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FIRE REGIMES AND FOREST STRUCTURE IN THE SIERRA MADRE OCCIDENTAL, DURANGO, MEXICO

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ABSTRACT

Frequent, low-intensity fire is a key disturbance agent in the long-needled pine forests of western North America, but little is known about the fire ecology of the Mexican forests which have been least affected by fire exclusion. We compared fire disturbance history and forest structure at four unharvested or lightly-harvested study sites differing in recent fire history. Frequent, low-intensity fires, recurring between 4 to 5 years for all fires and 6 to 9 years for widespread fires, characterized all the sites until the initiation of fire exclusion in the mid-twentieth century at three of the four sites. Although most fires in the study area are ascribed to human ignitions, evidence of both lightning and human-caused burning was observed on the study sites. A possible connection between fire occurrence and climate was indicated by a correspondence between regional fire years and positive extremes of the Southern Oscillation index, which is associated with cold/dry weather conditions. Forest ecosystem structures differed in ways consistent with the thinning and fuel consuming effects of fire. Two sites with extended fire exclusion were characterized by relatively dense stands of smaller and younger trees, high dead woody biomass loading, and deeper forest floors. In contrast, a site which had burned following a 29-year fire exclusion period, and the final site where frequent fires had continued up to the present, were both relatively open forests dominated by larger trees. The recently burned sites had lower dead woody biomass loading, especially of rotten woody fuels, and more shallow duff layers. The high regeneration density but low overstory density at the recently burned sites is also consistent with the thinning effect of low-intensity fire. Long-term management and conservation strategies for these forests should recognize the historic role of fire disturbance as well as the potential for changes in fire intensity and ecological effects following extended fire exclusion.

RESUMEN

El fuego frecuente de baja intensidad es un agente clave de disturbio en los bosques de pinos de hojas largas del occidente de Norteamérica; sin embargo, se conoce poco de la ecología del fuego en los bosques mexicanos que han sido menos afectados por la exclusión del fuego. Comparamos la secuencia de disturbio causado por fuego y la estructura forestal de cuatro sitios, poco o no explotados, que varían en su reciente historial de fuegos. Los frecuentes incendios de baja intensidad, que se presentaron periódicamente cada 4 a 5 años, si se toman en cuenta todos los incendios, y cada 6 a 9 años para el caso de los incendios extensos, caracterizaron todos los sitios hasta el inicio de la exclusión de fuego a mediados del siglo XX en tres de las cuatro parcelas de estudio. Aunque se considera que la mayoría de los incendios en el área son de origen humano, en los sitios de referencia se observaron evidencias de igniciones causadas tanto por las descargas eléctricas, como por el hombre. Una posible conexión entre la incidencia de los incendios y el clima quedó señalada por la correlación encontrada entre los años con incendios regionales y los extremos positivos del índice de oscilación meridional, asociados con condiciones de clima frío y seco. Las estructuras del ecosistema forestal diferían de acuerdo con los efectos del aclareo de árboles y del consumo de material combustible. Dos sitios con una larga exclusión de incendios se caracterizaron por masas relativamente densas de árboles más pequeños y jóvenes, altas cantidades de material combustible, y capas de humus más profundas. En contraste, tanto en la parcela que se quemó después de un período de exclusión de fuego de 29 años, como también en el último sitio donde los incendios frecuentes han continuado hasta el presente, se registraron bosques relativamente abiertos dominados por árboles más grandes y maduros. Las áreas de muestreo recientemente quemadas tenían menos material combustible, especialmente de madera podrida, y capas de humus menos profundas. La gran cantidad de renuevos en combinación con la baja densidad del estrato superior de los sitios recientemente quemados es también consistente con el efecto de aclareo de los incendios de poca intensidad. Las estrategias para la conservación y el manejo a largo plazo de estos bosques deben reconocer el papel histórico del disturbio causado por el fuego, así como el potencial de cambios en la intensidad de los incendios y los efectos ecológicos de una larga exclusión de fuego.

INTRODUCTION

Disturbance regimes of frequent, low-intensity fires are a keystone ecological process (sensu Holling, 1992) regulating the density, species composition, dead woody biomass, and forest floor structure of western long-needled pine ecosystems in North America (Weaver, 1943, 1951; Cooper, 1960; Covington et al., 1994; Arno et al., 1995; Swetnam and Baisan, 1996). These long-needled pines (*Pinus ponderosa*, *P. durangensis*, *P. engelmannii*, *P. arizonica*, *P. jeffreyi*, *P. washoensis*, and others) form a closely-related ecological group in the section *Ponderosae* with thick bark, insulated buds, and high capability to recover from crown scorch, all of which are considered adaptations to frequent fire (Conkle and Critchfield, 1988; McCune, 1988; Barton, 1993). The temperate coniferous forests of northern Mexico are the most biologically diverse of these long-needled pine ecosystems (Bye, 1995; Felger and Wilson, 1995) and the least affected by the disruption of frequent fire regimes (Leopold, 1937; Marshall, 1962; Fulé and Covington, 1994; Baisan and Swetnam, 1995). However, with a few exceptions (e.g., Minnich, 1993; Baisan and Swetnam, 1995; Minnich et al., 1995; Villanueva-Díaz and McPherson, 1995), little is known about the role of fire or the relationship between fire and ecosystem structure in the forests of northern Mexico.

Fire ecology studies in the coniferous forests of the southwestern United States have shown that frequent fire played a key ecological role in these ecosystems by maintaining open, park-like forests of fire-adapted species, controlling tree populations, limiting accumulation of dead woody biomass and deep forest floors, cycling nutrients stored in dead organic material, and encouraging herbaceous production (Weaver, 1951; Cooper, 1960; Biswell, 1972; Covington and Moore, 1994; Covington et al., 1994; Swetnam and Baisan, 1996). Regimes of frequent, low-intensity fire were disrupted throughout the southwestern United States between 1880 and, 1900, however, by heavy grazing of domestic animals, logging, and fire suppression associated with Euro-American settlement of the region (e.g., see Cooper, 1960; Swetnam and Baisan, 1996; Covington et al., 1994). In the absence of frequent fires, striking changes occurred: tree species less adapted to frequent fire (e.g., *Abies*, *Pseudotsuga*) have invaded mesic sites at the expense of other plants, and tree biomass, both live and dead, has steadily accumulated, contributing to increasing susceptibility to insect and disease epidemics and supporting a shift from frequent, low-intensity surface fires to increasingly larger crown fires (Cooper, 1960; Swetnam, 1990; Covington and Moore, 1994; Kolb et al., 1994; Swetnam and Baisan, 1996).

In contrast, the Mexican experience with fire has been quite different from that of the United States. Deliberate agricultural burning has long been practiced by both native and Hispanic peoples in Mexico. Despite a history of official opposition to wildfire dating back to pre-Columbian times, effective fire suppression --the combination of laws, anti-fire attitudes among the public and land managers, and adequate financial and infrastructure resources needed to systematically exclude fire-- has not been achieved in much of the country (Lumholtz, 1902; Leopold, 1937; González-Cabán and Sandberg, 1989; Rodríguez and Sierra, 1992; Chou et al., 1993). Striking differences between the dense, fire-excluded forests of the southwestern United States and the open, frequently burned forests of northern Mexico were noted as early as, 1937 by Leopold and later by Marshall (1957, 1962). They and others (González-Cabán and Sandberg, 1989; González et al., 1993; Bye, 1995; De Bano and Ffolliott, 1995) have called repeatedly for increasing fire ecology research in Mexico. Most recently, Minnich et al. (1995) found that coniferous forests of Sierra San Pedro Mártir (Baja California Norte), where unmanaged fire regimes continued, remained relatively open and dominated by pines. To our knowledge, however, the present study is the first to examine the relationship between fire regimes and ecosystem structures in the central Sierra Madre Occidental.[1]

STUDY QUESTIONS

The goal of the present study was to compare ecosystem structures at relatively undisturbed coniferous forest sites in northern Mexico which differed in recent fire history, ranging from extended fire exclusion to continuing frequent fires. We selected study sites and sampling procedures to address the following questions: What are the past and present patterns of fire occurrence on sites representing (a) extended fire exclusion, (b) fire exclusion followed by fire return, and (c) continued frequent fire? Are fires primarily of natural or human origin? Is there a relationship between climatic factors and fire occurrence? Have forest density, basal area, regeneration density, and dead biomass all increased with fire exclusion, consistent with the changes observed in long-needled pine forests of the western U.S. following fire exclusion? Finally, what are the implications of changing fire regimes for conservation and management of these forests?

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[1] Some data from two of the study sites presented here was compared by Fulé and Covington (1994) and an abbreviated summary discussing all four sites was published by Fulé and Covington (1996). The present study is comprehensive, containing detailed quantitative data and comparisons not previously published.

STUDY AREA

The Sierra Madre Occidental, part of the North American cordillera linking the Rocky Mountains with the central Mexican highlands, is an important migratory pathway and a center of endemism (Perry, 1991; Toledo and Ordóñez, 1993). The exceptional biological diversity of the Sierra Madre is comparable to that of tropical rainforests (Bye, 1995; Felger and Wilson, 1995). While much of the northern Sierra Madre in Durango and Chihuahua has remained relatively undeveloped even 400 years after Spanish colonization due to the rugged and remote landscape, rapid increases in population, infrastructure development, and commercial timber exploitation, are causing substantial shifts and raising international concerns about conservation of the existing social and ecological conditions (Weaver, 1993; Bojórquez-Tapia et al., 1995). The study area, a region approximately 2,000 km^2 in size of pine-oak forests of the Sierra Madre Occidental in northwestern Durango (Fig. 1), was selected because the relatively recent advent of large-scale timber harvesting (*circa* 1970) made it possible to find examples of unharvested forest tracts for sampling. Cooperation and assistance from local landowner and forest management organizations facilitated the study.

Four study sites representing apparently different recent fire regimes were chosen within the study area for fire history and forest structure sampling (Fig. 1). Selection criteria included no (or minimal) harvesting; evidence of past fires (such as fire scars and charred wood); and apparent differences in recent fire history ranging from extended fire exclusion to continuing frequent fire up to the present (evidenced by relative amount of fuels, forest density, and relative weathering of charred wood). The first two sites, sites AV (Arroyo Verde, lat. 25°05' N, lon. 106°13' W), with extended fire exclusion, and AL (Arroyo Laureles, lat. 24°57' N, lon. 106°13' W), with continuing frequent fire, were sampled in July and August, 1993. A preliminary comparison of these two sites showed that the initiation of fire exclusion at AV was associated with increased forest density and fuel accumulation (Fulé and Covington, 1994). The third site, SP (Salsipuedes, lat. 25°15' N, lon. 106°30' W), representing fire exclusion followed by fire return, was sampled in April, 1994. All three sites were unharvested pine-oak forests. The fourth site, CB (Cebadillas, lat. 24°53' N, lon. 106°00' W), also with extended fire exclusion, was a mixed pine-oak-*Abies-Pseudotsuga* forest, representing the mixed conifer type which is relatively rare in the Sierra Madre Occidental. Some of the larger pines had been selectively harvested at CB approximately 15 years prior to sampling in April, 1994. Table 1 summarizes study site characteristics.

Table 1. Study site characteristics.

