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Fire severity, size, and climate associations diverge from historical precedent along an ecological gradient in the Pinaleño Mountains, Arizona, USA

Christopher D. O'Connor[a,c,*], Donald A. Falk[a,c], Ann M. Lynch[b,c], Thomas W. Swetnam[c]

[a] University of Arizona, School of Natural Resources and the Environment, 325 Biosciences East Building, Tucson, AZ 85721, United States
[b] U.S. Forest Service, Rocky Mountain Research Station, 1215 E. Lowell St., Tucson, AZ 85721, United States
[c] University of Arizona, Laboratory of Tree-Ring Research, 1215 E. Lowell St., Tucson, AZ 85721, United States

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ABSTRACT

In recent decades fire size and severity have been increasing in high elevation forests of the American Southwest. Ecological outcomes of these increases are difficult to gauge without an historical context for the role of fire in these systems prior to interruption by Euro-American land uses. Across the gradient of forest types in the Pinaleño Mountains, a Sky Island system in southeast Arizona that experienced two relatively large high-severity fires in the last two decades, we compared fire characteristics and climate associations before and after the onset of fire exclusion to determine the degree of similarity between past and recent fires. We use a gridded fire scar and demography sampling network to reconstruct spatially explicit estimates of fire extent and burn severity, as well as climate associations of fires from individual site to landscape scales from 1640 to 2008 C.E. We found that patterns of fire frequency, size, and severity were relatively stable for at least several centuries prior to 1880. A combination of livestock grazing and active fire suppression after circa 1880 led to (1) a significant reduction in fire spread but not fire ignition, (2) a conversion of more than 80% of the landscape from a frequent, low to mixed-severity fire regime to an infrequent mixed to high-severity fire regime, and (3) an increase in fuel continuity within a mid-elevation zone of dry mixed-conifer forest, resulting in increased opportunities for surface and crown fire spread into higher elevation mesic forests. The two most recent fires affecting mesic forests were associated with drought and temperature conditions that were not exceptional in the historical record but that resulted in a relative proportion of high burn severity up to four times that of previous large fires. The ecological effects of these recent fires appear to be more severe than any fire in the reconstructed period, casting uncertainty upon the recovery of historical species composition in high-severity burn patches. Significant changes to the spatial pattern, frequency, and climate associations of spreading fires after 1880 suggest that limits to fuel loading and fuel connectivity sustained by frequent fire have been removed. Coinciding factors of high fuel continuity and fuel loading, projected lengthening of the fire season, and increased variability in seasonal precipitation suggest that large high-severity fires, especially in mixed-conifer forests, will become the predominant fire type without aggressive actions to reduce fuel continuity and restore fire-resilient forest structure and species composition.

1. Introduction

The average annual area burned in wildfires in the Western U.S. increased more than six fold over the past four decades (Westerling et al., 2006; Littell et al., 2009) with area burned in large wildfires increasing by an average of 355 km^2 annually from 1984 to 2011 (Dennison et al., 2014). The area affected by high-severity fire is increasing as well (Eidenshink et al., 2007; Miller et al., 2009; Cansler and McKenzie, 2014), although the proportional increase in high-severity fire, in which most or all overstory vegetation is killed, is less consistent among western ecoregions and forest types (Dillon et al., 2011). While the human and natural resource costs of recent fires are indisputable, the degree of long-term ecological change resulting from recent fires is not well understood. In many dry forests of the interior West, fire was a

* Corresponding author at: University of Arizona, School of Natural Resources and the Environment, 325 Biosciences East Building, Tucson, AZ 85721, United States. Tel.: +1 520 621 0059.
E-mail address: oconnorc@email.arizona.edu (C.D. O'Connor).

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keystone ecosystem process (Falk, 2006) that shaped forest structure and species composition from stand to landscape scales (Swetnam and Baisan, 1996b; Brown and Shepperd, 2001; Heyerdahl et al., 2001; Taylor and Skinner, 2003; Fulé et al., 2009; Falk et al., 2011). In topographically diverse forests of the southwestern U.S., steep vertical gradients maintained distinct species assemblages and different fire regimes in close spatial proximity and often limited the spread of fire across ecological boundaries (Grissino-Mayer et al., 1995; Stephens, 2001; Margolis and Balmat, 2009; Swetnam et al., 2009). Heterogeneity of forest types and fire regimes contributed to the landscape-scale resilience of forests within these systems by limiting the patch size of high-severity fire (Agee, 1998a; Taylor and Skinner, 2003).

Euro-American settlement of the western states in the mid to late 19th century led to intensive livestock grazing, timber harvesting, and mineral extraction that interrupted natural fire cycles (Bahre, 1998; Swetnam et al., 2004). These actions initiated a series of changes that ultimately homogenized the structure and species assemblages of forests adapted to frequent, moderate to low-severity fires via infilling of canopy gaps by shade tolerant species (Minnich et al., 1995; Allen et al., 2002; Fulé et al., 2003). Conditions in many of the western forests that were adapted to these frequent moderate to low-severity fires now represent forest structure of assemblages adapted to infrequent, high-severity fire with multistoried canopies that promote fire crowning behavior as a result of fuel loading and development of understory ladder fuels (Agee, 1998b; Allen et al., 2002). Recent fires in some western forests may represent a return to historical patterns of fire size and severity following a century of fire interruption (Marlon et al., 2012; Williams and Baker, 2012; Odion et al., 2014), or they may represent a new fire dynamic in which legacies of fire exclusion are interacting with changing climatic conditions to produce fires uncharacteristic of those prior to Euro-American settlement (Crimmins, 2011; Garfin et al., 2013; Williams et al., 2013; Fulé et al., 2013).

Forests of Madrean "Sky Island" systems are a microcosm of western forest types ranging from desert cactus and shrubland to subalpine spruce-fir forest. Vegetation is distributed along steep vertical gradients that mediate temperature and moisture regimes and maintain distinct species assemblages (Whittaker and Niering, 1975; Van Devender and Spaulding, 1979; McLaughlin, 1993). The Sky Island region is expected to experience the ecological impacts of climate warming sooner than many other parts of the west with similar forest ecosystems (Notaro et al., 2012; Garfin et al., 2013), providing an opportunity to observe changes to the dynamic interactions of forests, fire, and climate sooner than in other western forests. Historically, fire regimes in the Sky Islands and elsewhere in the southwestern North America bioregion were regulated by steep ecological gradients. For example high frequency, low-severity fires were associated with pine and dry mixed-conifer forests, and lower frequency, higher-severity fires were associated with mesic mixed-conifer and subalpine forest types (Grissino-Mayer et al., 1995; Swetnam et al., 2001, 2009; Barton et al., 2001; Margolis and Balmat, 2009; Margolis et al., 2011).

In recent decades the region has been the locus of multiple large, mixed-severity fires, providing opportunities to compare current and historical fire regimes. Here we reconstruct the spatial, temporal, and severity patterns of fire encompassing the entire montane zone of a mountain range, spanning gradients of elevation and changing forest composition. The reconstruction covers more than a three century period of record in an area that experienced two large wildfires during the prolonged drought that began in the mid-1990s. We use the historical and contemporary fire record to examine whether there were differences in fire characteristics before and after fire exclusion including (1) frequency of isolated and spreading fires at landscape scale and within forest types; (2) associations between fire spread and climate variations; and (3) fire size, spatial patterns, and proportion of low and high-severity fire. These comparisons allow us to evaluate whether recent high-severity fires are within or outside the historical range of variability in terms of frequency, severity, and spatial extents.

