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Fire history and fire–climate relationships along a fire regime gradient in the Santa Fe Municipal Watershed, NM, USA

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ABSTRACT

The Santa Fe municipal watershed provides up to 40% of the city's water and is at high risk of a stand-replacing fire that could threaten the water resource and cause severe ecological damage. Restoration and crown fire hazard reduction in the ponderosa pine (PP) forest is in progress, but the historic role of crown fire in the mixed-conifer/aspen (MC) and spruce-dominated forests is unknown but necessary to guide management here and in similar forests throughout the southwestern United States. The objective of our study was to use dendroecological techniques to reconstruct fire history and fire–climate relationships along an elevation, forest type, and fire regime gradient in the Santa Fe River watershed and provide historical ecological data to guide management. We combined systematic (gridded) sampling of forest age structure with targeted sampling of fire scars, tree-ring growth changes/injuries, and death dates to reconstruct fire occurrence and severity in the 7016 ha study area (elevation 2330–3650 m). Fire scars from 141 trees (at 41 plots) and age structure of 438 trees (from 26 transects) were used to reconstruct 110 unique fire years (1296–2008). The majority (79.0%) of fires burned during the late spring/early summer. Widespread fires that scarred more than 25% of the recording trees were more frequent in PP (mean fire interval $(\mathrm{MFI})_{25\%} = 20.8$ years) compared to the MC forest (31.6 years). Only 24% of the fires in PP were recorded in the MC forest, but these accounted for a large percent of all MC fires (69%). Fire occurrence was associated with anomalously wet (and usually El Niño) years preceding anomalously dry (and usually La Niña) years both in PP and in the MC forest. Fire in the MC occurred during more severe drought (mean summer Palmer Drought Severity Index; PDSI = −2.59), compared to the adjacent PP forest (PDSI = −1.03). The last fire in the spruce forest (1685) was largely stand-replacing (1200 ha, 93% of sampled area), recorded as fire scars at 68% of plots throughout the MC and PP forests, and burned during a severe, regional drought (PDSI = −6.92). The drought–fire relationship reconstructed in all forest types suggests that if droughts become more frequent and severe, as predicted, the probability of large, severe fire occurrence will increase.

1. Introduction

Large areas of forest throughout the southwestern United States (Arizona, New Mexico and adjacent areas) are unnaturally dense due a century of fire exclusion, and are consequently at high risk of historically unprecedented large crown fires (Covington and Moore, 1994; Allen et al., 2002). Given limited resources for treatment, a triage approach must be adopted to identify areas with high resource value or that are located strategically within the larger landscape. Historical ecological data describing the range of variability of disturbance regimes and their climatic controls are vital to guide forest restoration (Swetnam et al., 1999), particularly when facing the additional challenge of a changing climate (Millar et al., 2007).

The upper Santa Fe River watershed, New Mexico is arguably the most at risk, high-profile municipal watershed in the southwestern U.S. Santa Fe is the oldest state capital, founded on the Santa Fe River in the early 17th century (Debuys, 1985). Sitting at 2137 m elevation on the alluvial plane of a steep, forested, montane watershed, Santa Fe is inextricably linked to the ecosystem services (e.g., drinking water) and natural hazards (e.g., fire and floods) associated with the wildland urban interface. Surface water that originates high in the spruce-fir forests of the Pecos Wilderness Area is regulated through a system of reservoirs that provides up to 40% of the city's water supply (Grant, 2002). The population in Santa Fe County has tripled in recent decades (1970–2007; USCB, 2009), overtaxing the already limited water

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supply. Like much of the West, there has not been a widespread fire in the ponderosa pine (PP) and mixed-conifer (MC) forests of the watershed for 130 years, increasing the area at high risk of crown fire beyond the spruce-fir forests, where they naturally occur.

Recent, large, crown fires in near by watersheds have produced runoff and erosion events two orders of magnitude greater than pre-fire events (Veenhuis, 2002). This type of event in the Santa Fe watershed could destroy the water supply infrastructure and flood the historic heart of the city. The threat of catastrophic fire sparked years of contentious public debates, which ultimately led to U.S. Congressional earmarks of seven million dollars to fund planning and implementation of crown fire hazard reduction and forest restoration in the lower elevation PP forests (USDA, 2001). However, the ecological role of fire and the consequences of fire exclusion in the upper elevation mesic MC and spruce-fir forest types remain largely unknown, and it is these forests that cover the majority of the area that supplies the main reservoir.

1.1. Gradients: elevation, forest types and fire regimes

Gradients (e.g., elevation and vegetation) are common in terrestrial ecosystems and are a valuable way to study how ecological processes vary across a range of conditions (Whittaker and Niering, 1965; Whittaker, 1967). In the southwestern U.S. fire is a keystone ecological process that has affected vegetation across ecosystem gradients for hundreds to thousands of years (Swetnam and Baisan, 1996; Allen, 2002; Anderson et al., 2008). The size, frequency and severity of fire over time define the fire regime (Agee, 1993). Fire regimes are commonly classified by the extremes of the fire severity gradient (low severity or high severity). Recently, the term *mixed-severity fire regime* has been described as including a range of fire severities across a spatially complex mix of forest patches, including unburned, low, moderate, and high severity fire (Agee, 2005).

At landscape scales (1000–100,000 ha; watersheds to mountain ranges), fire can move across gradients of elevation, forest types and likely, between fire regimes. The PP forest type in the southwestern U.S. is a classic low severity, high frequency fire regime (Swetnam and Baisan, 1996). Subalpine spruce-fir forests in the southern Rocky Mountains exemplify the other extreme: a high severity, low frequency fire regime (Romme and Knight, 1981; Sibold et al., 2006). The steep topography of the southwestern U.S. juxtaposes these two forest types (representing the extremes of the fire severity gradient) in close proximity (<10 km separation) along a continuous elevation gradient with continuous fuels. MC forests are intermediately located between PP and spruce-fir (Dick-Peddie, 1993). Lower elevation, xeric, MC forests historically burned with low severity, but less frequently than PP (Dieterich, 1983; Brown et al., 2001). Some upper elevation, mesic MC forests have evidence of high severity fire (Fule et al., 2003; Margolis et al., 2007; Margolis, 2007). Historically, drought synchronized fire occurrence within and between low and high severity fire regimes regionally (Swetnam and Baisan, 1996; Margolis et al., 2007), but there is limited research examining connectivity between low and high severity fire regimes along a continuous forest gradient in a single, continuous landscape (Fule et al., 2003).

The implications of low and high severity fire regime connectivity are important given the well-documented changes associated with fire exclusion in ecosystems of the southwestern U.S. Over a century of fire exclusion in PP forests of the region has dramatically increased forest density and the risk of large crown fires (Covington et al., 1997; Allen et al., 2002). While there is historical evidence of high severity fire patches in some MC forests (Fule et al., 2003), increased forest density in other MC forests due to fire exclusion has increased the size of forest patches at risk of crown fire (Fule et al., 2003; Cocke et al., 2005; Heinlein et al., 2005; Margolis et al., 2007).

There is comparatively less information about the effects of fire exclusion on forest density, composition, and crown fire risk in the upper elevation spruce-fir forests of the region (Fule et al., 2003; Cocke et al., 2005). It is generally thought that a century of fire exclusion has not had dramatic impacts in these naturally dense forest types (Sibold et al., 2006), because high elevation, high severity forest fire regimes burn at long (centennial-scale) return intervals (Turner and Romme, 1994). Evidence of decreased fire frequency during the fire suppression period, compared to previous centuries has been observed in some subalpine forests of the Southern Rockies (Kipfmueller and Baker, 2000), but not others (Sibold et al., 2006). If forest ecosystems along steep elevation gradients are connected by fire spread across vegetation and fire regime gradients, then a century of fire exclusion in the lower elevation pine-dominated and MC forests is likely to have affected the high elevation, high severity forest fire regimes as well.

The semi-arid climate of the southwestern U.S. is highly variable, with frequent (2–7 years) wet/dry oscillations that are partially driven by multiple ocean-atmosphere oscillations, particularly the El Niño Southern Oscillation (ENSO; Diaz and Markgraf, 2000). Fire–climate analyses indicate that moisture variability largely explains patterns of fire occurrence in tree-ring reconstructed and contemporary records in low- and mid-elevation forests of the southwestern U.S. (Swetnam and Betancourt, 1990; Crimmins and Comrie, 2004). Warmer temperatures in recent decades have increased the length of the fire season, resulting in more large fires throughout the western U.S. (Westerling et al., 2006). The established link between climate variability and fire, coupled with predicted warmer global temperatures (IPCC, 2007) and drier conditions in the southwestern U.S. (Seager et al., 2007) has led to predictions of more large fires in the future (Westerling et al., 2006). Better understanding of the link between climate variability and fire occurrence along the elevation gradient of forest types and fire regimes is necessary to proactively manage our forests with a science-based approach, in the face of climate change.

The goal of the research is to provide essential historical ecological data across a gradient of forest types and fire regimes to guide management in the upper Santa Fe watershed and similar upper montane forest types in the region. Our first objective was to reconstruct fire history (frequency, severity, and size) along an elevation, vegetation and fire regime gradient in the upper Santa Fe Watershed. Our second objective was to reconstruct and compare historic fire–climate relationships between forest types. Our third objective was to test for evidence of direct connectivity of fire regimes along the fire severity gradient from low, to mixed, to high severity.

1.2. Study area

The study area encompasses the upper Santa Fe River watershed (7016 ha), which includes the headwaters located within the U.S. Forest Service Pecos Wilderness Area (Fig. 1). The watershed is located on the west slope of the Sangre de Cristo Mountains, northeast of the city of Santa Fe, NM, near the southern terminus of the Southern Rocky Mountains. The upper watershed has been closed to the public since 1932 to protect the water supply for the city of Santa Fe (USDA, 2001). Elevation ranged from 2237 m at the lowest point in the stream channel to 3847 m on the peaks that rise above tree-line and define the headwaters of the basin. Tree-ring samples were collected from 2328 m to 3650 m.

