Fire and Forest Structure Across Vegetation Gradients in San Juan National Forest, Colorado: A Multi-scaled Historical Analysis

Final Report to the Joint Fire Science Program

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Project Summary

We used tree-ring and forest structural data to document fire regimes, forest structure, and relationships between climate change, fires, and tree recruitment across gradients in forest types, elevation, and landscape physiography in the middle Piedra River watershed of San Juan National Forest in southwestern Colorado. Our principle objectives were: 1) to document changes in forest conditions - including species composition, stand densities, and disturbance processes - from pre- to post-Euro-American settlement; and 2) to derive inferences about the top-down (climate and land use changes) and bottom-up (vegetation and landscape physiography) spatiotemporal drivers of fire timing, behavior, and tree demography over the past several centuries. Forest types studied include ponderosa pine, mixed-conifer, aspen, and subalpine forests. Our working hypothesis at the beginning of the study was that changes in forest structure and composition over the recent century of fire exclusion varied largely as a function of fire history, with ponderosa pine and dry mixed-conifer forests that experienced frequent, episodic surface fires having undergone greater changes in fuels and forest structure as a result of fire exclusion than higher-elevation mesic mixed-conifer or subalpine forests that burned less often. We sampled and crossdated ~3700 trees in 122 plots across three landscapes to examine this hypothesis. However, contrary to initial expectations, we found that much of the mixed-conifer forest - and even some of the subalpine forest - has undergone significant shifts in structure at both stand and landscape scales that parallel those seen in ponderosa pine forests. These changes include increases in tree density, changes in stand composition (especially an increase in subalpine fir relative to other species), and substantial lengthening of fire return intervals. Study results have been published in one peer-reviewed paper (Brown and Wu 2005, Climate and disturbance forcing of episodic tree recruitment in a southwestern ponderosa pine landscape, *Ecology* 86:3030-3038) with two other papers being readied for publication at this time. Results also have been presented in several talks and presentations at professional meetings and to managers. The study has had and will continue to have direct applicability to both on-going and proposed forest restoration and fuels treatment programs in San Juan National Forest and adjacent Federal, state, and private lands by providing baseline data on historical patterns and current conditions associated with fire suppression and land use over the past century. Study results also are being used in Fire Regime Condition Class (FRCC) assessments and a new project designed to test simulation model results used in both FRCC and LANDFIRE GIS layers with empirically derived fire history data.

Introduction

Profound changes in forest structure in many forests of the western US have resulted from fire exclusion and land use that accompanied Euro-American settlement in the middle to late nineteenth century (e.g., Covington et al. 1997, Swetnam et al. 1999). However, these changes have not been uniform across all forest types. It is widely assumed that the degree to which fire exclusion has had an effect on different forest types varies depending largely on their fire and vegetation histories (e.g., Veblen 2000, Schoennagel et al. 2004). Ponderosa pine forests that experienced frequent surface fires have typically undergone greater changes in composition, structure, and function as a result of fire exclusion than forests that burned less often, such as higher-elevation mixed-conifer or subalpine forests or lower-elevation pinyon-juniper woodlands. However, such a generalization is most likely scale-dependant. Although individual stands in less fire-prone forest types may not be all together different from historical conditions in terms of fuel conditions or fire risk, these same conditions averaged across landscapes may show that fire exclusion has resulted in more homogenous landscape characteristics or increased risk from other disturbances, such as insect outbreaks. For example, increases in landscape homogeneity of stand ages, tree sizes, and tree densities - caused at least in part by fire suppression during the twentieth century - have been implicated in recent massive tree mortality from bark beetle outbreaks in both pinyon pine woodlands of the Four-Corners area (e.g., USDA Forest Service 2004, Burkett et al. 2005) and spruce and lodgepole pine forests in the Rocky Mountains (e.g., McCullough et al. 1998). Reduction in landscape coverage of seral species in subalpine forests, such as aspen, also has occurred as a result of successional trajectories that favor shade-tolerant and fire-intolerant conifer species with greater time since fire.

In this study, we used tree-ring and forest structural data to document changes in fire regimes, forest structure, and relationships between climate change, fires, and tree recruitment over the past several centuries across gradients in forest types, elevation, and landscape physiography in the middle Piedra River watershed of the San Juan National Forest (SJNF) in southwestern Colorado (Figure 1). Our first objective for the study was to document changes in forest conditions - including species composition, stand densities, and disturbance processes - from pre- to post-Euro-American settlement. Forest types studied include ponderosa pine, mixed-conifer, aspen, and subalpine forests. Our working hypothesis at the beginning of the project was that changes in forest structure and composition over the recent century of fire exclusion have varied largely as a function of fire history, with ponderosa pine and dry mixed-conifer forests that experienced frequent surface fires having undergone greater changes in fuels and forest structure as a result of fire exclusion than upper-elevation mesic mixed-conifer or subalpine forests that burned less often. Findings from this first objective have direct implications for forest management in this region by providing direction and justification for fuels treatments, forest restoration, and wildland fire

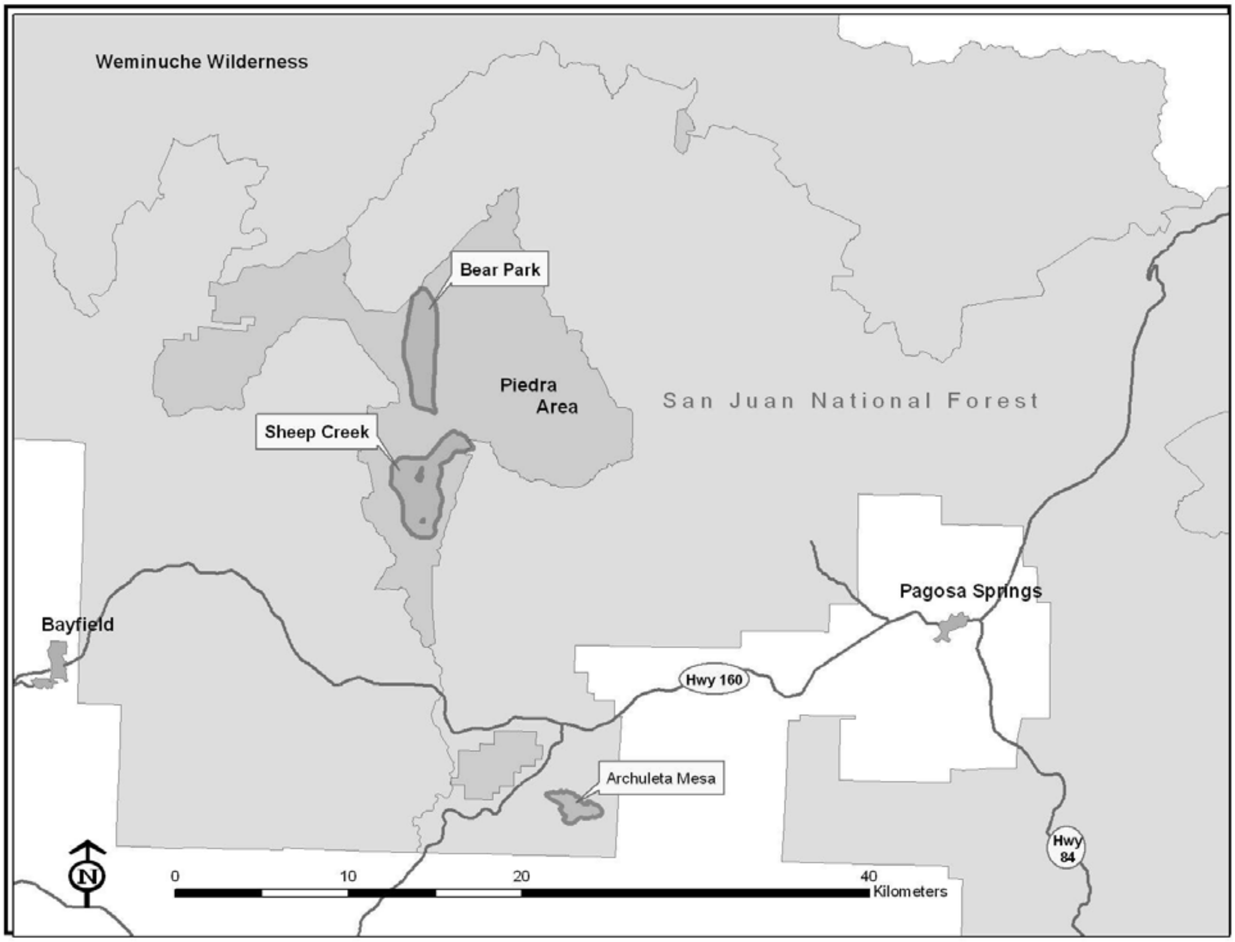


Figure 1. Location map of three study landscapes in SJNF, southwest Colorado.

use in differing forest types.

A second objective of the study was to derive inferences about the top-down (climate and land use changes) and bottom-up (vegetation and landscape physiography) drivers of fire timing and behavior and forest structure within and between forest types over the past several centuries (sensu Heyerdahl et al. 2001, Brown and Wu 2005, Brown 2006). Strong but relatively short (annual- to decadal-length) climatic anomalies can result in broad-scale effects that persist for decades to centuries in long-lived forest ecosystems (Allen and Breshears 1998, Swetnam and Betancourt 1998). Climate change impacts forest structure through both direct effects on tree demography and indirectly through effects on disturbance regimes (Overpeck et al. 1990). Our objectives are to compare timing of tree recruitment to variations in climate forcing and fire years, and to provide inferences about the contribution of both to development of forest structure over time. For this component of the project, we compare reconstructed chronologies of tree recruitment dates and fire years to proxy reconstructions of hydroclimate and global circulation indices to provide inferences about historical climate, fire, and vegetation dynamics across forested landscapes of southwestern Colorado.