2. Study area

The Pinaleño Mountains in southeast Arizona are the tallest of the Madrean Sky Island ranges, spanning a vertical gradient of more than 2100 m in just 8.6 km of horizontal distance, from Chihuahuan mixed-desert shrubland at 1150 m to spruce-fir forest up to 3268 m. Along this steep elevational gradient, the Pinaleño Mountains contain forest types representative of a latitudinal transect from Sonora Mexico to British Columbia Canada (Warshall, 1995). Forests are distributed along gradients of elevation and aspect (Fig. 1). In the study area above 2135 m, lower forests are comprised of ponderosa pine (*Pinus ponderosa* var. *scopulorum* Engelmann), Gambel oak (*Quercus gambelii* Nutt.), Arizona white oak (*Quercus arizonica* Sarg.), and silverleaf oak (*Quercus hypoleucoides* A. Camus) that transition to a dry mixed-conifer forest dominated by Douglas-fir (*Pseudotsuga menziesii* var. *glauca* (Mirbel) Franco), southwestern white pine (*Pinus strobiformus* Engelmann), and ponderosa pine, with minor components of white fir (*Abies concolor* (Gor. & Glend.) Lindl. ex Hildebr.) and aspen (*Populus tremuloides*). Above 2750 m mesic mixed-conifer forests are dominated by Douglas-fir and white fir with minor components of southwestern white pine, corkbark fir (*Abies lasiocarpa* var. *arizonica* (Hook.) Nutt.) and Engelmann spruce (*Picea engelmannii* Parry ex. Engelm.) (Martin and Fletcher, 1943). At the highest elevations and along north-facing slopes, Engelmann spruce and corkbark fir are dominant species interspersed with occasional Douglas-fir. The Pinaleño Mountains contain the southernmost extent of spruce-fir forest in North America, and dense upper elevation forests serve as critical habitat for one endangered and several threatened wildlife species (Stromberg and Patten, 1991; U.S. Fish and Wildlife Service, 2004; Koprowski et al., 2005, 2006; Sanderson and Koprowski, 2009). The upper spruce-fir forest also hosts the Mount Graham International Observatory, an astrophysical complex that includes the largest optical telescope in the Northern Hemisphere.

2.1. Disturbance history

Two fires in 1996 and 2004 burned a combined 14,160 ha in pine, mixed-conifer and spruce-fir forest, affecting 45% of the forested area above 2135 m (USDA Forest Service, 2013). Prior to the 2004 fire, a series of insect outbreaks in the spruce-fir forest resulted in mortality of 83% of Engelmann spruce and 63% of corkbark fir greater than seven cm DBH (Lynch, 2009; O'Connor, 2013). This mortality, combined with the 2004 fire, resulted in a 66% reduction in the area occupied by spruce and corkbark fir (O'Connor et al., 2013). More than half of the area of these fires burned at mixed- to high-severity based on the relative difference normalized burn ratio (RdNBR; Miller and Thode, 2007; MTBS, 2013). No information is available on the relative proportion of mixed and high burn severity for fires in 1956 and 1975, the only other known 20th century fires in the Pinaleño Mountains exceeding 2000 ha.

Prior to these recent fires, stand-scale studies ranging in size from 15 to 32 ha in two mixed-conifer and one spruce-fir site indicated that over the period 1575–1880 low-severity fire was frequent in parts of the mixed-conifer forest, occurring approximately every 4.2 years (Grissino-Mayer et al., 1995). No fire scars were found within the spruce-fir site and tree demographic evidence corroborated with fire scar dates from nearby mixed-conifer

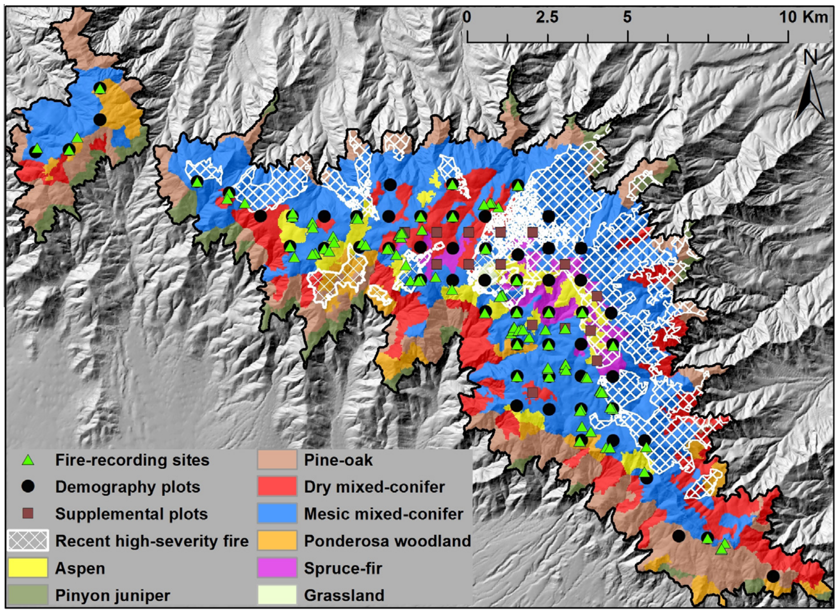


Fig. 1. Pinaleño Mountain forest types and sampling locations. Colored area denotes elevation above 2135 m. Underlying vegetation layer represents pre-1880 forest types generated from LANDFIRE biophysical setting (LANDFIRE, 2013). High-severity burn patches are calculated from the relative difference normalized burn ratio (MTBS, 2013) categorized according to Miller and Thode (2007). All raster values are generalized to a minimum patch size of 6.75 ha.

forest suggested that the sampled spruce-fir forest established after a stand-replacing fire in 1685 (Grissino-Mayer et al., 1995; Swetnam et al., 2009; Margolis et al., 2011). Grissino-Mayer et al. (1995) noted an abrupt decline in fire frequency in the mixed-conifer sites after 1880 and raised concerns that changes to forest structure and species could alter future fire behavior. Two other studies on the dynamics of the spruce-fir forest used tree age structure (Stromberg and Patten, 1991) and charcoal deposition (Anderson and Smith, 2009) to conclude that fire in parts of this forest type was rare, occurring at frequencies of 300 to more than 1000 years. Although these previous stand-level studies provided valuable insights, such selected and opportunistic-sampling designs have been argued to be biased and unrepresentative of broader landscape spatial patterns and temporal trajectories (e.g., Williams and Baker, 2012; Odion et al., 2014, but see Farris et al., 2013). This study expands upon the earlier dendroecology studies in the Pinaleño Mountains with a spatially unbiased, landscape-scale sampling of forest demography and fire history throughout the pine, dry and mesic mixed-conifer, and spruce-fir forests.

3. Methods

To reconstruct the spatial, temporal, and severity components of historical fire regimes in pine, dry and mesic mixed-conifer, and spruce-fir forest types, we used a systematic grid of 58 0.05-ha circular plots spaced one kilometer apart (Fig. 1). A secondary grid of 15 supplemental plots offset 500 m northwest of fixed area plots was used to increase sampling resolution in and around the spruce-fir forest. Gridded sampling designs have been shown to more efficiently capture spatial variability of simulated fire return intervals than random or stratified (targeted) sampling designs (Parsons et al., 2007; Farris et al., 2013). Plot locations were determined through a GIS overlay in advance of field sampling. Two plots that included roads or other highly modified areas were relocated 50 m in a direction perpendicular to the constructed feature. We collected up to three increment cores from live trees or a single cross section from snags and stumps with diameter at breast height (DBH) 19.5 cm or larger. Trees between one and 19.4 cm DBH were sampled on a nested sub-plot equal to one third the area of the full plot (0.017 ha). Samples were taken within 20 cm of the soil surface whenever possible to minimize the need for sampling height correction of pith dates for tree age. In supplemental plots, cores or cross-sections were collected from the 10 spruce and 10 non-spruce trees nearest plot center with DBH greater than 15 cm.

Fire-scarred material was collected from live trees, snags, and stumps within demography plots and while traveling among plots (Heyerdahl et al., 2014). All fire-scarred samples collected outside of demography plots were geo-referenced, and site characteristics were recorded and photographed. When appropriate, several samples from individual stumps and snags were collected to preserve as many fire dates as possible (Dieterich and Swetnam, 1984). The original collections of Grissino-Mayer et al. (1995) were combined with the landscape-scale fire history reconstruction and were in close proximity to four of the 73 demography sampling locations.

Increment cores and cross-sections were mounted and sanded with progressively finer grits until the wood cell structure was observable under 45× magnification (typically 400 grit). All samples were crossdated using a combination of visual pattern matching (Yamaguchi, 1991), skeleton plots (Speer, 2010), and statistical correlation analyses (Holmes, 1983; Grissino-Mayer, 2001a). On fire-scarred samples we recorded inner ring or pith date, year and season of fire (when determinable), outer year or bark date, scars of undetermined origin, and growth suppression or release dates. Scars of undetermined origin, injury-related growth suppressions, and outer ring (death) dates corresponding to fire scars recorded within 500 m of a sample were recorded as fire dates; otherwise they were excluded from fire history analyses (Farris et al., 2013; Heyerdahl et al., 2014).