The climate is semi-arid and continental. Precipitation peaks during summer monsoon convective storms (July–August), and winter snowpack is common except during extreme drought years

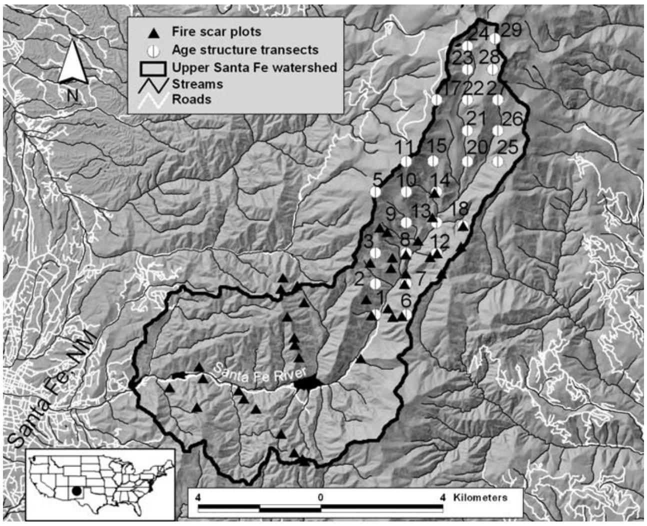

Fig. 1. Location of gridded age structure transects (numbered) and fire scar plots used to reconstruct fire history in the upper Santa Fe watershed, NM. The upper watershed, containing only age structure transects, is spruce-dominated forest. The lower watershed, containing only fire scar plots, is PP. The middle-elevation forest area where both age structure and fire scars overlap is MC.

(e.g., 2002). Temperature (1972–2005) at Santa Fe, NM (2060 m) ranged from an annual average minimum of 2.3 °C to an annual average maximum of 18.2 °C. Total annual average precipitation was 34.8 cm and total annual average snowfall was 44.2 cm (Western Regional Climate Center, www.wrcc.dri.edu). Fire occurrence records were available for 222 fires (1970–2003) from the ranger district containing the watershed and the adjacent district to the east (Española and Pecos/Las Vegas Ranger Districts). The majority (93%) of fires occurred between May and August, peaking in July, but monthly area burned peaks in May and June during the dry foresummer. Eighty percent of all fires were started by lightning (USFS, unpublished data).

Along the elevation gradient, forest types transitioned from PP dominated forests in the lower part of the study area, to MC in the middle elevations, to spruce-fir in the upper elevations. The spruce-fir type was composed of Engelmann spruce (*Picea engelmannii* Parry) and corkbark fir (*Abies lasiocarpa* [Hook] Nutt. var. *arizonica* [Merriam] Lemmon), but Engelmann spruce was dominant in all locations sampled and often present in pure stands. This general upper elevation forest type is hereafter referred to as "spruce-dominated." The MC forest was relatively diverse and species composition varied largely by aspect. The following species were present in various combinations in this forest type, listed in order of abundance of the dominant tree: Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco.), ponderosa pine (*Pinus ponderosa* Lawson), southwestern white pine (*Pinus strobiformis* Engelm), quaking aspen (*Populus tremuloides* Michx.), white fir (*Abies concolor* [Gord. and Glend.] Lindl. Ex Hildebr.), and Engelmann spruce. There were no conspicuous, large (>50 ha) stands dominated by quaking aspen that might indicate recent stand-replacing fire patches. The lower section of the study area was dominated by ponderosa pine, with associated species including Colorado pinyon pine (*Pinus edulis* Engelm.), Rocky Mountain juniper (*Juniperus scopulorum* Sarg.), Gambel oak (*Quercus gambelii* Nutt.), Douglas-fir, white fir, and southwestern white pine.

2. Methods

2.1. Tree-ring fire history methods

A combination of tree-ring methods was necessary to reconstruct fire history along the elevation, vegetation and fire regime gradient. Fire scar-based methods (Dieterich and Swetnam, 1984; Swetnam and Baisan, 1996) were used to reconstruct surface fire frequency, seasonality, and extent for the PP and dry MC portions of the watershed where fire scars were present. However, in the upper elevation spruce-dominated and mesic MC forests, fire-scarred trees were rare or non-existent because: (1) high severity, high intensity, stand-replacing crown fires destroy (kill and burn) direct tree-ring evidence of past fires, (2) the thin bark of spruce and fir species is more susceptible to being fatally girdled, even by low-intensity fire, thus leaving no evidence of the most recent fire (e.g., fire scars), and/or (3) long fire return intervals may allow the rare fire-scarred trees to heal over so that open fire scar wounds are not visible.

In forest types where fire scars are not abundant, age structure-based fire history methods are commonly applied (Heinselman, 1973; Agee, 1993; Johnson and Gutsell, 1994). These methods largely rely upon the establishment dates of tree cohorts that regenerate following stand-replacing fire events to date the fire and determine aerial extent of the stand-replacing patches. Thus, in the spruce-dominated forest type we used age structure sampling methods to reconstruct the age of the dominant, presumably oldest trees, thereby estimating the time since the last fire (Kipfmueller and Baker, 1998). Satellite imagery, aerial photography, and field observations were used to identify any potential post-fire forest patches.

Age structure data alone may not be sufficient to determine if the forest patch was a post-stand-replacing fire cohort and ultimately date the fire. Unlike lodgepole pine (*Pinus contorta*) or quaking aspen, spruce and fir trees may take years to decades to regenerate following stand-replacing fire (Antos and Parish, 2002). This is likely due to a combination of variability in seed sources, dispersal, and post-fire weather and climate. The precision of fire dates derived strictly from forest age will not be annual because of lagged regeneration. Decadal precision of fire dates can be sufficient when calculating area-based estimates of fire frequency (e.g., natural fire rotation; Heinselman, 1973), due to the long return intervals (100 years to >400 years) of forest crown fire regimes (Turner and Romme, 1994). Annually precise stand-replacing fire dates may be reconstructed if fire scars, fire-killed trees or injured trees are present in adjacent forest stands, or on the perimeter of unburned patches (Johnson and Gutsell, 1994; Margolis et al., 2007). Annually resolved fire dates are necessary for inter-annual fire–climate analyses, which can provide specific climate information associated with the relatively rare, but important, stand-replacing fire events.

In forest types such as MC, where a combination of low severity and high severity fire occurs (i.e., mixed-severity fire), it is necessary to use a combination of fire scar and age structure-based methods to reconstruct the fire history (Agee, 2005). For example, some stands within the MC zone had no fire scars or potentially fire-killed trees, and stand age was the best evidence of past fire. Alternatively, other stands had abundant fire scars (i.e., evidence of repeated, low severity surface fire) and no clear post stand-replacing fire cohorts.

2.2. Sampling design

In the PP forests of the lower portion of the study area we used a targeted approach to locate and collect fire scars. Targeted fire scar sampling in Southwestern ponderosa pine provides similar estimates of fire frequency compared to systematic sampling, particularly for widespread fires, with the added benefit of providing a longer record (Van Horne and Fule, 2006). Samples were primarily collected at 50 m-radius plots along two transects; a north-facing transect, and a south-facing transect. These transects were located in the middle of the PP zone and followed a series of ridges that extended from the river up to the respective watershed boundaries. From the ridgetop location we searched both slopes that descended from the ridge and adjacent slopes that could be seen from the ridge. Additional plots were located within the PP zone to provide broader spatial coverage of the PP forest fire history. One group of plots was located west (downstream) of the transects in the area surrounding the lower reservoir. An additional plot was located east (upstream) of the transects, above the second reservoir. The resulting spatial patterning of the plots was determined by a combination of our search effort, topography and the presence of fire-scarred material.

In the high elevation spruce-dominated forests, a systematic, gridded age structure sampling approach provided the best evidence of fire history (i.e., "time since last fire"; Johnson and Gutsell, 1994). We generated a 1 km grid beginning with a random location in the study area (Fig. 1). The grid was oriented along cardinal directions to facilitate navigation in the field. Two grid points (24 and 28) initially fell within unforested vegetation types and were relocated 50 m inside the nearest forested area.

In the middle-elevation, MC forest evidence of fire was present as both fire scars and post-fire tree regeneration cohorts. We extended the 1 km-spaced age structure grid into the MC zone, and because fire scars were only present at 5 of 12 MC age structure transects we used a targeted approach to locate and sample fire scars in this forest type. In the MC forest, fire-scarred trees were most abundant on the relatively flat ridges, apparently because of lower fire severity that allowed trees to survive fires that were otherwise stand-replacing on the adjacent steep slopes. We searched and sampled ridges with the goal of obtaining a relatively even spatial distribution of fire scar plots and to maximize the length of the fire history record. The final spatial distribution of the fire scar sample plots was ultimately determined by the location of fire-scarred trees, in part determined by topography and chance, and therefore is not evenly distributed in space.

In the topographically complex mountains of the semi-arid southwestern U.S., elevation and aspect can be important variables mediating vegetation type (e.g., Whittaker and Niering (1965)) and fire regimes (e.g., Iniguez et al. (2008)). To ensure that the distribution of aspect class (N, S, E, W) at our gridded, age structure transects was proportional to the relative abundance of aspect classes in the study area we stratified the sampling grid by aspect class. The percent of sample points in the four primary aspect classes was distributed similar to the percent of land area in each aspect (Table 1), with a slight over-sampling of east-facing slopes and under-sampling of the south-facing slopes.

2.3. Field sampling

Where multiple fire-scarred trees were present we used a plot-based field sampling approach. A plot was sampled where two or more fire-scarred trees were located less than 15 m apart. The plot center was located between the samples. Samples from multiple fire-scarred trees were collected within a 50 m search radius that defined the plot. Collecting multiple trees within a plot increased the probability of recording all fires that actually occurred in that area. This is necessary because trees are imperfect recorders of fire and individual trees may not record all fires (as fire scars) that burned around the tree (Dieterich and Swetnam, 1984). Wedges and cross-sections were collected from fire-scarred logs, stumps and rarely from live trees with a cross-cut saw in the MC forest (within the Pecos Wilderness Area) and with a chainsaw in the PP forests using standard procedures (e.g., Arno and Sneck (1977)).

To determine stand age at the gridded age structure transects in the spruce-dominated and MC zones we sampled the 20 largest (diameter at breast height (dbh)) trees along a 100 m by 20 m belt transect. The transect was centered on the grid point and the long axis was oriented parallel to the contour of the slope (i.e., sideslope). To determine tree age, increment cores were collected as close to the base of the tree as possible (<0.3 m). We angled the borer down to intersect the estimated location of the root crown in an attempt to sample all the years of tree growth. We re-sampled trees until we extracted a core containing rings estimated to be within 10 years of the pith.

