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Fire regimes and forest structure in a sky island mixed conifer forest, Guadalupe Mountains National Park, Texas, USA

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Abstract

Fire is a key disturbance agent in the fire-prone mixed conifer and ponderosa pine forests of the southwestern United States. Human activities (i.e., livestock grazing, logging, and fire suppression) have resulted in the exclusion of fire from these forests for the past century and fire exclusion has caused changes in forest structure and composition. This study quantifies spatial and temporal variability in fire regimes and forest change in a 1000-ha area of mixed conifer forest in Guadalupe Mountains National Park (GMNP), an area with an uncommon history of grazing and fire suppression. Dendroecological methods were used to quantify fire frequency, season, severity, and extent, as well as forest structural and compositional change. The mean composite fire return interval (CFI) for the study area was 4 years. Widespread fires were less frequent. The mean CFI for fires recorded in at least 10% of the samples collected was 9.2 years, and mean CFI for fires scarring at least 25% of samples was 16.3 years. Many of these widespread fires occurred in the 19th century. The mean point fire return interval (PFI) was longer at 24 years. Fire scars were primarily formed in the earliest portion of earlywood in annual rings, indicating that fires burned mainly in the spring, at the beginning of the growing season. The onset of grazing in the 1920s dramatically reduced fire frequency. An increase in tree density and a compositional shift from southwestern white pine (*Pinus strobiformis* Engelm.) to Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) coincides with the grazing era. In addition, the pre-ranching era was characterized by low-severity fires, while structural changes have resulted in a contemporary forest that is prone to high severity fire, as evidenced by two stand-replacing wildfires in GMNP in the 1990s.

Keywords: Fire regimes; Forest dynamics; Grazing; Mixed conifer forest; Sky islands; Guadalupe mountains; Southwest USA

1. Introduction

Fire is a key disturbance process that shapes the structure and composition of forest stands and forested landscapes in the American southwest (Cooper, 1960; Swetnam and Baisan, 1996). Spatial and temporal variation in fire regime characteristics are thought to contribute to maintenance of species and structural diversity in fire-prone ecosystems because species response to fire is strongly influenced by the season, frequency, severity, and extent of fire (Martin and Sapsis, 1992; Bond and van Wilgen, 1996; Barton, 1999; Fulé et al., 2003). In the southwest, the role of fire in forest ecosystem dynamics has changed since the late 19th century. Grazing, which reduces fine fuels that carry fire, and a Federal Policy of suppressing fire on public land since 1905 (Pyne, 1982), have greatly reduced the frequency and extent of fire, especially in low elevation ponderosa pine (*Pinus ponderosa* var. *scopulorum* Engelm.) forests (Baisan and Swetnam, 1990; Fulé et al., 1997). Fire exclusion in southwestern ponderosa pine forests has caused dramatic changes in forest structure and composition. Forests are now denser because seedlings and saplings are not thinned by low-intensity surface fires (Fulé et al., 1997, 2003). Also, stand-replacing fires have become more frequent and widespread in ponderosa pine forests because of these changes (Allen et al., 2002). Fire exclusion has also caused changes in forest conditions in higher elevation mixed conifer forests in some areas (e.g., Mast and Wolf, 2004), but not others (e.g., Cocke et al., 2005).

In the mountainous terrain occupied by mixed conifer forests, topography is a potentially important control on spatial variation in fire regimes. Topography (i.e., slope pitch, slope aspect, slope position, elevation) directly and indirectly influences fire behavior and fire regimes by affecting the nature and structure of fuels and the location of barriers to fire spread (Heyerdahl

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et al., 2001; Taylor, 2000; Taylor and Skinner, 2003; Cocke et al., 2005). For example, in mixed conifer forests in California (Beaty and Taylor, 2001) and Oregon (Heyerdahl et al., 2001), fire return intervals were shorter on south-facing rather than north-facing slopes because the period in which fuels are dry enough to carry fire each year is longer on south-facing slopes. Fire frequency also declines with elevation and with physiographic heterogeneity in mixed conifer forests in California (Taylor, 2000; Beaty and Taylor, 2001; Bekker and Taylor, 2001) and the southwest (Wolf and Mast, 1998; Brown et al., 2001). At higher elevation, short-needled species (i.e., white fir (*Abies concolor* [Gord. and Glend.]), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco)) are proportionately more abundant than long-needled pines (i.e., ponderosa pine, southwestern white pine [*Pinus strobiformis* Engelm.]). Dense litter beds of short-needled species slow fire spread (Albini, 1976; Rothermel, 1983) and fire frequency decreases over short distances when there is a change from long to short needled fuel beds (Stephens, 2001).

Fire frequency and extent in the southwest are also influenced by livestock grazing. Grazing reduces the abundance of grass fuels and fuel connectivity across the landscape, impeding fire spread. Consequently, the introduction of livestock in the 19th century is thought to be responsible for a region-wide reduction in the frequency and extent of fire in ponderosa pine and mixed conifers forests (Cooper, 1960; Madany and West, 1983). Moreover, the impact of grazing in the interior west may not be fully recognized because of the more visible effects of logging and fire suppression (Belsky and Blumenthal, 1997). In many parts of the southwest, the introduction of livestock was coincident with a period of extensive logging (Merriam, 1890; Bahre, 1991) and the establishment of fire suppression. Research in geographic locations with a unique land-use history provides an opportunity to evaluate the effects of individual land-use practices on change in forest ecosystem structure and dynamics (Swetnam et al., 1999).

The southern Guadalupe Mountains of west Texas have a different land use history than most areas in the Southwest. Livestock grazing did not occur in these mixed conifer forests until the mid 1920s, much later than other parts of the southwest. Rugged terrain and lack of water precluded livestock grazing until roads and a water system were constructed. Lands in the southern Guadalupe Mountains were also privately owned until 1972 when Guadalupe Mountains National Park (GMNP) was established (Fabry, 1988; Jameson, 1994). Consequently, federal fire suppression was not implemented on the ranch, and suppression of fire by ranch hands was limited. According to a former ranch employee, relatively little effort was spent on fighting fires in the early days of ranch operation (Kincaid, personal communication).

In this study, we characterize fire regimes and forest structure in GMNP to answer the following questions about the effects of land use on fire-vegetation relationships in mixed conifers forests: (1) How do fire regimes vary spatially and temporally? We expected fire frequency and extent to decline with the onset of ranching and not with the organized fire suppression that began 50 years later. We also expected spatial variation in fire regimes in this mountainous terrain to be related to topography (i.e., slope aspect, slope position, and forest composition). (2) How has forest structure and composition changed since Euro-American settlement? We expected forests to have increased in density and for the onset of that increase to coincide with fire exclusion. (3) Has the delay in Euro-American land use in the study area reduced the likelihood of high-severity fire? We expected high-severity fire to be less likely because changes in forest structure that promote more severe fire occurred 40 or 50 years later than in other parts of the southwest.