We used fire perimeter records from the Coronado National Forest fire atlas database for the period 1974–2008 (USDA Forest Service, 2013) and digitized the perimeter of the 1956 Nuttall fire from a hand drawn map from Forest historical archives. Fire perimeters prior to 1956 were reconstructed from the fire scar record. A map of logging activity from 1880 to 1970 was used to screen seedling recruitment for effects of logging activities.

4. Analysis

To differentiate ponderosa pine, dry mixed-conifer, mesic mixed-conifer and spruce-fir forest types we used the similarity of species assemblages, derived from Importance Value (IV) rankings (Cottam and Curtis, 1956; Taylor, 2000). IVs were calculated from the relative frequency and basal area of the six dominant conifer species in the year 1870 to account for changes to species composition, frequency, and basal area over the fire-interrupted period. Similarity of species assemblages, derived from IV rankings, was used to perform a cluster analysis of plots. Douglas-fir was the most abundant species throughout all types of mixed-conifer forest, so IVs of pine species were double-weighted to enhance the differentiation between dry and mesic mixed-conifer forest types (McCune et al., 2002). Hierarchical clustering was based on Ward's method of minimizing within-group variance among plots (Legendre and Legendre, 1998) using the *hclust* package in the R statistical computing environment (R Core Team, 2012). Prior to hierarchical clustering, the plot variable matrix was transformed to Jaccard distance to minimize the effect of zeros in the dataset on the clustering results (McCune et al., 2002). Demography plots without evidence of fire were excluded from the analysis unless fire history information was collected from plots on three adjacent sides, in which case they were grouped with nearest neighbors (Supplemental Fig. 1).

To observe patterns of fire spread, we used geo-referenced fire records composited at the plot level. Fire dates recorded within 500 m of a plot center were ascribed to a single plot to allow direct comparison of discrete 1 km^2 fire-recording sites (Dieterich, 1980; Farris et al., 2010). Composited fire records were filtered to include only fires recorded on two or more trees to reduce the possibility of incorporating non-fire related scars in the analysis. A composite record of fire years from multiple samples over a discrete spatial unit produces a more complete record of fire occurrence because individual trees and samples are imperfect recorders of fire, and scars are sometimes eroded or burned off in subsequent events (Dieterich, 1980; Dieterich and Swetnam, 1984; Falk et al., 2011; Farris et al., 2013). Compositing of fire records assumes topographic and ecological homogeneity within the specified scaling unit. While homogeneity at the kilometer scale cannot be assumed for all forest types and across the vertical gradient sampled in the Pinaleño Mountains, variability among one km^2 gridded cells was assumed to be greater than variability within cells. Compositing fire records by a standard spatial unit removes sampling bias generated by the natural aggregation of fire-recording sites and allows for an unbiased comparison of sites across a landscape (Farris et al., 2013).

The fire scar record is a conservative estimate of actual fire frequency at individual tree and composited site scales. Scar formation is a highly variable process dependent upon fire metrics such as tree-scale fire intensity, flame lengths, and residence time, individual tree properties such as bark thickness, diameter, and presence and condition of a prior scar, and site characteristics such as topographic position and time since last fire (Gutsell and Johnson, 1996; Swetnam and Baisan, 1996a; Baker and Dugan, 2013). In a study of scar formation following low and mixed-severity fires in ponderosa pine forests of northern Arizona, the probability of scar formation, and subsequent recording of a fire event was significantly different for trees with no previous fire scarring ($\sim$0.38) and trees with one or more previous fire scars ($\sim$0.89) (Baker and Dugan, 2013). This finding supports the definition of a recorder tree as defined by Swetnam and Baisan (1996a,b) and was the logic behind preferentially sampling trees with multiple fire scars where possible when traveling between fire and demography plot locations. Preferential sampling of multi-scarred trees reduces the possibility type two error (false negative) as noted by Baker and Dugan (2013), and reduces the number of samples necessary to approach the true fire frequency of a given site (Baker and Ehle, 2001).

4.1. Fire statistics by forest type

We calculated fire frequency statistics for composited sites over the pre-fire exclusion period when 20% or more of sites were recording, and during the post fire-exclusion period after 1880. Separate analyses were performed for pine, dry mixed-conifer, mesic mixed-conifer and the full study area. Mean and median fire intervals, Weibull median probability interval (WMPI), and minimum and maximum fire intervals were generated from the Fire History Analysis and Exploration System (FHAES) (Grissino-Mayer, 2001b; Sutherland et al., 2013). The Weibull model fitted to a frequency distribution of fire intervals has been shown to be a statistically more robust estimator of fire return interval in southwestern U.S. forests than mean interval values, because fire intervals tend to be right-skewed and include occasional extreme values (Grissino-Mayer, 1999; Falk and Swetnam, 2003). For the purposes of simplifying language, fire return intervals throughout this text are the Weibull median probability interval (WMPI) unless otherwise specified. For fire interval analysis, we used the convention of cumulative fire size classes with a lower size limit only. Fire intervals within forest types and at the full study area scale were calculated for all fires recorded in one or more sites, 10% or more of sites, 33% or more of sites, and 50% or more of sites (Table 1). For spatial analysis of fire size (Table 2), discrete fire sizes with lower and upper bounds were used to differentiate between small (3–10% of sites), spreading (11–49% of sites), and landscape scale ($\geqslant$50% of sites) fires. The standardized size of recording sites allows for a minimum spatial estimate of fire size based on the number of recording sites. We used a two tailed Student's *t*-test to identify significant differences ($p < 0.05$) in mean fire intervals between the pre-fire exclusion period (1640–1880) and the post fire-exclusion period (1881–2008) in each forest type and fire size class across the study area (FHX2 software package, Grissino-Mayer, 2001b).

To test for changes in patterns of fire size before and after Euro-American settlement, we standardized the total number of fires recorded during the two periods of analysis to 100 and then rescaled the number of fires recorded in each size class. We used a one-tailed t-test with unequal variance (Gotelli and Ellison, 2004) to test for significant differences in the total number of fires in each size class during the pre and post-fire exclusion periods.

4.2. Climate associations with fire size

Temporal relationships between drought conditions, ocean-atmospheric oscillations, and fires at the mountain range scale were identified with superposed epoch analysis (SEA) (Lough and Fritts, 1987; Swetnam, 1993). SEA tests for significant departures from the range of annual values in a continuous climate variable, in relation to a series of event years. In fire history analysis, values of a climate variable prior to and during individual fire years are compared with the distribution of values for the full time domain of the climate series. We tested the statistical significance of the fire year correlations to climate variables with 1000 bootstrapped random event years compared to actual event years (Holmes and Swetnam, 1994). Fire size classes used for fire-climate analysis were the same used for analysis of changes to fire size (small, spreading, landscape scale). Climate variables included reconstructed summer (June–August) Palmer Drought Severity Index (PDSI) for Southeast Arizona (Cook and Krusic, 2004, grid point

Table 1
Fire interval statistics by forest type and the full study area. Fire metrics are based on 241 fire-scarred trees distributed along a grid of 43, 1-km^2 composited sites with a minimum of two trees scarred. Fire intervals are filtered by the minimum percentage of sites recording as a proxy for fire size (Swetnam and Baisan, 1996a). Filtered fire statistics for the full study area correspond roughly to small (10%), spreading (33%), and landscape (50%) scale fires. WMPI is Weibull median probability interval.