2.4. Lab methods

All tree-ring samples were sanded with progressively finer sandpaper until the ring structure was visible and then crossdated using standard dendrochronological procedures (Stokes and Smiley, 1968). For fire scar samples, we determined the calendar year of the scar and the season of fire occurrence by analyzing the relative position of each scar within the annual growth ring: dormant season, early earlywood, middle earlywood, late early-wood, latewood, or unknown (Baisan and Swetnam, 1990). Predominant occurrence of spring or early summer fires in northern New Mexico and the southwestern U.S. is widely supported by fire seasonality data from observed 20th century fires in the region (Barrows, 1978), locally, and from hundreds of tree-ring reconstructed fires (Swetnam and Baisan, 1996). Based on our observations and conventional season of montane fire occurrence in the region, all fire years with fire scars recorded only in the dormant season were assigned to the spring/summer of the next year (ring).

For age structure samples, we estimated the date of the first year of growth (pith) for increment cores that did not contain the pith ring, using a concentric circle pith estimator (Applequist, 1958). Cores that were estimated to be greater than 30 years from the pith ring or that had no curvature in the inner rings were not included in the age structure data. Because cores were collected at a downward angle to intercept the root crown, the error associated with the age to core height was assumed to be within the resolution of the age class bins (10 years) and was not estimated.

A qualitative description of the initial tree-ring growth of cored trees (open, average, or suppressed) was recorded to provide information regarding the growth environment when trees established (Romme and Knight, 1981). Spruce and fir species are shade tolerant and are able to survive in low light conditions under canopies, but the growth rates in these conditions can be very slow (i.e., "suppressed"). Growing conditions for trees germinating in an open forest, such as following a stand-replacing fire, would be more favorable and should be indicated as relatively wide initial ring widths (i.e., open). This information was combined

Table 1
Aspect class of land area and age structure transect grid in the MC and spruce-dominated forests.

Aspect class	Area (ha)	Area (% of total)	Age structure transect (#)	Age structure transect (% of total)
Flat (0% slope)	0.22	0.01	0	0
N (315–45°)	221.11	8.29	3	11.54
E (45–135°)	773.59	28.99	9	34.62
S (135–225°)	852.73	31.95	6	23.08
W (225–315°)	821.03	30.77	8	30.77

with tree ages and fire scar dates to determine if trees were likely part of post stand-replacing fire cohorts.

2.5. Data analysis

The fire scar data were entered into a database and analyzed using FHX2 software (Grissino-Mayer, 2001). Because fire scar return intervals are rarely normally distributed and more commonly fit a Weibull distribution (Grissino-Mayer, 1999), we tested for the fit of the Weibull model (Kolmogorov–Smirnov (K-S) test) and estimated the Weibull Median Probability Interval (WMPI). Central tendency parameters (mean, median and WMPI) of fire frequency were calculated for five "filtered" subsets of the composite fire history data for (1) the PP forest and (2) the MC forest. The following filtered subsets of reconstructed fires were used for the analysis: (1) all fires, (2) fires recorded by a minimum of 2 trees, (3) a minimum of 2 trees and >10% of recording trees, 10% scarred, (4) a minimum of 2 trees and >20% of recording trees, 20% scarred and (5) fires recorded by a minimum of 2 trees and >25% of recording trees, 25% scarred. "Recording trees" refers to previously fire-scarred samples that have intact wood (i.e., not burned away or missing pieces) and an open wound (not covered by bark) during the time period in question. Many montane conifers have thick bark that protects trees from damage to the cambium by fire. These full-bark trees may not record fires as fire scars, while the same fire is recorded on adjacent trees with pre-existing open "cat face" fire scar wounds.

Filtering the fire scar data by the percent of recording trees scarred is used to infer relatively large, spreading fires, as compared to less widespread fires that only scar a relatively small number (percent) of trees (see discussion in Swetnam and Baisan, 2003). Widespread fires are thought to be more ecologically important because of the extent of the effects. Too few fire-scarred trees were present on the landscape and/or collected to confidently allow plot-based fire interval analysis (e.g., Iniguez et al., 2008). In addition, high severity fire in parts of the MC forest killed and burned evidence of prior fires at individual plots, so fire dates from all plots were combined to make a site composite for each forest type (Dieterich, 1980). We also chose not to analyze fire intervals for individual trees (point intervals), because our attempt to extend the record back in time by targeting remnant wood resulted in many samples having an incomplete record due to being burned and/or eroded. This was particularly the case in the MC zone of the wilderness area, where a majority of samples were remnant wood. Given these limitations of a relatively long record, we still are confident that the percent of trees recording fire is a good indicator of widespread vs. localized fires and that the widespread fires that we focus on are the most robust to variability in sampling (Van Horne and Fule, 2006). Because fire intervals vary over time with changes in fuels and climate (e.g., Swetnam (1993)), central tendency statistics (e.g., MFI) oversimplify historic fire regimes. We report additional statistics (e.g., minimum and maximum fire intervals) and interpret these data in terms of fire management to provide a better understanding of the historic range of variability of the fire regime.

To test for differences in historical fire frequency between the PP and MC forests we used the Student's *t*-test to compare MFI, the Folded-*f* test to analyze differences in variance, and the K-S test to analyze differences in distributions (FHX2, Grissino-Mayer, 2001). Because these tests assume that the data are normally distributed, the data are transformed to the standard normal distribution (i.e., mean of zero and a standard deviation of one) before the comparisons (Grissino-Mayer, 2001). To quantify synchrony of burning (i.e., fire spread) between the PP and the MC forests we counted the number of coincident fire years between the two forest types, and calculated the percent of all fire years in each forest type that were synchronous between forest types. As a more robust test of synchrony we used Chi-squared analysis to test for independence between MC fire years and PP fire years (1550–1880) for all filtered subsets of fire years.

We used superposed epoch analysis (SEA; Baisan and Swetnam, 1990) to test for inter-annual relationships between variability in four tree-ring reconstructed measures of climate and fire occurrence in (a) the PP forest and (b) the MC forest. The tree-ring reconstructed climate variables included (1) Palmer Drought Severity Index (PDSI), (2) annual precipitation from El Malpais, NM, (3) an index of El Niño/Southern Oscillation (ENSO), and (4) an index of the Pacific Decadal Oscillation (PDO).

PDSI is a commonly used measure of available moisture (Palmer, 1965). Summer (June–August) PDSI is a good indicator of moisture conditions prior to and during the southwestern U.S. fire season and is highly correlated with variability in historical fire occurrence (Swetnam and Baisan, 2003) and 20th century fire occurrence (Crimmins and Comrie, 2004). A 2.5° gridded tree-ring reconstruction of summer PDSI exists for much of North America and in the southwestern U.S. it extends hundreds of years prior to the 20th century instrumental climate data (Cook et al., 2004). PDSI gridpoint 133 is nearest to our study site and is used in the SEA analysis. A tree-ring based precipitation reconstruction from El Malpais National Monument, in west-central NM (Grissino-Mayer, 1996), was also used as a sub-regional climate variable.

Indices of Pacific Ocean-atmosphere oscillations (e.g., ENSO and PDO) that have been shown to affect climate variability in the southwestern U.S. (Diaz and Markgraf, 2000; Brown and Comrie, 2002) were also used as variables in the SEA analysis. As a proxy index for ENSO we used the tree-ring reconstructed Niño3 index (Cook, 2000) of winter (December–February) sea surface temperature (SST) from the eastern equatorial Pacific Ocean (5°N–5°S, 90°–150°W). Positive (negative) Niño3 index values represent warm (cool) SST's - El Niño (La Niña).

We used the (D'Arrigo et al., 2001) annual PDO index reconstruction derived from temperature sensitive tree-ring sites from coastal Alaska (5) and Oregon (1), and two tree-ring reconstructed PDSI grid points in northern Mexico. Positive (negative) index values of PDO correspond with warm (cold) phases of the primary mode of variability in Pacific Ocean SST's polewards of 20°N (Mantua et al., 1997).

To test whether drier conditions were associated with fire in the MC forest than in PP, we compared mean PDSI during widespread (25% scarred) and "all fire" years with a *t*-test. To test whether widespread fires occurred on drier years than "all fire" years we compared mean PDSI between fire years for each vegetation type with a *t*-test.

3. Results

3.1. Fire scars—PP

In the PP zone (1600 ha search area) we crossdated a total of 442 fire scars from 76 trees at 20 locations, for a total of 99 unique fire years (Tables 2 and 3). The PP fire scar record covers 709 years (1296–2004), with fire scars recorded from 1331 to 1966 (Fig. 2, Table 2). The period 1550–1880 was chosen for fire interval analysis as the best compromise between record length and sample depth.

The season of fire occurrence was determined for 331 (75%) of the fire scars (Table 4). The remaining fire scars were in poor condition or were in rings too narrow to accurately determine the season. When fire scar season could be determined, the most frequent occurrence (69%) was in the dormant (D) season (i.e., between ring boundaries). All but 3% of the remaining fire scars were recorded in the earlywood (E) portion of the ring and the

Table 2
Fire scar record statistics.

Forest type	Search area (ha)	Plots (#)	Fire-scarred trees (#)	Fire scars (#)	Unique fire years (#)	Full record (years)	Fire scar record (years)	Fire interval analysis (years)
PP	1600	20	76	442	99	1296–2004	1331–1985	1550–1880
MC	1200	21	65	139	35	1337–2008	1339–1879	1495–1880
Spruce-dominated	1200	26	0	–	–	–	–	–

Table 3
Upper Santa Fe watershed fire scar dates (all fires). Fire years recorded in both forest types indicated in bold.