Site	Size	Elevation in m	Slope	Aspect	Forest Type
AV	70 ha	2200-2500	37%	SW	Pine-oak unharvested
CB	30 ha	2850-2950	32%	N	Pine-oak-*Abies-Pseudotsuga* selectively harvested
SP	30 ha	2300-2500	47%	W (SW & NW)	Pine-oak unharvested
AL	70 ha	2200-2500	54%	SW	Pine-oak unharvested

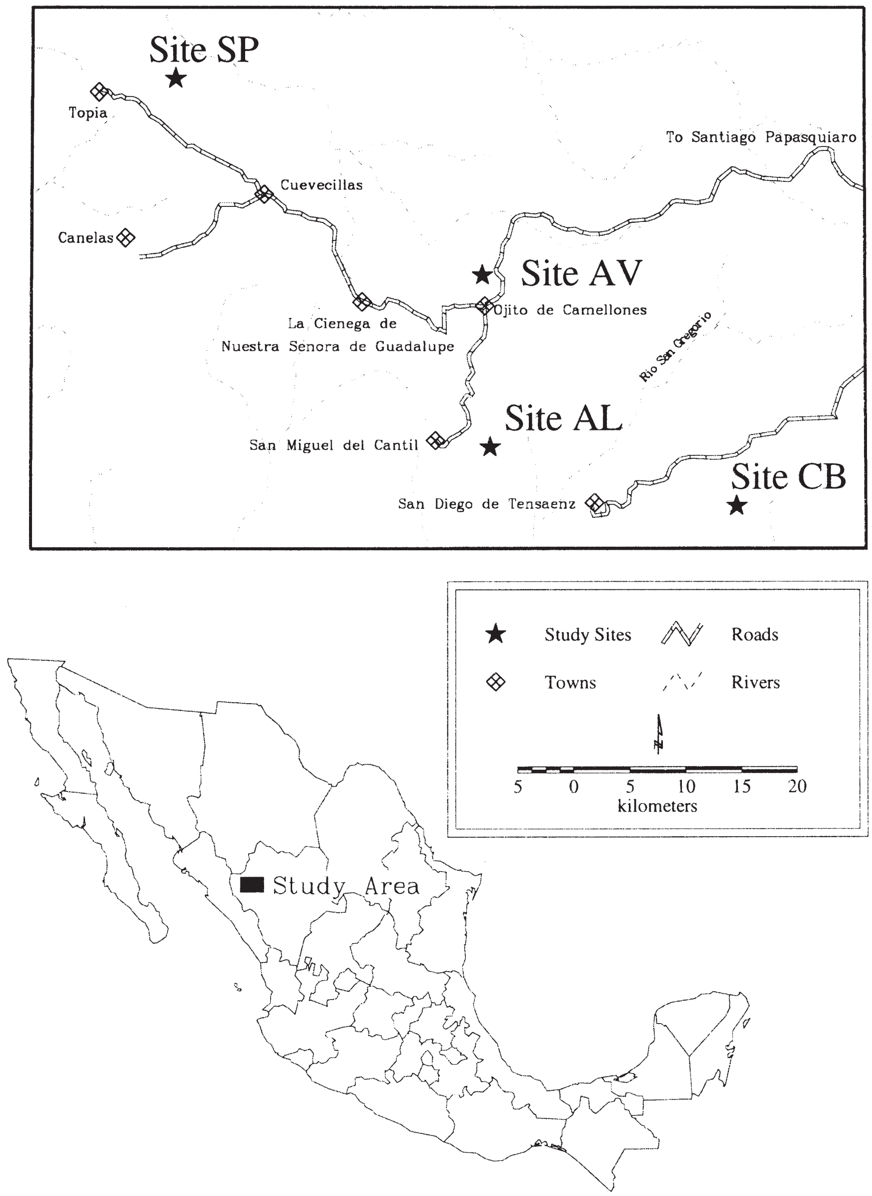
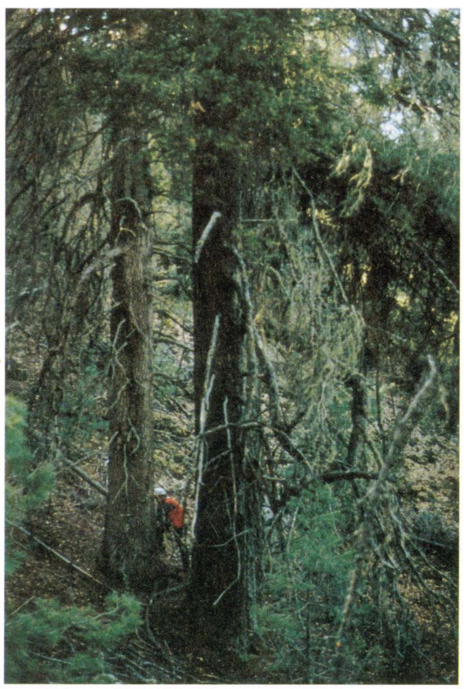


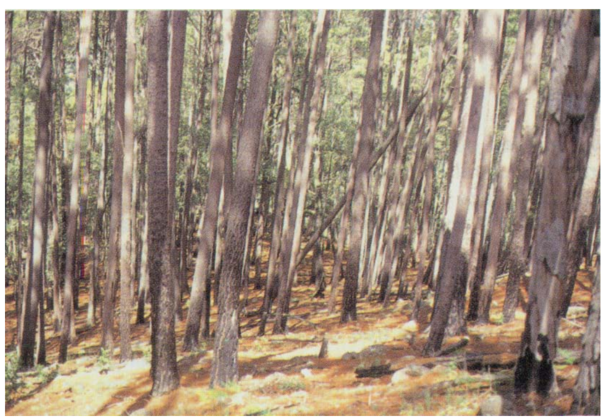
Fig. 1. Map of study area and sampling site locations.



Site AV: Scene from the northwest corner of the Arroyo Verde (AV) study site in July, 1993. The foreground shows a small clearing for cultivation, made by cutting and burning a group of trees. Similar activities appear to be the cause of some of the recent fires at several study sites, although lightning ignitions may have been a more important fire source over long periods of time. In the background is a very dense stand of small-diameter trees, primarily *Pinus lumholtzii*.

Site CB: Large *Psuedotsuga menziesii* and *Abies durangensis* trees dominate the Cebadillas (CB) site, photographed in April, 1994. Tree density appears to have increased following the exclusion of fire after 1951. Selective cutting of larger pines has enhanced the dominance of the Douglas-fir and fir in the overstory, while permitting the accumulation of dead woody biomass on the forest floor. The combination of heavy fuel loads together with continuous vertical fuels in the form of interlinked tree crowns suggests that much of this site will support a high-intensity canopy fire under dry, windy conditions.

Site SP: The effects of widespread recent fires, following 29 years of fire exclusion, are shown in this scene of the Salsipuedes (SP) site in April, 1994. Fuel accumulation over the fire-free period led to relatively intense fires in 1984 and 1993, damaging or killing many of the mature trees such as the pine in the center. Alders and oaks sprouted vigorously after the 1984 fire, but many were topkilled in 1993, as shown by the dead, small-diameter stems in the foreground. The sprouting species are resilient to these relatively intense recent fires, however, while the conifers which reproduce from seed appear to be declining.



Site AL: Periodic fires continued up to the sampling date at the Arroyo Laureles (AL) site, shown in July, 1993. Although this scene shows a relatively dense pine stand, the forest floor is relatively free of undergrowth and the lower tree boles have been pruned by repeated burning. Little dead woody biomass appears on the forest floor. Recent fires here had intense behavior at localized points, torching individual trees and even crowning through an area of several ha in the northwest corner of the site. However, the low average forest density and fuel loading maintained by the frequent fire regime indicate that this site was relatively resistant to large-scale crownfires. The AL site was selectively harvested in 1994, after our sampling.

All four study sites were communally owned, sites AV and AL by the Ejido Salto de Camellones, site SP by the Ejido Topia, and site CB by the Ejido Hacienditas. Adjacent lands included both *ejidos* and private holdings. Most surrounding land with accessible commercially-valuable timber had been selectively harvested at least once. Evidence of natural resource utilization observed on the study sites, in addition to the pine harvest at site CB, included grazing, scattered tree cutting for firewood, poles, shingles, and access to beehives. At the three pine-oak sites, one to several clearings for illicit cultivation were found. The clearings were small, ranging from 200 m^2 to 500 m^2, created by felling a patch of trees, igniting the dried vegetation, and cultivating the plot for one or more seasons. Clearings were deliberately located in steep, remote, forested areas; no firebreaks existed to prevent the burns from passing into the surrounding forest.

The regional climate pattern is dry spring conditions followed by a summer rainy season. Annual precipitation at Ojito de Camellones (elevation 2400 m) is 2200 mm (9 year average), with 80% occurring between June and October. Other weather stations at lower elevations average less annual precipitation: 1540 mm (11 year average) at San Miguel del Cantil (2250 m elevation), 1150 mm (5 year average) at Topia (1800 m elevation), and 740 mm (11 year average) at San Diego de Tensaenz (1550 m elevation). Records of temperature are less available than those of precipitation, but 1992 maximum and minimum temperatures at La Cienega, 12 km west of Ojito de Camellones at 2500 m elevation, were 18° and -8°C in January, and 28° and 7°C in August. Soils originated from igneous rocks, primarily rhyolitic with scattered basaltic and granitic outcrops (Guízar et al., 1992). The topography is characterized by high-elevation mesas dissected by steep canyons.

METHODS

Field Sampling

Sampling was carried out on systematic grids designed to simultaneously measure fire history and forest structural characteristics. The grid spacing was 150 m X 150 m at sites AV and AL (total site size 70 ha each) and 100 m X 100 m at the smaller sites SP and CB (30 ha each). Fig. 2 shows the sampling layouts at each study site.

Thirty sampling points were established at each study site and used as the center points of nested sampling plots varying in size. Fire scars were sampled at the largest scale, represented by the 150 or 100 m interplot distance. Following a search of 75 or 50 m radius centered on each sampling point, the tree catfaces apparently containing the oldest scars and/or the best record of multiple fires were selected for sampling (Arno and Sneck, 1977; Swetnam and Baisan, 1996). Because of the ongoing fire regimes or recent onset of fire exclusion at the study sites, most fire scar samples were taken as partial cross sections cut from living trees, although samples from snags, logs, and stumps were also collected. The geographic location of each fire scar sample was referenced (distance and bearing) to the sample point center. Fig. 2 maps fire scar sample locations relative to the plot sampling grid and lists sample identification numbers.