2. Material and methods

2.1. Study area

Mixed conifer forests were studied in a 1000-ha area in GMNP. GMNP is located at the southern tip of the Guadalupe Mountains in the basin and range physiographic province. Forests in the Guadalupe Mountains occur between the elevations of 2200–2700 m on top of a plateau that rises 600 m above the surrounding Chihuahuan Desert. The Guadalupe Mountains are composed of Permian-aged limestone, and soils are mostly shallow, well-drained loam to sandy loam that formed in residuum (Kittams, 1972). There were no perennial streams in the study area, but ephemeral streambeds may have acted as fuel breaks and impeded the spread of fire.

The climate is semi-arid with cool winters and hot summers. Mean January and July temperatures at Carlsbad Caverns (1340 m) 55 km northeast of the park are −5.6 °C and 27.2 °C, respectively. Average annual precipitation at Carlsbad Caverns is 35 cm and most (78%) falls from May to October during the southwest monsoon (Kittams, 1972). Lightning-ignitions peak in mid- to late spring, before the onset of monsoon showers (Ahlstrand, 1980).

Mixed conifer forests in the study area occupy a range of topographic settings, and forest composition varies with slope aspect and position. The most xeric sites are not forested. Instead, the vegetation is dominated by shrubs, stem succulents, leaf succulents, grasses, and forbs. Forests on south-facing slopes are dominated by ponderosa pine (*Pinus ponderosa* var. *scopulorum* Engelm.), alligator juniper (*Juniperus deppeana* Steud. var. *deppeana*), Gambel oak (*Quercus gambelii* Nutt.), and pinyon pine (*Pinus edulis* Engelm.). Cooler, more mesic sites on north-facing slopes, at higher elevation, or on shaded, lower slopes, are dominated by Douglas-fir (*Pseudotsuga menziesii* var. *glauca* [Beissn.] Franco) and southwestern white pine (*Pinus strobiformis* Engelm.) (Ahlstrand, 1980). Shrubs or small trees such as hophornbeam (*Ostrya knowltonii* Cov.), bigtooth maple (*Acer grandidentatum* Nutt.), and serviceberry (*Amelanchier utahensis* Koehne.) are common in the forest understory.

People have used lands in or near GMNP for at least 10,000 years (Fabry, 1988). Prior to Euro-American settlement, GMNP was used by the Mescalero Apache. The Mescalero Apache were subjugated in ca. 1870, and permanent European-American settlement at the base of the mountains occurred shortly thereafter. Use of the high-country for livestock grazing did not occur until the 1920s with the establishment of the Guadalupe Mountain Ranch and the installation of a water system. With permanent water, large herds of sheep and goats were grazed year round, but grazing ceased when the area became a National Park in 1972 (Jameson, 1994; Fabry, 1988). Fire suppression was implemented when the area became a National Park. Several wildfires and prescribed fires have burned in the study area since the establishment of GMNP. Since fires consume evidence (i.e., fire scars and trees) of past forest structure and fire regimes (Swetnam et al., 1999), recently burned areas were avoided for sampling.

2.2. Forest composition

Variation in forest composition was identified by first stratifying forested sections of the 1000-ha study area by slope aspect (north, south, east, west) and slope position (valley bottom, mid-slope, ridgetop), and distributing sixty 400 m^2 (20 m $\times$ 20 m) plots among the strata. Slope aspect and position were determined using a topographic map, and plots were randomly placed among the strata in proportion to the amount of forest occupying each stratum. Plot selection was made using UTM coordinate pairs, and navigation to these points was accomplished with the aid of a hand-held GPS receiver. Slope aspect and position were verified on the ground at each site. If a site was unsuitable (e.g., not forested or recently burned by wildfire), the plot was moved in 100 m increments along the contour of the slope until a suitable replacement (i.e., at least 40% forest cover and no evidence of recent disturbance) location was found. In each plot, all live trees (stems >5 cm dbh), standing dead trees, and downed logs that were rooted in the plot were identified by species and their diameters were measured at breast height (dbh). Live saplings (stems >1.4 m tall and <5.0 cm dbh) and seedlings (0.5–1.4 m tall) of each species were tallied. We also recorded the elevation, slope aspect, slope pitch, slope configuration, and topographic position of each plot. The last four topographic variables were used to calculate each plot's topographic relative moisture index (TRMI), a measure of relative site moisture availability that ranges from 0 to 60 (Parker, 1982).

We identified plots with similar forest composition using cluster analysis. Plots were grouped by first calculating species importance values (IV) as the sum of relative density and relative basal area (maximum 200) for each species in each plot (Husch et al., 2003). Species IV in each plot were then clustered using Ward's method and relative Euclidean distance. Ward's method minimizes within group variance relative to between group variance (Gauch, 1982; McCune and Mefford, 1999).

2.3. Fire regimes

Fire regime parameters (frequency, season, severity, extent) were quantified using fire scars in partial wood cross-sections removed from fire-scarred trees. Partial cross-sections were collected from dead and live trees throughout the 1000-ha study area (Arno and Sneck, 1977). Trees with the most visible scars (the longest fire record) were targeted for collection from the population of all fire-scarred trees, and preference was given to trees that had solid wood around the scars suitable for collection with a chainsaw. Additionally, dead material (logs and snags) was favored in order to minimize damage to live trees in the park. The location of each fire-scarred tree sampled was identified with a GPS receiver and recorded on a topographic map (Fig. 1).

The dates of fires recorded in each sample were determined by first sanding each partial cross-section to a high polish and then visually cross-dating the sample's tree ring series by comparing lists of marker years in the sample to marker years of a tree-ring chronology for the site (Stokes and Smiley, 1968; Yamaguchi, 1991). We used a tree-ring record of Douglas-fir from Guadalupe Peak (Stahle et al., 1992) for comparison of marker years. The calendar year of each annual ring with a fire scar in it was then recorded as the fire year. Fire years for scars that occurred at the ring boundary were assigned to the following year.

2.3.1. Seasonality

The season of each burn was inferred from the position of fire scars within annual growth rings (Baisan and Swetnam, 1990). The positions of scars within a ring were classified as: (1) dormant (boundary between rings), (2) early earlywood (first 1/3 of earlywood), (3) middle earlywood (middle 1/3 of earlywood), (4) late earlywood (last 1/3 of earlywood), and (5) latewood. In ponderosa pine and mixed conifer forests in the southwest, dormant season scars usually represent fires that burn early in the spring before the onset growth for the year, rather than burns in late fall that occur after trees have stopped growth for the year (Baisan and Swetnam, 1990; Caprio and Swetnam, 1995; Brown et al., 2001).

2.3.2. Temporal and spatial patterns

Temporal variation in fire occurrence that may be related to land-use change was identified by comparing a composite record of fire occurrence for the pre-Euro-American (1700–1880), settlement (1881–1922), and grazing (since 1922) periods using a *t*-test and FHX2 software (Grissino-Mayer, 2001). A composite record of fires was used because they are more sensitive to temporal changes in burning patterns than samples from a more restricted area (Dietrich, 1980).