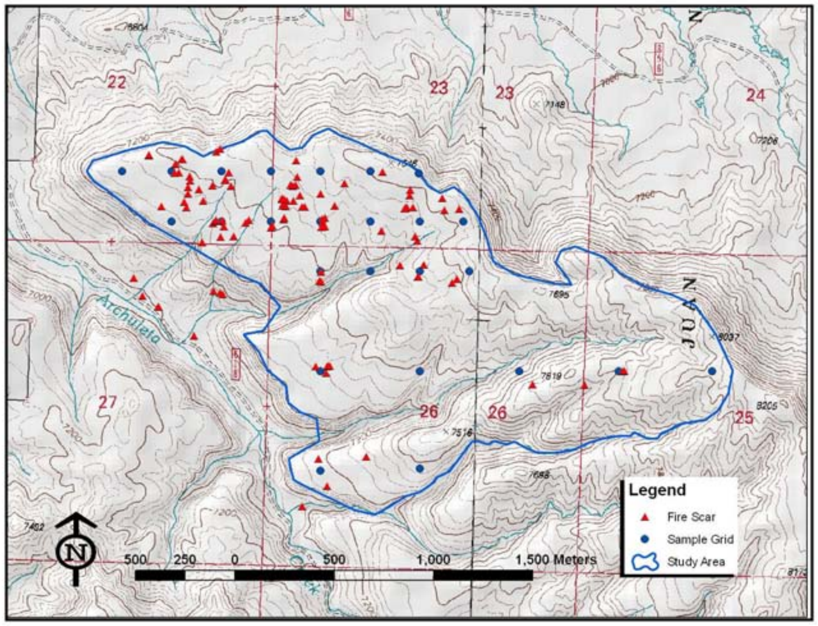
Study Area and Methods

The study was conducted in the middle Piedra River watershed on the SJNF in southwestern Colorado (Figure1). Areas that we selected for the study both contain representative forest types present in SJNF and adjacent forests of the 4-Corners area, and are largely unharvested, allowing for more accurate characterization of pre- and post-settlement forest structure than areas that have been harvested and missing smaller diameter trees (Brown and Cook 2006).

Much of the work for the project was carried out through a cooperative agreement between SJNF and Rocky Mountain Tree-Ring Research (RMTRR), which was in place by September, 2002. Initial fieldwork was conducted in early fall, 2002, with the majority of fieldwork conducted during field seasons in 2003, 2004, and 2005. We sampled a total of ~3700 trees from 122 plots spread across three landscape units (Figure 1). Landscape units were selected to represent a broad range of forest types and physiographic features present in this area of southwestern Colorado and the Four-Corners region. Archuleta Mesa (ACH) is a 760 ac landscape of mainly ponderosa pine forest located on a relatively level mesa southeast of Chimney Rock (Figure 2; Brown and Wu 2005). Sheep Creek (SHP) is a 3210 ac landscape of dry mixed-conifer to subalpine forest extending across south- and north-facing slopes on the east side of the Piedra River (Figure 3). Bear Park (BPK) is a 2570 ac landscape of mesic mixed-conifer to subalpine forests extending up Bear Mountain Ridge above the First Fork of the Piedra River (Figure 4). The Bear Park area also was chosen since it is the site of previous fire-scar sample collection by Wu (1999).

A 500 m grid was established across each landscape to systematically select sample plot locations (Figures 2-4). Plot centers were designated for each grid point and located in the field with a GPS. We collected two types of tree-ring evidence from plots to reconstruct stand-level fire and tree recruitment histories: 1) fire scars formed during burning that does not kill a host tree; and 2) dates of tree recruitment that may have either post-dated stand-opening fires or other disturbances or that were the result of other forcings, such as climate synchronization of tree recruitment. We use the term "tree recruitment" to refer to trees that established in the overstory and have persisted to the present. At each grid point, we used *n*-tree density-adapted sampling methods (Jonsson et al. 1992, Lessard et al. 2002, Brown and Wu 2005, Brown and Cook 2006, Brown 2006) to sample the nearest 30 remnant

Figure 2. Archuleta Mesa study landscape. Dots are locations of plots sampled for tree recruitment (a 250 m grid was used on the north end of the mesa to sample additional older ponderosa pine in this area; see text in Brown and Wu 2005) and triangles are locations of fire-scarred trees sampled outside of plots.



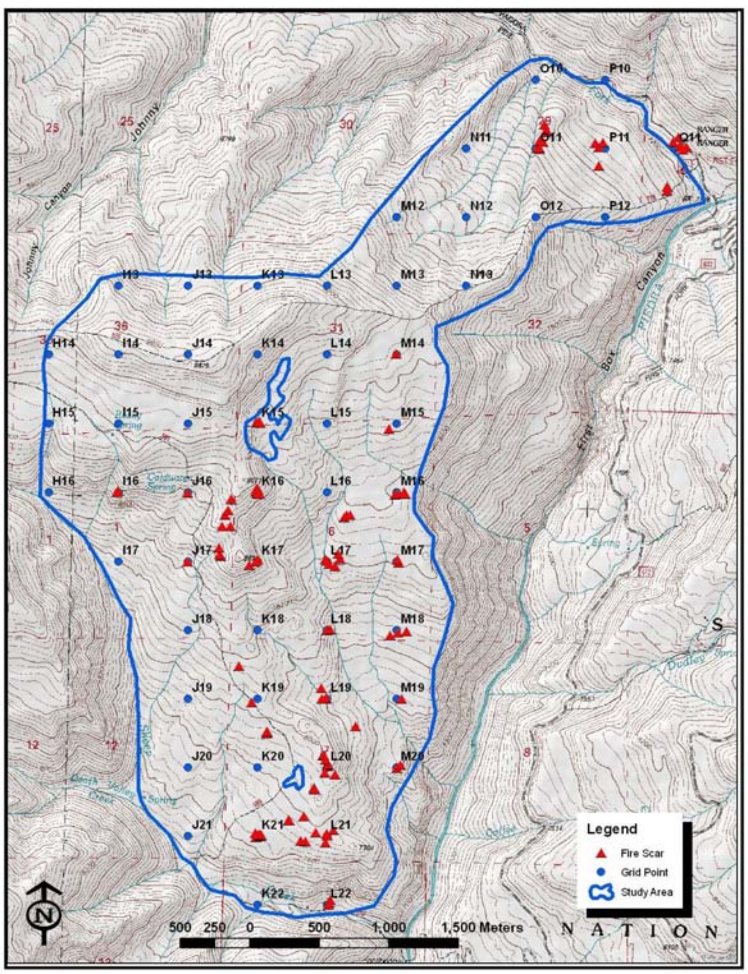


Figure 3. Sheep Creek study landscape. Dots are locations of plots sampled for tree recruitment and triangles are locations of fire-scarred trees sampled outside of plots.

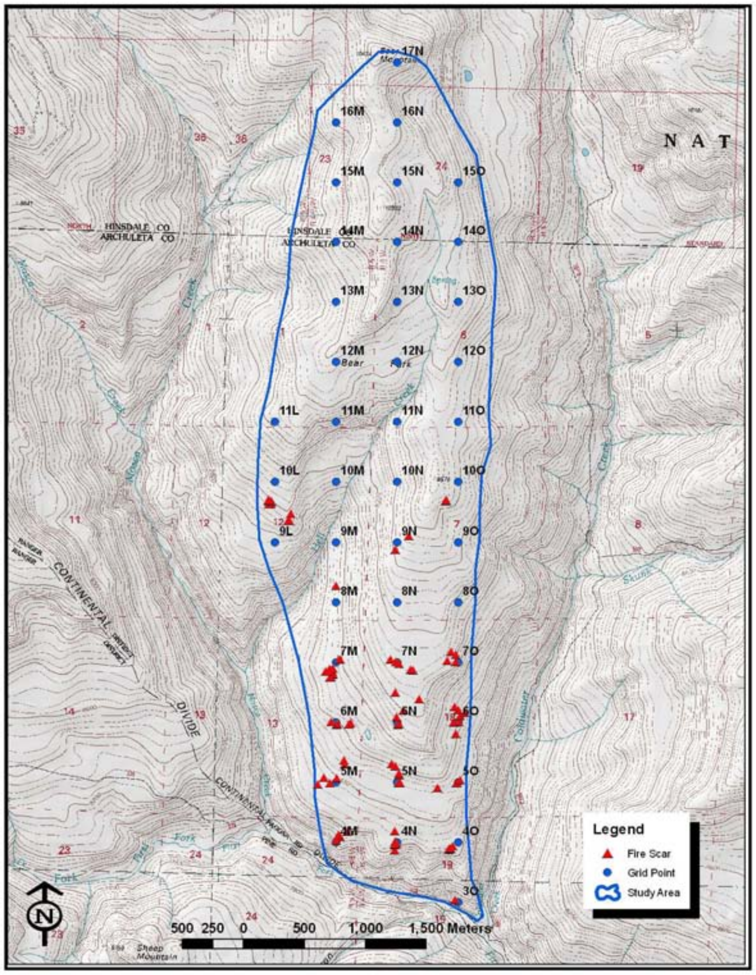


Figure 4. Bear Park study landscape. Dots are locations of plots sampled for tree recruitment and triangles are locations of fire-scarred trees sampled outside of plots.

(logs, snags, and occasional stumps) or living trees ≥ 20cm diameter-at-breast-height (dbh) to each plot center. We did not sample living trees < 20 cm dbh since our interest was in reconstructing pre-twentieth century stand structure and we assumed that smaller trees established during the past ~100 years. The maximum plot radius was set at 40 m (~0.5 ha in area) and most plots were < 0.15 ha in size. Increment cores were removed from 10 cm height above ground level on living trees and cross sections were cut from remnant trees such that one surface was 10 cm height above estimated root-shoot boundary. Sampled cores had to be no more than a field-estimated ten years from pith to minimize pith offset when assessing pith date. Tree diameters at 10 cm height were measured on living trees or estimated for remnant trees missing bark, sapwood, and often heartwood. We also measured distance and azimuth of trees for later mapping of stand structural patterns. In most plots, four standard fuel transects (Brown 1974) also were measured for additional characterization of current fuel conditions across the study area.

In addition to tree recruitment in plots, we also collected cross sections from fire-scarred logs and snags. We did not collect cross sections from living trees because most fire-scarred living trees were old-growth ponderosa pines that we did not want to structurally compromise. This did not influence the completeness of our final fire chronology because our collection from logs and snags was extensive enough to compensate for their omission. Within each plot we collected cross sections from any fire-scarred log and snag regardless of the number of scars per sample. We also searched for additional fire-scarred samples within ~80m of plot centers and cut cross sections from these (Figures 2-4). We also opportunistically sampled any fire-scarred samples encountered between plots. We collected samples that exhibited multiple fire scars in usable wood (many fire-scarred logs in some areas were too decayed for sampling). In areas with high fire-scarred sample concentration, we selected samples containing multiple scars to maximize the time span of fire chronology (Swetnam and Baisan 2003). In other areas where fire scars were rare (especially subalpine stands), we collected all fire-scarred samples encountered in plots, in plot vicinities, and between plots. All fire-scarred samples were GPSed.