Panel A: Fire Statistics 1640 to 1880										
	Pine-dominated (11 sites)					Dry mixed-conifer (20 sites)				
Sites recording	*N*	Mean FI	Median FI	WMPI	Min/Max FI	*N*	Mean FI	Median FI	WMPI	Min/Max FI
All scars	97	2.4*	2	2.1	1/11	98	2.4	2	2.0	1/11
10%	48	4.8	4	4.1	1/18	69	3.4*	3	2.9	1/12
33%	23	9.6	9	9.1	3/21	24	9.3	10	9.0	3/18
50%	12	18.0	17	17.2	4/34	19	10.7	11	10.4	3/20
	Mesic mixed-conifer (12 sites)					Study area (43 sites)				
Sites recording	*N*	Mean FI	Median FI	WMPI	Min/Max FI	*N*	Mean FI	Median FI	WMPI	Min/Max FI
All scars	56	4.1*	3	3.4	1/18	141	1.7*	1	1.5	1/11
10%	48	4.8	4	4.1	1/18	64	3.6*	3	3.2	1/12
33%	18	11.9	11	10.4	2/32	20	10.8	11	10.5	4/20
50%	9	23.9	24	21.6	4/44	11	19.6	16	18.6	8/44
Panel B: Fire Statistics 1881 to 2008										
	Pine-dominated					Dry mixed-conifer				
Sites recording	*N*	Mean FI	Median FI	WMPI	Min/Max FI	*N*	Mean FI	Median FI	WMPI	Min/Max FI
All scars	25	4.6*	4	3.9	1/15	28	4.3	2	3.0	1/27
10%	7	10.3	4	6.2	1/35	10	12.3*	6	8.3	1/40
33%	1	–	–	–	–	1	–	–	–	–
50%	1	–	–	–	–	0	–	–	–	–
	Mesic mixed-conifer					Study area				
Sites recording	*N*	Mean FI	Median FI	WMPI	Min/Max FI	*N*	Mean FI	Median FI	WMPI	Min/Max FI
All scars	20	5.9*	5	5.1	2/17	54	2.3*	1	2.0	1/8
10%	1	–	–	–	–	6	21.8*	25	18.2	2/40
33%	0	–	–	–	–	0	–	–	–	–
50%	0	–	–	–	–	0	–	–	–	–

– Indicates not enough fires to calculate statistics or compare periods.
* Indicates significant change in fire interval between analysis periods ($\alpha < 0.05$).

Table 2
Change in the relative proportion of small, spreading, and landscape-scale fires before and after fire exclusion. Fire size classes used here are distinct subsets of total fires from 1640 to 1880 ($n = 141$) and 1881–2008 ($n = 54$).

Classification	Sites recording	Proportion of sites (%)	Size (ha)	Count of fires	Fires per year	% fire years
Panel A: Fire size statistics 1640–1880						
Single site	1	⩽2	<100	6	0.03*	4
Small	2–5	3–10	100–500	104	0.43*	74
Spreading	6–21	11–49	600–2100	20	0.08*	14
Landscape	⩾22	⩾50	>2,200	11	0.05	8
Panel B: Fire size statistics 1881–2008						
Single site	1	⩽2	<100	24	0.19*	44
Small	2–5	3–10	100–500	27	0.21*	50
Spreading	6–21	11–49	600–2100	3	0.02*	6
Landscape	⩾22	⩾50	>2200	–	–	0

–Indicates not enough fires for comparison.
* Indicates significant difference between periods.

105) and Niño3 index from Mexico and Texas (Cook et al., 2009) at annual lags of one to six years prior to a fire year. In the Southwest, drought in the spring and prior winter are correlated strongly with tree-ring reconstructed summer PDSI (St. George et al., 2010) which has been shown to influence fire probability during the pre-monsoon period (Baisan and Swetnam, 1990; Swetnam and Baisan, 1996a; Westerling et al., 2003). The Niño3 index is a tree-ring reconstructed proxy for the El Niño Southern Oscillation (ENSO) winter sea surface temperature in the Pacific Ocean (5°N–5°S, 90°–150°W). A positive Niño3 index indicates warm sea surface temperatures associated with moist El Niño winter conditions in the Southwest U.S. and subsequent high snow accumulation and reduced likelihood of spreading fire in the spring. Conversely, a negative Niño3 index correlates with dry La Niña winter conditions in the Southwest U.S. and conditions favorable for spreading fire (Swetnam and Betancourt, 1990a,b; Diaz and Markgraf, 2000; Margolis and Swetnam, 2013).

To meet the statistical assumption of inter-annual independence for SEA analysis, we used an autoregressive model to remove year-to-year autocorrelation in PDSI and Niño3 index values (Heyerdahl et al., 2011; Margolis and Swetnam, 2013). The autoregressive model residuals are a "prewhitened" version of the original index series that meets the statistical requirements for SEA analysis. There were no statistically significant differences in fire–climate relationships between raw and "prewhitened" climate indices. We compared fire–climate relationships in the pre-fire exclusion period (1640–1880) to the post fire-exclusion period (1881–2008) to determine changes attributable to fire exclusion and recent warming trends.

4.3. Spatial reconstruction of fire size and severity

Fire severity terminology used in this study is based on the degree of overstory tree mortality, in which low-severity fire is limited primarily to surface mortality of seedlings, saplings, and shrubs, mixed-severity fire includes patches of canopy tree mortality but effects are primarily on surface vegetation, and high-severity fire describes complete or near complete mortality of canopy trees (Turner et al., 1999). To reconstruct the size and severity of past fires we used a combination of composited fire-scar site locations (Iniguez et al., 2009; Farris et al., 2010) and demography plots (Heyerdahl, 1997; Heyerdahl et al., 2001; Brown and Wu, 2005; Margolis et al., 2007) (Supplemental Fig. 2). The spatial extent of fire reconstructions was determined by applying a 750 m buffer to the grid of demography plots to generate a continuous fire reconstruction surface between spatially discrete composited fire records (Hessl et al., 2007; Swetnam et al., 2011). Interpolated surfaces of individual fire sizes and severities were based on inverse distance weighting of four nearest point locations with a power function of 2 and a raster cell size of 30 m (ESRI Inc., 2012).

Patterns of fire spread were examined by reconstructing the spatial pattern of mean fire return intervals for all fires recorded in two or more sites during the pre-settlement and post-settlement periods. A fire interval surface was interpolated over the sampled area through inverse distance weighting of mean fire interval values assigned to each point on the gridded surface. We used the interpolated fire interval surface to identify patterns of fire return intervals in relation to forest types, landscape features, and period of analysis. Prior to 1955, fire perimeters were estimated by interpolating the area covered by the gridded sampling network. Fires after 1955 were mapped from fire atlas data.

To reconstruct severity of historical fires we used a combination of fire scars, tree establishment cohorts, and tree death dates to ascribe a fire severity class to each gridded fire composite site for each fire year. The formation of a tree cohort, defined as four or more trees recruiting in a 20-year period after a fire (adapted for 0.05 ha plots from Heyerdahl et al., 2011), was one of several criteria used to differentiate between fire severity classes. Severity was classified as low (value "1") if fire scars were recorded but there was no evidence of cohort formation, mixed (value "2") if a combination of fire scars and death dates and one or more cohorts were present, and high (value "3") if there was a single post-fire cohort, no record of recruitment prior to the cohort, and no evidence of fire scar formation with or without tree death dates. Sites were coded as "No Data" until they began recording fire and "0" if they did not record fire during a specific fire year. For a discussion of the assumptions of multiproxy reconstruction of historical fire severity see Heyerdahl et al. (2011, 2014). To calculate patch size, the continuous interpolated severity surface was binned such that values greater than 2.6 were high-severity, 1.5–2.6 were mixed-severity, 0.3–1.5 were low-severity, and <0.3 were no fire. Thresholds for fire-severity bins were calibrated to satellite-derived vegetation burn severity thresholds of RdNBR (MTBS, 2013) from Miller and Thode (2007) for plots located within the burn perimeters of the 1996 and 2004 fires. Satellite-derived fire severity classes for recent fires were verified by seedling recruitment and tree survivorship within demography plots.

5. Results

Spatial and temporal reconstructions of historical fires were based on 1201 crossdated fire scars collected from 241 trees at 130 fire recording locations, and 1222 tree establishment dates from 2178 crossdated samples. We identified 231 unique fire years over the period 1403–2008, but limited the spatial reconstruction of fires to the period 1640–2008 when 20% or more of sites were recording (Fig. 2a). The 10 mesic spruce-fir sites comprising 19% of the study plots had no fire-scars recorded over the study period. Fires affecting two or more sites were common over the period 1640 to 1880, but became less frequent in the 20th century when the average interval between small fires doubled and landscape-scale fires ceased (Fig. 2b and c).

5.1. Fire size and fire interval at forest type and landscape-scales

Return intervals for fires recorded in 10% or fewer of sites (<500 ha in size) varied little across pine and mixed-conifer forests prior to 1880 (Table 1A). Fires on the order of 100–200 ha typically occurred every 2–4 years in pine, dry mixed-conifer and mesic mixed-conifer forest types. Larger fires affecting up to 33% of sites typically occurred at 9–11 year intervals and were recorded on up to 600 ha within individual forest types and 1400 ha at the scale of the entire study area. Fires recorded in more than half of sites were more variable, occurring most frequently in dry mixed-conifer forest (WMPI 10.4 years), somewhat less frequently in pine-dominated forest (WMPI 17.2 years), and least frequently in mesic mixed-conifer forest (WMPI 21.6 years). Over the entire study area, the majority of fires (74%) were approximately 100–500 ha, and 8% of fires were greater than 2200 ha and were recorded in at least two forest types (Table 2A).