Overstory vegetation was sampled on circular plots 200 m^2 in size (radius = 7.98 m) centered on each sampling grid point. In the context of this study, all woody plants over

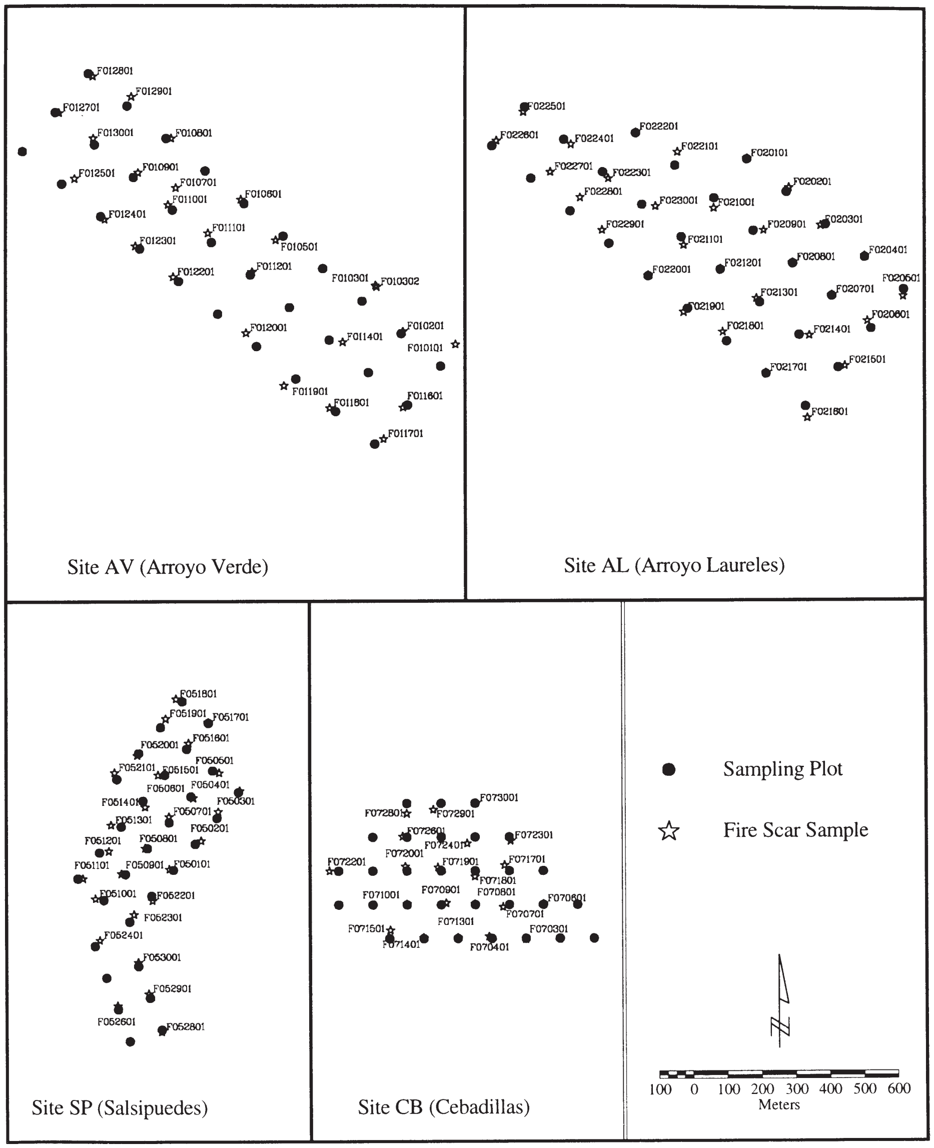


Fig. 2. Map of forest sampling plots and fire scar sample locations.

1.3 m in height were considered "overstory" vegetation, as contrasted with the woody seedlings described below. Species, condition, height, live crown ratio, presence of lightning scars, and diameter at breast height (dbh, 1.3 m above ground level) were recorded for all live and dead trees over 1.3 m in height, as well as for stumps and dead and downed trees which surpassed breast height while alive. Condition classes were assigned based on a tree, snag and log classification system originally developed in ponderosa pine/mixed conifer forests of the Blue Mountains of Oregon (Maser et al., 1979; Thomas et al., 1979) and widely applied in ponderosa pine forests (Rogers et al., 1984; Covington and Moore, 1994a; Lundquist, 1995). Condition classes were (1) live; (2) declining; (3) recent snag; (4) loose bark snag; (5) clean snag; (6) snag broken above dbh; (7) snag broken below dbh; (8) downed dead tree; and (9) cut stump. Increment cores were taken at 45 cm above ground level from all living conifers over 6 cm dbh.

Woody regeneration, defined as tree seedlings below 1.3 in height, was sampled on nested circular plots 40 m^2 in size (radius = 3.57 m). The number of woody stems on each plot was tallied by species. Percent cover by herbaceous species was also estimated visually on each 40 m^2 plot.

Dead woody biomass and forest floor depth were sampled along a 15-m planar transect laid out in a randomly-selected direction from the center of each sample plot (Brown, 1974). Intercepts of the transects with woody fuels in the diameter classes 0-0.6 cm, 0.6-2.5 cm, and 2.5-7.6 cm were tallied along the first 5 m; diameters of sound and rotten fuels larger than 7.6 cm were measured along the entire transect. Depth of litter and duff was recorded every 5 m.

Analysis

Dendrochronology-fire scar samples

In the laboratory, fire scar samples were surfaced, mounted on plywood backing, and sanded with increasingly finer belts up to 400 grit abrasive. A total of 105 fire scar samples was crossdated (Stokes and Smiley, 1968) with master tree-ring chronologies previously developed in the region (Harlan, 1973) and others developed in the present study. Most samples were less than 200 years old and could be crossdated visually using characteristic patterns of narrow marker years, especially in the twentieth century: 1860, 62, 87, 1902, 04, 09, 25, 34, 43, 51, 56, 63, 71, 74. Several samples from dead trees were initially dated as floating tree-ring width series with the COFECHA program (Grissino-Mayer and Holmes, 1993), then dating was confirmed visually on each sample. After dating, ring widths of all samples were measured and dating was checked with COFECHA as well as by independent visual crossdating of a 20% to 30% subset of samples by another dendrochronologist. To measure the season of occurrence of fires (Baisan and Swetnam, 1990), the relative position of each fire scar within the ring was recorded using the following categories: EE (early earlywood), ME (middle earlywood), LE (late earlywood), L (latewood), and D (dormant season, scar occurring between the cessation of latewood growth and the beginning of the next year's earlywood growth). The assumption that dormant season scars are generally spring fires in the southwestern United States (Baisan and Swetnam, 1990) appeared valid for the similar climate patterns and spring fire season of the central Sierra Madre Occidental, although the possibility of dormant scars forming in the fall following latewood growth cannot be excluded. From the four study sites, a total of 637 fire scars were dated to the year and season (if determinable) of origin.

Forest structure analysis

Overstory age structure was determined from ring counts of surfaced and mounted increment core samples, using known patterns of narrow marker years to correct ring counts by visual crossdating whenever possible. The ring counting provides a rapid age estimate highly accurate within 10-year age classes, adequate for the purposes of this study in which approximately 1,000 cores were collected for age determination. For cores which missed the pith, a pith locator consisting of a transparent overlay of concentric circles was used. An appropriate circle set was selected based on the curvature of the rings nearest the center and the average growth rate of rings near the center. "Tree age" in the following sections refers to total age at the 45 cm coring height.

Dead woody biomass was calculated from the planar transect intercept tallies using the method of Sánchez and Zerecero (1983), which is analogous to the method of Brown (1974) and is commonly applied in Mexican temperate forests. Basal area (m^2/ha) and density (number/ha) were calculated for living overstory trees, recent snags, older snags, dead and downed trees, and stumps, and density of regeneration was also calculated.

Statistical analysis

Statistical analysis of the fire history data was facilitated by the capabilities of the FHX2 fire analysis software developed by Grissino-Mayer (1995). Fire return intervals were analyzed statistically beginning in the first year in which an adequate sample depth of recording trees occurred at each site (Grissino-Mayer et al., 1994). The minimum sample depth was taken to be the first fire year with three recording trees (10% or more of the total sample size at each site). Fire return intervals were analyzed statistically in different sub-categories. First, all fire years, even those represented by a single scar, were considered. Then only those fire years were included in which respectively 10% or more, and 25% or more, of the recording samples were scarred. These categories help define "widespread" fires which may have been larger in area or more intense (Grissino-Mayer, 1995). The statistical analysis of fire return intervals includes several measures of central tendency: the mean fire interval (average number of years between fires), the median, and the Weibull median probability interval (WMPI). The latter statistic is a central measure in the Weibull distribution, which is useful to model asymmetric fire interval distributions and to express fire return intervals in probabilistic terms (Johnson, 1992; Grissino-Mayer et al., 1994; Swetnam and Baisan, 1996). Finally, the maximum hazard fire interval (100% Hazard) is the time in years at which the 100% probability level is reached in the Weibull distribution. Theoretically, the hazard interval represents the maximum fire-free period possible in the modeled distribution, and may be compared with the actual maximum fire return intervals (see Grissino-Mayer (1995) for discussion of the properties of the Weibull distribution).

Statistical analysis of forest structure was carried out with the SYSTAT software (Wilkinson, 1988). Alpha level for all analyses was .05. Multivariate analysis of variance

(Tabachnick and Fidell, 1983, p. 222-291) was used to determine differences between sites on intercorrelated measured variables, such as the suite of dead woody biomass variables. If a significant difference was found between sites (Wilk's lambda statistic), then Tukey's honest significant difference (HSD) test (Zar, 1984, p. 186; Day and Quinn, 1989) was used for multiple comparisons among means at the four study sites.

RESULTS

Fire Disturbance History

The fire history results confirmed that all four study sites burned frequently in the past but periods of extended fire exclusion in the twentieth century occurred at three of the sites, while the fourth continued to burn frequently up to the present. The fire disturbance histories at the four study sites are summarized in the master fire charts shown in Fig. 3a-d and compared graphically in Fig. 4. The horizontal lines in each chart represent individual fire scar samples. The sample identification numbers at the right are the same as those shown in the fire scar sample map (Fig. 2). The horizontal lines are dashed when the sample is not considered a recording tree, usually before the first scar forms an open wound susceptible to repeated scarring from subsequent fires, or during periods in which fire scars have been burnt away completely (Grissino-Mayer, 1995; Swetnam and Baisan, 1996). Continuous lines indicate recording periods. Short vertical bars show the years in which each sample was scarred. Beneath the individual samples, a composite line lists all the fire years at the site. The time span of the fire histories is relatively short (maximum 196 years) compared to fire histories from the southwestern United States (Grissino-Mayer, 1995; Swetnam and Baisan, 1996); this is due to a combination of the relatively short lifespan of trees at the study sites (see age distribution below) as well as to the fact that recurring fires up to the mid-twentieth century or to the present have consumed relict wood which often is used to extend U.S. fire histories into the past (Baisan and Swetnam, 1990).

Dates of fire regime disruption, following the last widespread fires at sites AV, CB, and SP were identified in the fire histories (Fig. 3a-d). Prior to fire regime disruption, all four sites had mean fire return intervals of less than 5 years (Table 2) when all fire years are included, and mean fire return intervals of 9 or fewer years for widespread fires (scarring 25% or more of the recording trees). The three measures of central tendency in Table 2 agreed closely when the number of fire return intervals was high (e.g., the all-scar and 10%-scarred distributions), but showed less agreement when the number of intervals was low (e.g., the 25%-scarred distribution). Similarly, the 100% hazard value of Weibull distribution agreed less closely with the observed maximum fire-free interval when the number of intervals was low (e.g., the 25%-scarred distribution).

The areal extent and intensity of past fires affect the scale and nature of fire effects on ecosystem structures, but spatial fire patterns are difficult to estimate in ecosystems characterized by frequent, low-intensity fire where forest stand boundaries do not correspond with fire edges (Swetnam and Dieterich, 1985; Swetnam and Baisan, 1996). A direct relationship between the percent of fire scarring and the size or intensity of a given fire cannot be drawn because fires are frequently not recorded even on trees with open fire scars

Table 2. Fire return intervals at the four study sites. Statistical analysis was carried out in three categories: (1) all fire years, including those represented by a single fire scar; (2) fire years in which 10% or more of the recording trees were scarred; and (3) fire years in which 25% or more of the recording trees were scarred. For those sites where a disruption of the long-term fire regime was identified (sites AV, CB, and SP), fire return intervals are presented both for the pre-disruption period as well as the full historical recording period.