All cores and cross sections collected were dendrochronologically crossdated to derive absolute dates for tree-ring series and any fire scars recorded within them. For both tree recruitment dates and fire scars, dendrochronological dating of tree rings was a critical step to provide absolute dates for fire events and age structure that can be compared across spatial scales and with other temporally resolved data such as land use history and climate reconstructions. We used standard dendrochronological methods to crossdate all cores and cross sections against master chronologies we developed for the study areas. We used visual matching of ring characteristics and correlated measured ring widths to assure absolute pith and fire-scar dates. Samples for which we could not determine absolute dates were not used in subsequent analyses. Intra-annual positions of fire scars also were noted to assess season of fire occurrence. On increment cores and cross sections that did not include pith but inside ring curvature was visible, pith dates were estimated using overlaid concentric circles of varying diameters that take into account both average inside ring widths and an estimated distance to pith. Tree recruitment dates and fire-scar dates were compiled into chronologies using the program FHX2, an integrated package for graphing and statistical analyses of fire and forest histories (Grissino-Mayer 2001).

To estimate fire frequency across landscapes and within plots, we determined mean, median, and Weibull median fire intervals to describe and compare fire frequency using different subsets of fire years. Fire frequency analysis is based on composited fire years (Dieterich 1980) derived from fire scars recorded on all trees within plots, grids and across each landscape. We consider fire scars only as proxy evidence (i.e., the event recorded in a natural archive) for dates of fire events, and not as a true representation of burning across each landscape (sensu Baker and Ehle 2001). We assume that during any fire year there were many trees that had fire at their base but did not record a fire scar (Falk 2004) that invalidate assessment of precise spatial patterns of burning from fire-scar evidence. However, we assume that percentages of trees recording fire scars during fire years are representative of the relative spatial scales of burning (Swetnam and Baisan 2003).

We evaluated tree recruitment, fire year, and climate relationships by graphically and statistically comparing tree recruitment and fire year chronologies with independently derived tree-ring based reconstructions of precipitation, drought, and global circulation indices, including measures of the El Niño-Southern Oscillation (ENSO), Pacific Dedacal Oscillation (PDO), and the Atlantic Multidecadal Oscillation (AMO). ENSO is a coupled atmosphere/ocean feature of the equatorial Pacific. PDO is an index of sea surface temperature (SST) anomalies in the North Pacific basin. Both ENSO and PDO have been associated with fire occurrence across the western US through synoptic control of annual droughts that synchronize fire timing (Heyerdahl et al. 2002, Hessl et al. 2004, Brown and Wu 2005, Taylor and Beaty 2005, Sibold and Veblen 2006, Brown 2006, Kitzberger et al. in review). AMO is an index of North Atlantic basin (0 to 70°N) SST anomalies that recently has been associated with droughts (McCabe et al. 2004, Sutton and Hodson 2005) and fire occurrence (Sibold and Veblen 2006, Brown 2006, Kitzberger et al. in review) across the central and western US. We used superposed epoch analysis (SEA; Swetnam 1993) to compare average annual climate conditions for the set of fire years recorded on 2 or more trees to climate for the entire period of record. We also used SEA to compare climate during years prior to fire years to assess antecedent climate conditions. Significant climate conditions were determined using bootstrapped confidence intervals based on average annual climate values with the same number of years as fire year data sets. We exclude fire years recorded on only one tree across each landscape from this analysis since there also may be false positives, scars not caused by fire but assumed to be fire scars (Falk 2004), when assigning fire years. We assume that use of fire-scar dates recorded on two or more trees minimizes the likelihood of false positives since it is less likely that other possible scarring mechanisms (e.g., lightning, fell-tree abrasions) affected more than one tree during the same year.

Here, we report on results from Archuleta Mesa (Brown and Wu 2005); other results will be forthcoming in publications that are being readied for submission for peer-review in early fall, 2006. For the Archuleta data, we used four independently derived tree-ring based climate reconstructions in SEA: 1) annual precipitation from northern New Mexico (Grissino-Mayer 1996); 2) summer Palmer drought severity indices (PDSI) from the 4-corners area (Cook et al. 2004); 3) Southern Oscillation Index (SOI; Stahle et al. 1998), and 4) Niño3 sea surface temperature (SST) index (Cook 2000). SOI is a commonly used measure of ENSO, and is the difference in surface air pressure between Darwin, Australia, and Tahiti. The Niño3 SST index is also a measure of ENSO and is the average sea surface temperature from mid-tropical Pacific recording stations, the region that has the largest variability in sea-surface temperature on El Niño (3 to 4 yr) time scales. The SOI reconstruction only goes back to 1706 and we used the SST index reconstruction to extend our inferences about ENSO forcing of fire years for the entire 500 yr tree-ring record.

Results and Discussion

In this section, we mainly describe published results from the ponderosa pine landscape at Archuleta Mesa (Brown and Wu 2005). After this section, we briefly discuss results from the other two landscapes, but final results and discussion are in process for two papers that will be submitted for peer-review. Once accepted, we will submit copies of these as appendices to this report to the Joint Fire Science Program.

At Archuleta Mesa (Figure 2), We were able to crossdate 574 of 730 trees collected, of which 515 had pith or on which a pith date could be estimated with confidence. Many of the trees we were not able to crossdate were junipers (65 trees, 37% of the total juniper trees collected), mainly because of false and missing rings. Other samples had too few rings, were too complacent (i.e., not enough variability in ring patterns to cross-match against the master chronology), were too slow-growing (i.e., rings too tight), or contained injuries, branches, or decay that made accurate crossdating impossible.

Recruitment dates are summarized and graphically compared with fire years and reconstructions of annual precipitation, PDSI, and ENSO indices in Figure 5. Very few trees (all ponderosa pine) predate a multi-year megadrought centered in the 1580s (Figure 5b,d). This prolonged drought, the most severe in at least the last 1000+ years in this region (Grissino-Mayer 1996), has been identified in tree-ring chronologies from throughout the western US and northern Mexico (Stahle et al. 2000). Some seedlings established shortly before or during the megadrought but these were very slow growing until moisture regimes became more favorable during a prolonged pluvial in the early 1600s. Abundant ponderosa pine recruitment also occurred during the early 1600s and was likely a result of the combination of openings from drought-caused tree mortality as well as moisture conditions optimal for seedling germination and sapling growth in this dry ponderosa pine forest. The effects of the 1580s megadrought and following pluvial on forest age structure in the Southwest have been noted elsewhere (Swetnam and Brown 1992). Comparatively few trees in forests throughout Arizona and New Mexico predate the late 16th century but it is easy to find trees that established during the early 17th.

It is doubtful that severe fires were responsible for stand opening during the megadrought since we found no fire scars on any trees surviving from this period. It is, however, possible that other disturbance factors, such as bark beetles, contributed to widespread and synchronized mortality during the megadrought. Massive forest dieback occurred in many

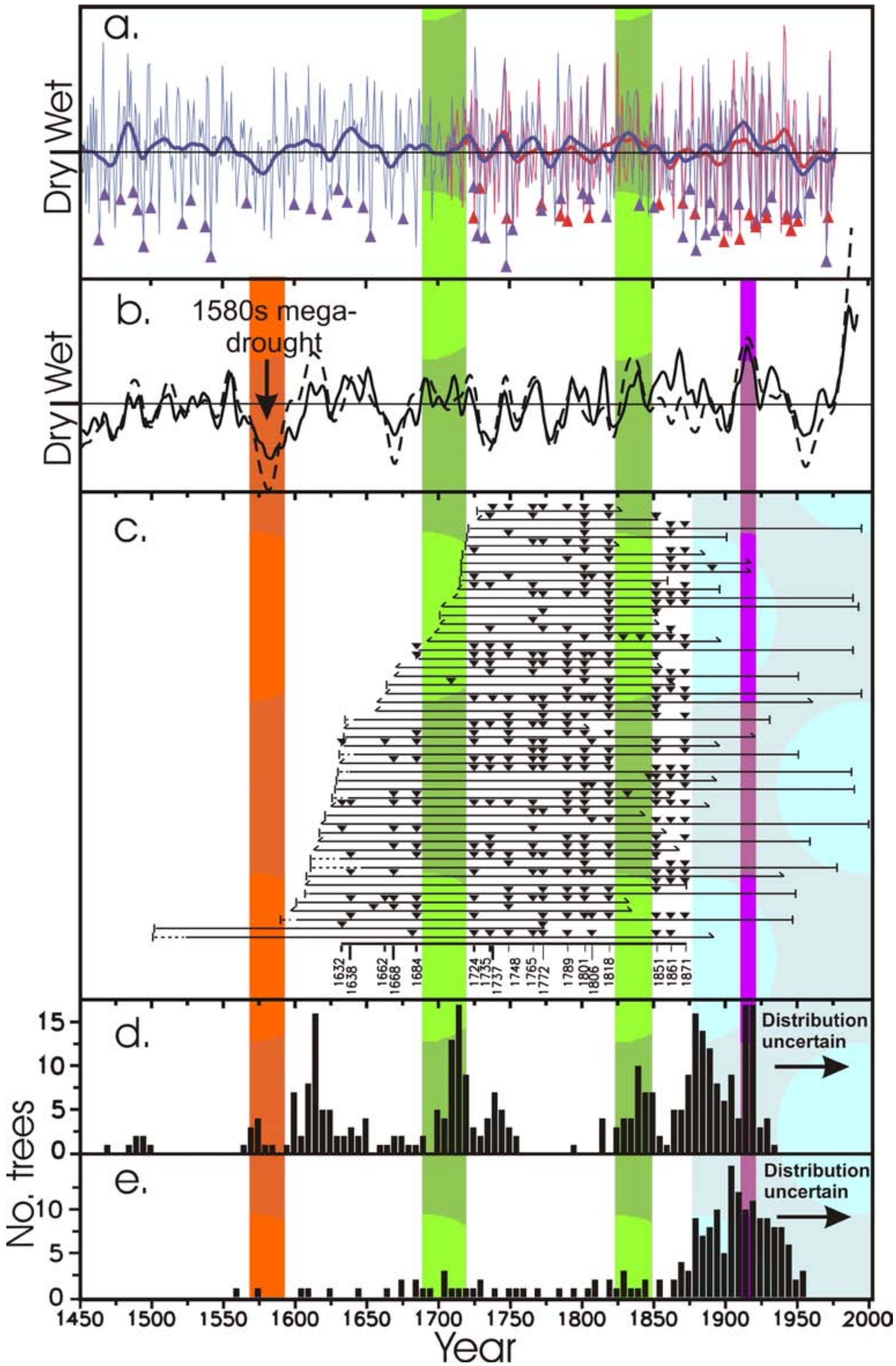


Figure 5. Comparison of ENSO, hydroclimate, fire-year, and tree recruitment chronologies: a) ENSO time series: blue, Niño3 SST index (Cook 2000); red, SOI (Stahle et al. 1998). SOI is reversed to be consistent with other moisture indices. Heavy lines are