Within individual forest types with fire recording trees, the frequency of small and spreading fires varied considerably. Approximately half of fires in pine forests (51%) were small, recorded in fewer than 10% of sites and with a return interval of approximately four years ($n = 49$, WMPI 2.1–4.1 years) (Table 1A). Fires recorded in 33% or more of pine sites occurred at intervals of 10–17 years. In dry and mesic mixed-conifer forests where spatial connectivity among sites was greater, the majority of fires were recorded in 10% or more of sites (70% of dry mixed-conifer fires and 86% of mesic mixed-conifer fires). In dry mixed-conifer forest, fire return intervals were smaller and fire sizes were larger than in any other forest type (WMPI 2–10.4 years, for fires recorded in 1–50% or more of sites respectively). In mesic mixed-conifer forest, fire return intervals were generally longer than those of other forest types (WMPI 3.4–21.6 years for fires recorded in 1–50% or more of sites respectively) and the proportion of fires recorded in more than half of sites (16%) was intermediary between pine (12%) and dry mixed-conifer (20%) forest types. Variability of fire return intervals at individual sites in mesic mixed-conifer forest appears to have been influenced by proximity to dry mixed-conifer or spruce-fir forest. Sites abutting dry forests typically recorded fires at 1–18 year intervals, whereas sites abutting the cool, moist spruce-fir forest recorded fires at 17–44 year intervals. Within the upper elevation spruce-fir forest, 10 sites had no evidence of fire scars but had tree demographic evidence of high-severity stand replacing fire in 1685 and 2004.

After 1880 the proportion of fires recorded in 10% or more of sites across the study area dropped from 45% to 11%, and no fires were recorded in 33% or more of sites (Table 1B). Within individual forest types and at the landscape scale, variability in WMPI increased significantly (t-test for difference in variance of the interval distributions, $p < 0.01$) and return intervals for fires recorded in more than 10% of sites increased six fold (Table 1 parts A and B). There were not enough fires recorded in 33% or more of any forest type to calculate fire statistics. Fires recorded in fewer than 10% of sites accounted for 72%, 67%, and 95% of fires recorded in pine, dry mixed-conifer, and mesic mixed-conifer forests respectively.

5.2. Change in the proportion of fire sizes

The proportions of small, spreading, and landscape-scale fires were significantly different in the pre and post fire-exclusion periods (Table 2). The proportion of fires confined to a single site (<100 ha) increased 11-fold from 4% of fires prior to 1880, to 44% of fires after 1880. Fires recorded in 10% or less of the study area (fires <600 ha) comprised 78% of all fires prior to Euro-American settlement and 94% of fires after settlement. The proportion of spreading fires affecting more than 10% of sites (fires >600 ha) decreased by more than threefold after 1880 from 22% to 6%. Landscape-scale

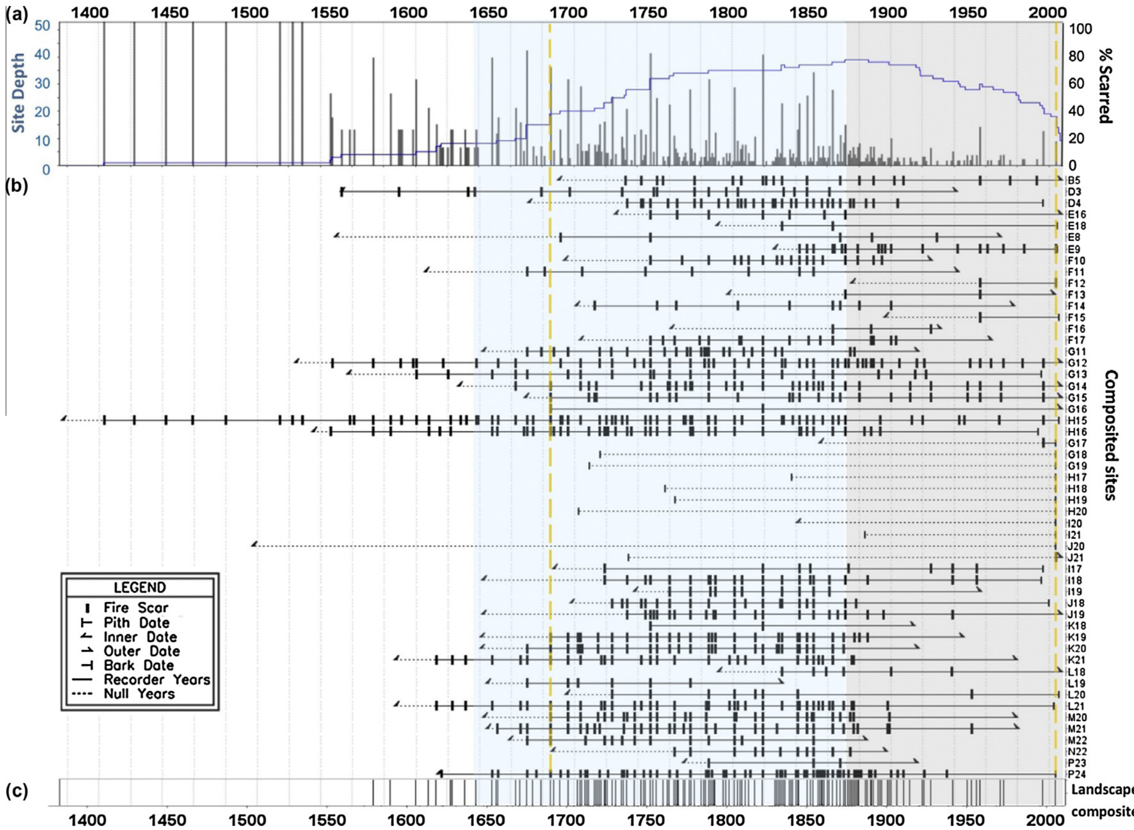


Fig. 2. Landscape-scale fire history of the Pinaleño Mountains above 2135 m elevation. Chart is based on 43 composited 1-km^2 fire recording sites and 10 additional spruce-fir sites that recorded only the most recent fire in 2004. (a) Sample depth and percent of sites scarred. (b) Chronology of fires recorded at each composited site location; horizontal lines are time spans and vertical tick marks are fires recorded by two or more trees within each site. (c) The composite record depicts all fires events recorded in two or more sites. Periods of analysis from 1640 to 1880 and 1881 to 2008 are shaded in blue and gray, respectively. Stand replacing fires in 1685 and 2004 are identified by a dashed orange line. Sites are arranged along a spatial continuum from northwest (top) to southeast (bottom). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

fires recorded on more than 2200 ha (50% of the study area) comprised 8% of total fires prior to 1880 and no fires after 1880.

5.3. Fire severity

Evidence of high-severity fire at the plot-scale was recorded during three events in the reconstructed period in 1685, 1996, and 2004. High-severity fire in 1685 is recorded as establishment of fire-recording trees (pith dates and inner ring dates) shortly after 1685 with no evidence of tree survival at the plot scale prior to 1685 (Fig. 2b). High-severity fire in 1996 and 2004 is recorded as tree death dates (outer bark) in their respective fire years with no evidence of tree survival beyond the year of fire at the plot scale (Fig. 2b). The number of sites recording high-severity fire, a proxy for high severity patch size, was greatest in 1685 (12 sites total, 10 sites adjacent to one another), followed by 2004 (12 sites total, 9 sites adjacent to one another) and 1996 (5 sites total, 3 sites adjacent to one another). The lack of fire scar and demography records prior to 1685 in four sites that subsequently recorded fire for several hundred years (Fig. 2B) suggests that severity of the 1685 fire in mesic and dry mixed-conifer forest was anomalous in the period of reconstruction. Demographic and fire scar evidence of mixed-severity fire was recorded in 1685 (2 sites), 1733 (1 site), 1773 (2 sites), 1785 (1 site), 1819 (1 site), and 1974 (1 site).