Spatial variation in fire return intervals (FRI) in the study area was determined by comparing both point and composite FRI for samples in each slope aspect and slope position group. Composite FRI included all recorded fires for samples in a group. Point FRI is the record of consecutive fires in a single sample. Point FRI tend to be longer than composite FRI because they reflect the time dependence of fuel accumulation at a single point (Dietrich, 1980; Kitzberger et al., 2001). Comparisons among groups were made using a Kruskal–Wallis *H*-test.

Statistical description of fire-return intervals for both point and composite fire chronologies includes the mean fire interval, median fire interval, and the Weibull median probability interval (WMPI) as measures of central tendency. Distributions

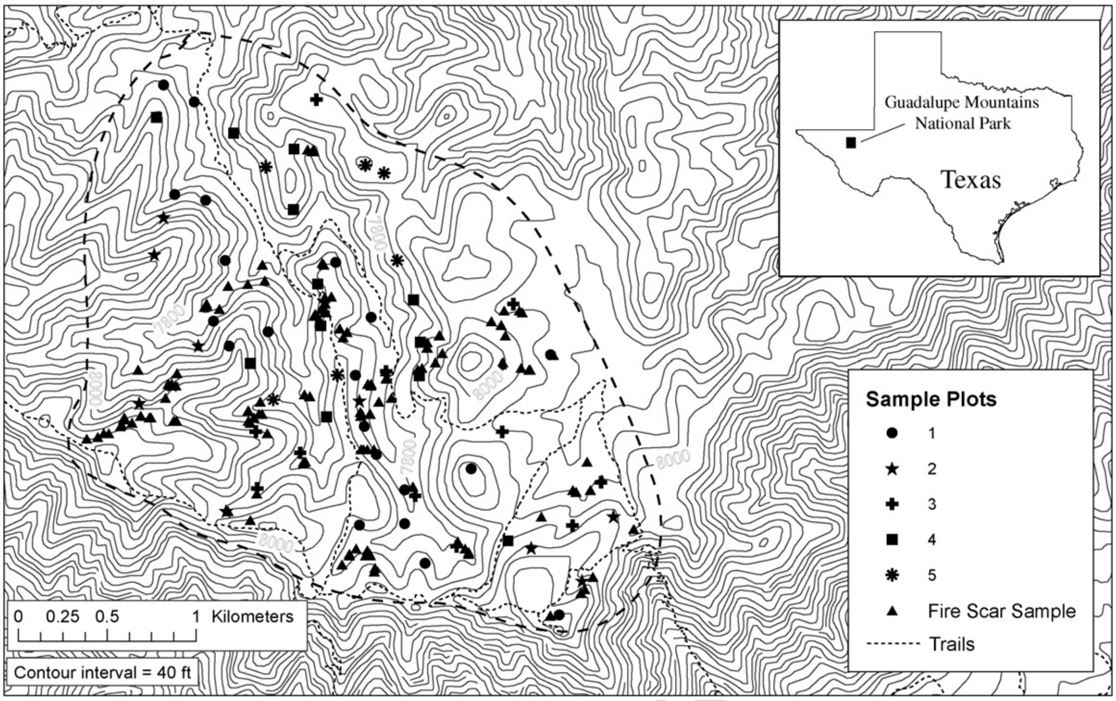

Fig. 1. Locations of the study area, sample plots, and fire scar samples in Guadalupe Mountains National Park, Texas. Numbers and symbols in the legend refer to forest types given in Table 1.

of fire intervals are asymmetrical, and the WMPI is a measure used to describe central tendency in asymmetrical distributions (Grissino-Mayer, 2001).

2.4. Fire extent

Fire extent was not measured directly by drawing fire boundaries (e.g., Taylor, 2000) because the network of fire scar samples was too sparse. Instead, the frequency of fires of varying extent were estimated using an index derived from three different composite fire chronologies: all detected fires; all fires recorded by 10% or more; and all fires recorded by 25% or more of the fire scar samples. This analysis distinguishes the frequency of years of widespread fires from years with fires of limited extent.

2.4.1. Fire severity

Fire severity was interpreted from stand age structure. As fires burn across a landscape, they may kill many trees in some stands and few in others. This variation is reflected in the age structure of tree populations. For example, stands that experienced a severe disturbance, such as high-severity fire, are usually even-aged, whereas stands that experience moderate-severity fires are usually multi-aged. Forests that burn frequently with low-severity are multi-aged, but the age classes may not correspond clearly with fire occurrence (Agee, 1993; Taylor and Skinner, 2003). The age structure of trees was determined by coring all stems >20 cm dbh and an average of 11 trees >5 and <20 cm dbh in each plot. All trees were cored at a height of 30 cm above the soil surface with an increment borer. Tree ages were determined by sanding each core to a high polish, cross-dating the annual rings in each core (Stokes and Smiley, 1968; Yamaguchi, 1991), and assigning the pith date as tree age. To identify variation in the number of years for a tree to grow to coring height we collected 28 seedlings (12 southwestern white pine, 12 Douglas-fir, 4 ponderosa pine) from sites throughout the study area. Stems of the seedlings were cut off at the root-shoot interface and at a height of 30 cm. Both cross sections were sanded to a high polish, and the difference in the ring counts for the disks was used to determine age at coring height. The average number of years trees took to grow to coring height was 17 years (range 10–29). Given the variation in initial growth rates, we report tree ages as age at coring height rather than adding a correction factor to the ages of cored trees. Tree ages were then placed in 20-year classes, and we counted the number of classes occupied by trees in each plot. Presumably, plots with trees in many 20-year age classes experienced less severe fire than those with stems in fewer age classes. Finally, we calculated the mean number of occupied age classes for each forest compositional group.

The size of a surviving tree when it is first scarred by fire is an indicator of past fire intensity (Kilgore and Taylor, 1979). If surviving fire-scarred trees were small in diameter when first scarred by a fire, then fires must have been low in intensity because they damaged the cambium but did not kill the tree (Agee, 1993). To assess the intensity of past fires, we measured the distance from the pith to the juncture of the earliest fire scar on all stem cross-sections that included the pith year ($n = 74$). Radial distance was then used to estimate the diameter of the tree when it was first scarred by a fire. All fire-scarred cross-sections were extracted from stems at a height of 30–50 cm above the soil surface so measurements of diameter at the time of first fire scarring are below breast height.

2.5. Forest age structure and compositional change

Twentieth-century forest changes were identified by examining recent patterns of tree establishment. If forest changes were related to livestock grazing during the ranching era, then a pattern of increased tree establishment and survival beginning about 80 years ago should be discernible in the age structure of tree populations (e.g., Kaufmann et al., 2000; Taylor, 2000). However, if forest changes are related only to active fire suppression, increased tree recruitment and survival should have begun about 30 years ago. Age structures were developed for each forest compositional group by combining all stem ages for plots in each forest compositional group into 20-year age classes.