annual series smoothed with 20 yr cubic smoothing splines. Years of significant triennial wet/dry oscillations (y_t - y_{t-2}) identified by superposed epoch analyses (SEA) in Figure 6c,d are shown by arrows centered on the drought years. Biennial oscillations (y_t - y_{t-1}) also were tested and found to be largely absent during the fire-quiescent periods of 1684-1724 and 1818-1851. b) Reconstructed hydoclimate time series, smoothed with 20 yr cubic splines: solid line, annual precipitation in northeastern New Mexico (Grissino-Mayer 1996); dashed line, Palmer drought severity index for the 4-corners area (Cook et al. 2004). c) Fire-year chronology for Archuleta Mesa: horizontal lines mark time spans of individual trees with fire scars designated by inverted triangles. Fire years at bottom are those recorded on $\geq$2 trees and used for SEA in Figure 3. d,e) Tree recruitment dates by 5yr periods for: top, ponderosa pine; bottom, other tree species. Recruitment dates are truncated towards the present since we only collected data on trees $\geq$20 cm. Red vertical bar marks the 1580s megadrought, green bars mark fire-quiescent periods, blue bar marks recent decades of fire exclusion after Euro-American settlement, and purple bar marks the early 20th century wet period and pulse of recruitment in ponderosa pine during 1919 (Savage et al. 1996).

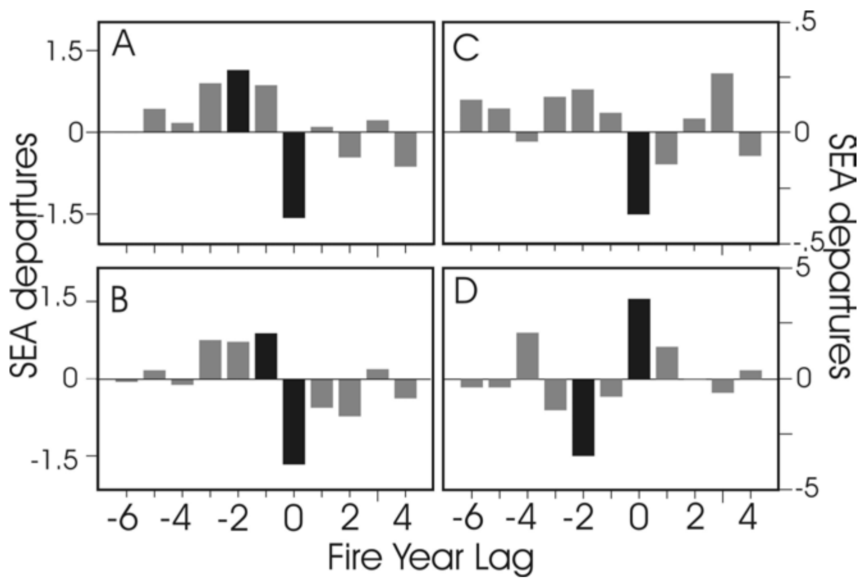


Figure 6. Superposed epoch analyses of hydroclimate and ENSO anomalies for fire years in Figure 5c: a) precipitation from northern New Mexico (Grissino-Mayer 1996); b) PDSI from the 4-corners region (Cook et al. 2004); c) Niño3 SST index (Cook 2000); and d) SOI (Stahle et al. 1998). Black bars mark significant departures (P<0.01) based on bootstrapped confidence intervals. SOI is reversed in terms of moisture relative to other proxies (dry years are high SOI). Fire year lag 0 is the average climate anomaly for all fire years, with antecedent conditions indicated by negative lags.

forests of the 4-corners region during 2002-2004 as a combined result of severe drought and extensive beetle outbreaks (USDA Forest Service 2004). However, no evidence of bark beetles, such as blue stain in sapwood, persists from the megadrought period. Any trees that would have died in the megadrought have long since decayed from the landscape.

In addition to an extended pluvial, the early 17th century was also a period with few fires (Figure 5c). Seedlings and smaller saplings are readily killed by surface burns, and longer intervals between fires would have allowed more trees to grow to a height where they could survive subsequent fires. After the early 1600s, two distinct pulses of tree recruitment occurred during extended fire-quiescent periods, the first from 1684 to 1724 (40 yrs) and the second from 1818 to 1851 (33 yrs). Excluding these two periods, fires occurred on average every 11 yrs (range 2 to 24 yrs) from 1632 to 1871 (Table 1).

Table 1. Fire frequency estimates for fire years at Archuleta Mesa. All values are in years.

Period	MFI ± SD[1]	Median	WMI[2]	Range
1632 - 1871	14.1 ± 10.1	11	12.4	2 to 40
1632 - 1871[3]	11.1 ± 5.7	10	10.6	2 to 24

[1] Mean fire interval ± standard deviation.
[2] Weibull median interval.
[3] Excluding the 2 fire-quiescent intervals from 1684-1724 and 1818-1851.

It appears that climate change likely affected fuel conditions that were less favorable for burning during the fire-quiescent periods. SEA documents that fires occurred predominately during dry years, and were preceded by 1 to 3 wet years (Figure 6a,b). Fire years on average also were La Niña years, which are typically dry years in the Southwest, and often preceded by El Niño years, which typically are wet (Figure 6c,d). This pattern has been interpreted as a buildup of surface fuels (grasses and forbs) during exceptionally wet years which then burn more extensively and readily during subsequent drought years (Swetnam and Baisan 1996, Swetnam and Betancourt 1998, Brown et al. 2001, Westerling et al. 2003). This pattern has been found in both tree-ring and modern fire-atlas datasets. Coupled inter-annual oscillations of El Niño/La Niña years of the magnitudes apparently necessary for a pattern of fuel buildup and drying were notably absent during the 1684-1724 and 1818-1851 fire-quiescent periods (Figure 5a). These results suggest that a dampening of interannual moisture and ENSO variability was as crucial a factor for modifying fire occurrence and causing tree recruitment as extended wet conditions, depending on the period in question. Contemporaneous fire-quiescent periods and tree cohorts occurred in other ponderosa pine forests in the Southwest (White 1985, Swetnam and Betancourt 1998, Mast et al. 1999; T.W. Swetnam, personal communication) providing supporting evidence for a hypothesis of regional climate synchronization of fire timing and related episodes of successful tree recruitment. However, more recruitment and fire year datasets from throughout the Southwest are needed to better assess broad-scale direct and indirect (mediated by fire timing) effects of climate on recruitment patterns.

Density-independent recruitment in ponderosa pine forests contrasts with density-dependent recruitment in closed-canopy forests. Cohorts of even-aged trees in closed-canopy forests are often evidence of severe crown fires or other major disturbance events (Agee 1993, Johnson and Gutsell 1994). However, even-aged cohorts in ponderosa pine forests likely have little to do with episodic mortality in the overstory (sensu Ehle and Baker 2003), but rather relate primarily to extended fire-quiescent periods and favorable climatic episodes. White (1985) proposed that spatially episodic recruitment in ponderosa pine forests was restricted to "safe sites", locations missed by burning during patchy surface fires. Our results suggest that more important effects on broader-scale forest structure were "safe periods", extended fire-quiescent intervals that resulted in temporally distinct recruitment episodes. Regional differences in ponderosa pine forests make generalizations difficult (Allen et al. 2002), but since virtually all ponderosa pine forests show evidence of surface fires in the form of fire scars it is likely that cohort structures in many areas were more dependent on episodic recruitment opportunities rather than episodic mortality events that reduced overstory density.

In addition to surface fires, is it possible that so-called mixed- or variable-severity fires also occurred on Archuleta Mesa? Mixed-severity fires were primarily surface burns that contained patches of fire-caused tree mortality (Agee 1993, Ehle and Baker 2003). A few stands exhibit "truncated" recruitment dates (i.e., the oldest trees in the stand are much younger than the oldest trees present on the landscape; Table 2), and it has been suggested that such variation in stand ages provides evidence for mixed-severity fires (Ehle and Baker 2003). However, it is impossible to know why some portions of the forest are older than others. Variation in ages of surviving trees in any one stand is the result of hundreds of years of patch dynamics that included both chronic as well as episodic tree mortality by any number of factors, only one of which may have been lethal fire but also including drought stress or other disturbances such as insects, pathogens, windthrow, or lightning. What is clear from our results is that by scaling up from individual plots to the entire landscape an emergent pattern appears in which cohort structure is uncoupled from any single mortality event and instead appears to be the result of broader scale climate forcing of fire timing that resulted in successful recruitment episodes (Figure 5). This conclusion furthermore suggests that there are minimum spatial and temporal scales over which tree recruitment and fire histories must be assessed to adequately encapsulate relevant patterns and develop robust inferences (see also Falk 2004). Cohesive patterns resulting from climate forcing only emerge at broader spatial and longer temporal scales. While this study does not address what minimal scales for fire and recruitment histories should be, our data strongly suggest that such scales should encompass more than only a few stands and one to two hundred years of tree recruitment and fire history data (e.g., Ehle and Baker 2003).

Table 2. Spatial variation in oldest recruitment dates in plots.

Oldest date	No. of plots
≤1600	13
1601 - 1650	6
1651 - 1700	2
1701 - 1750	1
1751 - 1800	0
1801 - 1850	0
1851 - 1900	1

Land use changes that accompanied Euro-American settlement in the latter half of the 1800s led to cessation of surface fires from virtually all ponderosa pine forests across the western US (Covington and Moore 1994, Swetnam and Betancourt 1998, Brown et al. 2001, Friederici 2003; Figure 5c). The proximal cause of fire cessation in the late 19th century was widespread sheep and livestock grazing that was later followed by active fire suppression in the 20th century (Allen et al. 2002). The 135 year-long fire-free period from 1871 to 2005 is more than 3x as long as the longest historical fire-quiescent period of 40 years from 1684 to 1724. Establishment of both ponderosa pine and less fire-tolerant species has been unchecked by fire-caused mortality of seedlings with the result that current forests are much denser than in the past (Covington 2000; Figure 5d,e). Optimal climatic conditions for ponderosa pine regeneration and establishment in the early 20th century also contributed to denser forests, especially during 1919 when a tremendous pulse of seedlings established across apparently much of the Southwest (Pearson 1933, Savage et al. 1996; Figure 5d). Increased post-settlement tree density has resulted in altered canopy fuel structures, including formation of "ladder" fuels that allow wildfires to burn more severely. Crown fire has largely replaced surface fire in many ponderosa pine forests (Covington 2000, Allen et al. 2002). Record-setting fire sizes and severities in ponderosa pine and closely related forests during fire seasons in 1996, 2000, 2002, and 2003 have been attributed in large part to changes in forest structure and fuel conditions over the period of fire exclusion (Covington 2000, Romme et al. 2003).