5.4. Fire and climate relationships at landscape and local scales

Over the period 1640–1880, spreading and landscape-scale fires were both significantly associated with regional drought and the onset of cool phase La Niña conditions (Fig. 3a–d). Smaller fires (not shown) were associated with moderate drought years but were not significantly correlated with PDSI or ENSO over this period. Spreading fires were more strongly associated with prior wet conditions, which probably indicates the importance of fuel accumulation prior to fire in dry forests (Swetnam and Baisan, 1996a) (Fig. 3a). Landscape scale fires were associated with extreme winter and spring drought the year of fire, as indicated by strong associations to PDSI (St. George et al., 2010), and a weaker pattern of antecedent wet conditions up to five years prior to the fire (Fig. 3b). Spreading fires were associated consistently with the first year of strong sea surface cooling (La Niña conditions) following a prolonged 3–5 year period of warm-phase El Niño conditions (Fig. 3c and d).

During the period of fire exclusion after 1880, spreading fires were correlated significantly with drought the year of fire and persistent drought 1–2 years prior to fire (Fig. 3e), but were no longer associated with prior wet or cool phase La Niña conditions (Fig. 3f). Climate associations of small fires were unchanged following fire interruption, maintaining associations with moderate drought but no significant correlation with PDSI or the Niño3 index (data not shown).

5.5. Climate associations of high-severity fires

The fire that burned into and across the spruce-fir forest in 1685 occurred during very different drought and ENSO conditions than recent fires with similar high-severity effects on the spruce-fir system in 2004, and to a lesser extent in 1996 (Fig. 4). Conditions

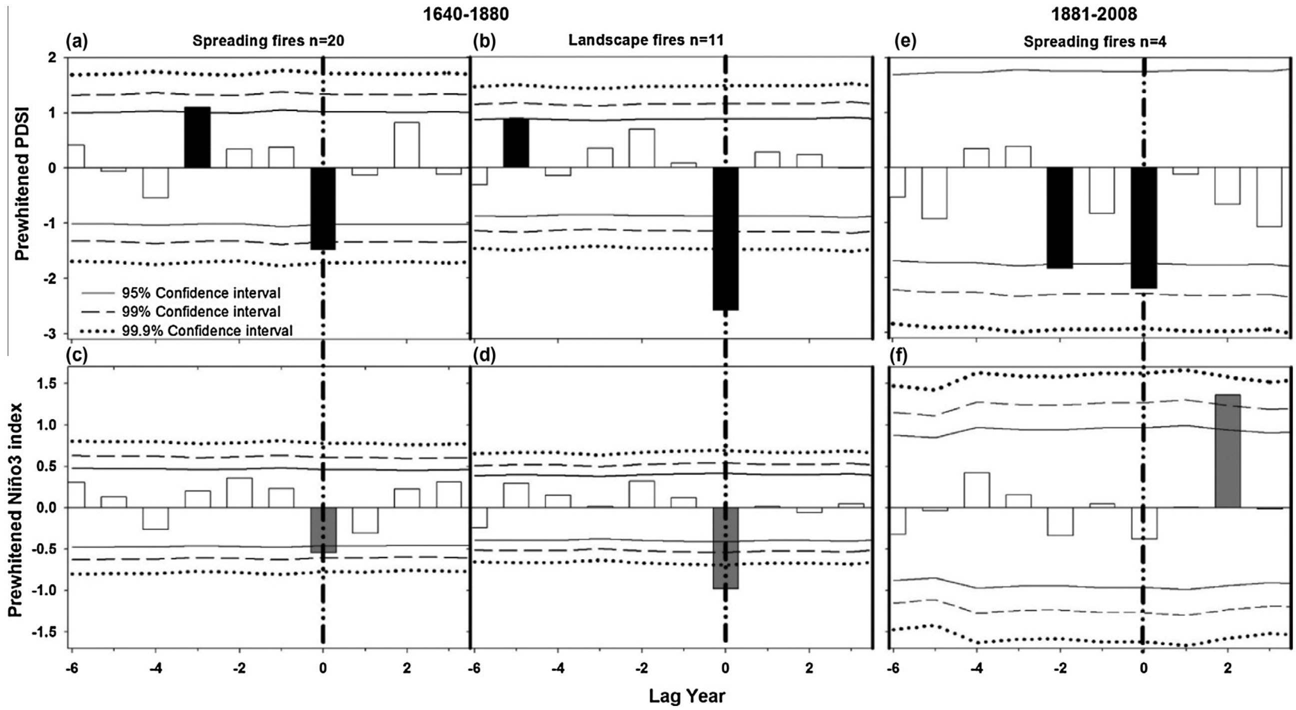


Fig. 3. Fire-climate associations for spreading fires 1640–1880 and 1881–2008. Fires are reconstructed from 43 1-km^2 composited fire-recording sites. Climate relationships are lagged up to six years; vertical dashed lines indicate year of fire. Colored bars indicate significant relationship surpassing the 95% confidence interval. Palmer Drought Severity Index (PDSI) and Niño3 index were prewhitened to remove inter-annual autocorrelation. Spreading fires are recorded in 11–49% of sites and two forest types (20 fires pre-1880, 4 fires post 1880). Landscape fires are recorded in 50% or more of sites and three or more forest types (11 fires). No landscape scale fires are recorded after 1863.

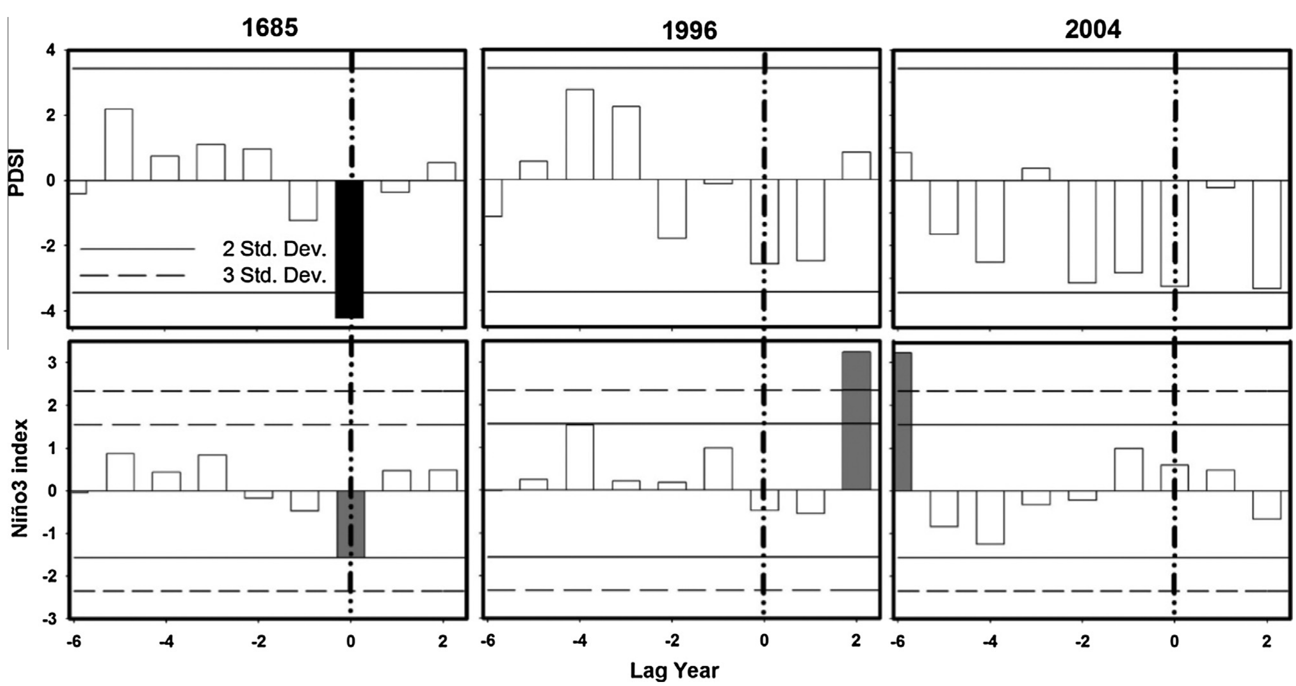


Fig. 4. Drought and ENSO conditions before and during the three high-severity fires affecting the spruce-fir forest (1685, 1996, and 2004). Vertical dashed line indicates year of fire. Palmer Drought Severity Index (PDSI) and Nino3 index are raw series from Cook and Krusic (2004) and Cook et al. (2009) respectively. Reference lines indicate two and three standard deviations in the climate series over the period 1684–2006.

prior to and during the 1685 fire were extreme examples of conditions associated with 10 other widespread landscape fires prior to 1880, characterized by fire occurrence the first year of severe drought following a pluvial period and coinciding with a switch from El Niño to La Niña conditions. The cold season drought in 1685 was the most severe in the more than 400-year period from 1586 to 2008 (PDSI-4.2) and followed a four-year pluvial period (Cook and Krusic, 2004). The Niño3 index indicates that 1685 was the eighth strongest La Niña year (Niño3 index-1.6) over the same period. The fire in 1996 coincided with drought and Niño3 index patterns comparable to but less severe than 20 spreading fires that were recorded in less than half of the study area prior to 1880, and was the only such fire to enter into the spruce-fir forest type. In contrast, the 2004 fire occurred four years into a persistent drought and did not follow a pluvial period. PDSI in 2004 indicated moderate winter drought conditions (PDSI-3.3, 13th most severe drought during past 400 years) during the fifth most severe persistent drought in 400 years (3-year mean of PDSI-3.1). The 2004 fire was the only fire during persistent drought that was recorded in the spruce-fir forest. Additionally, the 2004 fire was the only fire recorded in more than 10% of sites that occurred during an El Niño event (in this case, a relatively weak event, Niño3 index 0.6).