Century	<1500	1500s	1600s	1700s	1800s	1900s
PP (1296–2004)	1331, 1398, 1415, 1434, **1445**, 1479, **1495**	1503, **1516**, **1522**, 1532, **1542**, 1551, 1558, **1562**, 1573, 1580, **1587**, 1591	1600, 1604, 1605, 1606, **1608**, 1612, 1616, 1617, **1619**, **1622**, 1623, **1624**, 1628, 1631, 1633, 1636, 1638, 1642, 1644, 1646, 1648, **1654**, 1656, 1659, 1661, 1664, 1672, 1676, 1683, **1685**, 1687, 1696,	**1700**, 1705, **1715**, 1719, 1724, 1725, **1729**, **1737**, 1739, 1742, **1748**, 1763, **1773**, 1778, 1779, 1784, 1786, 1788, 1794, **1795**	1803, 1805, 1808, 1809, 1810, 1814, **1819**, 1823, 1825, 1826, 1831, 1835, **1842**, 1858, **1860**, 1864, 1867, 1877, **1879**, 1883, 1885, 1886, 1893	1902, 1904, 1911, 1931, 1946, 1966
MC (1337–2008)	1399, 1444, **1445**, **1495**	1500, **1516**, **1522**, **1542**, 1546, **1562**, 1579, **1587**, 1599	**1608**, 1614, **1619**, **1622**, **1624**, **1654**, **1685**	**1700**, **1715**, 1716, **1729**, 1730, **1737**, **1748**, **1773**, **1795**	**1819**, 1820, **1842**, 1857, **1860**, **1879**	

Table 4
Fire scar seasonality reconstructed from the relative position of the fire scar in the tree-ring. Period of record: PP, 1296–2006 and MC, 1337–2008.

Scar position	Number of fire scars (PP/MC)	Percent of scars with season determined (PP/MC)
Dormant	229/39	69.2/42.4
Early earlywood	40/27	12.1/29.3
Middle earlywood	35/15	10.6/16.3
Late earlywood	18/11	5.4/12.0
Latewood	9/0	2.7/0.0

majority of those were in the first third of the earlywood (early earlywood, EE). The remaining 3% of the fires were recorded in the latewood (A) portion of the tree-ring. Overall, 81% of the fires in the PP zone were burning in the beginning of the growing season (May or June; D or EE).

The fire frequency of the reconstructed PP fire regime was highly variable through time (Fig. 2), and cannot adequately be described by one metric (e.g., MFI). The fire interval data (1550–1880) were not normally distributed (K-S d-statistic = 0.438, $p < 0.001$) and were fit with the Weibull model (K-S d-statistic = 0.132, $p = 0.144$). Increasingly exclusive filters increased the fire interval central tendency statistics by eliminating the (small) fires recorded by only a few trees, such that the WMPI increased from 3.8 years (all fires) to 18.8 years (25% scarred; Table 5). MFI was similar and ranged from 4.3 years (all fires) to 20.8 years (25% scarred). Thus, somewhere within the 1600 ha PP search area there was a fire recorded by at least one tree approximately every four years, on average, and relatively widespread fires scarring more than 25% of the trees occurred approximately every 18–21 years, on average. The minimum fire interval ranged from 1 year (all fires) to 7 years (25% scarred). The maximum fire interval ranged from 16 years (all fires) to a fire-free period of 63 years (1779–1842, 25% scarred). No widespread fires (25% scarred) occurred in the 20th century.

3.2. Fire scars—MC

In the mixed–conifer/aspen forests (1200 ha search area) we crossdated a total of 139 fire scars from 65 trees at 21 locations, for a total of 35 unique fire years (Tables 2 and 3). The MC fire scar record covers 672 years (1337–2008) with fire scars recorded between 1399 and 1879 (Fig. 2, Table 2). The period from 1495 to 1880 was chosen for fire interval analysis.

The season of fire occurrence was determined for 92 (66%) of the fire scars (Table 4). Based on the observed dominance of earlywood fires and the absence of latewood fires we used the same convention as in the ponderosa zone to assign fires only recorded in the dormant season to the spring/summer of the next year ($n = 7$). The majority (72%) of the fire scars dated to the season in the MC zone were burning in the spring or early summer (May or June; D or EE).

The MC fire interval data (1495–1880) were fit with the Weibull model (K-S d-statistic = 0.103, $p = 0.897$). The WMPI ranged from 10.3 years (all fires) to 27.8 years (25% scarred, Table 5). MFI was similar and ranged from 12.4 years (all fires) to 31.6 years (25% scarred). Minimum fire intervals ranged from 1 year for all fires, to 6 years for widespread (25% scarred) fires. Maximum fire intervals ranged from 31 years for all fires, to 94 years for widespread fires. No widespread fires (25% scarred) occurred in the 20th century. Further comparisons of fire intervals among the 5 filtered datasets and between vegetation types are discussed later in the paper.

Table 5
Fire interval analysis statistics for the PP (1550–1880) and the MC forests (1495–1880) for five filtered subsets of fire years (e.g., 20% = fires recorded by >20% of the recording trees).

Filter	Intervals (#) PP/MC	Mean fire interval (years) PP/MC	Median fire interval (years) PP/MC	Weibull median probability interval (years) PP/MC	Minimum interval (years) PP/MC	Maximum interval (years) PP/MC
All fires	76/31	4.3^{+}/12.4^{+}	4.0/12.0	3.8/10.3	1/1	16/31
>2 Trees	48/18	6.8^{+}/21.3^{+}	5.0/16.5	5.8/18.9	1/6	20/71
10%	34/18	9.1^{+}/21.3^{+}	7.0/16.5	8.0/18.9	1/6	25/71
20%	17/14	17.1^{+}/27.4^{+}	15.0/22.5	15.0/24.4	7/6	63/94
25%	14/11	20.8/31.6	15.5/25.0	18.8/27.8	7/6	63/94

$^{+}$ Indicates significantly different ($p < 0.05$) MFI between PP and MC (Student's t-test).

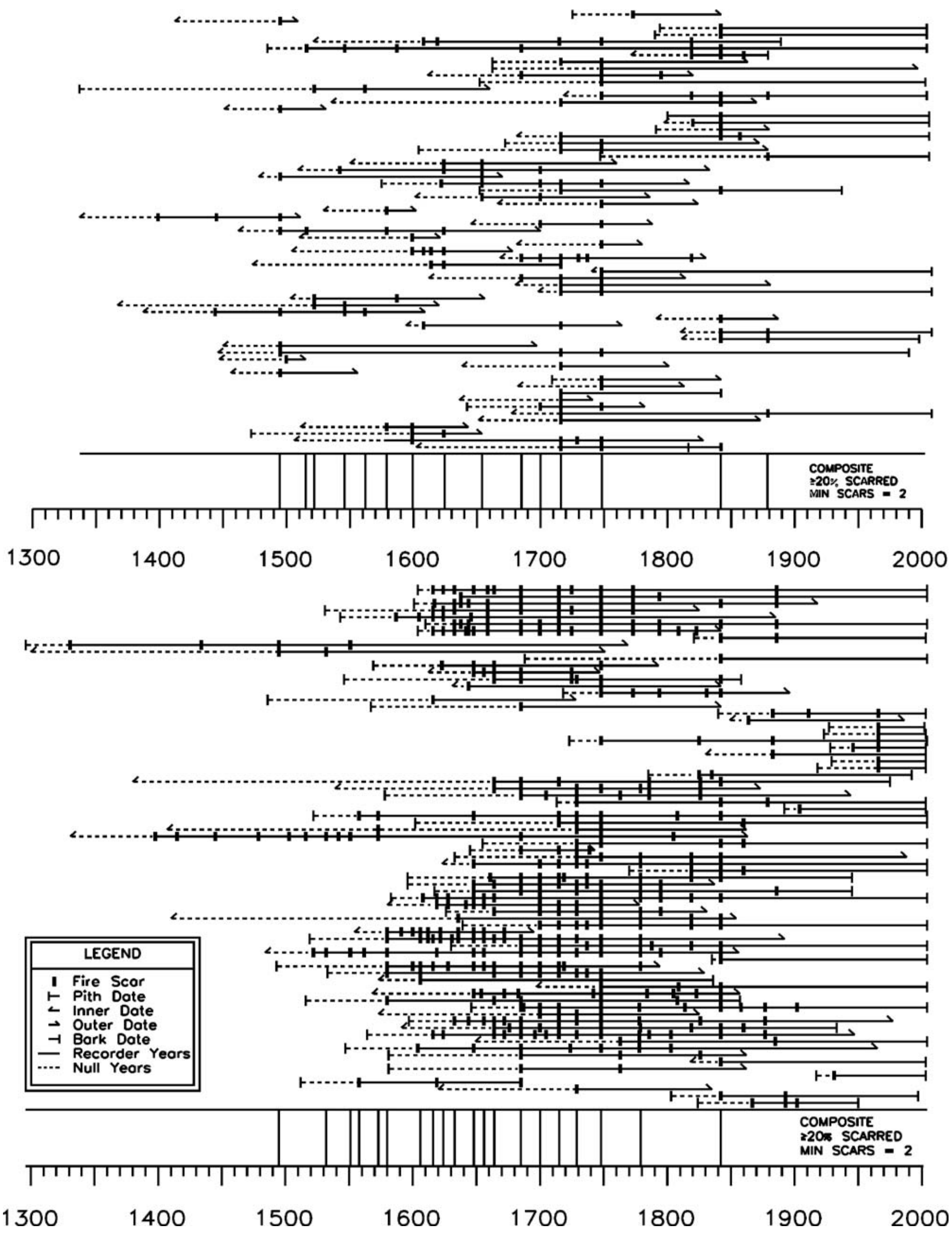

Fig. 2. Historical fire occurrence recorded by fire scars (1296–2004) in the PP forest (bottom) and MC forest (top) of the Santa Fe watershed. Each horizontal line is a tree and each vertical line is a dendrochronologically crossdated fire scar. The fire occurrence composite (bottom of each fire chart) indicates "widespread" fires recorded by a minimum of 2 trees and at least 20% of the trees recording fire.

3.3. Fire scars—spruce-dominated forest

No fire scars or any other direct evidence of fire (e.g., charred wood) were encountered at or between the age structure transects in the spruce-fir zone (1200 ha search area). Fire history in this vegetation type is presented in the age structure section.

3.4. PP vs. MC

The number of fire scars and individual fire years in the PP zone was approximately three times greater than that in the MC forest (Table 2). Historic fire intervals (1550–1880) in the PP zone were significantly shorter than in the MC forest for four of the five filtered subsets of fire years (all fires, $\geq$2 trees scarred, 10% scarred, and 20% scarred, Table 5). Although the MFI in the MC zone for the 25% scarred class (31.6 years) was approximately 10 years longer than in the ponderosa zone (20.8 years), the values were not statistically different (Student's t-test with equal variance, t-statistic = -1.780, $p = 0.092$).