Shifts in forest composition that may be related to live stock grazing or fire exclusion were identified using stand structure data and ordination. We ordinated the average density (stems ha^{-1}) of understory (<20 cm dbh) and overstory (≥20 cm dbh) stems of each species in each compositional group using detrended correspondence analysis (DCA) (Gauch, 1982). This approach assumes that differences in the composition and abundance of understory versus overstory stems represent recent shifts in regeneration patterns related to livestock grazing or fire exclusion. Compositional changes were displayed in DCA species' space by joining the understory and overstory in each group with a vector.

3. Results

3.1. Forest composition

Five forest compositional groups were identified based on cluster analysis of species IV, and they are segregated by elevation, slope position, slope configuration, and total potential soil moisture (Kruskal–Wallis H-test, $P < 0.05$) (Table 1). The Douglas-fir/oak/mixed conifer group (PSME-QUGA-MC) is dominated by Douglas-fir with southwestern white pine, ponderosa pine, gambel oak, and bigtooth maple being important associates (Table 1). Most stands in this group occur on middle slope positions on north-facing slopes (Fig. 1). The Douglas-fir group (PSME) occupies high elevation north-facing slopes and is strongly dominated by Douglas-fir; gambel oak and southwestern white pine are important associates. The southwestern white pine/Douglas-fir group (PIST-PSME) is co-dominated by southwestern white pine and Douglas-fir, and gambel oak and ponderosa pine are important associates. Stands in this group occupy lower slopes and valley bottoms on north and east-facing slopes. The juniper/ponderosa pine/Douglas-fir compositional group (JUDE-PIPO-PSME) is dominated by alligator juniper and ponderosa pine, and Douglas-fir and pinyon pine are important associates. Stands in this group typically occupy upper slope positions on west-facing slopes. Stands in the pinyon pine/juniper compositional group (PIED-JUDE) occupy the driest forested sites in the study area, on or near ridge-tops. The group is dominated by pinyon pine and alligator juniper, and ponderosa pine is an important associate.

3.2. Fire regimes

3.2.1. Fire record

A total of 854 fire scars were successfully cross-dated from 133 partial cross-sections from live (n = 23) and dead (n = 110) trees. Most samples were extracted from southwestern white pine (n = 119); others were from ponderosa pine (n = 14). This species selection is proportional to the survey of fire-scarred

Table 1
Mean importance value (IV, maximum 200), basal area (BA) (m^2 ha^{-1}) and density (ha^{-1}) of trees (>5.0 cm dbh) for forest types identified by cluster analysis of species IV in Guadalupe Mountains National Park

Species	PSME-QUGA-MC (n = 21)			PSME (n = 10)			PIST-PSME (n = 11)			JUDE-PIPO-PSME (n = 12)			PIED-JUDE (n = 6)		
	IV	BA	Density	IV	BA	Density	IV	BA	Density	IV	BA	Density	IV	BA	Density
Douglas-fir*	77.9	10.6	657.1	133.6	23.3	662.5	56.4	10.5	352.3	27.6	2.5	81.3	6.0	1.0	12.5
Southwestern white pine*	27.7	4.9	172.6	19.3	2.8	117.5	97.1	18.2	600.0	3.7	0.4	8.3	0.0	0.0	0.0
Ponderosa pine*	19.5	4.1	82.1	6.5	1.3	27.5	14.6	2.1	115.9	66.8	7.9	156.3	11.6	1.7	29.2
Colorado piny on pine*	1.7	0.1	19.0	0.0	0.0	0.0	0.6	0.1	4.5	19.1	0.6	79.2	116.6	10.3	495.8
Alligator juniper*	11.6	2.5	46.4	1.9	0.4	5.0	2.5	0.3	20.5	81.9	11.2	158.3	65.8	10.8	137.5
Gambel oak*	54.6	6.4	517.9	36.9	2.7	325.0	25.8	3.5	211.4	0.9	0.0	4.2	0.0	0.0	0.0
Knowlton hophornbeam	2.8	0.2	35.7	1.6	0.1	15.0	2.9	0.2	29.5	0.0	0.0	0.0	0.0	0.0	0.0
Bigtooth maple	4.2	0.2	53.6	0.3	0.0	2.5	0.2	0.0	2.3	0.0	0.0	0.0	0.0	0.0	0.0

	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.
Elevation (m)*	2343.0	80.0	2447.0	97.0	2390.0	33.0	2362.0	55.0	2360.0	37.4
Aspect	14.0	4.0	16.0	3.0	13.0	5.0	9.0	4.0	4.0	2.3
TRMI*	33.0	6.0	35.0	6.0	39.0	8.0	23.0	6.0	18.0	4.4
Slope position*	10.0	5.0	11.0	4.0	15.0	5.0	5.0	4.0	5.0	5.4
Slope pitch (o)	19	6.6	23.5	7.6	19.1	10.5	20.3	6.5	19.7	12.2
Slope configuration*	5.1	0.7	5.3	0.9	6.1	1.5	4.8	0.9	5	0

Forest types are Douglas-fir-Gambel oak-mixed conifer (PSME-QUGA-MC). Douglas-fir (PSME), southwestern white pine/Douglas-fir (PIST-PSME), alligator juniper-ponderosa pine-Douglas-fir (JUDE-PIPO-PSME), and pinyon pine/alligator juniper (PIED-JUDE). n = Number of samples in each forest type. TRMI varies between 0 (xeric) and 60 (mesic) and is a scalar index of four slope parameters: slope position (0–20), slope aspect (0–20), slope pitch (0–10), slope configuration (0–10). Variables with and asterisk (*) had values that were significantly different among forest types ($P < 0.05$, Kruskal–Wallis H-test).

Table 2
Fire return interval (FRI) statistics for point and composite samples on north ($n = 71$), east ($n = 19$), south ($n = 4$), and west ($n = 39$) slope aspects

Slope aspect	Mean	Median	WMPI	Range	S.D.	No. of intervals
Point fire return intervals						
North	24.6	22.0	22.4		15.4	
East	22.7	20.5	20.3		15.9	
South	23.6	15.0	17.9		26.1	
West	23.1	21.0	20.7		15.8	
All slopes	24.0	21.0	21.6		16.0	
Composite fire return intervals, all scarred						
North	4.5	2.0	2.9	1–54	6.8	84
East	8.5	5.0	5.0	1–68	12.9	47
South	11.5	9.0	9.3	1–36	9.3	25
West	6.0	3.0	3.8	1–46	8.6	65
All slopes	4.0	2.0	2.4	1–54	7.2	115
Composite fire return intervals, 10% scarred						
North	8.3	5.0	5.7	1–68	11.0	45
East	9.7	6.0	6.3	1–68	13.5	41
South	11.5	9.0	9.3	1–36	9.3	25
West	9.6	6.0	6.7	1–46	10.6	41
All slopes	9.2	6.0	6.0	1–68	12.1	50
Composite fire return intervals, 25% scarred						
North	15.3	13.0	13.9	3–40	9.9	20
East	18.1	12.5	15.1	3–68	16.2	22
South	11.5	9.0	9.3	1–36	9.3	25
West	18.1	15.0	15.6	1–46	12.3	21
All slopes	16.3	14.0	15.1	6–46	10.1	24

trees in the study area (southwestern white pine $n = 261$; ponderosa pine $n = 31$). The fire record included 139 fire dates recorded between A.D. 1530 and 1990. During this period, the mean CFI for all samples was 4 years. Fires recorded in 10% and 25% of the samples were less frequent with mean CFIs of 9.2 years and 16.3 years, respectively. The mean point fire return interval of all samples was 24 years (Table 2).