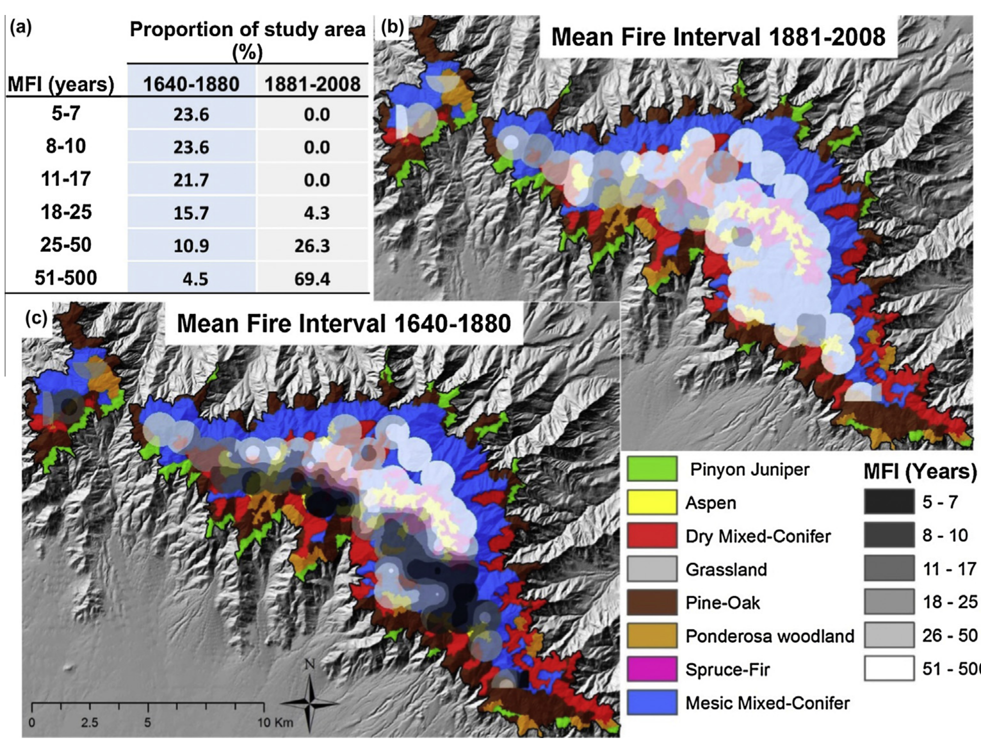
5.6. Spatial reconstruction of fire

We reconstructed fire occurrence from fire-scar evidence in 43 plots and from demographic evidence in an additional 10 plots. Composite fire scar records averaged 5.2 fire-recording trees per plot (range 2–35 trees). Comparisons of relative fire sizes before and after the availability of 20th century fire maps is limited to the 6470 ha area sampled for fire scar and demography plot evidence of fire.

Fire frequency and spatial distributions changed considerably after 1880 (Fig. 5). Prior to 1880, the predominately southwest-facing pine and dry mixed-conifer sites recorded the most fire, forming a nearly contiguous corridor of fire spreading at 5–10 year intervals along the central plateau of the range (Fig. 5a and c). Mesic drainages and mixed-conifer stands abutting the spruce-fir forest served as barriers to fire spread in the majority of fire years. Of 31 fires >600 ha over the 240 year period, only the 1685 fire breached the mesic mixed-conifer forests and burned into the spruce-fir zone. After 1880, fire intervals over more than 90% of the study area more closely resembled those of the mesic mixed-conifer and spruce-fir forest that comprised less than 16% of the study area prior to 1880 (Fig. 5a and b). Fire spread was no longer contiguous along the central plateau of the range, although fires continued to burn at lower frequency in predominately northwestern dry mixed-conifer and pine sites. Sites that sustained fire at 2–10 year intervals along the central and southeastern parts of the range prior to 1880 were affected by a maximum of two fires in the past 130 years, and nearly half of the pine and dry mixed-conifer forest experienced no fire after 1900.

From 1640 to 2008, 11 landscape fires each burned more than 50% of the study area, affecting most of the pine and dry mixed-conifer sites and occasionally burning into mesic mixed-conifer and the edge of the spruce-fir zone (Supplemental Fig. 3). These large fires appear to have spread from the southeast to northwest, following the prevailing wind direction for spring and early summer in this region (Crimmins, 2011; WRCC, 2012), and burning across dry forests along south-facing aspects. A somewhat isolated pine forest on the northwest end of the range rarely burned in large, widespread fires but burned frequently in smaller local fires, and these events were generally asynchronous with fires in the pine and mixed-conifer forest on the main portion of the mountain range (Supplemental Fig. 3).

The spatial footprint of the 1685 fire was similar to that of other landscape-scale fires with the exception of an approximately 1940 ha high-severity patch in the spruce-fir and surround high-elevation mesic mixed-conifer forest (Fig. 6a). The 1685 fire is recorded in fire scars as a low to mixed-severity event in pine and dry mixed-conifer forest types, and as a distinct pulse of seedling establishment in spruce-fir and adjoining mesic mixed-conifer forest, indicating stand-replacing severity in these sites (O'Connor,



(a)

MFI (years)	Proportion of study area (%) 1640-1880	1881-2008
5-7	23.6	0.0
8-10	23.6	0.0
11-17	21.7	0.0
18-25	15.7	4.3
25-50	10.9	26.3
51-500	4.5	69.4

Fig. 5. Changes to mean fire interval from 1640 to 1880 and 1881 to 2008. Tabular summary of percent of landscape comprising each fire interval class (a) and spatial distribution of mean fire intervals superimposed on historical forest types from 1881 to 2008 (b) and 1640 to 1880 (c). Mean fire interval (MFI) surface is based on inverse distance weighting of 53 composited fire-recording sites using four nearest neighbors with a power factor of two. Fire intervals are calculated from fires recorded in two or more sites from 1640 to 1880 (n = 104) and 1881 to 2008 (n = 27). Underlying vegetation layer represents pre-1880 forest types generated from LANDFIRE biophysical setting (LANDFIRE, 2013) with raster values generalized to a minimum patch size of 6.75 ha.