Twenty-four fire years were synchronous between the two forest types (Table 3). Multiple synchronous fire years occurred every century from the 1400s to the 1800s. The number of synchronous fire years was greater than that would be expected by chance for all fire years ($\chi^2 = 39.22$, $p < 0.005$) and widespread (25% scarred) fire years ($\chi^2 = 29.15$, $p < 0.005$, with Yates correction for continuity). Sixty nine percent of all fires in the MC forest were also recorded in the PP zone. Only 24% of all fires in the PP forest were recorded in the MC zone.

3.5. Age structure

All of the age structure transects were located in the MC and spruce-dominated forest. Age structure transects were classified as spruce-dominated ($n = 14$) if the plurality of dominant trees was

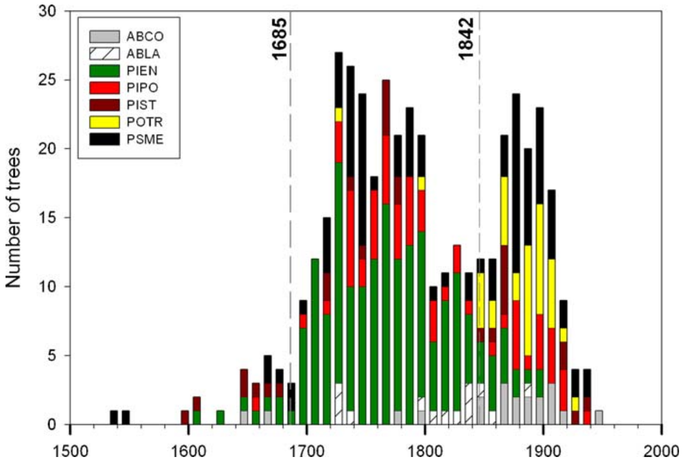

Fig. 3. Age structure and species composition of the dominant trees from the MC and the spruce-dominated forests. Data are from all age structure transects, in 10-year classes (plotted on the last year of the decade), and presented as estimated pith dates. Quaking aspen (POTR) was sub-dominant, but was sampled as a potential indicator of high severity fire. The last widespread fires with a stand-replacing component in the spruce-dominated (1685) and the MC forests (1842) are indicated as dashed lines.

Engelmann spruce. The remaining transects were classified as MC ($n = 12$). We collected 594 cores from 488 trees at 26 age structure transects (Fig. 1). We were not able to collect cores from all 20 dominant trees at 5 transects due to decomposed wood near the tree center and inclement weather. We were able to estimate pith dates for 438 (90%) of the sampled trees. Cores from the remaining 10% of the trees had no curvature in the inner rings or were estimated to be greater than 30 years from the pith so the number of rings to pith could not be estimated. The major cause for inadequate cores for pith estimation was decomposed wood near pith.

The collective age structure of dominant trees at all 26 transects in the MC and spruce-dominant forest has two recruitment peaks (i.e., a bi-modal distribution, Fig. 3). Less than 3% of the dominant trees established prior to 1650. A change in recruitment occurred in the late 1600s, increasing from a local minima of three trees (1681–1690) to the mode of 27 trees only 40 years later (1721–1730). This recruitment peak is dominated by Engelmann spruce. A second major tree recruitment pulse occurred in the mid-1800s. This younger recruitment peak is dominated by MC species. The recruitment peaks follow the last widespread fires in the MC (1842) and the spruce-dominated forests (1685) and there are relatively few trees dating to the decades prior to these two widespread fires.

The age structure at the individual transects illustrates both commonality and variability within and between the MC and the spruce-dominated forests (Figs. 4 and 5). The youngest MC stands all established after 1850 (1, 2, 6, 7) and were located nearest to the PP zone. The two oldest MC stands established circa 1600 (14, 18) and were located on rocky, relatively fire-protected sites near the upper MC/spruce ecotone. The youngest spruce-dominated stand began regenerating in the 1760s and the oldest trees date to the 1530s. The average age of the dominant trees in the spruce-dominated stands (mean [median] estimated pith date = 1769 [1763]) was approximately 60–100 years greater than in the MC forest (1829 [1861]).

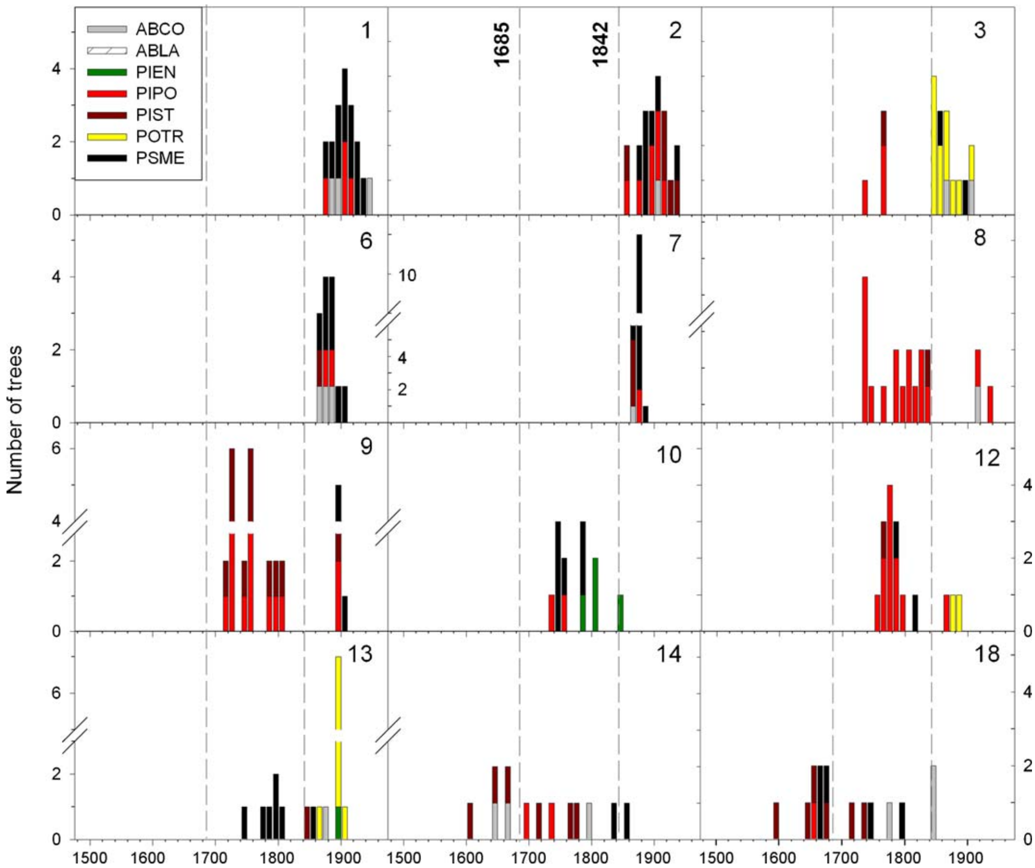

Fig. 4. Age structure and species composition of the dominant trees at individual age structure transects (e.g., 1) from the MC forest. Tree age data (estimated pith dates) are in 10-year classes. Note different scale for transects 7, 9, and 13.

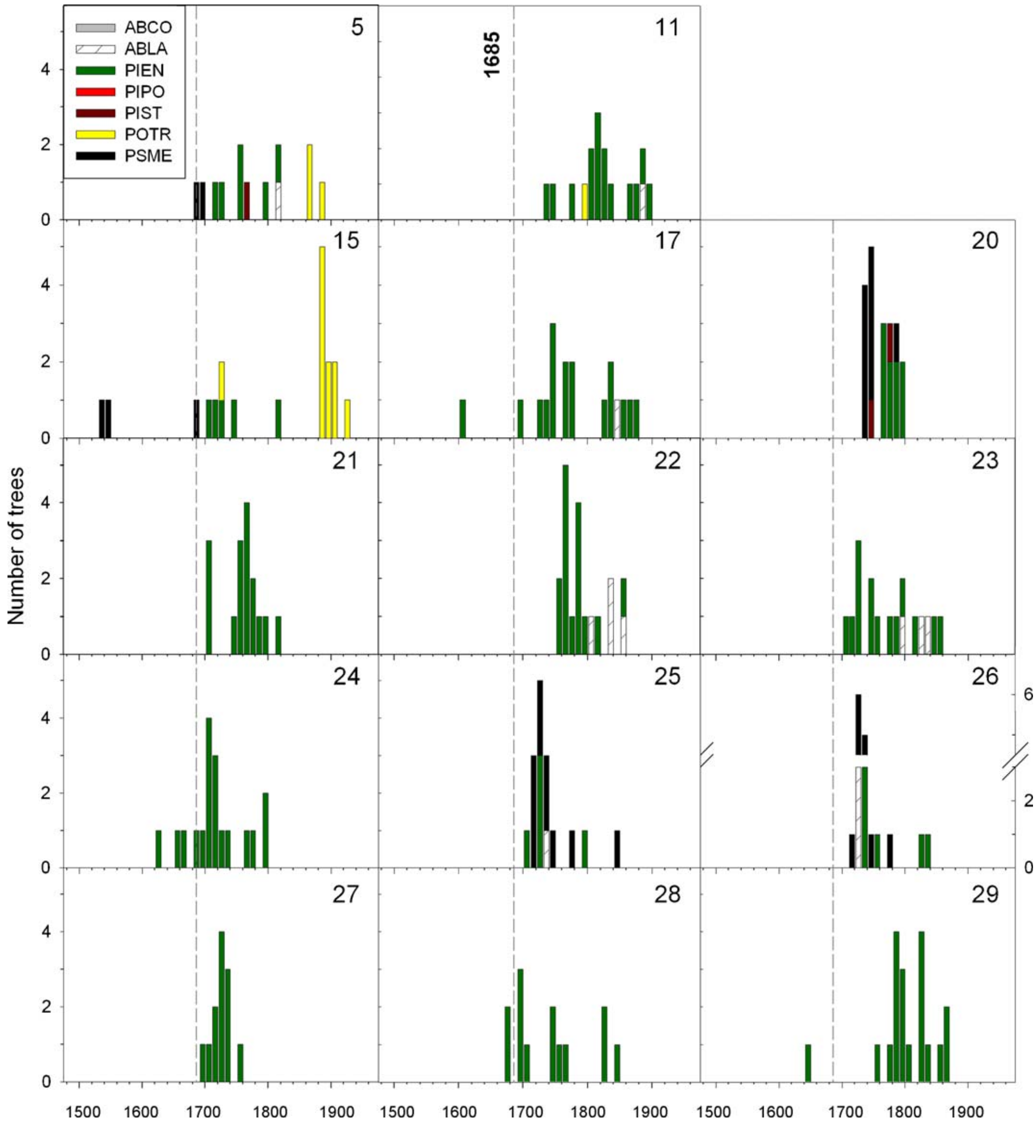

Fig. 5. Age structure and species composition of the dominant trees at individual age structure transects (e.g., 11) from the spruce (co-) dominated forest. Tree age data (estimated pith dates) are in 10-year classes. The lack of trees pre-dating the 1685 fire at 9 of the 14 transects suggests that this fire was largely stand-replacing in the upper, spruce-dominated portion of the watershed. Four of the five transects with trees surviving the fire (15, 17, 24, 28) had growth changes or injuries (i.e., traumatic resin ducts) in the tree-rings in 1685 (see Fig. 6).

