel loading, and fire intensity. Unless current land use and fire use practices are changed, fire has the potential to transform large areas of tropical forest into scrub or savanna.

Fire is recognized as a historic but infrequent element of the Amazonian disturbance regime (*1*, *2*). Currently, however, fires in Amazonian forests are frequent because of the accidental spread from nearby pastures and the increased susceptibility of partially logged or damaged forests (*3*–*6*). Here, positive feedbacks associated with accidental forest fires are reported; these constitute a threat to the integrity of a large part of the Amazonian forest.

Field studies were concentrated in the Tailândia region (Fig. 1). Ten 0.5-ha plots (eight fire-affected and two control), spread over 100 km^2, were established in 1996 to study fire impacts on forest structure, biomass, and species composition (*3*). These plots were recensused after the dry season of 1997, during which eight of the plots burned to varying degrees. Fire recurrence, tree mortality, and biomass combustion levels within forests of different burn histories were quantified. In addition, combustible fuel mass was assessed with the planar intersect method (*7*) as adapted by Uhl and Kauffman (*8*, *9*).

We also examined characteristics of fires while they were occurring in four forest types (previously unburned, once-burned, twice-burned, and more than two previous burns) in December 1997. Direct observations of fires were made at widely scattered locations within a 150-km^2 area south of Tailândia. For each observed fire, flame heights and depths (the width of the flaming front) were measured or estimated (*10*). The time the fireline took to move across a known distance was used to calculate the rate of spread and was combined with flame depth data to calculate the average range of flame residence times at a point. Flame height was used as a conservative estimate of total flame length for the calculation of fireline intensity (*11*) because wind and slope were minimal (*12*).

The first fire to enter a forest usually moves slowly along the ground (Table 1) and is similar to a prescribed burn ($<$50 kW m^{-1}) in intensity (*13*). These fires consume little besides the dry leaf litter, but because of the characteristically thin tree bark [7.3 ± 3.7 mm for $>$20 cm diameter at breast height (dbh) (*8*)] protecting the cambium tissues, they still kill roughly 95% of the contacted stems $>$1 cm dbh. Large, thicker barked trees survive. After the fire, a rain of combustible fuels of all sizes falls from the standing dead trees (Table 1) (*14*). Fire damage and windthrow in these thinned forests continue to cause mortality for at least 2 years after the fire (*4*, *15*). Fuel levels rise substantially and the open canopy (50 to 70% cover) allows greater solar heating and air movement to dry out the forest fuels. Previously burned forests thus become susceptible to fire during common dry season weather conditions (*3*).

Previously burned forests were much more likely to burn than were unburned forests in 1997 (Table 1). Burned forests are often adjacent to fire-maintained pasture and agricultural plots and are therefore frequently exposed to sources of ignition. Second fires are faster moving and much more intense. We estimate heat release (*12*) of $<$7500 kW m^{-2} in first burns but of 75,000 kW m^{-2} or more in subsequent burns. Because of the increased flame depth, the residence time increases despite faster rates of spread, resulting in greater tree mortality. Large trees have little survival advantage during these more intense fires. Fire-induced tree mortality can be modeled as a function of bark thickness and fire residence time (*16*). For the observed fire characteristics and bark thickness distribution (*8*), no more than 45% of trees over 20 cm dbh are susceptible to fire-induced mortality in the initial fires. However, in recurrent fires, up to 98% of the trees become susceptible to fire-induced mortality.

The impacts of recurrent fires are much worse than those of initial fires. Higher mortality results in a very open canopy (10 to 40% cover), large inputs of combustible fuels, and faster drying. During the 1997 fires, substantial amounts of carbon were released to the atmosphere, with combustion reducing onsite biomass by approximately 15, 90, and 140 Mg ha^{-1} in first, second, and recurrent burns, respectively. Invading grasses and weedy vines add highly combustible live fuels to the already

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