Site/Analysis Period Scar Category	No. of Intervals	Mean (MFI)	Median	Standard Deviation	Minimum	Maximum	WMPI	100% Hazard
Site AV / 1801-1992								
All scars	42	4.55	4.0	3.38	1	17	4.08	14
10% scarred	33	5.79	5.0	4.81	1	24	5.08	26
25% scarred	17	11.24	7.0	10.48	4	47	9.63	120
Site AV / 1801-1945								
All scars	38	3.79	3.0	2.20	1	12	3.58	7
10% scarred	30	4.80	4.0	2.96	1	16	4.49	12
25% scarred	16	9.00	6.5	5.15	4	18	8.47	34
Site CB / 1797-1993								
All scars	33	5.94	5.0	4.96	1	23	5.15	33
10% scarred	30	6.53	5.0	5.02	1	23	5.79	33
25% scarred	19	10.32	6.0	10.27	1	42	8.31	590
Site CB / 1797-1951								
All scars	31	4.97	4.0	3.14	1	14	4.59	14
10% scarred	28	5.50	5.0	3.19	1	14	5.16	15
25% scarred	18	8.56	5.5	7.03	1	31	7.43	121
Site SP / 1812-1993								
All scars	40	4.53	4.0	2.91	1	17	4.21	10
10% scarred	33	5.49	4.0	4.70	2	29	5.00	12
25% scarred	24	7.54	6.0	5.34	3	29	7.08	18
Site SP / 1812-1955								
All scars	35	4.09	4.0	1.90	1	9	3.96	7
10% scarred	31	4.61	4.0	1.98	2	9	4.53	7
25% scarred	22	6.50	6.0	2.84	3	15	6.41	10
Site AL / 1879-1992								
All scars	24	4.67	4.0	2.87	2	14	4.44	8
10% scarred	22	5.09	4.0	2.88	2	14	4.89	9
25% scarred	17	6.59	6.0	3.32	3	15	6.43	11

(Dieterich and Swetnam, 1984), but the extent of scarring is roughly related to the areal extent of fire spread and/or intensity of burning. Furthermore, because sampling systematically covered each study area, the grid-based fire scar sampling design (Fig. 2) helps support spatial interpretations of fire spread since sampling covers the entire area (see Arno et al. (1993) for a similar approach).

Spatially homogeneous fire regimes prevailed at all four sites, consistent with the lack of topographic or vegetative barriers to fire spread on the sites and the fact that fires could have persisted for extended periods in the absence of fire suppression activities. Spatial homogeneity of the fire regimes was examined by dividing each area along its shortest axis into two roughly equal sections. Fire return intervals and percentage of scarring between the two sections of each study site for the all-scar, 10%-scar, and 25%-scar distributions were tested for significantly different means (t-test), variances (F-test), and distributions (Kolmogorov-Smirnov test). Alpha level for all tests was .05. No consistent pattern of significant differences between sections within study sites was found, although a few individual test statistics were marginally significant at site AL (see discussion below). In addition, the synchroneity of fire years in the two sections of each site was tested (chi-square test, 2 X 2 contingency table compared observed and expected values of fire occurrence at either or both sites). The null hypothesis of statistically independent fire chronologies between sections was rejected for all three test distributions at all sites. Although no spatial differences were observed at this scale of comparison, some smaller-scale spatial patterns of fire were detected at sites AV and AL (see below).

Seasonal patterns of fire occurrence (Table 3) show that 60-77% of fires burned in the spring and the remainder in the summer. At no site did late earlywood scars exceed 10% of the total and no latewood fires were observed. This seasonal distribution is similar to patterns observed in Arizona (Baisan and Swetnam, 1990) and New Mexico (Allen et al., 1995; Grissino-Mayer, 1995).

Site AV (Arroyo Verde): 47 years fire exclusion

The fire history at site AV was characterized by a disruption of the fire regime after 1945 (Fig. 3a). Although the first recorded fire occurred in 1764, the analysis period begins with 1801, the first fire year for which three recording trees exist in the data set. A pattern of frequent fires is seen from 1801 to 1945 (MFI for all scars = 3.79 years, Table 2), although an unusually long fire-free interval of 12 years occurred between 1929 and 1941. Since the 1941 fire was only recorded on a single sample, the effective fire-free period comprised the 16 years between 1929 and 1945. However, the 1945 fire burned across the entire site, scarring 63% of the recording trees. The site has not experienced a widespread burn scarring 25% or more of recording trees in the 47 years from 1945 to the sampling date, although fires scarring over 10% of the recording trees did burn in 1962 and 1986. A comparison of the sample identification numbers in the master fire chart (Fig. 3a) with the fire sample map (Fig. 2) shows that the scarred trees in 1962 and 1986 were confined to the northwest quadrant of the study site, an area where evidence of 2-3 clearings was encountered during sampling.[2]

[2] A previous comparison of fire regimes at the Arroyo Verde and Arroyo Laureles sites (Fulé and Covington, 1994) was based on fewer fire scar samples so fire interval results were slightly different.

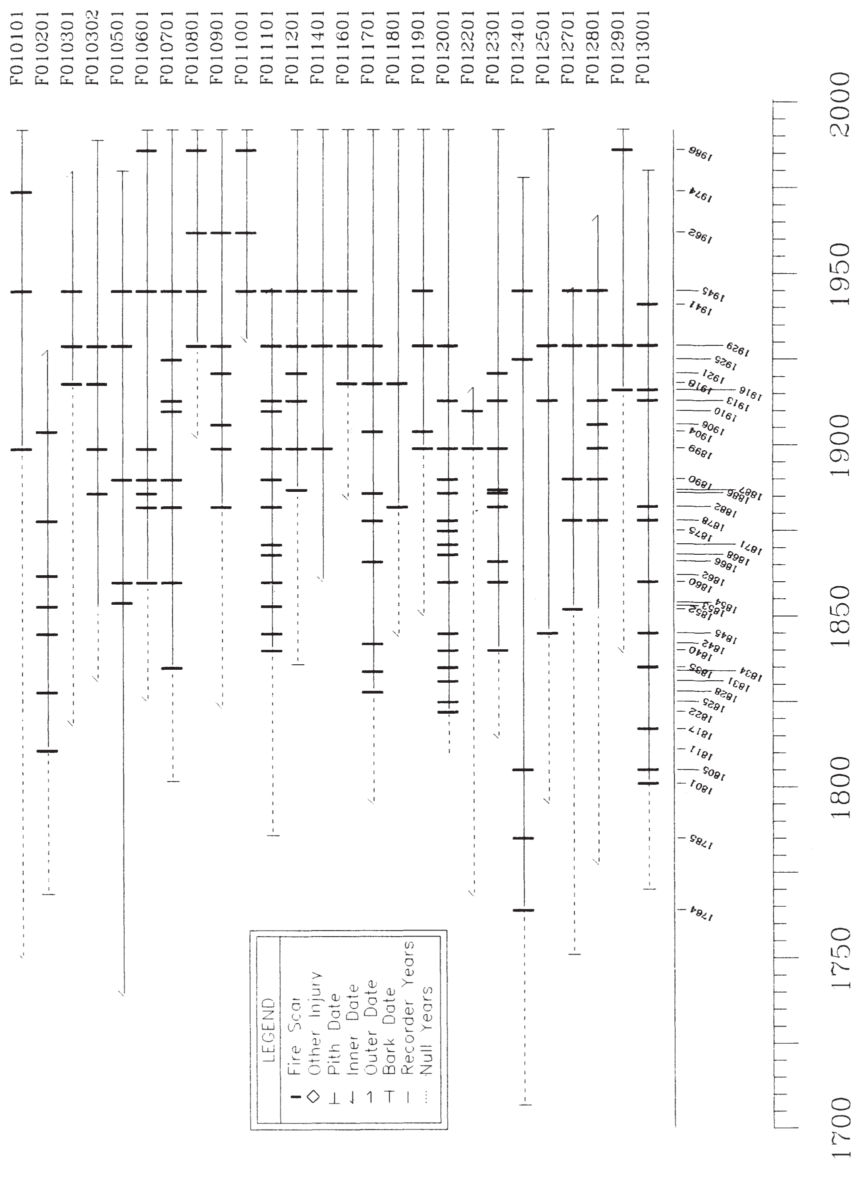


Fig. 3a. Composite master fire chart for Site AV (Arroyo Verde). The last widespread fire at this site occurred in 1945. Fires in 1962 and 1986 were limited to the north west corner of the study site.

Table 3. Seasonal distribution (number and percent) of fire scars at the four study sites based on the position of the fire injury within the scarred tree ring.

Site	AV	CB	SP	AL
Season Determined	102 (68%)	66 (62%)	166 (70%)	114 (78%)
Season Undetermined	48 (32%)	40 (38%)	68 (30%)	33 (22%)
Dormant	67 (66%)	28 (42%)	92 (55%)	48 (42%)
Early Earlywood	11 (11%)	15 (23%)	26 (16%)	20 (18%)
Middle Earlywood	20 (20%)	17 (26%)	47 (28%)	39 (34%)
Late Earlywood	4 (4%)	6 (9%)	1 (1%)	7 (6%)
Latewood	0	0	0	0
Dormant + Early (= spring fires)	78 (77%)	43 (65%)	118 (71%)	68 (60%)
Middle + Late (= summer fires)	24 (23%)	23 (35%)	48 (29%)	46 (40%)

Patterns of fire exclusion are commonly "messy": Swetnam and Baisan (1996) compared 63 fire histories from the southwestern United States which frequently evidenced a period of reduced fire frequency and sporadic post-disruption burns even after the regional trend of fire exclusion had become firmly established. The identification of a fire regime disruption at site AV is not unequivocal, as would be the case if no fire scars at all were found after 1945. The year 1929 could also be reasonably selected as a disruption date, because of the long fire-free interval which followed. However, the conclusion that the 1945 fire had important ecological impacts on the ecosystem, while the 1962 and 1986 fires did not impact most of the site, is supported by the forest structure data, particularly the age and size distributions described below.

Site CB (Cebadillas): mixed conifer site, 42 years fire exclusion

Site CB, the high-elevation site with *Abies* and *Pseudotsuga*, had clear evidence of recent fire regime disruption, with a fire-free period of 42 years to the present following the last widespread fire in 1951 (Fig. 3b). The only evidence of post-1951 fire is a single fire scar in 1974. As with site AV, there is also evidence of some earlier fire-regime disruption due to the gap of 22 years between the 1924 and 1946 fires, broken only by the 1934 fire

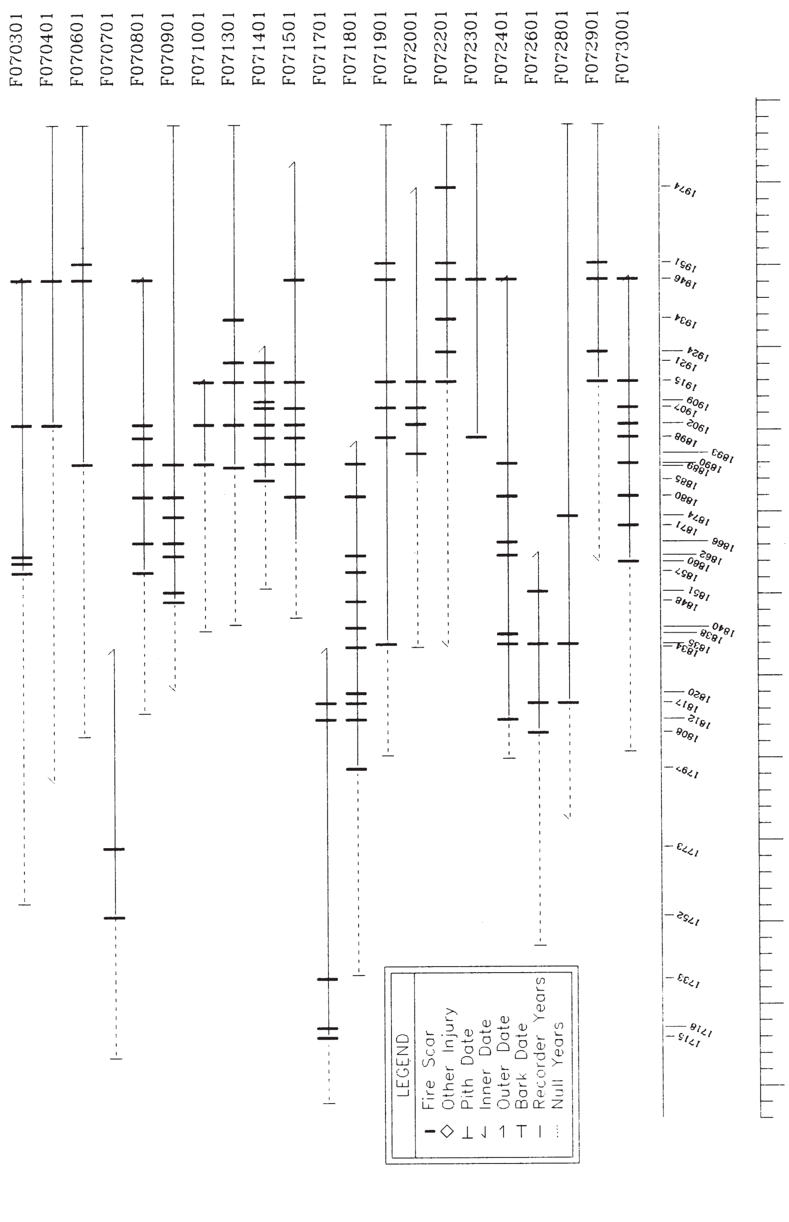


Fig. 3b. Composite master fire chart for Site CB (Cebadillas). Frequent fires were interrupted at this site after 1951. This forest area, supporting fir and Douglas-fir in addition to pine and oak, was selectively harvested for pines circa 1970.