3.6. Evidence of stand-replacing fire

The 1685 fire was recorded as fire scars by 57% (n = 35) of the recording fire-scarred trees at 68% (n = 19) of the recording fire scar plots throughout the MC and PP zones. Nine of the 14 spruce-dominated age structure transects and 10 of the 12 MC transects had no living trees that pre-date 1685. Four out of the five spruce-dominated transects that pre-date 1685 (15, 17, 24, and 28) had trees with growth changes or injuries/resin ducts in the tree-rings in 1685 (e.g., Fig. 6). The combination of age structure, growth changes/injuries, and widespread fire scar evidence indicates that the 1685 fire was relatively large and stand-replacing in the upper elevation forest.

The interpolated area of the 1685 fire within the upper Santa Fe watershed based on the spatial distribution of tree-ring evidence was 4730 ha. Approximately 25% of the reconstructed fire area was stand-replacing (1200 ha), all within the spruce-dominated zone (Fig. 7). It is likely that some of the younger forest stands below the spruce-dominated zone also burned with stand-replacing severity in 1685, but subsequent fires killed and burned any evidence of prior post-fire cohorts. We were conservative when reconstructing fire area and included these younger age structure transects as "not recording." The gaps between polygons in the reconstructed 1685 fire area are likely due to this lost record of fire.

Other fires that were widespread throughout the watershed (i.e., recorded by >50% of recording fire scar plots in the MC and PP forests, 1748, 1842; Fig. 2) were not recorded in the spruce-dominated forest. Age structure transects with many trees that

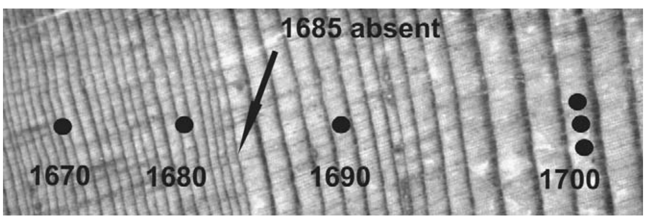

Fig. 6. Tree-ring growth release in a Douglass-fir core inferred to be a result of reduced competition due to tree mortality following the 1685 high severity fire.

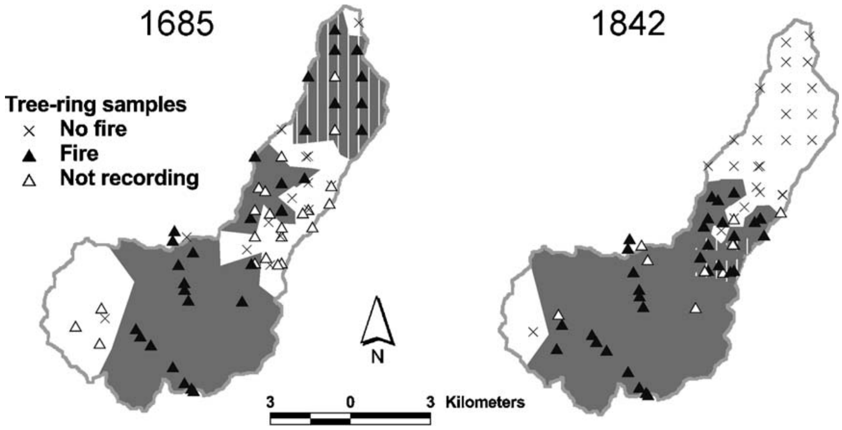

Fig. 7. Reconstructed fire area (gray) derived from thiesen polygon interpolation of tree-ring fire history data (fire scars, death dates, growth changes/injuries, and forest age structure). Areas with vertical white lines indicate stand-replacing fire patches.

pre-date these fires suggest that although these fires were widespread in the mid-elevation MC and lower pine forests, climate and/or fuel conditions were not suitable for fire spread into the mesic upper elevation spruce-dominated forest. It is possible however, that some widespread fires (e.g., 1716) may have burned with localized stand-replacing severity in the lower spruce-dominated forest and may explain the lack of trees in the early 1700s at some transects (e.g., 20).

3.7. Mixed-severity fire

There was evidence of mixed-severity fire in the MC zone. We use the term "mixed-severity" to indicate that some forest stands experienced high severity, stand-replacing fire (recorded as a tree recruitment pulse with no surviving trees) and other, adjacent stands experienced low-severity surface fire (recorded as fire scars). The landscape structure in the lower MC zone is such that north- and south-facing slopes are located on opposite sides of ridges. The youngest stands in the watershed (transects 1, 2, 6 and 7; Figs. 1 and 4) were on the more productive north- and east-facing slopes in this zone, near the ecotone with PP. These stands established in the mid-to-late 19th century and had the highest percentages of trees with "open" inner-ring growth (85–95%).

The 1842 fire was recorded as fire scars by 82% ($n = 24$) of the recording plots and 57% ($n = 42$) of the recording fire-scarred trees in the MC and PP forests. In addition to the four transects with no trees surviving the 1842 fire, three transects (3, 9, and 12) had growth changes or injuries/resin ducts in the tree-rings in 1842. Transect three had an aspen recruitment pulse beginning immediately following 1842 and the dominant PP trees that survived the fire had multi-year growth suppressions in the tree-rings beginning in 1843. The fire scar plot located less than 200 m southwest of age structure transect six had no samples post-dating 1842 and one of the fire-scarred trees had a bark-ring date of 1841.

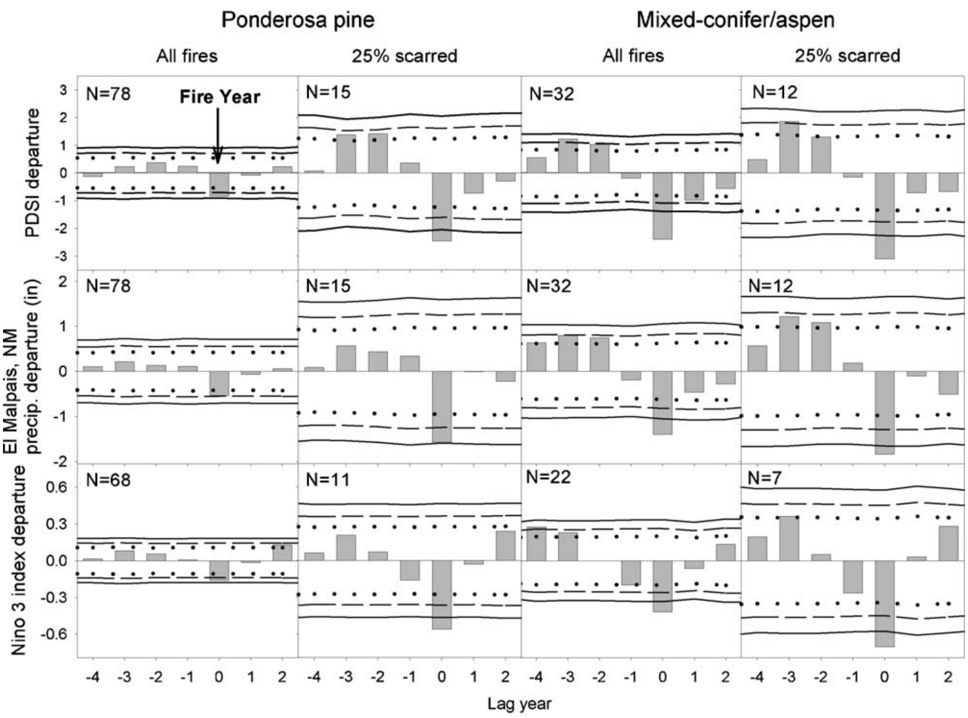

Fig. 8. Superposed epoch analysis for the PP and MC forests illustrating departure from the mean of reconstructed climate indices (PDSI, El Malpais, NM precipitation, and Niño3 index) for all fires and widespread (25% scarred) fires. Dotted, dashed and solid lines represent 95, 99, and 99.9% confidence intervals derived from 1000 Monte Carlo simulations; n, number of fire years.

The proximity of high severity (age structure) and low-severity (fire scar and tree-ring growth change) evidence within the lower MC zone indicates that the 1842 fire burned with mixed-severity within this forest type. Based on the multiple lines of fire evidence presented above, the reconstructed 1842 fire area within the upper Santa Fe watershed was 4642 ha (Fig. 7). The reconstructed stand-replacing fire area was 182 ha, consisting of multiple patches ranging from 34 ha to 110 ha.

3.8. Fire–climate

The results of the SEA indicate that all four filtered subsets of fire scar data (all fires, 10% scarred [not shown], 20% scarred [not shown], and 25% scarred) from both the PP and MC forests were significantly associated with negative (dry) departures during the fire year from mean summer PDSI and El Malpais, NM precipitation (Fig. 8). Fire occurrence in both forest types was also associated with positive (wet) departures from mean summer PDSI two to three years prior to the fire year. Fire occurrence was also associated with positive (wet) departures from mean annual precipitation at El Malpais, NM two to four years prior to the fire year in the MC forest. All sets of fire scar data in both forest types were associated with negative (cool ocean phase—La Niña) SST departures from the mean Niño3 index during the fire year. All fires and widespread (25% scarred) fires in the MC forest and 10% scarred fires in the PP forest (not shown) were associated with positive (warm ocean phase—El Niño) SST departures three to four years prior to the fire year. Fire occurrence in both forest types was not associated with inter-annual variations in PDO (results not shown). The period of analysis was the same used in the fire interval analyses, except when the reconstructed climate series was limiting (i.e., earliest date for reconstructed Niño3 index, 1600 and PDO index, 1700).