3.2.2. Temporal variability

Fire occurrence in GMNP varied over time (Fig. 2). The mean composite fire return interval for all recorded fires during the pre-EuroAmerican period (1700–1879) was 2.7 years (range: 1–15), and it was longer at 8.4 years (range 5–13 years) (t-test, $p < 0.05$) during the settlement period (1880–1922). Fire occurrence dropped dramatically at the beginning of the ranching era and the last widespread fire occurred in 1922. Two samples recorded a fire in 1990; they were scarred by a wildfire in the northeast part of the study area.

3.2.3. Spatial variability

There was no spatial variation in composite FRI by slope aspect or slope position (Kruskal–Wallis H-test, $p > 0.05$) for all fires, fires that burned 10% or more, or 25% or more of the samples (Table 2). Similarly, point fire return intervals did not vary (Kruskal–Wallis H-test, $p > 0.05$) by slope aspect or slope position (Table 3).

3.2.4. Fire extent

The frequency of fires of different extent was inferred from the number of samples that recorded a fire. Small fires that burned one or more samples had the shortest FRI (median = 2.0 years), and they were most frequent between A.D. 1650 and 1800 (Table 3, Fig. 2). Intermediate sized fires recorded by 10% or more of the samples had longer FRI (median = 6.0 years) (Table 4). The largest fires that scarred 25% or more of the samples had the longest FRI (median = 14.0 years), and they occurred with equal frequency throughout the fire record.

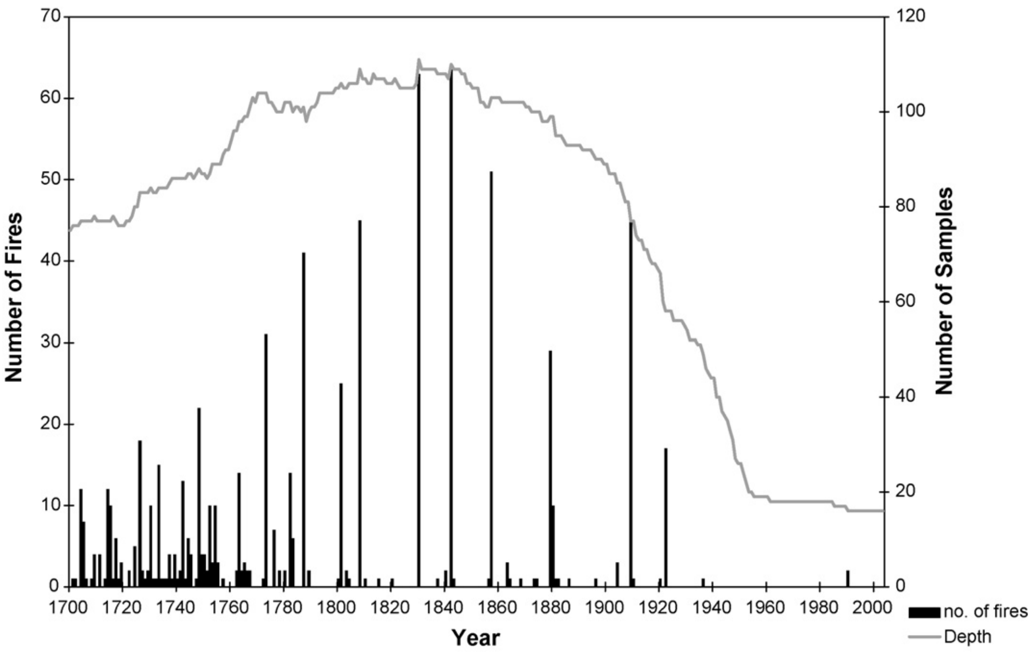

Fig. 2. Composite fire chronology for mixed conifer forests for the period 1700–2003 in Guadalupe Mountains National Park, Texas. Bars represent the number of samples scarred and the line is sample depth.

Table 3
Fire interval statistics for point and composite fire return intervals by slope position

Slope position	Mean	Median	WMPI
Point fire return intervals			
Lower	23.6	22.0	22.1
Mid	23.2	19.0	20.8
Upper	25.6	21.0	21.9

Slope position	Mean	Median	WMPI	Range
Composite fire return intervals, all scarred				
Lower	507	3.0	3.9	1–43
Mid	4.5	2.0	2.8	1–54
Upper	7.4	5.0	4.7	1–86
Compostie fire return intervals, 10% scarred				
Lower	9.5	7.0	7.4	1–43
Mid	8.1	5.0	5.3	1–68
Upper	10.6	6.0	7.4	1–86
Compostie fire return intervals, 25% scarred				
Lower	15.9	14.0	13.4	1–43
Mid	12.7	12.0	11.3	1–33
Upper	17.8	13.5	14.7	3–86

Lower = the lower 1/3 of a slope facet or a valley bottom ($n = 31$), mid = the middle 1/3 of a slope facet ($n = 66$), and upper = the upper 1/3 of a slope facet or a ridge-top ($n = 31$).

3.2.5. Seasonality

The positions of fire scars within annual growth rings indicate that most fires burned early in the growing season. Scars occurred most frequently in the first (67%) and second (24%) 1/3 of the earlywood. Few scars were present in latewood or at the ring boundary when trees were dormant (Table 5).

3.2.6. Fire severity

Trees occupied a large number of age classes in the forest compositional groups (Fig. 3). The mean number of 20-year age classes ($n = 16$) occupied by trees in the Douglas-fir/oak/mixed conifer group was 5.6 (range 4–8) and it was 5.6 (range 2–7) in the Douglas-fir group too. Plots in the southwestern white pine/Douglas-fir group had trees in, on average, 6.0 (range 4–9) 20-year age classes, while the means for the alligator juniper/ponderosa pine/Douglas-fir and pinyon pine/alligator juniper groups were 4.3 (range 2–7) and 5.2 (range 3–7) age-classes, respectively.

Fire-scarred trees were small when they were first scarred. The mean diameter of stems ($n = 74$) at the time of first scarring was 10.1 cm (range 1.6–28.6 cm). Moreover, 57% of the samples were < 10 cm, and 26% were <5 cm in diameter (sampling size) when they were first scarred.