Higher elevation landscapes exhibit more complicated relationships between tree recruitment, stand-opening fires and other severe disturbances, and possible climate forcing (Figure 7). Interpretations of patterns present in both the Sheep Creek (Figure 7) and Bear Park (not shown) landscapes are on-going, and papers are in preparation on these data that will be submitted by early fall, 2006. These papers will be submitted to JFSP as part of this study once they are accepted for publication.

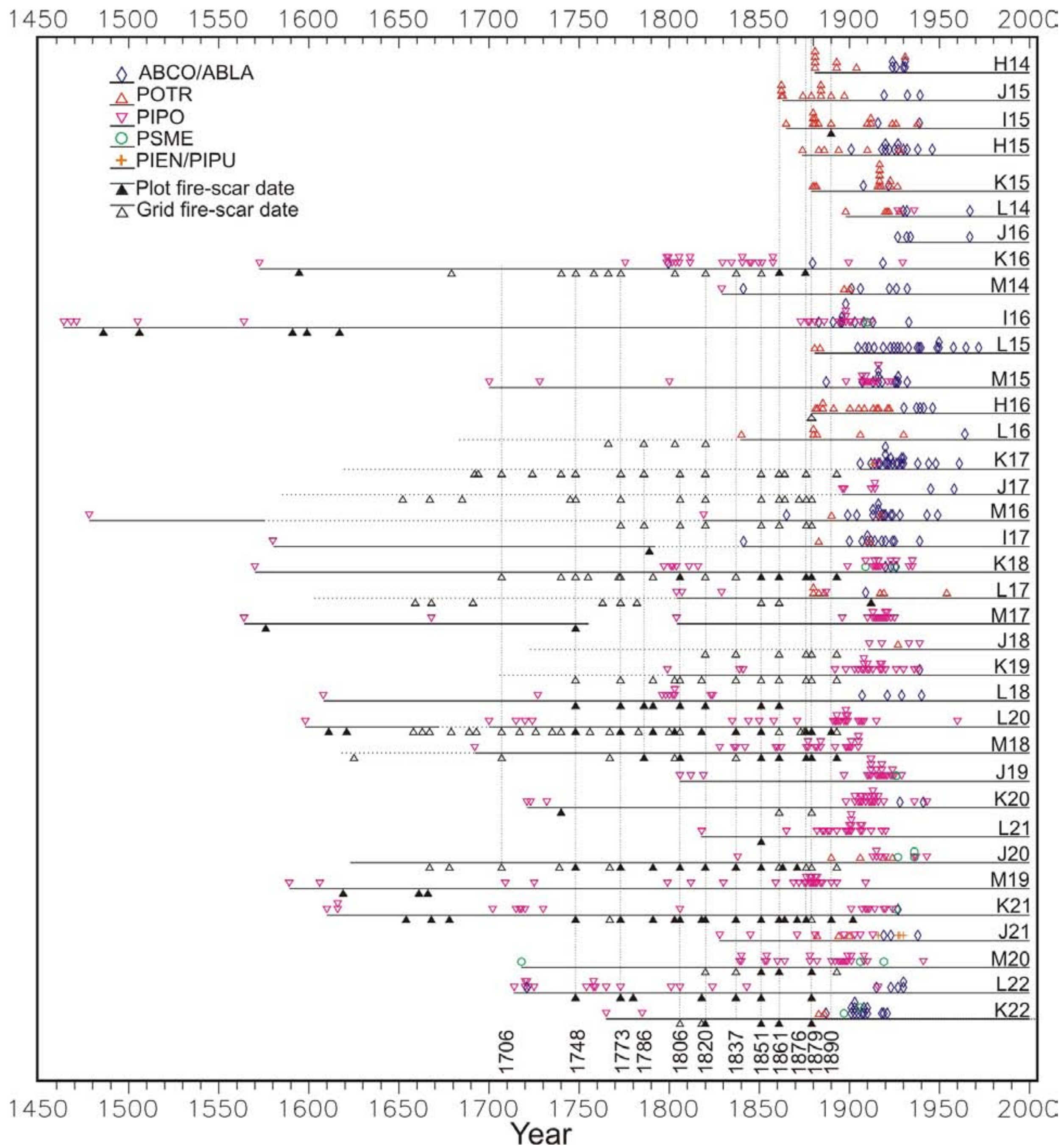


Figure 7. Tree recruitment by species and fire-scar dates for plots sampled on the Sheep Creek landscape (Figure 3). Time spans of trees within plots are marked by solid lines and with trees sampled in plot vicinities marked by dashed lines. Symbols above lines are pith dates of plot trees by species, with inverted triangles below lines marking dates of fire scars recorded on plot trees (solid triangles) or in the plot grid (open triangles). Note the abundant recruitment of especially fir and ponderosa pine after fire cessation in 1890.

Project Deliverables

- Project web site: http://www.rmtrr.org/SJNF/SJNFfirehistory.html
- Publications:
  - Brown, P.M., and R. Wu. 2005. Climate and disturbance forcing of episodic tree recruitment in a southwestern ponderosa pine landscape. Ecology 86:3030-3038. (PDF included in Appendix A)
  - Brown, P.M., and R. Wu. In preparation. Historical fire regimes, stand structure, and climate forcing across forest and physiographic gradients in the San Juan Mountains, Colorado. To be submitted to Ecological Applications, early fall, 2006.
  - Brown, P.M., and R. Wu. In preparation. Historical stand structure in a southwestern ponderosa pine landscape. To be submitted to Forest Ecology and Management, early fall 2006.
- Presentations using data developed from this project:
  - 11/2003, 2nd International Wildland Fire Ecology and Fire Management Congress, Orlando, FL (*Reconstructing spatiotemporal patterns in fire regimes from fire-scar and tree-origin data in southwestern ponderosa pine forests*)
  - 4/2004, Joint Fire Sciences PI Meeting, Phoenix, AZ (*Between drought and fire; climate and disturbance forcing of episodic tree recruitment in the San Juan National Forest*)
  - 5/2004, Mountain Climate Science Symposium, Lake Tahoe, CA (*Between drought and fire: Climatic and disturbance forcing of episodic tree recruitment in ponderosa pine forests*)
  - 11/2004, Presentation at University of Idaho, Department of Forest Resources, Moscow, ID (*Reconstructing fire, climate, and fire climatology from tree rings*; invited)
  - 11/2004, Conference on Mixed Severity Fire Regimes: Ecology and Management, Spokane WA (*Defining scales: time, place, and bottom-up effects on fire regimes*; invited plenary)
  - 12/2004, American Geophysical Union Annual Meeting, San Francisco, CA (*Fire and climate: Past, present [and future?]*; invited)
  - 2/2005, Region 2 Ecologists and Botanists Meeting, Ft. Collins, CO (*HRV data from tree rings: Methods and issues*; invited)
  - 4/2005, American Association of Geographers Annual Meeting, Denver, CO (*Spatial and temporal variation in importance of antecedent and fire year climate for fire occurrence*)
  - 5/2005, Conference on Ecology and Management of Pinyon-Juniper and Sagebrush Communities, Montrose, CO (*Fire history studies in the Rocky Mountains*; invited plenary)
  - 5/2005, Workshop on Fire History and Climate Synthesis in Western North America, Flagstaff, AZ (*Fire severity vs. fire effects: what are we reconstructing?*; invited)
  - 2/2006, Presentation at the Missoula Fire Lab, Rocky Mountain Research Station,

Missoula, MT (*Fire, climate, and tree recruitment: Mixed-severity fires vs. mixed effects from multiple factors*; invited)

- 6/2006, 7th International Conference on Dendrochronology, Beijing, China (*Climate and disturbance effects on tree recruitment and mortality*)

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Appendix A: Publications from the study

Ecology, 86(11), 2005, pp. 3030–3038

CLIMATE AND DISTURBANCE FORCING OF EPISODIC TREE RECRUITMENT IN A SOUTHWESTERN PONDEROSA PINE LANDSCAPE

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Abstract. Strong but relatively short (annual to decadal length) climate change can have broad-scale and long-lasting effects on forest communities. Climate impacts forests through direct effects on tree demography (mortality and overstory recruitment) and indirect effects on disturbance regimes. Here, we compare multicentury chronologies of tree recruitment from a 307-ha ponderosa pine forest in southwestern Colorado to reconstructions of fire years, hydroclimate, and the El Niño-Southern Oscillation (ENSO). Few trees predate a regional multiyear megadrought centered in the 1580s. A prolonged pluvial in the early1600s resulted in a pulse of tree recruitment that corresponds to recruitment seen over much of the Southwest. Other cohorts in the early 1700s and mid-1800s established during multidecadal fire-quiescent periods. These periods correspond to shifts in ENSO that apparently resulted in dampening of interannual wet/dry oscillations responsible for fuel buildup and drying. Fires, mediated by stochastic climate variation, acted as a density-independent regulation on tree populations since establishment was not limited by overstory tree density, but rather by fire-caused mortality of seedlings and saplings during periods of more frequent fires. Even-aged cohorts in ponderosa pine forests likely have little if anything to do with episodic mortality caused by more severe fires, but rather relate mainly to episodic recruitment opportunities. Fire cessation after Euro-American settlement in the late 1800s resulted in an increase in tree density and changes in forest composition, which are major factors that have contributed to recent severe wildfires in other Southwestern forests. Our results document clear linkages between synoptic climate forcing, fires, and recruitment episodes, and highlight the importance of regional historical processes on contemporary forest composition and structure.

Key words: dendroecology; density-independent population dynamics; drought; El Niño-Southern Oscillation; fire regimes; tree demography; tree recruitment.