Although the results of the SEA indicate surprisingly similar inter-annual fire–climate relationships between the MC and the PP forest types, there were some differences. Mean summer PDSI associated with all fires in the MC forest (-2.59) was significantly drier than in the PP (-1.03; $t = 3.428$, $p < 0.001$, t-test with equal variance; SPSS 16.0). Widespread fire years (25% scarred) in the mixed/conifer aspen forests occurred on drier years (mean PDSI = -3.22) than in the PP forest (-2.57), but the difference was not statistically significant ($t = 0.798$, $p = 0.432$). Widespread fires occurred during drier years on average compared to all fires in the PP forest ($t = 2.498$, $p = 0.014$). The same was true in the MC forest, but the difference was not significant ($t = 0.896$, $p = 0.375$). The PDSI during the one reconstructed stand-replacing fire (1685) in the spruce-fir zone was -6.92.

4. Discussion

Fire in the upper Santa Fe River watershed historically spread between forest types and fire regimes. Low severity fires burned frequently in the PP forests. During sufficiently dry conditions fire spread up the watershed into the MC forests and burned with mixed-severity. During an extreme drought (1685), fire continued to spread into the highest elevation spruce-dominated forests and burned primarily with high severity. The connectivity of forests through fire, the removal of this important process, and historical evidence of large (100–1200 ha) stand-replacing fire patches in MC and spruce-dominated forests have important implications for both fire and water management in the upper Santa Fe watershed and similar forests throughout the region.

4.1. Human influence on the fire regime

Santa Fe was settled by the Spanish earlier than other locations in the southwestern U.S. (1600s), making this site unique. The most striking feature of the Santa Fe watershed fire scar record is the lack of widespread fire since the mid-to-late 19th century (Fig. 2). Fires stopped earlier (i.e., last widespread fire in the PP and MC, 1842) compared to the general pattern of circa 1900 fire exclusion in the southwestern U.S. (Swetnam and Baisan, 1996, 2003). The start of fire exclusion at a particular site has been linked to the timing of intensive land use practices (e.g., grazing and fuel wood collecting) by the Spanish and Anglo-American settlers (Savage and Swetnam, 1990; Baisan and Swetnam, 1997). Sheep herding in the vicinity of Santa Fe began in the 1600s, became a stable industry regionally by the mid-1700s, and peak numbers in the pre-American Civil War era were recorded in the 1820s and 1830s (Baxter, 1987). This early, intensive land use may have created a pattern of anomalously early fire exclusion (e.g., early 1700s, Sandia Mountains, NM; Baisan and Swetnam, 1997) on the east side of the Rio Grande valley along the Camino Real Spanish travel and settlement route. A long gap between widespread fires in the PP and MC forest in the Santa Fe watershed beginning in the 1700s may indicate initial effects of early grazing, but may also have a climatic explanation.

In specific locations in the southwestern U.S. the fire scar record has revealed periods of anomalously high fire frequency (e.g., repeated 1-year fire intervals) or a change in the seasonality of fire occurrence, indicating possible human ignitions (e.g., Chiricahua Mountains, Arizona; Seklecki et al. (1996)). Very few (<2%) latewood fires were recorded in the Santa Fe watershed and there was not evidence of anomalously high fire frequency, despite the long record of settlement. The high percentage of lightning-caused fires (80%, $n = 178$, 1970–2003) in the local area supports the general premise that sufficient lightning ignitions occur in the southwestern U.S. to account for the reconstructed frequency of fire occurrence (Allen, 2002).

4.2. Spruce-fir fire history

Very little fire history and/or age structure data exist for old-growth spruce-fir forests of Arizona and New Mexico. Fule et al. (2003) reconstructed a mixed-severity fire regime with surprisingly frequent small fires ($MFI_{all\ fires} = 2.6$ years) and less frequent widespread fires ($MFI_{25\%} = 31.0$ years) in a relatively low elevation (<2800 m) spruce-fir forest that contained a mix of species (including PP) on the north rim of the Grand Canyon, AZ. A higher elevation spruce-fir forest (average elevation 3200 m) in the San Francisco Peaks, AZ, has not burned catastrophically for over 200 years based on the age of the oldest trees (Cocke et al., 2005). Other high elevation (>3000 m) pure spruce-fir forests in the southern sky island region (Pinaleño Mountains, AZ, and Mogollon Mountains in the Gila Wilderness, NM) had not experienced significant stand-replacing disturbance for at least 300 years prior to the recent crown fires beginning in the late 1990s (Grissino-Mayer et al., 1995; Margolis, 2007). Multiple lines of tree-ring evidence suggest that the Pinaleño spruce-fir stand regenerated after a stand-replacing fire in 1685 (Grissino-Mayer et al., 1995; Margolis, 2007; Swetnam et al., 2009), the same year as the upper Santa Fe watershed. Drought conditions in 1685 were remarkably severe and widespread throughout the southwestern U.S. (Cook et al., 2004). This climate event synchronized these rare stand-replacing fire events, and potentially others, hundreds of kilometers apart.

4.3. Comparing the PP and MC fire regimes

Historical MFI was significantly shorter in PP compared to the higher elevation MC forest in four of the five filtered subsets of fire years (Table 5). Widespread fires in the MC forest occurred on average at intervals that were 10 years (50%) longer than in PP

(Table 5; PP $MFI_{25\%}$ = 20.8 years, MC $MFI_{25\%}$ = 31.6 years). The difference in fire frequency might be partially explained by a larger area in the PP zone (PP, 1600 ha vs. MC, 1200 ha), different sampling intensity (PP, 76 trees: MC, 65 trees) or the spatial distribution of samples. However, these sampling differences are relatively small and with sufficient sample numbers, 25% scarred MFI is robust to differences in sampling (Van Horne and Fule, 2006) and thus likely does not account for the magnitude of observed fire frequency differences. Regionally, MC forests burned less frequently than pine-dominant forests based on comparisons from dozens of Southwestern fire history studies (Swetnam and Baisan, 1996; Heinlein et al., 2005). $MFI_{25\%}$ of widespread fires at six other MC sites in New Mexico ranged from 16.0 years to 26.4 years (Swetnam and Baisan, 1996), which is shorter than the Santa Fe watershed (MC $MFI_{25\%}$ = 31.6 years). The relatively long MFI could be a result of settlement and land-use (e.g., grazing) by the Spanish beginning in the 1600s (Debuys, 1985), which could have reduced fine fuels and consequently fire occurrence in the watershed earlier than in other locations (e.g., Savage and Swetnam, 1990; Baisan and Swetnam, 1997).

An inverse relationship between fire frequency and elevation exists broadly across the montane forests of the western U.S. (Martin, 1982) and at individual sites (Caprio and Swetnam, 1995; Brown et al., 2001)), but site-specific topographic factors may weaken the relationship in some locations (Brown et al., 2001). A hypothesized mechanism for this pattern relates to increased moisture in the higher elevation forests and consequently less frequent occurrence of drought conditions severe enough to dry fuels sufficiently to sustain fire spread. Our results indicate that, on average, fires in the MC forest occurred during drier conditions compared to the adjoining lower elevation PP, providing quantitative support for this hypothesis (Fig. 8). Specifically, the grassy understory of the drier, relatively open PP forest was more likely to carry fire, even if fuels in the mesic mixed-conifer zone were not primed by drought for widespread fire.

4.4. Fire–climate relationships

The relationship between fire occurrence in MC and PP forests and drought during the fire year is intuitive and commonly observed in fire history reconstructions across fuel types in the southwestern U.S. (Fig. 8; Swetnam and Baisan, 1996). The relationship between fire occurrence and wet conditions in prior years is less intuitive, but also well replicated in pine-dominant forests of the southwestern U.S. from fire history studies (Baisan and Swetnam, 1990; Swetnam and Baisan, 1996) and the instrumental record (Crimmins and Comrie, 2004; Baisan and Swetnam, 1990) hypothesize that wet years increase fine fuels (e.g., grass and pine needles) that carry fire, which are burned during subsequent dry years.

This antecedent wet-year relationship is not present in high elevation sub-alpine forests and upper montane seral MC forests of the Southern Rockies (e.g., Sibold et al., 2006; Margolis et al., 2007). A similar drought-only fire–climate relationship exists at multiple MC fire history sites in the region (Swetnam and Baisan, 1996; Touchan et al., 1996). These more mesic, higher elevation forest types are generally not fuel-limited, but require more severe drought for fire occurrence than lower elevation forests.

Based on this prior research, the relationship between fire occurrence and antecedent wet years in the MC forests of the Santa Fe watershed was somewhat surprising (Fig. 8). This result suggests that variability in fine fuels may have been important for fire occurrence (i.e., the system was fuel-limited). But how can a fire regime with a 20- to 30-year mean return interval for widespread fires be fuel-limited? Twenty years in a MC forest should be sufficient to produce enough fuel to sustain fire spread, even in the semi-arid southwestern U.S. It is possible that due to the topographic heterogeneity of the landscape (opposing north and south-facing slopes), wet conditions followed by drought were needed to produce sufficient fuel on the drier south aspects to connect the more productive forest patches and allow fire to burn across aspect and forest types. Grazing could amplify the aspect-driven fuel discontinuity by further reducing fuels on the drier, grassy, south-facing slopes.

A second factor that may explain the wet lags in the MC SEA results is the connectivity of the MC forest to the adjacent, large, frequent burning PP forest. The PP forest in the Santa Fe watershed, similar to others throughout the southwestern U.S., had an herbaceous understory that fueled the frequent fires. As expected, historical fire occurrence in the Santa Fe watershed PP forest was associated with prior wet years that replenished this herbaceous fuel layer (Fig. 8). Prevailing wind direction and the tendency for fire to move upslope would push fires from the PP into the MC forest. Based on our analysis of fire synchrony, 24% of the PP fires spread to the MC forests, but these accounted for a large proportion (69%) of all fires in the MC forest (Table 3). Thus, if fires in PP were in part fueled by prior wet years, and it was sufficiently dry during the fire year, fires would continue to spread up the "fireshed" into the MC forest. The connectivity between forest types would indirectly link fire occurrence in the MC zone to antecedent wet years.