3.2.7. Age structure and compositional change

A large pulse of tree establishment coincided with the onset of livestock grazing in all forest compositional groups (Fig. 3). In the Douglas-fir/oak/mixed conifer group, there were pulses of mostly Douglas-fir establishment 40 and 80 years ago. In the Douglas-fir and southwestern white pine/Douglas-fir groups, most trees established between 60 and 120 years ago. Both the juniper/ponderosa pine/Douglas-fir and pinyon pine/juniper groups had regeneration pulses between 20 and 60 years ago, with peaks between 40 and 60 years ago.

The ordination of overstory (<20 cm dbh) and understory (≥20 cm dbh) stems indicates that forests are undergoing compositional change (Fig. 4). The vector length between understory and overstory stems is proportional to the magnitude of compositional difference between these layers. In groups 1 and 3, there has been a compositional shift from southwestern white pine and ponderosa pine to Douglas-fir. Group 4 has an overstory dominated by ponderosa pine and an understory of alligator juniper, and group 5 is undergoing a compositional shift from juniper to pinyon pine. Group 2 exhibits little compositional change. Both layers are dominated by Douglas-fir. Gambel oak was excluded from the analysis because it rarely exceeds 15 m in height and seldom reaches the overstory. Overall, Douglas-fir and pinyon pine have increased relative to all other tree species in GMNP.

4. Discussion

Mixed conifer forests in GMNP varied widely in composition. Forest species distribution and abundance patterns were controlled by elevation and topographically influenced patterns

Table 4
Mean and median composite fire return intervals for all fires and widespread fire events by time period

	Pre-settlement (1700–1879)			Settlement (1880–1922)			Fire exclusion (1923–2003)		
	Mean	Median	Range	Mean	Median	Range	Mean	Median	Range
All scarred	2.74	2	1–15	8.4	8	5–13	*	*	*
25% Scarred	15.3	14.5	7–25	*	*	*	*	*	*

* Indicates that there were not enough intervals to analyze.

Table 5
Seasonal distribution of fires based on the position of fire-caused lesions within annual growth rings

Fires	Seasonality of burn		Fire scar position within annual growth ring				
	Determined	Undetermined	Dormant	Early	Middle	Late	Latewood
Number	475	379	31	319	112	4	9
Percentage	55.6	44.4	6.5	67.2	23.6	0.8	1.9

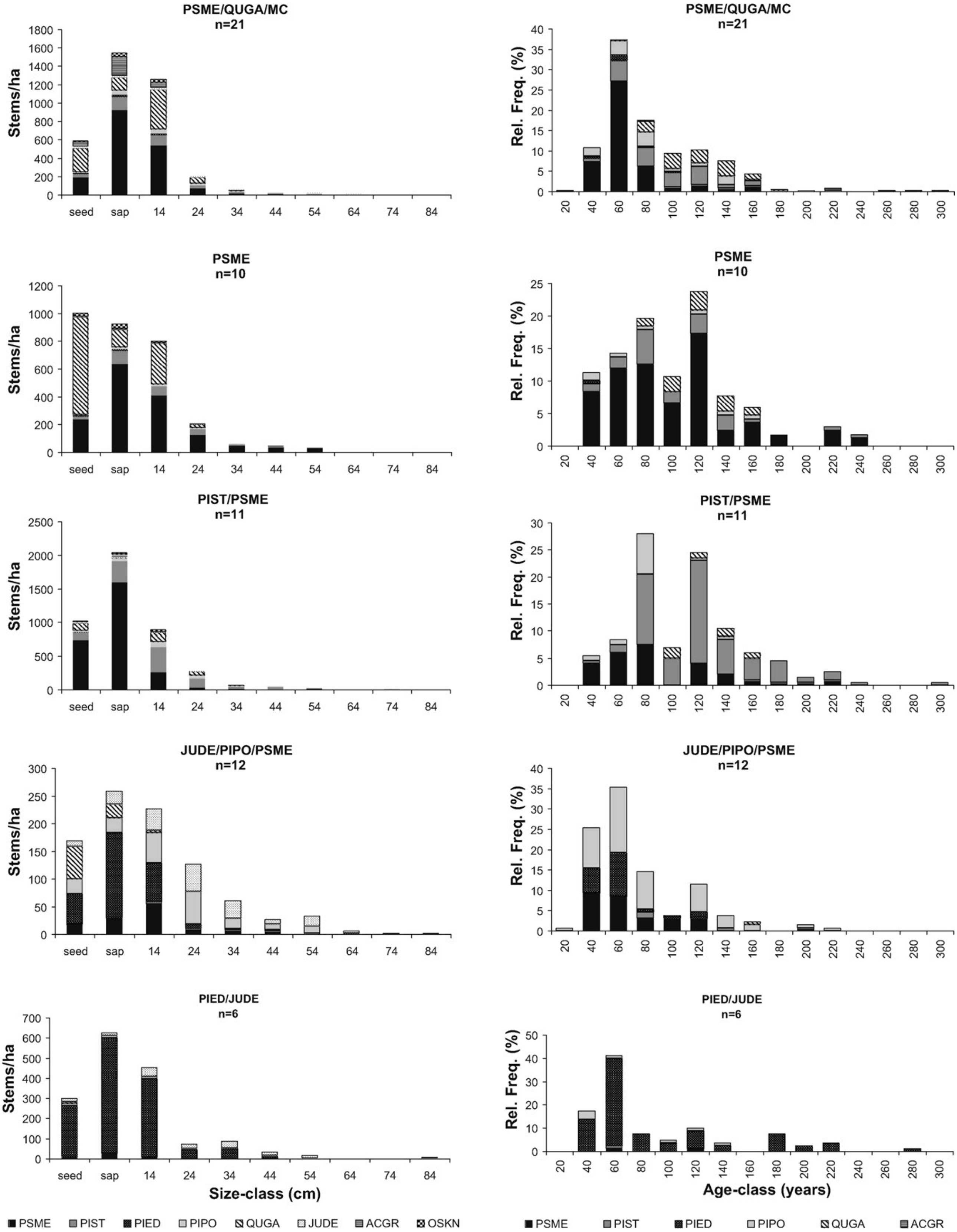

Fig. 3. Size and age structure of tree populations in size structural groups identified by cluster analysis of seedlings, saplings, and trees in 10 cm dbh classes (ha^{-1}) from 60 sample plots in Guadalupe Mountains National Park, Texas. Only a subsample of trees were aged in each plot, and the *y*-axis is the relative frequency of all aged stems in a 20-year age class. Species are bigtooth maple (ACGR), alligator juniper (JUDE), gambel oak (QUGA), ponderosa pine (PIPO), pinyon pine (PIED), southwestern white pine (PIST), Douglas-fir (PSME). Note that values on the *x*-axis are upper limits of age and size classes, and the vertical scale is not the same on each graph.