which scarred 13% of the recording trees. Over the analysis period 1797-1951 (Table 2), the MFI for all fires (4.97 years) was about 60% less than the MFI for fires scarring 25% or more of the recording trees (8.56 years). These values are only slightly higher than the comparable pre-disruption fire return intervals at the pine-oak sites (Table 2), showing that the mixed-conifer site CB, on a high, relatively mesic, north-facing slope, burned at nearly the same frequency as the lower-elevation, relatively xeric sites on southwestern and western aspects. This result contrasts with the longer fire return intervals, averaging 52 years, for mixed conifer forests of the Sierra San Pedro Mártir in Baja California Norte (Minnich, 1993); however, this study was based on identification of fire perimeters from a time series (1942-1991) of aerial photographs, a method that may be less likely to identify frequent, low-intensity fires. In the southwestern United States, fire history studies based on fire scars in mixed conifer forests have shown pre-disruption (presettlement) MFI values for widespread fires (scarring 20% or more of recording trees) of 22 years (Dieterich, 1983), 11 years (Grissino-Mayer et al., 1995), and 9.9 years (Baisan and Swetnam, 1990). Grissino-Mayer et al. (1995) concluded that the fire regime at a mixed-conifer site in southern Arizona was very similar to that of lower-elevation ponderosa pine forests.

Site SP (Salsipuedes): fire exclusion and return

A pattern of recurring fire at site SP (1812-1955 MFI = 4.09 years, Table 2) was interrupted by a 29-year period with no widespread fires after 1955 (Fig. 3c). Single recording trees were scarred in 1963, 1980, and 1982, but widespread fire returned to site SP in 1984, scarring 95% of the recording trees. Nine years later, in 1993, another widespread fire burned the site. Evidence of several clearings, including one with a fence surrounding the planted area, was observed at site SP during sampling in 1994. Since site SP is the least accessible of the four study sites, it may have been favored in recent years for illicit cultivation, possibly causing the 1984 and 1993 fires.

Site AL (Arroyo Laureles): frequent fire up to the present

Site AL is the only site which has burned up to the present without a clear period of fire exclusion (Fig. 3d), providing one of the few examples of a continued frequent forest fire regime in North America (for comparison, see Dieterich, 1983; Swetnam and Baisan, 1996). Even at site AL, relatively long gaps of 14 years and 11 years appear between 1951-1965, and 1969-1980, respectively. But these gaps do not exceed the pre-disruption maximum fire-free intervals at the other three sites, suggesting that these periods remain within the range of natural variability of undisrupted fire regimes. However, although the 1980 and 1986 fires were widespread (Fig. 3d), they did not scar trees in approximately the northwestern third of the site. Several of the individual test statistics for spatial differences in fire regime at site AL showed marginal statistical significance (e.g., $p \approx .03$-$.04$), the only site where some statistical evidence of spatial heterogeneity was observed. If this portion of the study site did not burn between 1965 and 1991, the accumulation of fuel over 26 years may explain the high-intensity 1991 fire behavior evidenced by tree mortality adjacent to the northern site boundary (Fulé and Covington, 1994). Site AL was commercially logged in 1994.

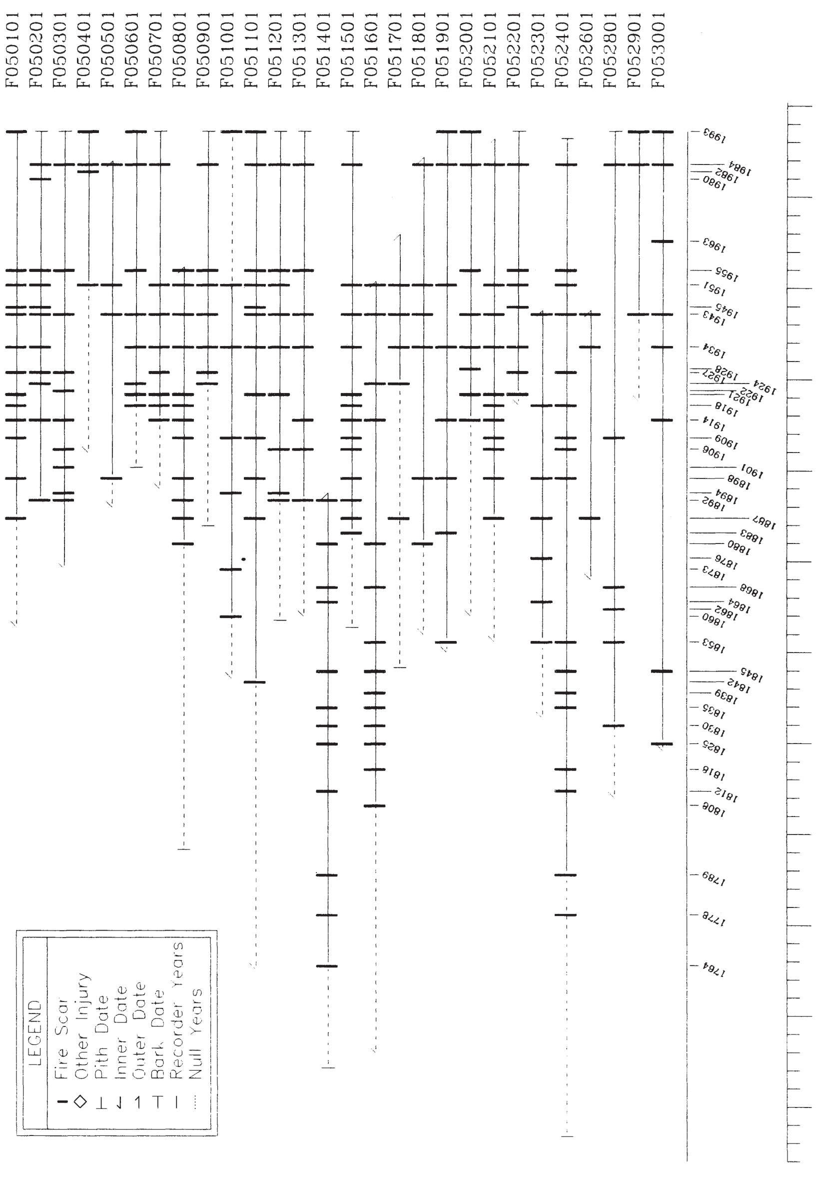


Fig. 3c. Composite master fire chart for Site SP (Salsipuedes). Frequent fires were interrupted at this site after 1955. After a fire-free period of 29 years, widespread fires returned in 1984 and 1993.

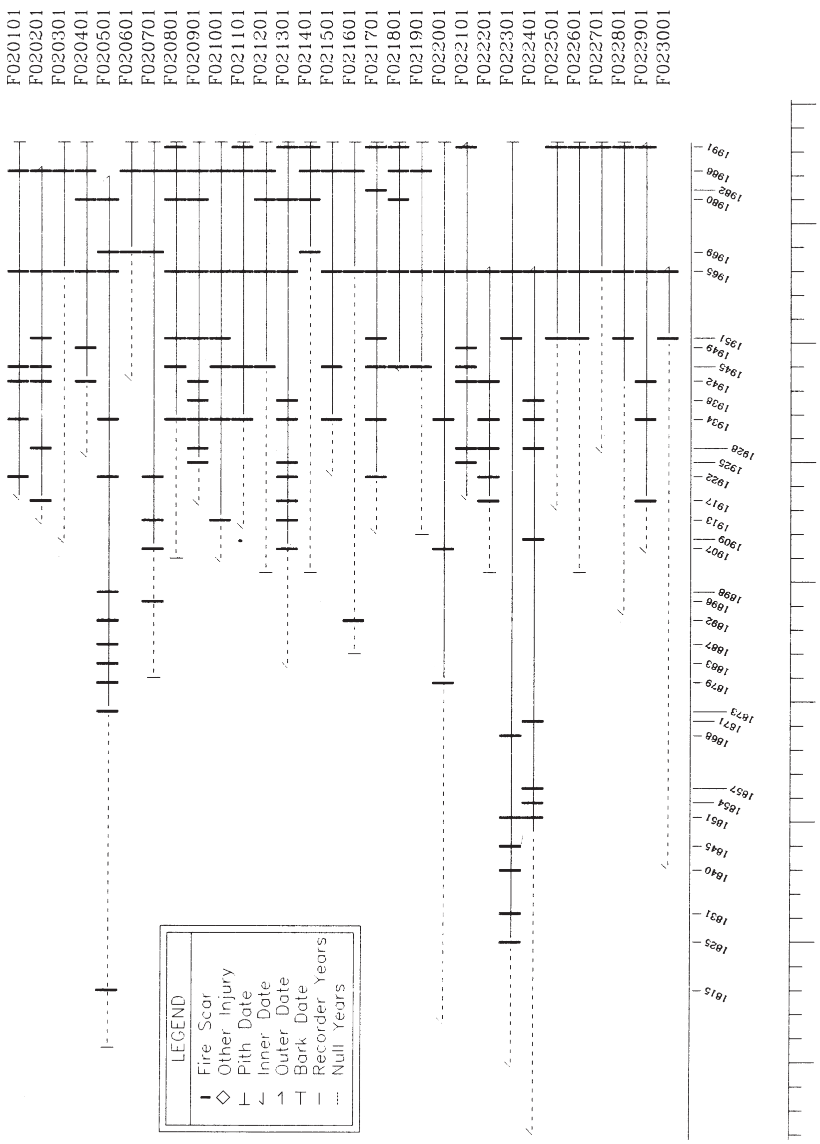


Fig. 3d. Composite master fire chart for Site AL (Arroyo Laureles). This site has continued to burn frequently up to the present, although there is some evidence of fire regime changes in the twentieth century, notably the 14-years gap from 1951 to 1965.

Site comparison

Fire histories at the four study sites are compared graphically in Fig. 4, with twentieth-century fire exclusion increasing in impact from the all-scar comparison (top), which shows a minimal level of recurring fire at most sites up to the present, through the 25%-scarred comparison (bottom), which shows the complete exclusion of widespread fire from sites AV and CB as well as the 29-year exclusion period at site SP. The period 1955-1980 was particularly free of widespread fire, except for the 1965 fire at site AL (Fig. 4). The regional fire interval analysis including all sites (Table 4) for the period 1797-1993 shows that the MFI for fire scarring at least one tree on at least one site was 2.11 years, rising to 15.40 years for fires scarring at least three of the four sites. The ten years in which three or all four of the sites burned are listed as major regional fire years, from 1835 to 1951, in Table 5.