4.5. Landscape scale connectivity of fire regimes

By reconstructing fire history along an elevation, vegetation and fire regime gradient we were able to reconstruct evidence of the transition of fire regimes (and individual fires) from surface fire, to mixed-severity fire (e.g., 1842 fire), to widespread stand-replacing fire (e.g., 1685 fire) in a single watershed. We present multiple lines of evidence of connectivity between forest types and fire regimes through fire as a continuous process that moves across artificially drawn fire regime and vegetation boundaries (Caprio and Swetnam, 1995; Fule et al., 2003). An important implication of this connectivity is that by altering the fire regime in one location (forest type) there may be effects in other forest types. The disruption of the surface fire regime in the mid-elevation, pine-dominated forest throughout the southwestern U.S. (Swetnam and Baisan, 1996, 2003) may not only have serious consequences for that vegetation type (Allen et al., 2002), but is also likely to have effects all along vegetation/elevational gradients. In the Santa Fe watershed, early fire exclusion in PP (i.e., last widespread fire, 1842) from grazing followed by active fire suppression removed an important source of fires for the MC and the spruce-dominated forests. As a result, fire frequency was dramatically reduced in the upper elevation MC forest (Fig. 2).

4.6. Mixed-conifer/aspen forest change due to fire exclusion

Over 120 years of fire exclusion in the MC forest has contributed to changes in structure and composition similar to what occurred regionally and locally in PP and MC forests (Fig. 9). We present age structure data from two fire sensitive species (white fir and quaking aspen) as examples of changes in species composition in the MC forest that occurred coincidently with fire exclusion. Seventy-five percent of the dominant white fir in the MC zone recruited since the last widespread fire (1842; Figs. 3–5). Young white fir has thin bark, making them particularly sensitive to even low-intensity surface fire. In the absence of fire these trees survived to occupy a dominant canopy position, and because they are shade tolerant, they have continued to recruit in the understory, creating ladder fuels, and increasing crown fire hazard. This pattern has been documented in PP dominated systems (Allen

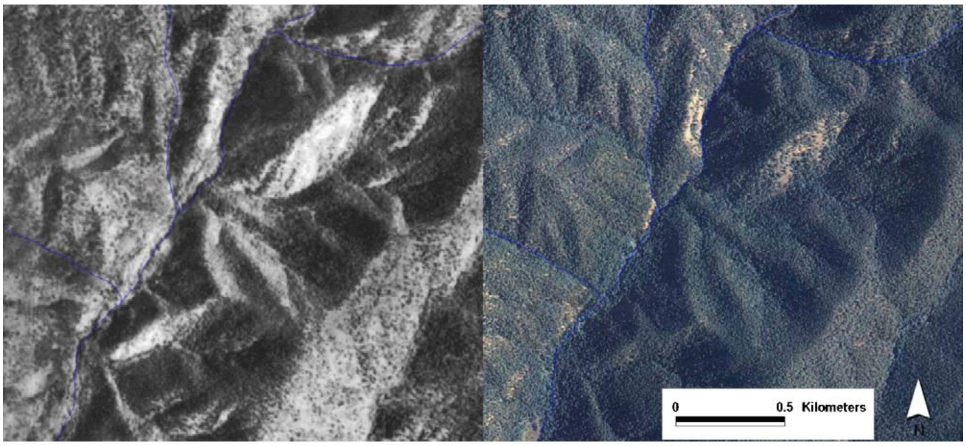

Fig. 9. Comparison of aerial photos (1935 on the left, 2005 on the right) from the MC forest of the Santa Fe watershed indicating a dramatic increase in forest cover on south- and southeast-facing slopes. Images encompass age structure plots 8, 12, and 7. Photos from the U.S.F.S. Santa Fe National Forest, courtesy of Julie Luetzelschwab.

et al., 2002) and other southwestern U.S. MC forests (Mast and Wolf, 2006).

Fire was historically an important determinant of quaking aspen mortality and natality in many upper elevation forests across the western U.S. (Kulakowski et al., 2006; Margolis, 2007; Margolis et al., 2007) and the cessation of fire has been identified as one cause of widespread stand-deterioration throughout its range (Bartos and Campbell, 1998; Kashian et al., 2007). In the Santa Fe watershed, only one (2.5%) quaking aspen stem pre-dated the last widespread fire (1842, Figs. 3–5). Quaking aspen recruitment pulses occurred at three transects following the last fire (3, 13, and 15). Conifers survived the fire at these locations, indicating mixed-severity fire effects by species (i.e., quaking aspen were top-killed and re-sprouted while the overstory conifers survived). This evidence of fire killing and regenerating quaking aspen stems at multiple locations throughout the MC forest illustrates the substantial effect of fire (occurrence and exclusion) on quaking aspen age structure.

4.7. Spruce-fir forest: potential for fire exclusion effects

In the high elevation spruce-fir forests of the region, limited research has assessed the potential for changes related to fire exclusion (Fule et al., 2003; Cocke et al., 2005). Cocke et al. (2005) recorded increased density in spruce-fir since 1876, but this is consistent with natural succession in this forest type. Because *Picea* and *Abies* species are shade tolerant and fires are infrequent, these forests naturally increase in density through time. Different approaches (e.g., examining effects of fire interval length on successional pathways) may be necessary to evaluate potential effects of fire exclusion in this forest type.

Changes in the length of fire-free intervals, even if they were naturally long, may affect successional pathways and forest composition (Romme and Knight, 1981; Kipfmueller and Kupfer, 2005). For example, Romme and Knight (1981) found that sites with naturally longer fire-free intervals and more rapid succession were dominated by spruce-fir forests, compared to sites with more frequent fire and slower succession, which were dominated by lodgepole pine. In the southwestern U.S., quaking aspen is the upper elevation tree species most likely to be sensitive to changes in the length of fire intervals. Seral quaking aspen in the upper montane forests of the region depend on stand-replacing fire for widespread regeneration and long-term perpetuation of the stand (Margolis et al., 2007). Following fire in these seral stands, the aspen-conifer successional pathway proceeds and shade tolerant conifer species regenerate under the canopy, eventually overtopping and shading out the aspen stems in the absence of fire (Dick-Peddie, 1993). Lengthening fire-free intervals in seral aspen stands beyond the life of the above-ground stems and the below-ground clonal root resources could potentially remove aspen from the site, affecting the long-term forest composition.

We hypothesize that although fire intervals were naturally long in high elevation forests of the southern Rocky Mountains, because fire historically spread between forest types, fire exclusion in the lower elevation forests has likely affected some high elevation forests. Future research should be designed to test for changes (e.g., altered successional pathways) resulting from fire exclusion. Upper elevation spruce-fir forests are naturally dense, so although forest density has been an indicator of change in PP and MC forests, it is not likely the best variable to test for change in the spruce-fir zone.

4.8. Will Santa Fe flood?

Large patches of high severity fire (>100 ha) historically occurred on some north-facing slopes in the MC forests of the Santa Fe watershed. The dramatic increase in forest density and canopy cover in these forests, evident from repeat photos (Fig. 9), has very likely increased the size of forest patches at risk of high severity fire. Areas that historically burned with mixed-severity (i.e., 100 ha patches of high severity fire adjacent to equally large low-severity patches) now are likely to burn as larger, contiguous high severity patches. This increased area of forest at risk of stand-replacing fire could subsequently result in a larger, historically unprecedented post-fire hydrologic response in this vital municipal watershed (e.g., Veenhuis, 2002).

One approach to evaluating post-fire flood risk would be to use a combination of our historical fire reconstructions and a hydrological model. The 1685 fire was the worst-case scenario in the spruce-dominated forest; 93% of the sampled spruce forest burned with stand-replacing severity (∼1200 ha). The reconstructed spatial extent and location of low and high severity fire patches from this fire and others (e.g., 1842) could be used to populate a GIS-based hydrologic model such as The Automated Geospatial Watershed Assessment Tool (Goodrich et al., 2006). Alternatively, fire behavior and fire spread models (e.g., FARSITE) could be used to estimate the range of high severity patch sizes under current forest conditions for comparison with reconstructed patch size. The different fire scenarios (modeled and reconstructed) could then be used to populate the hydrologic model. Modeled post-fire runoff and erosion output would provide the best possible answer to the big question in the Santa Fe watershed: what will happen to the water supply when the forest burns?

5. Conclusions

Historical fire in the upper Santa Fe River watershed burned across gradients of elevation, forest types and fire severity. Widespread fires that burned up to 80% of the MC forest area occurred on average at intervals 10 years longer ($MFI_{25\%}$ = 31.6 years) than in the adjacent, lower elevation PP forest ($MFI_{25\%}$ = 20.8 years). The historical MC fire regime is best described as mixed-severity, where patches of stand-replacing fire greater than 100 ha were located adjacent to stands with evidence of repeated surface fire. The upper elevation spruce-dominated forest last burned in 1685 in a climate-driven stand-replacing fire that affected greater than 93% (1200 ha) of the sampled spruce forest and at least 68% of the MC and PP forests (total fire area, 4730 ha). This history of fire that includes natural stand-replacing patches in the upper elevation forests presents challenges for fire management in the watershed. Restoring the aspect-driven heterogeneity of fuels in the MC forest is both ecologically sound and would reduce the area at risk of crown that could threaten the water supply. Given the natural occurrence of large (>1000 ha) stand-replacing fire patches in the spruce-fir zone of the Pecos Wilderness Area, where fire hazard reduction treatment options are limited and would be ecologically unsound, hydrologic models should be used to develop a contingency plan for a large, high severity fire.

Climate variability has strongly influenced fire regimes for centuries in the montane forests of the southwestern U.S. (Swetnam and Betancourt, 1990; Swetnam and Baisan, 1996) and more broadly across western North America (Kitzberger et al., 2007). Fire synchrony between the MC and the PP forest during 24 individual fire years (69% of all MC fires) indicates both top-down control of fire occurrence by climate and connectivity between forest types and fire regimes. More severe drought was required on average for the higher elevation MC forest to burn (sometimes with mixed-severity), compared to the lower PP forest. The worst single-year drought in over 700 years (1685) was associated with the last major fire in the upper elevation spruce-dominated forests of the Santa Fe watershed and synchronized high severity fire in the upper elevations of multiple, distant mountain ranges. This evidence of a direct relationship between drought severity, fire occurrence, and fire severity in MC and spruce-dominated forests suggests that if temperatures continue to increase (IPCC, 2007) and droughts become more frequent and severe as predicted (Seager et al., 2007), the probability of large and severe fire occurrence will increase (Westerling et al., 2006). This emphasizes the urgency for creative and science-based fire and watershed planning and management in this and other fire prone, vitally important watersheds across the West.

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