INTRODUCTION

Strong but relatively short (annual to decadal length) climatic anomalies can result in broad-scale effects that persist for decades to centuries in long-lived forest ecosystems (Allen and Breshears 1998, Swetnam and Betancourt 1998). Climate change impacts forest structure through both direct effects on tree demography and indirectly through effects on disturbance regimes (Overpeck et al. 1990). Since climate changes and resulting effects are nonstationary and probably nonlinear, prediction of vegetation dynamics is difficult without basic understanding of past transient versus persistent climatic effects on plant community composition and structure.

Dendrochronological analysis of static (i.e., present-day) tree ages combined with disturbance and climate histories can elucidate long-term climate/vegetation/disturbance coupling (Mast et al. 1998, Swetnam and Betancourt 1998, Villalba and Veblen 1998, Heyerdahl et al. 2001). Static age structure reflects tree survivorship resulting from the combination of natality and mortality over time. Climate directly affects age structure through favorable conditions for tree establishment or through unfavorable conditions (e.g., droughts) that result in mortality. Climate indirectly affects age structure through control of both disturbance severity, which influences scale and magnitude of mortality, and disturbance frequency, which limits establishment to periods between disturbance events. Disturbances of varying scales also are crucial processes in many forests to open up canopy space for new canopy recruitment to occur.

Confounding effects may limit inferences that can be made about climatic vs. disturbance forcing of forest patterns. In closed-canopy forests, the presence of even-aged tree cohorts is often evidence of past severe crown fires (Agee 1993, Johnson and Gutsell 1994). This evidence relies on coupled but distinct ecological/demographic processes: fire that kills trees followed by synchronous tree recruitment into canopy openings. However, in open-canopy or climatically marginal forests and savannas, episodic recruitment also occurs as a result of transient moisture or temperature conditions optimal for new recruitment to occur (e.g., Peet 1981).

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PLATE 1. Old-growth ponderosa pine forest and gamble oak community on Archuleta Mesa, southwestern Colorado. Photo credit: P. M. Brown.

Climatic vs. disturbance effects should tend to be scale-dependant. Fire-caused cohorts are more likely to be relatively local in size (only a few stands burned catastrophically in a past fire, while others were not affected or burned less intensively), while climatically forced recruitment should be synchronous across larger areas because of the broader-scale ''footprints'' of climatic anomalies.

The dominant historical disturbance regime in ponderosa pine (*Pinus ponderosa* Laws) forests across western North America consisted of recurrent surface fires (Covington and Moore 1994, Mast et al. 1998, Swetnam and Betancourt 1998, Brown et al. 1999, Heyerdahl et al. 2001, Allen et al. 2002, Ehle and Baker 2003, Friederici 2003, Swetnam and Baisan 2003, Grissino-Mayer et al. 2004). Mature ponderosa pine trees are well adapted to survive surface burning, with thick bark that protects vascular cambium from girdling and high crowns that reduce the likelihood of fatal crown scorch. Ponderosa pine also has relatively large, heavy seeds that are not well adapted to rapid recolonization after extensive crown fires. Dates and locations of surface burning are reconstructed from proxy fire-scar records, distinctive injuries caused by localized cambial death from heating and recorded in tree-ring series. Networks of fire-scar chronologies document regionally synchronous fire years that correspond to seasonal droughts induced by interhemispheric synoptic climate forcings, such as the El Niño-Southern Oscillation (ENSO; Swetnam and Betancourt 1998, Heyerdahl et al. 2002, Westerling and Swetnam 2003).

Less clear is how demographic patterns in ponderosa pine forests relate to climate variation. Most seedlings and smaller saplings were killed during recurrent fires, and forests were generally open as a result (White 1985, Covington and Moore 1994, Allen et al. 2002). Overstory recruitment in many ponderosa pine forests was apparently highly episodic, related both to optimal climate conditions for seed production, seedling germination, and sapling growth (e.g., Pearson 1933, Savage et al. 1996) and to longer intervals between surface fires, which allowed more seedlings and saplings to reach a stage where they were relatively immune from subsequent burns (White 1985, Grissino-Mayer and Swetnam 2000). These factors are correlated, since extended wet periods would have resulted in more years conducive to seedling germination and tree growth in typically xeric ponderosa pine forests and fewer years during which fuels were dry enough for fire ignition and spread. However, other studies suggest that even-age cohort structure in at least some ponderosa pine forests was the result of severe fires opening up canopies for new recruitment to occur (Ehle and Baker 2003). Unfortunately, few tree-ring studies in ponderosa pine forests have collected the tree-establishment data needed to assess regional climatic vs. local disturbance effects on recruitment patterns.

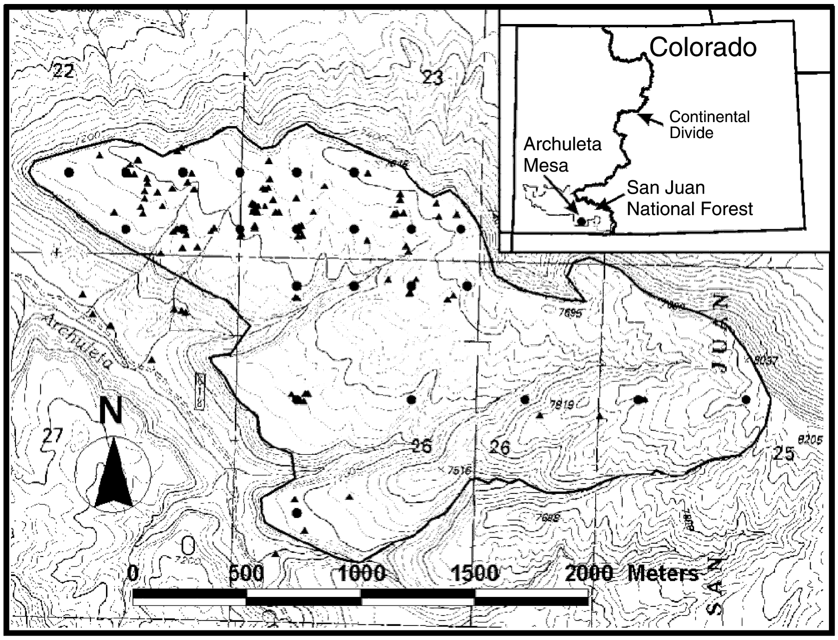

FIG. 1. Location of tree recruitment plots (circles) and fire-scarred trees (triangles) sampled on Archuleta Mesa.

In this study, we constructed chronologies of tree recruitment dates and compared them to proxy reconstructions of fire years, hydroclimate, and ENSO indices to provide inferences about historical climate, fire, and vegetation dynamics across a ponderosa pine landscape in southwestern Colorado. The climate reconstructions document large annual- and decadal-scale variations over the past five centuries, the period covered by fire year and tree recruitment chronologies. Our objectives are to compare timing of tree recruitment to variations in climate forcing and fire years, and to provide inferences about the contribution of both to development of forest structure over time.

METHODS

Study area

Archuleta Mesa is an isolated mesa in southwestern Colorado ~25 km southwest of Pagosa Springs (see Plate 1, Fig. 1). Elevations range from 2225 m to 2450 m, sloping down from the southeast to the northwest. Soils are sandy clays. The forest on the mesa top is an unlogged, old growth forest of mainly ponderosa pine. Other tree species include Rocky Mountain juniper (*Juniperus scopulorum*), Utah juniper (*J. osteosperma*), piñon (*Pinus edulis*), and Douglas-fir (*Pseudotsuga menzesii*). There is abundant oak (*Quercus gambelii*) shrub cover in patches of typically <0.1 ha in size. Ground cover consists of grasses and herbaceous species. Instrumental climate averages from nearby Pagosa Springs (elevation 2200 m) for the period 1906 to 1998 were: January maximum temperature, 3.3°C, minimum −17.0°C; July maximum temperature, 28.4°C, minimum 7.3°C; annual precipitation, 513 mm.

Reconstructing tree recruitment dates and fire years

We designated a 307-ha study landscape on the mesa top, within which we established a 500-m grid for tree recruitment sample plot locations (Fig. 1). We use the term "tree recruitment" to refer to trees that established in the overstory and have persisted to the present. We increased plot density to a 250-m grid on the northwestern tip of the mesa to increase sampling of larger ponderosa pine trees in this area. A total of 23 plots were sampled. We used *n*-tree density-adapted sampling methods (Jonsson et al. 1992, Lessard et al. 2002) to sample the nearest 30 remnant (logs and snags) or living trees ≥20 cm diameter at breast height (dbh) to each plot center. The maximum plot radius was set at 40 m (~0.5 ha in area) and most plots were <0.25 ha in size. Increment cores were removed from 10 cm height above ground level on living trees and cross sections were cut from logs and snags such that one surface was 10 cm above the estimated root–shoot boundary. Sampled cores were no more than a field-estimated 10 yr from pith to minimize pith offset when assessing pith date. Tree diameters at 10 cm height were measured on living trees or estimated for remnant trees missing bark, sapwood, and often heartwood.

We collected cross sections from an additional 70 fire-scarred trees for reconstruction of fire frequency and timing (Fig. 1). We collected samples from living trees, logs, and snags that both exhibited sequences of multiple fire scars and from which we were able to obtain a usable cross section (many fire-scarred logs were too decayed for sampling). Multiple-scarred trees were selected to maximize the time span of fire years (Swetnam and Baisan 2003). While this is not a complete spatial census of fire-scarred trees, we assumed this dataset to be representative of temporal patterns of past fire years across the study landscape for comparison to climate reconstructions and tree recruitment dates.

We used standard dendrochronological methods to crossdate all cores and cross sections against a master chronology for Archuleta Mesa. We used visual matching of ring characteristics and correlated measured ring widths to assure absolute pith and fire-scar dates. Questionable dates were not used in subsequent analyses. Intra-annual positions of fire scars also were noted to assess season of fire occurrence. On increment cores and cross sections that did not include pith but inside ring curvature was visible, pith dates were estimated using overlaid concentric circles of varying diameters that take into account both average inside ring widths and an estimated distance to pith. Pith dates at 10 cm height were corrected to germination dates by subtracting 5 yr, the average time we estimated for seedlings to grow from germination to 10 cm height (P. M. Brown, *unpublished data*). Tree germination dates (referred to as tree recruitment dates in this paper) and fire-scar dates were compiled into chronologies using program FHX2, an integrated package for graphing and statistical analyses of fire and forest histories (Grissino-Mayer 2001).