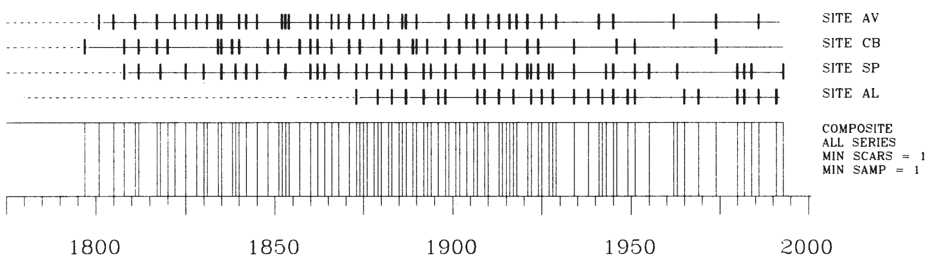
Table 4. Regional fire interval analysis comparing fire years over all four sites. The analysis period is 1797-1993.

	No. of Intervals	Mean (MFI)	Median	Standard Deviation	Minimum	Maximum	WMPI	100% Hazard
All fires	93	2.11	2.0	1.27	1	7	1.97	2
Two or more sites	42	4.67	4.0	3.78	1	23	4.13	16
Three or more sites	10	15.40	11.5	10.74	2	38	13.68	1337

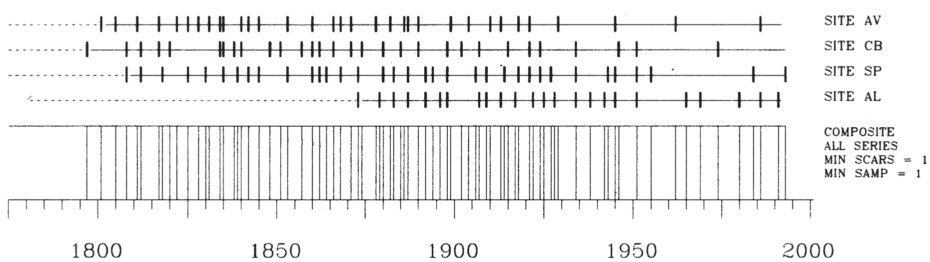
Table 5. Major regional fire years based on the percentage of fire occurrence at the four study sites.

Year	Number of Sites Scarred	Number of Recording Sites	Percent of Sites Burned	Fire Interval
1835	3	3	100%	.
1860 [A]	3	3	100%	25
1862 [A,B]	3	3	100%	2
1887 [A,B]	3	4	75%	25
1898	3	4	75%	11
1909 [A,B]	3	4	75%	11
1921 [A,B]	3	4	75%	12
1934 [A,B]	3	4	75%	13
1945	3	4	75%	11
1951 [A]	3	4	75%	6

[A] Positive Southern Oscillation Index (SOI) extreme reconstructed by Stahle and Cleaveland (1993).
[B] Positive SOI extreme reconstructed by both regression and classification methods (Table 1 in Stahle and Cleaveland, 1993).



Site Comparison: all fire years.



Site Comparison: fire years in which 10% or more of recording samples were scarred at each site.

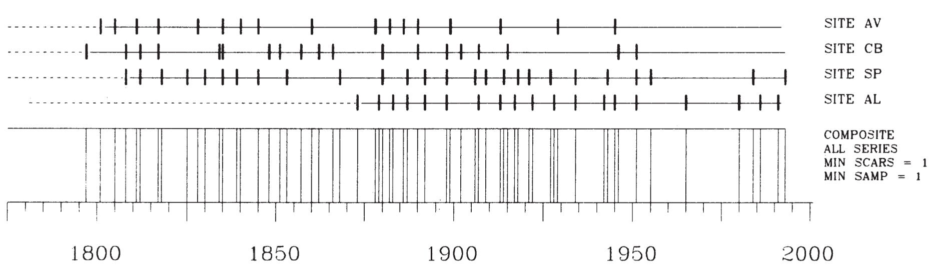


Fig. 4. Comparison of fire occurrence at all four study sites in three categories: (1) all fire years; (2) years in which 10% or more of recording samples were scarred; and (3) years in which 25% or more of recording samples were scarred at each site.

Forest Structure

Species composition

A variety of tree species was encountered on the sampling plots (Table 6). For purposes of analysis, trees were grouped as (1) pines (family Pinaceae, including *Abies* and *Pseudotsuga*), (2) oaks (family Fagaceae), and (3) others. Pines, oaks, and *Arbutus* (madrone) were found at all four study areas; other species occurred sporadically.

Table 6. Tree species encountered on the study sites. Nomenclature follows González et al. (1991).

Family	Species	Common Name
Betulaceae	*Alnus* spp.	Alder
Cupressaceae	*Juniperus depeanna* Steud.	Alligator juniper
	Juniperus spp.	Juniper
Ericaceae	*Arbutus* spp.	Madrone
Fagaceae	*Quercus coccolobifolia* Trel.	Encino roble
	Q. laeta Liebm.	Encino blanco
	Q. sideroxyla Humb. & Bonpl.	Encino prieto
	Quercus spp.	Encino
Pinaceae	*Abies durangensis* Mtz.	Durango fir
	Pinus arizonica Engelm.	Arizona pine
	P. ayacahuite K. Ehrenb.	Pino blanco
	P. durangensis Mtz.	Durango pine
	P. engelmannii Carr.	Apache pine
	P. herrerai Mtz.	
	P. leiophylla Schlecht. & Cham.	Chihuahua pine
	P. lumhotzii Rob. & Fern.	Pino triste
	P. teocote Schlecht. & Cham.	
	Pseudotsuga menziesii (Mirb.) Franco	Douglas-fir
Rosaceae	*Prunus serotina* Ehrh.	Cerezo
	Unknown (*Fraxinus velutina*?)	Fresno

Live tree structure

Live overstory trees differed significantly in density between sites in a pattern consistent with the thinning effects of fire: sites AV and CB, with extended fire exclusion, had the highest mean densities, up to 2733 trees/ha, while sites SP and AL, with recent fires, had much lower mean densities (Table 7). However, basal area was not correlated with density: the most dense site, AL, had the lowest basal area, while there was no statistically significant difference between basal areas at the other three sites (Table 7). Site AV was therefore dominated by numerous small trees (quadratic mean diameter 10.4 cm) while sites SP and AL are characterized by fewer, larger trees (quadratic mean diameters 29.0 cm and 27.1 cm respectively). Site CB falls in between (quadratic mean diameter 17.7 cm). The diameter distributions of live trees at the four sites (Fig. 5a-d) confirm these relationships, with the fire-excluded sites AV and CB having distinct reverse-J shaped distribution curves. Diameters at sites SP and AL, though still positively-skewed, appeared more normally distributed. All three analysis groups (pines, oaks, and other) generally shared the same relative patterns of diameter distribution at each site (Fig. 5a-d).

Table 7. Forest structural characteristics (basal area and density) of living overstory trees and regeneneration density at the four study sites. Within-row means not sharing a letter are significantly different ($p < .05$). $N = 30$ at all sites.

	Site AV		Site CB		Site SP		Site AL	
	Mean	S.E.M.	Mean	S.E.M.	Mean	S.E.M.	Mean	S.E.M.
Live Basal Area				m^2/ha				
Pine	12.9 a	1.2	14.2 ac	1.8	24.8 b	3.3	22.6 bc	3.1
Oak	8.9 a	0.9	18.1 b	3.3	8.9 a	1.3	13.5 ab	1.9
Other	1.6 ab	0.4	5.7 a	2.1	3.1 ab	0.9	1.1 b	0.4
Total	23.4 a	1.6	38.0 b	3.6	36.8 b	3.6	37.2 b	2.9
Live Tree Density				number/ha				
Pine	1499 a	235	842 b	96	288 c	40	275 c	41
Oak	980 a	107	373 b	61	167 b	26	287 b	47
Other	253 a	38	327 a	54	103 b	24	85 b	29
Total	2733 a	264	1541 b	146	558 c	67	647 c	68
Regeneration Density				number/ha				
Pine	1525 a	695	1041 a	254	575 a	134	3542 b	940
Oak	3675 a	618	7083 a	1205	7442 a	1348	4200 a	1119
Other	675 a	141	1358 a	257	3975 b	929	983 a	293
Total	5875 a	695	9483 ab	1247	11992 b	1897	8725 ab	1333

Rates of lightning scarring of live overstory trees were below 3% at all sites, even considering only trees larger than 20 cm dbh. No lightning-scarred trees were encountered at site CB.

Age structure of conifers over 6 cm dbh at the four sites (Fig. 6a-d) shows the relatively young age of the trees. The oldest individual (a *Pinus durangensis*) dated to the mid-1600's and the great majority of trees were less than 100 years old. Some effects of fire and fire exclusion are reflected in the age distributions, but the patterns are mixed. For example, the overwhelming majority of conifers at site AV (Fig. 5a) established after the

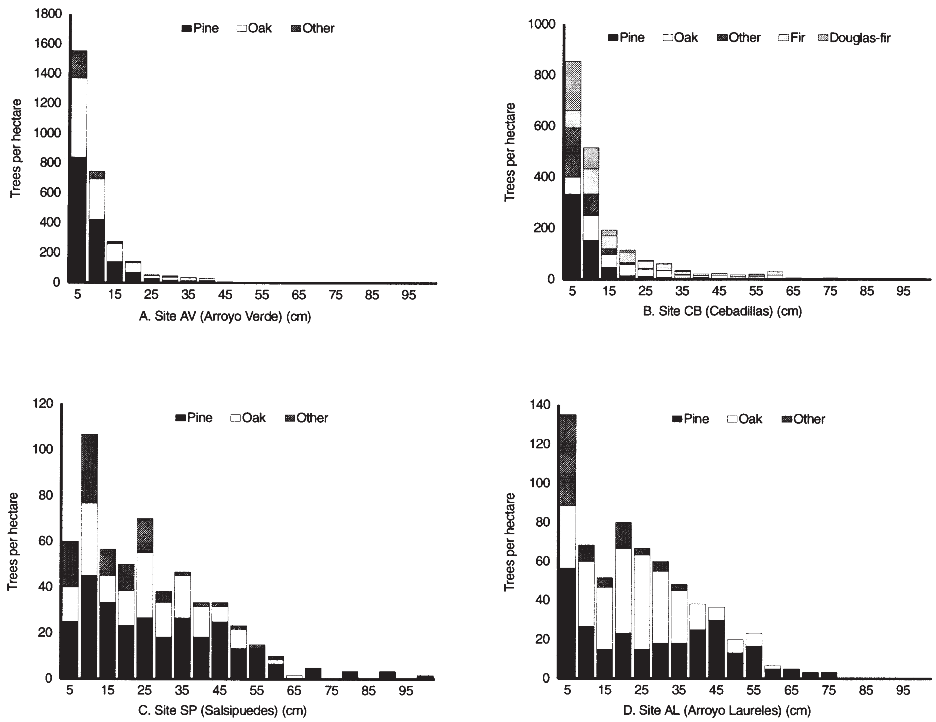


Fig. 5. Live overstory tree diameter distribution at sites (a) AV (b) CB (c) SP and (d) AL. X-axis shows endpoints of 5-cm dbh classes (e.g., 15 = 10.1 - 15 cm).

last widespread fire in 1945, but most conifers at site CB (Fig. 5b) were already well-established before fire exclusion began following the 1951 fire. The gap in the age distribution at site AL (Fig. 5d) in the mid-twentieth century may be an effect of the widespread 1965 fire killing trees in the younger age classes, but no analogous gap appears in the age distribution at site SP (Fig. 5c) after the 1984 fire. The age distributions appear more heterogeneous at the burned sites because the proportions of younger to older trees are much higher at the unburned sites. All four sites were actually similar in terms of the densities of trees older than 50 years, approximately corresponding to the period of fire exclusion, ranging from 63 trees/ha at CB to 97 trees/ha at AL (densities of conifers from which increment cores could be sampled). By 1993, however, the tree densities increased substantially at the unburned sites, altering the forest age structure toward greater dominance by younger trees. The ratio of trees under 50 years to trees over 50 years was 2.7 at AV and 1.9 at CB. The ratio of young:old trees at the burned sites, in contrast, was 0.2 at AL and 0.6 at SP.