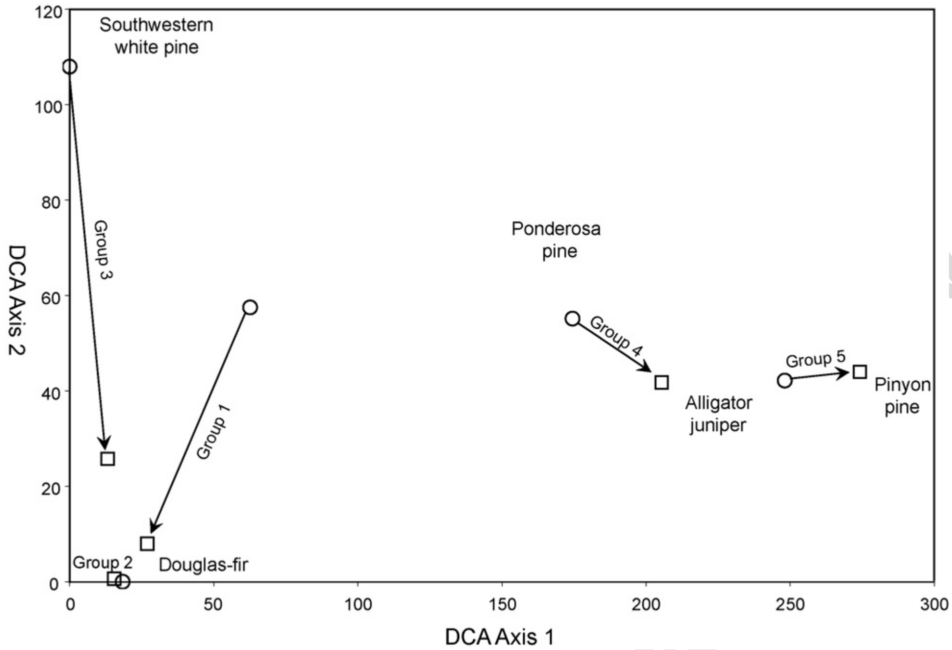

Fig. 4. Compositional differences in the density of the understory (<20 cm dbh) stems (squares) and overstory (≥20 cm dbh) stems (circles) in five mixed conifer forest compositional groups identified by cluster analysis in Guadalupe Mountains National Park, Texas. Vectors show the direction and magnitude of compositional difference between the two size classes for each group in DCA species space. The position of species labels represents the regions of relative dominance. Group 1 = Douglas-fir/mixed conifer, group 2 = Douglas-fir, group 3 = southwestern white pine/Douglas-fir, group 4 = alligator juniper/ponderosa pine/pinyon pine, group 5 = pinyon pine/alligator juniper.

of soil moisture. Pinyon pine, alligator juniper, and ponderosa pine were most abundant on dry upper slopes and ridgetops on west facing slopes. In contrast, Douglas-fir, southwestern white pine, and gambel oak were most abundant on mesic lower slopes and valley bottoms, particularly on north and east facing slopes. Overall, the topographic controls on tree species distribution and abundance patterns we identified in our study area are similar to those for montane forests elsewhere in the southwestern USA, except for slope aspect (Whittaker and Niering, 1965; Niering and Lowe, 1984; Kaufmann et al., 1998; Cocke et al., 2005). South and west aspects in our study area were dominated mainly by shrubs or stem succulents, but forest occurred on lower shaded slopes in this incised terrain. Consequently, tree species distribution and abundance were more strongly associated with slope position and configuration than slope aspect.

Variation in slope aspect, species composition, and elevation is a potentially important control on spatial patterns of fire frequency in forested landscapes. In mixed conifer forests in the Pacific Northwest, fire frequency is higher on south-facing slopes than north-facing slopes (Beaty and Taylor, 2001; Taylor and Skinner, 2003). This spatial variation in fire frequency is related to several factors that affect the production, moisture, arrangement, structure, and flammability of fuels (Biswell, 1989). First, differences in temperature and duration of snowpack among slope aspects combine to favor dominance by long-needled pines on south-facing slopes, and short-needled fir (*Abies*, *Pseudotsuga*) on north-facing slopes, respectively. Fire intensity and spread are greater in the low-density fuel beds of pine than fir (Albini, 1976; Rothermel, 1983; Fonda et al., 1998; van Wagtendonk, 1998). Second, south-facing slopes are snow-free earlier in spring, and the period in which fuels are dry enough to burn is longer each year than on north-facing slopes. The longer the period in which fuels are dry each summer on south-facing slopes increases the probability of ignition and spread of fire (Agee, 1993). Third, production of fine fuels is higher in pine than fir-dominated mixed conifer forests (Agee et al., 1978; Stohlgren, 1988; van Wagtendonk, personal communication), so a fire can burn again sooner on a south rather than a north-facing slope.

In GMNP, forest composition varied with elevation and topographically controlled patterns of soil moisture, but there was no spatial variability in fire frequency related to topography or forest composition. Mean point and composite FRI were similar among forest types, despite topographically related variation in species composition. Moreover, mean FRI did not vary with slope aspect or slope position. In the southwest, topographic controls such as slope aspect and slope position are weakly expressed in most ponderosa pine and mixed conifer forests. Only small differences in fire frequency between ponderosa pine and mixed conifer forests were identified for 63 sites in Arizona and New Mexico (Swetnam and Baisan, 1996). Similarly, there was little variation in fire frequency across elevation and forest compositional gradients in ponderosa pine and mixed conifer forests in the nearby (<200 km) Sacramento Mountains (Brown et al., 2001). However, in the Sacramento Mountains, variation in fire frequency was related to physiographic heterogeneity that reduced fuel continuity and potential for fire spread (Brown et al., 2001). Topography features such as rock outcrops, soils with low productivity, and streams are known to act as barriers to fire spread and reduce fire frequency (Taylor, 2000; Taylor and Skinner, 2003). In

Oregon mixed conifer forests, fire frequency only varied with slope aspect in terrain with barriers to fire spread (Heyerdahl et al., 2001). In watersheds without fuel breaks, where slopes of different aspect converged, fire frequency did not vary with slope aspect. There were no obvious breaks in fuel continuity in our study area, and even the non-forested parts of south and west-facing slopes had a cover of grass and shrubs that could carry fire. Thus, the similarity in FRI among forest types, slope aspects, and slope positions in GMNP is probably related to high fuel connectivity.