Evaluating fire frequency

We determined mean, median, and Weibull median fire intervals to describe and compare fire frequency using different subsets of fire years. Fire frequency analysis is based on composited fire years (Dieterich 1980) derived from fire scars recorded on all trees. We consider fire scars only as proxy evidence (i.e., the event recorded in a natural archive) for dates of fire events, and not as a true representation of burning across the landscape (sensu Baker and Ehle 2001). We assume that during any fire year there were many trees that had fire at their base but did not record a fire scar (Falk 2004) that invalidate assessment of precise spatial patterns of burning from fire-scar evidence. However, we assume that percentages of trees recording fire scars during fire years are representative of the relative spatial scales of burning for those years (Swetnam and Baisan 2003).

Evaluating tree recruitment, fire year, and climate relationships

We graphically and statistically compared tree recruitment and fire year chronologies with independently derived tree-ring based reconstructions of precipitation, drought, and ENSO to assess climatic forcing of fire years and recruitment episodes. We used superposed epoch analysis (SEA; Swetnam 1993) to compare average annual climate conditions for the set of fire years recorded on two or more trees to climate for the entire period of record. We also used SEA to compare climate during years prior to fire years to assess antecedent climate conditions. Significant climate conditions were determined using bootstrapped confidence intervals based on average annual climate values with the same number of years as fire year data sets. We exclude fire years recorded on only one tree from this analysis since there also may be false positives, scars not caused by fire but assumed to be fire scars (Falk 2004), when assigning fire years. We assume that use of fire-scar dates recorded on two or more trees minimizes the likelihood of false positives since it is less likely that other possible scarring mechanisms (e.g., lightning, fell-tree abrasions) affected more than one tree during the same year. We used four independently derived tree-ring based climate reconstructions in SEA: (1) annual precipitation from northern New Mexico (Grissino-Mayer 1996); (2) summer Palmer drought severity indices (PDSI) from the four-corners area (Cook et al. 2004); (3) Southern Oscillation Index (SOI; Stahle et al. 1998), and (4) Niño3 sea surface temperature (SST) index (Cook 2000). SOI is a commonly used measure of ENSO, and is the difference in surface air pressure between Darwin, Australia, and Tahiti. The Niño3 SST index is also a measure of ENSO and is the average sea surface temperature from mid-tropical Pacific recording stations, the region that has the largest variability in sea-surface temperature on El Niño (3–4 yr) time scales. The SOI reconstruction only goes back to 1706 and we used the SST index reconstruction to extend our inferences about ENSO forcing of fire years for the entire 500-yr tree-ring record.

Results and Discussion

We were able to crossdate 574 of 730 trees collected, of which 515 had pith or on which a pith date could be estimated with confidence. Many of the trees we were not able to crossdate were juniper (65 trees, 37% of the total juniper trees collected), mainly because of false and missing rings. Other samples had too few rings, were too complacent (i.e., not enough variability in ring patterns to cross-match against the master chronology), were too slow growing (i.e., rings too tight), or contained injuries, branches, or decay that made accurate crossdating impossible.

Recruitment dates are summarized and graphically compared with fire years and reconstructions of annual precipitation, PDSI, and ENSO indices in Fig. 2. Very few trees (almost all ponderosa pine) predate a multiyear megadrought centered in the 1580s (Fig. 2b, d). This prolonged drought, the most severe in at least the last 1000+ years in this region (Grissino-Mayer 1996), has been identified in tree-ring chronologies from throughout the western United States and northern Mexico (Stahle et al. 2000). Some seedlings established shortly before or during the megadrought, but these were very slow growing until moisture regimes became more favorable during a prolonged pluvial in the early 1600s. Abundant ponderosa pine recruitment also oc-

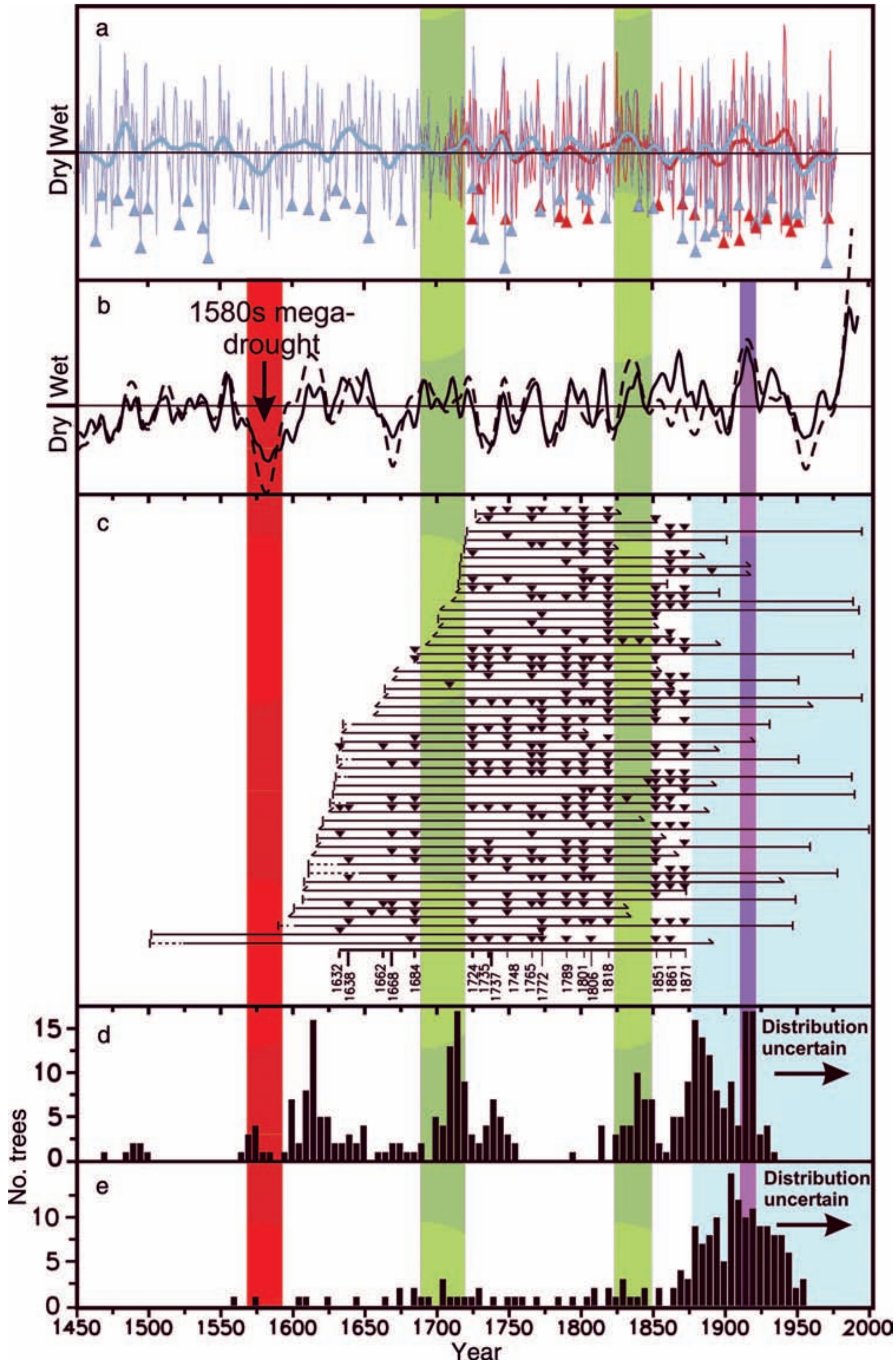

FIG. 2. Comparison of ENSO, hydroclimate, fire-year, and tree recruitment chronologies. (a) ENSO time series (blue, Niño3 SST index [Cook 2000]; red, SOI [Stahle et al. 1998]). SOI is reversed to be consistent with other moisture indices. Heavy lines are annual series smoothed with 20-yr cubic smoothing splines. Years of significant triennial wet/dry oscillations ($y_t - y_{t-2}$) identified by superposed epoch analyses (SEA) in Fig. 3c, d are shown by arrows centered on the drought years. Biennial oscillations ($y_t - y_{t-1}$) also were tested and found to be largely absent during the fire-quiescent periods of 1684–1724 and 1818–1851. (b) Reconstructed hydoclimate time series, smoothed with 20-yr cubic splines. The solid line shows the annual precipitation in northeastern New Mexico (Grissino-Mayer 1996), and the dashed line shows the Palmer drought severity index for the four-corners area (Cook et al. 2004). (c) Fire-year chronology for Archuleta Mesa. Horizontal lines mark time spans of individual trees, with fire scars designated by inverted triangles. Fire years at bottom are those recorded

curred during the early 1600s and likely was a result of the combination of openings from drought-caused tree mortality as well as moisture conditions optimal for seedling germination and sapling growth in this dry ponderosa pine forest. The effects of the 1580s megadrought and following pluvial on forest age structure in the Southwest have been noted elsewhere (Swetnam and Brown 1992). Comparatively few trees in forests throughout Arizona and New Mexico predate the late 16th century but it is easy to find trees that established during the early 17th century.

It is doubtful that severe fires were responsible for stand opening during the megadrought, since we found no fire scars on any trees surviving from this period. It is, however, possible that other disturbance factors, such as bark beetles, contributed to widespread and synchronized mortality during the megadrought. Massive forest dieback occurred in many forests of the four-corners region during 2002–2004 as a combined result of severe drought and extensive beetle outbreaks (USDA Forest Service 2004). However, no evidence of bark beetles, such as blue stain in sapwood, persists from the megadrought period. Any trees that would have died in the megadrought have long since decayed from the landscape.

In addition to an extended pluvial, the early 17th century also was a period with few fires (Fig. 2c). Seedlings and smaller saplings are readily killed by surface burns, and longer intervals between fires would have allowed more trees to grow to a height where they could survive subsequent fires. After the early 1600s, two distinct pulses of tree recruitment occurred during extended fire-quiescent periods, the first from 1684 to 1724 (40 yr) and the second from 1818 to 1851 (33 yr). Excluding these two periods, fires occurred on average every 11 yr (range 2 to 24 yr) from 1632 to 1871 (Table 1).