High regeneration density at all sites (several thousand/ha), compared with the low overstory density at the burned sites SP and AL (Table 7), is consistent with the thinning effect of recent fires which appear to have reduced the density of trees surviving to the overstory. At the fire-excluded sites AV and CB, dense young trees have been able to grow into the overstory without thinning from fire. Although all tree seedlings below 1.3 m were tallied together in this study, qualitative differences were observed between the sites. Regeneration at the fire-excluded sites AV and CB consisted mainly of dense seedlings and sprouts between 0.5 and 1.3 m, while regeneration at sites SP (last fire 1993) and AL (last fire 1991) typically consisted of very small, young seedlings and sprouts. Numerous pine seedlings at site AL were still shedding seed coats at the time of sampling in August, 1993.

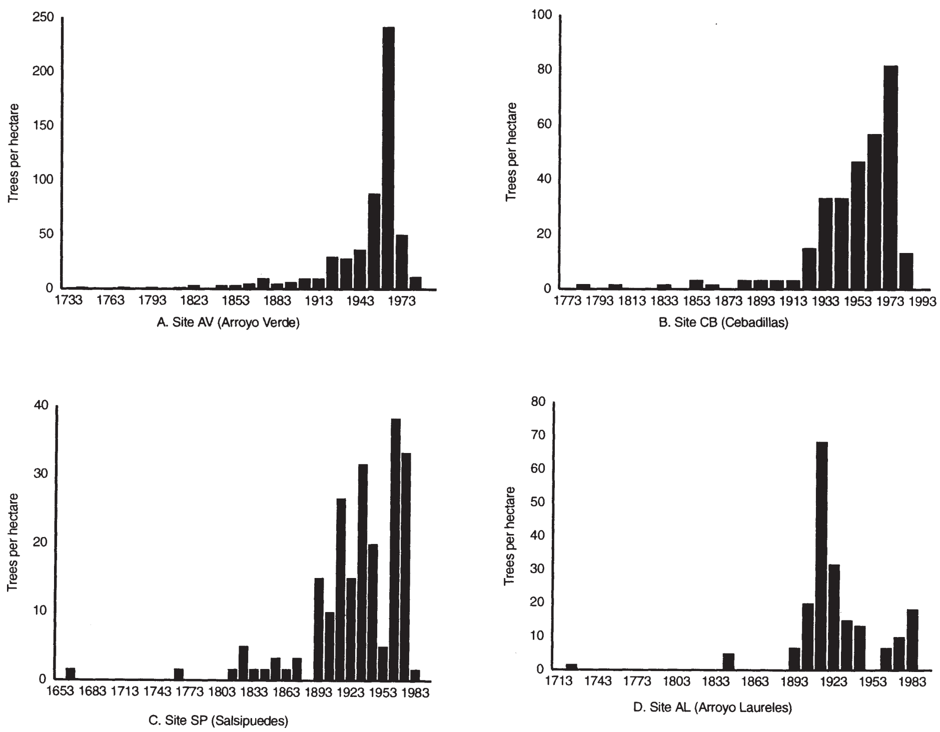


Fig. 6. Center date distribution of conifers over 6 cm dbh at sites (a) AV (b) CB (c) SP and (d) AL. X-axis shows endpoints of 10-year age classes (e.g., 1803 = 1794 - 1803).

Dead tree structure

Dead tree structure (snags, dead and downed trees, and stumps) is summarized in Table 8. Density of recent snags, those with branches and bark still intact, was significantly highest at site SP, where it was evident in the field that the return of fires in 1984 and 1993 after a 29-year fire-free period killed numerous trees. In some parts of site SP, mature oaks or alders appeared to have been killed in 1984. The subsequent thickets of sprouts around the snags were then killed or thinned by the 1993 fire. Older snags were more dense at the fire-excluded sites, reflecting the combustion of older dead material by recent fires at sites SP and AL. Stump density was generally low since three of the four sites had never been harvested on a large scale. Pines at the fourth site, CB, had been selectively harvested at a mean density of 13.3 trees/ha (Table 8). Although the majority of dead trees at all sites was in smaller diameter classes (Fig. 7a-d), the densities of the smallest snags were sharply higher at the recently burned sites SP and AL, consistent with the thinning effect of fire.

Table 8. Forest structural characteristics (basal area and density) of snags, dead/down trees, and cut stumps at the four study sites. Within-row means not sharing a letter are significantly different ($p < .05$). N = 30 at all sites. "Recent" snags have intact bark and branches (condition class 3); "older" snags are in more advanced conditions of decay (condition classes 4 through 7).

	Site AV		Site CB		Site SP		Site AL	
	Mean	S.E.M.	Mean	S.E.M.	Mean	S.E.M.	Mean	S.E.M.
Recent Snag Basal Area			m^2/ha					
Pine	1.0 a	0.3	0.03 a	0.01	1.0 a	0.3	1.2 a	0.4
Oak	0.2 a	0.2	0.05 a	0.03	0.4 a	0.2	0.1 a	0.1
Other	0.01 a	0.01	0.02 a	0.02	0.5 b	0.2	0.2 ab	0.1
Total	1.2 ab	0.4	0.1 a	0.04	1.9 b	0.5	1.5 ab	0.5
Recent Snag Density			number/ha					
Pine	105.4 a	30.3	20.0 b	7.4	68.3 ab	17.9	73.3 ab	18.1
Oak	8.3 a	4.2	10.0 a	5.0	151.7 a	79.1	31.7 a	11.9
Other	6.7 a	3.2	3.3 a	3.3	136.7 b	51.5	30.0 a	15.6
Total	120.4 a	30.0	33.3 a	10.5	356.7 b	99.6	135.0 a	30.5
Older Snag Basal Area[A]			m^2/ha					
Pine	1.5	0.5	2.0	1.1	1.2	0.6	0.8	0.6
Oak	0.2	0.1	0.4	0.3	0.3	0.3	0.01	0.004
Other	0.01	0.01	0.5	0.5	0.1	0.1	0.2	0.2
Total	1.7	0.5	2.9	1.5	1.6	0.9	1.0	0.6
Older Snag Density			number/ha					
Pine	45.8 a	10.2	11.7 b	5.7	15.0 b	4.3	11.7 b	5.2
Oak	6.7 a	4.0	11.7 a	5.2	1.7 a	1.7	3.3 a	2.3
Other	1.7 a	1.7	3.2 a	2.3	1.7 a	1.7	1.7 a	1.7
Total	54.2 a	11.9	26.7 ab	10.6	18.3 b	5.1	16.7 b	6.0

Table 8. Continuation.

	Site AV		Site CB		Site SP		Site AL	
	Mean	S.E.M.	Mean	S.E.M.	Mean	S.E.M.	Mean	S.E.M.
Dead/Down Basal Area[A]			m²/ha					
Pine	2.4	1.0	1.0	0.7	1.3	0.7	1.8	0.7
Oak	0.04	0.04	0.4	0.3	0.3	0.3	0.3	0.2
Other	0.04	0.04	0	0	0	0	0.5	0.3
Total	2.5	1.0	1.4	0.7	1.6	0.7	2.6	0.8
Dead/Down Density			number/ha					
Pine	68.3 a	25.1	6.7 b	3.2	10.0 b	4.4	31.7 ab	10.6
Oak	3.3 a	2.3	3.3 a	2.3	1.7 a	1.7	11.7 a	5.2
Other[B]	0.8	0.8	0	0	0	0	11.7	4.6
Total	72.5 a	25.4	10.0 b	3.7	11.7 b	4.6	55.0 ab	13.4
Stump Basal Area[A]			m²/ha					
Pine	0.02	0.02	3.0	1.7	0	0	1.2	1.2
Oak	0.05	0.05	0	0	0	0	0	0
Other	0	0	0.2	0.2	0	0	0	0
Total	0.06	0.06	3.2	1.7	0	0	1.2	1.2
Stump Density[A]			number/ha					
Pine	1.7	1.7	13.3	5.8	0	0	8.3	6.8
Oak	1.7	1.7	0	0	0	0	0	0
Other	0	0	1.7	1.7	0	0	0	0
Total	3.3	3.3	15.0	6.4	0	0	8.3	6.8

[A] No significant difference found by manova between sites.
[B] One or more groups has no variance.

Dead woody biomass and herbaceous cover

Dead woody biomass ranged from a low of 7.1 metric tons/ha at site SP to a high of 53.2 metric tons/ha at site CB (Table 9). The fire-excluded sites had higher dead woody biomass loads than the recently burned sites, although only the very high loading at site CB was statistically significantly different. The biggest difference was in rotten woody biomass, where the fire-excluded sites had loadings several times those of the recently burned sites. Litter depth was similar at all sites, but duff depth was significantly lower at the recently burned sites. These characteristics, particularly the rotten woody loading and duff depth, match the expected effects of fire in consuming these fuels. The comparable levels of forest litter suggest that burned and unburned sites both have continuous fine fuels which can support fire, and that even recently burned sites (site SP burned 1 year before sampling, site AL 2 years before sampling) can quickly recover fine fuel loadings similar to those of unburned sites. The heavy fuel loadings at the unburned sites, particularly of flammable rotten fuels, indicate that the fire-excluded sites have accumulated fuel beyond the range normally found in burned forests, fuels which can support high-intensity fire. Herbaceous cover was relatively low at all four sites, ranging from 10 to 20%, but tended to be higher at the recently burned sites, as expected, given their lower overstory densities.

Table 9. Dead woody biomass, forest floor, and herbaceous cover at the four study sites. Withinrow means not sharing a letter are significantly different ($p < .05$). N = 30 at all sites.

	Site AV		Site CB		Site SP		Site AL	
	Mean	S.E.M.	Mean	S.E.M.	Mean	S.E.M.	Mean	S.E.M.
Dead woody biomass			metric ton/ha					
00.6 cm diameter	0.1 a	0.02	0.3 a	0.08	0.3 a	0.04	0.1 a	0.02
0.62.5 cm diam.	0.6 a	0.1	0.9 a	0.1	0.7 a	0.08	0.5 a	0.1
2.57.6 cm diam.	2.9 a	0.5	3.8 a	0.8	1.8 a	0.4	3.0 a	0.8
>7.6 cm diam. sound	0.5 a	0.3	18.0 b	8.1	3.6 ab	2.0	4.3 ab	1.5
>7.6 cm diam. rotten	11.7 a	2.3	30.3 b	9.2	0.9 a	0.3	2.6 a	1.1
Total woody biomass	15.8 a	2.6	53.2 b	11.8	7.1 a	2.0	10.6 a	2.3
Forest floor depth			cm					
Litter	2.1 a	0.1	2.2 a	0.1	2.0 a	0.2	2.2 a	0.2
Duff	2.2 a	0.2	2.7 a	0.3	0.9 b	0.1	1.1 b	0.1
Herbaceous cover			percent					
	16.5 ab	1.78	10.6 a	1.6	19.7 b	2.1	20.0 b	3.0

DISCUSSION

Patterns of Fire Occurrence

The high frequency of fire in the Sierra Madre Occidental indicates that sources of ignition, weather patterns favorable to fire spread, and adequate contiguous fuels exist across the region. Evidence of both lightning and human-caused ignition (clearings) was observed directly on the study sites, except for site CB. The rates of these sources of direct ignition were apparently relatively low compared to the high frequency of fire (less than 3% lightning scarring of trees; one to several clearings per site), but the general absence of natural or artificial firebreaks around the study sites would permit fires to enter from outside the sites and cross large areas without impediment. The perennial streams forming the downslope boundaries of sites AV and AL were narrow enough (5-10 m) for fires to spot across readily under windy conditions. Roads came close to sites AV, CB, and AL, but they are of recent construction (post-1970) and are also relatively narrow (≈ 3-4 m) and often covered with fine fuels consisting of leaves and needles from the surrounding forest.