The response of species to fire is strongly influenced by the season of burn (Biswell, 1989), and the position of fire scar lesions within annual growth rings in the GMNP study area indicate that most fires (91%) were early growing season burns. Spring, or early season fires, are a hallmark of ponderosa pine and mixed conifer forests in the southwest, but the fires often burn before trees start growth for the year (dormant season) (Swetnam and Baisan, 1996; Swetnam and Betancourt, 1990). In mixed conifer forests in the Sacramento Mountains, burns were most frequent ($\geq$40%) in the dormant season (Brown et al., 2001). The pattern of later season burns in GMNP may be related to landscape structure and fuel connectivity. In the Sacramento Mountains, mixed conifer forests are connected to lower elevation pinyon pine-juniper woodland and ponderosa pine forests. Fuels in these lower elevation forests dry earlier in the year, and fires in these forests could spread into higher elevation mixed conifer forests. In our study area, mixed conifer forests are isolated from lower elevation vegetation by a nearly 500 m vertical escarpment that is sparsely covered with vegetation. Consequently, early season burns that start in lower elevation pinyon-juniper woodland or Chihuahuan desert scrub may have rarely spread into higher elevation mixed conifer forest.

Variation in fire severity creates heterogeneity in forest structure at landscape scales because burns may kill all trees in some stands and few in others. Forests that experience high-severity fire are even-aged while those that experience mainly low and moderate-severity fires have stems in a wide range of age classes because fires kill few trees in a stand (Agee, 1993). Mixed conifer forests in our study area were multi-aged. When trees were assigned to 20-year age-classes, an average of 5.3 age-classes were occupied in each plot. Moreover, 48% of the plots had stems in 6 or more different 20-year age classes, and 57% of the plots had trees $\geq$160 years old. Mixed conifer forests with a multi-aged or multi-sized forest structure that experienced frequent fire have been identified in other parts of the southwest (Fulé et al., 2003; Mast and Wolf, 2004). Our sample of fire-scarred southwestern white pine indicates that stems were small in diameter when they were first scarred and these stems survived repeated fires. Southwestern white pine has thin bark and is more sensitive to scarring by fire than ponderosa pine or Douglas-fir (Ahlstrand, 1980), and fire scars on ponderosa pine or Douglas-fir were infrequent in our study area. The age structure and fire-scar data suggest that pre-fire exclusion fires in GMNP mixed conifer forests were mainly low or moderate-severity burns.

Fire frequency and extent in GMNP varied over time and shifted from a regime of frequent small burns before 1800, to less-frequent larger burns after 1800, and then no fires burned after 1922. The shift c. 1800 to less frequent fires with greater synchrony among samples was a sudden change in the fire regime in GMNP, and it may have been caused by a decline in the population of native Americans or by a change in climate.

Mescalero Apache settlements were present in the Sacramento Mountains and near springs at the base of the Guadalupe escarpment in the19th century, and Apache hunted in the forested high-country (Ahlstrand, 1980; Jameson, 1994). Euro-American settlement, disease, and finally a military campaign eliminated the local Apache populations. This may have reduced ignitions in the high country and resulted in less frequent, but more widespread burns caused by lightning ignitions. However, the size of the Mescalero Apache population in the region and the timing and magnitude of the native population decline remains uncertain.

The reduction of fire frequency c. 1800 also coincided with reduced fire frequency in ponderosa pine forests in New Mexico (Touchan et al., 1995; Grissino-Mayer and Swetnam, 2000) and mixed conifer forests in northern Mexico (Stephens et al., 2003). Moreover, Kitzberger et al. (2001) found a similar shift in fire frequency and extent in South American forests and attributed the fire regime change to inter-hemispheric changes in the frequency and amplitude of the El Niño – Southern Oscillation (ENSO). ENSO is a high frequency (i.e., 2–7 years) coupled ocean-atmosphere process in the eastern and central equatorial Pacific Ocean and, through teleconnections with mid-latitude climate systems, is the primary driver of North American inter-annual climate variability (Diaz and Markgraf, 2000). The similarity and synchronicity of the fire regime change in GMNP as compared with other sites in the southwest suggest that climate variation contributed to the fire regime change.

The cessation of fire in GMNP after 1922 coincides with the introduction of livestock. Although Euro-Americans first settled in the area in the late 19th century, use of the high-country was limited until the establishment of the Guadalupe Mountain Ranch in the 1920s when large herds of sheep and goats were grazed in the high country (Jameson, 1994). Livestock grazing eliminates fine fuels (grass, shrubs) that support fire spread, and introduction of livestock grazing in the 19th century has been implicated as the initial cause of a decline in fire frequency in ponderosa pine and mixed conifer forests throughout the southwest (Savage and Swetnam, 1990; Swetnam and Betancourt, 1998; Touchan et al., 1995). Sheep and goats in GMNP maintained low levels of fuels into the 1960s, and then a policy of suppressing forest fires was implemented when the area became a national park in 1972.

Forest changes caused by fire exclusion are well documented in southwestern ponderosa pine forests (Covington and Moore, 1994; Fulé et al., 1997). Forest density began to increase in the late 19th century with elimination of fire, caused mainly by livestock grazing. Mixed conifer forests in GMNP experienced a similar change, although four decades later than in other southwestern forests. Tree establishment began to increase after fires ceased in 1922, and tree establishment peaked in the period (1940–1960) when livestock grazing was most intense (Jameson, 1994). Reduction of competition between grass and tree seedlings caused by livestock grazing can result in high tree seedling establishment (Cooper, 1960). The observed peak in recruitment may have occurred earlier since tree age is reported as age at coring height. The estimated average age to coring height was 17 years (range 10–29), so trees would be unlikely to shift more than one age class if true germination dates could be obtained.

The increase in tree density identified in GMNP is greater than in other mixed conifer forests in the southwest. For instance, Cocke et al. (2005) recorded tree densities of 820 trees ha^{-1} for the San Francisco Peaks, Arizona. In GMNP, the average density of three of the mixed conifer forest types exceeded 1000 trees ha^{-1}. The density difference may be related to differences in species composition. In the San Francisco Peaks, quaking aspen (*Populus tremuloides* Michx.) and limber pine (*Pinus flexilis* James) account for a large percentage of the forest density. In GMNP, small and young Douglas-fir and Gambel oak account for a large percentage of the density. Moreover, the high density of Douglas-fir in the understory will likely shift composition of the overstory from mixed dominance by southwestern white pine and ponderosa pine to Douglas-fir.

The forest density increase caused by elimination of fire may be a key factor contributing to a shift to a high-severity fire regime in GMNP mixed conifer forests. Two wildfires in GMNP (1990, 1994) burned large areas of mixed conifer forest at high-severity resulting in nearly 100% tree mortality in most places. These fires burned under extreme weather in dense surface and aerial fuels. Our fire history and forest structure data indicate that high-severity fires are unusual with respect to the pre-fire exclusion period. The degree to which climate and fuel accumulation each contribute to the occurrence of high severity fire cannot easily be evaluated in GMNP. GMNP and the entire southwest have experienced some of the hottest and driest years on record in the last decade, but there has also been a dramatic increase in forest fuels since Euro-American use of the land began in the early 20th century. Despite a fire exclusion period that is 40 or more years shorter than similar forests elsewhere in the southwest, there was a comparable shift in forest conditions with a new risk of high-severity fire (Allen et al., 2002).

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