It appears that climate change likely affected fuel conditions that were less favorable for burning during the fire-quiescent periods. SEA documents that fires occurred predominately during dry years, and were preceded by one to three wet years (Fig. 3a, b). Fire years, on average, also were La Niña years, which are typically dry years in the Southwest, and often preceded by El Niño years, which typically are wet (Fig. 3c, d). This pattern has been interpreted as a buildup of surface fuels (grasses and forbs) during exceptionally wet years which then burn more extensively and readily during subsequent drought years (Swetnam and Baisan 1996, Swetnam and Betancourt 1998, Brown et al. 2001, Westerling et al. 2003). This pattern has been found in both tree-ring and modern fire-atlas data sets. Coupled interannual oscillations of El Niño/La Niña years of the magnitudes apparently necessary for a pattern of fuel buildup and drying were notably absent during the 1684–1724 and 1818–1851 fire-quiescent periods (Fig. 2a). These results suggest that a dampening of interannual moisture and ENSO variability was as crucial a factor for modifying fire occurrence and causing tree recruitment as extended wet conditions, depending on the period in question. Contemporaneous fire-quiescent periods and tree cohorts occurred in other ponderosa pine forests in the Southwest (White 1985, Swetnam and Betancourt 1998, Mast et al. 1999; T. W. Swetnam, *personal communication*) providing supporting evidence for a hypothesis of regional climate synchronization of fire timing and related episodes of successful tree recruitment. However, more recruitment and fire year data sets from throughout the Southwest are needed to better assess broad-scale direct and indirect (mediated by fire timing) effects of climate on recruitment patterns.

Climate forcing of recurrent surface fires affected density-independent control on tree recruitment in ponderosa pine forests. Recruitment in many forests tends to be density limited because of competitive limitations for light, nutrients, water, and space imposed on seedlings by the existing overstory. However, at Archuleta Mesa, recruitment occurred largely during periods when surface fires were restricted by climate conditions less conducive to burning. Fires burned mainly in understory grasses and herbaceous fuels and, since they were affected by seasonal weather conditions, would have occurred irregardless of overstory tree or seedling density at the time of burning. During periods of shorter-interval fires, most seedlings and saplings were killed before they had a chance to establish and forests were typically open as a result. Historic accounts and early settlement photographs of ponderosa pine forests

Table 1. Fire frequency estimates for fire years at Archuleta Mesa, southwestern Colorado, USA.

Period	Fire interval (mean ± SE)	Median	WMI†	Range
1632–1871	14.1 ± 10.1	11	12.4	2–40
1632–1871‡	11.1 ± 5.7	10	10.6	2–24

Note: All values are in years.
† Weibull median interval.
‡ Excluding the two fire-quiescent intervals from 1684–1724 and 1818–1851.

←

on ≥2 trees and used for SEA in Fig. 3. (d, e) Tree recruitment dates by 5-yr periods for (d) ponderosa pine and (e) other tree species. Recruitment dates are truncated toward the present, since we only collected data on trees >20 cm. The red vertical bar marks the 1580s megadrought, green bars mark fire-quiescent periods, the blue bar marks recent decades of fire exclusion after Euro-American settlement, and the purple bar marks the early 20th-century wet period and pulse of recruitment in ponderosa pine centered on 1919 (Savage et al. 1996).

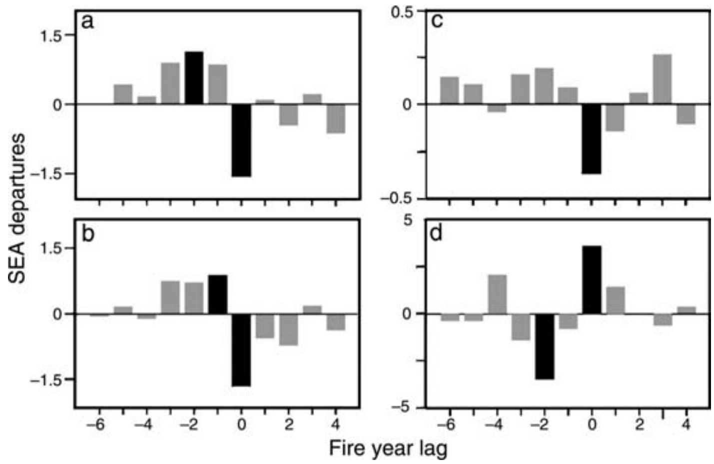

FIG. 3. Superposed epoch analyses of hydroclimate and ENSO anomalies for fire years in Fig. 2c: (a) precipitation in northern New Mexico (Grissino-Mayer 1996); (b) PDSI from the four-corners region (Cook et al. 2004); (c) Niño3 SST index (Cook 2000); and (d) SOI (Stahle et al. 1998). Black bars mark significant departures ($P < 0.01$) based on bootstrapped confidence intervals. SOI is reversed in terms of moisture relative to other proxies (dry years are high SOI). Fire year lag 0 is the average climate anomaly for all fire years, with antecedent conditions indicated by negative lags.

throughout the West document often ''park-like,'' open, multi-aged forest stands, composed of mainly large trees scattered across grassy understories (Covington and Moore 1994, Allen et al. 2002, Friederici 2003). Timing of fire intervals and recruitment suggests that trees had to have been on the order of 10–40 yr old before they experienced their first surface fire to successfully recruit into the overstory and survive to the present (Fig. 2d).

Density-independent recruitment in ponderosa pine forests contrasts with density-dependent recruitment in closed-canopy forests. Cohorts of even-aged trees in closed-canopy forests are often evidence of severe crown fires or other major disturbance events (Heinselmann 1973, Agee 1993, Johnson and Gutsell 1994). However, even-aged cohorts in ponderosa pine forests likely have little to do with episodic mortality in the overstory (sensu Baker and Ehle 2002, Ehle and Baker 2003), but rather relate primarily to extended fire-quiescent periods and favorable climatic episodes. White (1985) proposed that spatially episodic recruitment in ponderosa pine forests was restricted to ''safe sites,'' locations missed by burning during patchy surface fires. Our results suggest that more important effects on broader-scale forest structure were ''safe periods,'' extended fire-quiescent intervals that resulted in temporally distinct recruitment episodes. Regional differences in ponderosa pine forests make generalizations difficult (Allen et al. 2002), but since virtually all ponderosa pine forests show evidence of surface fires in the form of fire scars it is likely that cohort structures in many areas were more dependent on episodic recruitment opportunities rather than episodic mortality events that reduced overstory density.

TABLE 2. Spatial variation in oldest recruitment dates in plots.

Oldest date	No. plots
Before 1600	13
1601–1650	6
1651–1700	2
1701–1750	1
1751–1800	0
1801–1850	0
1851–1900	1

In addition to surface fires, is it possible that so-called mixed- or variable-severity fires also occurred on Archuleta Mesa? Mixed-severity fires were primarily surface burns that contained patches of fire-caused tree mortality (Agee 1993, Ehle and Baker 2003). A few stands exhibit ''truncated'' recruitment dates (i.e., the oldest trees in the stand are much younger than the oldest trees present on the landscape; Table 2), and it has been suggested that such variation in stand ages provides evidence for mixed-severity fires (Ehle and Baker 2003). However, it is impossible to know why some portions of the forest are older than others. Variation in ages of surviving trees in any one stand is the result of hundreds of years of patch dynamics that included both chronic as well as episodic tree mortality by any number of factors, only one of which may have been lethal fire but also including drought stress or other disturbances such as insects, pathogens, windthrow, or lightning. What is clear from our results is that by scaling up from individual plots to the entire landscape an emergent pattern appears in which cohort structure is uncoupled from any single mortality event and instead appears to be the result of broader scale climate forcing of fire timing that resulted in successful recruitment episodes (Fig. 2). This conclusion, furthermore, suggests that there are minimum spatial and temporal scales over which tree recruitment and fire histories must be assessed to adequately encapsulate relevant patterns and develop robust inferences (see also Falk 2004). Cohesive patterns resulting from climate forcing only emerge at broader spatial and longer temporal scales. While this study does not address what minimal scales for fire and recruitment histories should be, our data strongly suggest that such scales should encompass more than only a few stands and one to two hundred years of tree recruitment and fire history data (e.g., Ehle and Baker 2003).

Land use changes that accompanied Euro-American settlement in the latter half of the 1800s led to cessation of surface fires from virtually all ponderosa pine forests across the western United States (Covington and Moore 1994, Swetnam and Betancourt 1998, Brown et al. 2001, Friederici 2003: Fig. 2c). The proximal cause of fire cessation in the late 19th century was widespread sheep and livestock grazing that was later followed by active fire suppression in the 20th century (Allen et al. 2002). The 135 yr long fire-free period from 1871 to 2005 is more than three times as long as the longest historical fire-quiescent period of 40 yr from 1684 to 1724. Establishment of both ponderosa pine and less fire-tolerant species has been unchecked by fire-caused mortality of seedlings with the result that current forests are much denser than in the past (Covington 2000: Fig. 2d, e). Optimal climatic conditions for ponderosa pine regeneration and establishment in the early 20th century also contributed to denser forests, especially during 1919 when a tremendous pulse of seedlings established across apparently much of the Southwest (Pearson 1933, Savage et al. 1996: Fig. 2d). Increased post-settlement tree density has resulted in altered canopy fuel structures, including formation of ''ladder'' fuels that allow wildfires to burn more severely. Crown fire has largely replaced surface fire in many ponderosa pine forests (Covington 2000, Allen et al. 2002). Record-setting fire sizes and severities in ponderosa pine and closely related forests during fire seasons in 1996, 2000, 2002, and 2003 have been attributed in large part to changes in forest structure and fuel conditions over the period of fire exclusion (Covington 2000, Romme et al. 2003).

Previous modeling of forest dynamics has tended to discount rates of tree recruitment relative to rates of tree mortality (e.g., Shugart 1998). Climatically forced mortality can cause rapid changes in forest structure or composition while recruitment has been viewed as a much slower process (Allen and Breshears 1998). Yet, after the 1580s megadrought, southwestern ponderosa pine forests were able to recover fairly rapidly in following decades that were both relatively climatically optimal for seedling establishment and growth and that saw few fires. This has important implications for prediction of vegetation response after recent massive mortality caused by interactions between drought and insect populations over millions of hectares of piñon and ponderosa pine forests in the southwestern United States (USDA Forest Service 2004). Over multicentury time scales, broad-scale dynamics in the Southwest have included abrupt and synchronized mortality that, as least once before in the relatively recent past, was followed by fairly rapid community recovery. Recruitment data highlight the importance of historically contingent events in formation of forest structure and composition, and the dynamic interaction of climate, disturbances such as fire, and forest structure over both short and long time scales. Historical data aid in modeling possible future responses of plant communities to climate change and accompanying effects in disturbance regimes (Overpeck et al. 1990).

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