Foresters and landowners in the study area believe that most fires are human-caused, resulting from burning of pastures and agricultural fields, clearings, campfires, and smoking. This is a widely-held view in Mexico. Both official statistics (González-Cabán and Sandberg, 1989; Anonymous, 1994) as well as some botanical and ecological studies of Mexican forests (e.g., Loock, 1950; Perry, 1991; Fisher et al., 1995) reflect this attitude, grounded in documented cases of deforestation caused by repeated agricultural burning, primarily in the heavily-populated central Mexican states (Martínez, 1948; Perry, 1991). Prior to Spanish colonization, indigenous inhabitants of the study area, including Acaxee, Xixime, and Tepehuán peoples (Pennington, 1969; Gerhard, 1982), also used fire. Ethnographic studies of the historic Tepehuán (who no longer inhabit the study area) and Rarámuri (Tarahumara) people, who live in similar forest and canyon ecosystems within 100 km of the study area, describe fire use for clearing agricultural fields, hunting, and warfare (Lumholtz, 1902; Pennington, 1963, 1969). Lightning is broadly considered a minor source of ignition both at the national level (Anonymous, 1994) and in the study area (J.G. Paredes Pérez, Forester, personal communication, 1993) because it is generally accompanied by heavy rain and high humidity.

However, lightning is the predominant ignition source in long-needled pine ecosystems of the southwestern U.S. (Swetnam and Betancourt 1990, Swetnam and Baisan 1996) and the role of natural ignitions is probably very significant in maintaining frequent fire regimes in the central Sierra Madre Occidental as well. Lightning ignitions in the study area can smolder until favorable burning conditions return (P.Z. Fulé, personal observation, 1994). Furthermore, the human population of the study area has changed substantially since Spanish colonization, with the indigenous population in the region of the study area falling from an estimated 10,000 in 1,500 A.D. to 500 by 1,700 A.D. (Gerhard, 1982). Although the Spanish city of Santiago Papasquiaro was founded in 1593 A.D. and mining exploration in the Sierra Madre began shortly afterward, Hispanic settlement across much of the central Sierra Madre Occidental was limited until the expansion of *ejido*s after the Second World War. Considering the relatively stable pattern of pre-disruption fires found at the four study sites in this study from approximately 1800 through the 1920's to 1950's, despite low and fluctuating human populations, it seems likely that lightning has been an important ignition source. When longer fire chronologies are developed from the Sierra Madre, more detailed comparisons of changes in fire frequency and human presence can be carried out.

While recent human activities are probably related to the patterns of fire exclusion at the four study sites, specific reasons for the differing patterns of fire exclusion are not known. Fire exclusion at sites AV and CB, and the 29-year fire exclusion period at site SP, may be due to changes in grazing, roads, or agricultural practices and may be associated with the establishment of ejidos. At least some of the recent fires at sites AV, AL, and SP, however, appear to be due to the deliberate burning of clearings.

Climatic factors appear to influence fire regimes in the study area in a manner similar to climate-fire relations in the southwestern United States. Seven of the ten major fire years in Table 5 correspond to years in which positive winter Southern Oscillation index (SOI) extremes were reconstructed from dendrochronological data in the Sierra Madre Occidental by Stahle and Cleaveland (1993). The positive SOI extremes are associated with cold/dry weather conditions, unfavorable for tree growth but possibly conducive to fire. None of the major fire years corresponded to years with negative (warm/wet) SOI extremes (Stahle and Cleaveland, 1993). The major fire years do not match the six major regional fire years identified by Swetnam and Baisan (1996) in the southwestern United States during the overlapping period of their analysis, 1835-1879, possibly reflecting different patterns of climate and ignition as well as the small sample size (4 sites) of the present study. But the results are generally in agreement with the findings of Swetnam and Betancourt (1990), who showed that area burned in the southwestern United States was greatest in years with high positive SOI values and least in low SOI (El Niño) years. The spring-summer fire season (Table 3) is also similar to southwestern patterns (Swetnam and Baisan, 1996). As additional fire history data are collected in northern Mexico, a more detailed analysis of the climate-fire relationship should be developed to increase understanding of past and present ecological conditions and to improve prediction of short-term and long-term changes in fire frequency and intensity.

Fire and Forest Structure

Forest ecosystem structures appeared to be closely linked to fire patterns at the four study sites. The descriptive comparison among different sites forming a chronosequence of fire exclusion in this study cannot statistically separate fire effects from the variety of other factors influencing forest structure and dynamics. In particular, the mesic site characteristics of site CB (high elevation, north aspect) are distinctly different from those of the drier pine-oak sites, influencing at least the species composition, and probably also affecting tree density and dead woody biomass. However, several broad trends related to recent fire regimes are clear across all sites. First is the trend from relatively open forests of large trees at the recently burned sites AL and SP to the relatively dense forests of small trees at the fire-excluded sites AV and CB. Other differences, discussed above, include regeneration (older understory trees at unburned sites, seedlings at burned sites), age and diameter distributions (dense young trees at unburned sites), dead tree structure (more dead trees, especially small-diameter trees, at burned sites), dead woody biomass loadings (higher fuel loading at burned sites), and herbaceous cover (higher at burned sites). These trends are consistent with the effects of frequent fire in thinning small, young trees and maintaining low levels of woody fuels and duff.

Differences between the recently burned and unburned study sites follow patterns similar to the changes observed in ponderosa pine forest structure of the southwestern U.S. following fire exclusion. Ponderosa pine density at a number of sites in Arizona and New Mexico ranged from 7 to 116 trees/ha prior to Euro-American settlement (summary of reconstructive studies and early forest inventories cited in Covington and Moore [1994b]). Due to the exclusion of fire and reduced herbaceous competition, modern tree densities are much higher. The average ponderosa pine density in Arizona in 1985 was 776 trees/ha (Garrett et al., 1990), but an unmanaged ponderosa "natural area" increased far more, from 61 trees/ha in 1876 to 3,098 trees/ha in 1992 (Covington et al., 1997). The probable changes in forest density due to fire exclusion inferred from the present study have a smaller range, with the two recently burned sites (AL and SP) averaging 602 trees/ha and the two fire-excluded sites (AV and CB) averaging 2,137 trees/ha, about 3.5 times higher. But the direction of change in density and other associated structural attributes, such as increased fuel loading, parallel those of the dense U.S. ecosystems which now support high-intensity, stand-replacing wildfires over thousands of hectares (Covington et al., 1994).

Fire exclusion impacts have received relatively little attention in Mexico. Although beneficial aspects of fire as a silvicultural tool and as a natural ecological disturbance factor have been recognized within the scientific community (Sánchez and Dieterich, 1983; Rodríguez and Sierra, 1992; Alanís-Morales, 1996), public perceptions have been influenced by anti-fire publicity (Rodríguez and Sierra, 1992). The present situation in northern Mexico may be analogous to that of the western U.S. about 50 years ago. Several observers had already noted deleterious effects of fire exclusion (e.g., Leopold 1924, Weaver 1943), but these reports were often met with skepticism by foresters. For example, when Harold Weaver (1951) called for increased application of prescribed fire, the Journal of Forestry found the topic sufficiently controversial to solicit a rebuttal. In recent decades, however, the attention of land managers and the public in the U.S. has been focused on the hazards of fire exclusion as a result of increasingly large, destructive, and costly wildfires in fire-excluded ecosystems (Swetnam, 1990; Covington et al., 1994).

By contrast, in Mexico large high-intensity fires are still rare, perhaps as a consequence of low fuels from frequent burning, relatively short periods of fire exclusion, and a mosaic of forest patches burned at different times, analogous to the chaparral patterns in Baja California Norte (Minnich, 1983). A recent government report on national wildfire status (Anonymous, 1994) described Mexican forest fires as affecting the herbaceous vegetation but only rarely the tree crowns.[3] Because of the relatively fewer firefighting resources available in Mexico, compared with the high historic investment in fire suppression in the U.S., fire exclusion periods extending to 100 or more years may not become common in the Sierra Madre Occidental. But wildfires are likely to become larger and more severe as fire exclusion periods lengthen and fuels accumulate. The high overstory densities and heavy fuel loading observed at the fire-excluded sites in this study are likely to support high-intensity fire with extensive tree mortality. Taking site SP as an example of the return of fire to a fire-excluded ecosystem, if the densities of recently-killed trees and living trees are summed as a rough approximation of forest density after 29 years of fire exclusion (total 914 trees/ha), the 1984 and 1993 fires killed up to 40% of the trees at the site. Despite this tree mortality, the ecosystem at site SP remains a pine-oak forest. The outcome of the next wildfire at sites AV and CB may be different, as these fire-excluded sites have overstory densities 150% to 300% that of site SP, as well as very high woody fuel loading, so high-intensity fire could lead to heavy mortality or deforestation. The hazard is augmented through fuel structures such as flammable rotten wood, deep forest floors which burn slowly with extended periods of lethal temperatures, and fuel ladders composed of smaller and mid-sized trees that carry fire into the overstory crowns.

Further fire ecology research and increased application of innovative fire management practices such as prescribed burning will be invaluable in helping to reconcile the paradoxical nature of fire, seen both as an essential ecological process and as the destroyer of the forest. The contradictory nature of these perspectives on fire is evident in the Sierra Madre Occidental. As populations rise and as the infrastructure needed for timber exploitation is developed, landowners and foresters have become increasingly aware of timber values and have initiated extensive campaigns to prevent, detect, and suppress forest fires. *Ejido* members at the studied areas are proud of their contributions to fire prevention and suppression crews (P.Z. Fulé, personal observation, 1994). However, the regime of frequent,

[3] "En México los incendios forestales son de tipo superficial; es decir, afectan vegetación herbácea y rara vez la copa de los árboles."

low-intensity fire which has prevailed till recently in these forests is a key ecological process contributing to the present health and low destructive potential of fuels in many of these forests today (Leopold, 1937, Marshall, 1962, Fulé and Covington, 1994, Baisan and Swetnam, 1995). Another example of a problematic fire paradigm is expressed in a recent study which concluded that these northern states are becoming deforested because the area reforested (i.e., planted) is a tiny fraction of the area burned (Fisher et al., 1995); this deforestation argument is based on the unsubstantiated assumption of complete tree mortality from fire. While the authors appear to be motivated by a wish to preserve these unique ecosystems, the present study and numerous others suggest that the unintended consequence of a fire exclusion approach to preservation is likely to be the replacement of frequent low-intensity fires with infrequent, high-intensity, stand-killing fires. We recommend further research in the ecology of fire and the potential for application of prescribed fire as a tool for management and conservation of these unique and diverse ecosystems.

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