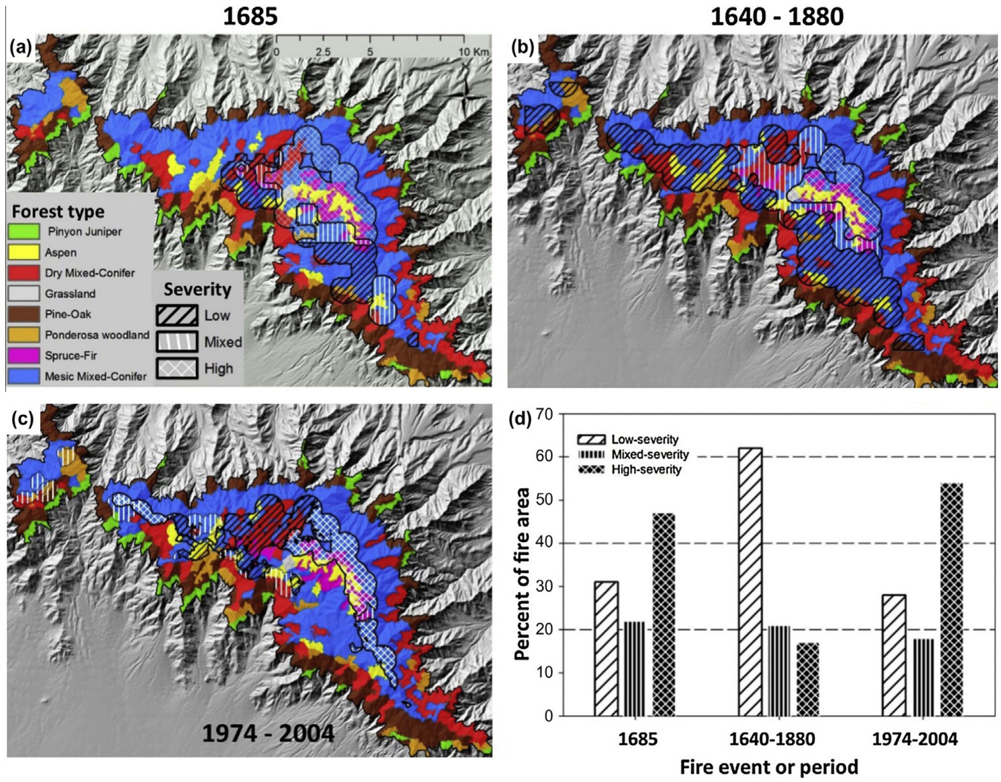


Fig. 6. Spatial extent and severity of historic and contemporary fires. Severity of fires prior to 1974 is reconstructed from 41 composited fire-scar locations, and 12 demography plots indicating post-fire cohort recruitment (parts a and b). Polygons are interpolated from inverse-distance weighting of four nearest neighbors with a power function of two. Modern fire severities (c) are reclassified from relative difference normalized burn ratio for fires after 1984 (Miller and Thode, 2007; MTBS, 2013), and fire scar and demographic evidence associated with fire perimeters (Coronado NF GIS 2013) from 1975–1983. Percent fire area classified by burn severity is specific to a single fire in 1685 and is the mean value of fires recorded in two or more sites from 1640–1880 and 1974–2004 (part d). The underlying vegetation layer represents pre-1880 forest types generated from LANDFIRE biophysical setting (LANDFIRE, 2013) with raster values generalized to a minimum patch size of 6.75 ha.

2013). Several small clusters of Douglas-fir trees dispersed throughout the spruce-fir forest survived the 1685 fire, but no sampled spruce or corkbark-fir predated the event.

The proportion of total fire area that was classified as high-severity in 1685 (47%) is more than double the landscape proportional area of high burn severity over the period 1640–1880 (17%) that includes the 1685 fire effects; but is below the proportion of high burn severity from 1974 to 2004 (54%) (Fig. 6d). In addition, the spatial distribution of high burn severity was not consistent between historical and contemporary fires. High-severity burn patches of up to 1046 ha within the study area and 2386 ha within the perimeter of the 2004 fire extend into dry and mesic mixed-conifer forest (Fig. 6c) farther than any fire in the pre-fire exclusion period (Fig. 6a and b).

6. Discussion

Prior to fire exclusion, small and widespread fires followed regular patterns of frequency, spatial pattern, severity, and climate associations. Under all but the most extreme drought conditions, fire frequently burned across pine and mixed-conifer forests at low to mixed-severity but was excluded from the higher elevation mesic spruce-fir forest. The consistency of fire return intervals across vegetation types and strong pattern of fire-climate associations suggest that forest conditions and fire regimes were relatively stable for at least several centuries prior to 1880. Following Euro-American settlement, fire frequency and spread were altered substantially. Throughout the first half of the 20th century, small fires continued to burn at individual sites but spreading fires ceased. Fire-adapted pine and dry mixed-conifer forests of the Pinaleño Mountains would have been the most immediately affected by interruption of ground fires beginning in the late 1870s by livestock grazing, road construction, and logging activities (Bahre, 1998). Spreading fires were completely absent from 1884 to 1955, and the few fires recorded in pine, dry, and mesic mixed-conifer forests since 1955 were smaller, less frequent, and higher severity than fires in the pre-fire exclusion period. Patterns of fire size and frequency that were distinct between pine, dry, and mesic mixed-conifer forests prior to 1880 were no longer discernable after 1880. The fuel-limited corridor of pine and dry mixed-conifer forest that experienced frequent fire along the south-facing plateau of the range, began to accumulate fuels (both living and dead trees) over more than 70 years without fire (O'Connor, 2013). Accumulated fuels and increased canopy connectivity during the 20th century removed what was essentially a "fuel-break" at the middle elevations. Frequent surface fires in these forests rarely accumulated sufficient ladder fuels to allow spread into the tree canopy, thereby limiting fire spread into the more mesic, higher elevation forests prior to Euro-American settlement (Swetnam et al., 2009). Estimates of historical fire frequency (in terms of MFI, fire rotation, etc.) in the extensive, unbiased sampling design reported here are consistent with the fire frequency estimates in dry pine and mixed-conifer forests estimated in other gridded, random, and systematic studies at stand to landscape scales in the Southwest in these forest types (i.e., Falk and Swetnam, 2003; Van Horne and Fulé, 2006; Farris et al., 2010, 2013). Fire frequencies were not consistent with the much higher "corrected fire interval" estimates suggested by Baker and Ehle (2001) as a means of compensating for a lack of information on fine-scale spatial patterns of historical fires. The filtered composite fire scar records collected over a gridded sampling network in this study provided the fine-scale spatial sampling necessary to reconstruct spatial variability of mean fire return intervals without relying on the series of statistical assumptions implicit in the multiplicative "fire interval correction" method of Baker and Ehle (2001). Moreover, fire frequency estimates from this study were consistent with targeted/opportunistically sampled stand-level studies of Grissino-Mayer et al. (1995) in this mountain range. This finding demonstrates that, at least in this landscape, the suggestions of Williams and Baker (2012) and Odion et al. (2014) that existing fire history studies are unrepresentative (biased) is not supported.

6.1. Fire-climatology

Along the steep ecological gradients of the Pinaleño Mountains, fire-climate associations prior to 1880 were representative of those across southwestern forest types. Swetnam and Baisan (1996a) identified a pattern of wetter than average conditions prior to years with widespread fire that is thought to have increased fine fuels and connectivity in fuel-limited systems. Several studies in other pine and dry mixed-conifer forests throughout the southwest, including in this study prior to fire exclusion, have identified similar patterns (Touchan et al., 1996; Brown and Shepperd, 2001; Brown and Wu, 2005; Margolis and Balmat, 2009). In the Santa Fe Watershed in New Mexico, Margolis and Balmat (2009) found that spreading fires in pine-dominated stands adjoining mixed-conifer forest occurred consistently during moderate drought only after two or more years of wet conditions but that fire spread into mesic mixed-conifer forest required more intense drought conditions during the year of fire. A similar conclusion can be drawn from the present study in which spreading fires in pine and dry mixed-conifer forest were associated with prior wet conditions and larger fires affecting mesic forests were more strongly associated with extreme drought conditions the year of fire. A similar pattern of severe drought-driven spreading fires was noted in upper-elevation forests of Arizona and New Mexico (Margolis and Swetnam, 2013).

The 28% reduction in frequency of all fires and cessation of spreading fires shortly after 1880 occurred before a multidecadal switch from relatively dry to moist conditions throughout the region after 1905 that lasted approximately 40 years (Biondi et al., 2011; Griffin et al., 2013). A similar reduction in fire frequency and spread did not occur during other prolonged moist periods from 1743–1771 and 1826–1871 (Cook and Krusic, 2004), which include two of the most widespread fire years throughout the Southwest in 1748 and 1863. This suggests that human activities and not climatic conditions were responsible for the change in fire dynamics. Cessation of spreading fires was further aided by active fire suppression after ~1910, although the effectiveness of fire suppression is difficult to separate from the effects of livestock grazing and other land-uses in the Sky Islands during early decades of the 20th century. After more than 70 years without spreading fires, a shift back to drier climate conditions in the 1950s, coupled with increased stand densities and accumulated fuels, resulted in the few spreading fires that have occurred after 1950. Spreading fires of the last six decades were associated with persistent drought mediated by warmer than average winter and spring conditions (PRISM, 2013), with no significant associations to antecedent moist conditions, as was the case historically.

The shift toward fires associated only with persistent drought suggests that the legacy of fire suppression (i.e., high fuel accumulations in dry forests) removed the fuel-limited distinction between dry and mesic forest types (Margolis and Balmat, 2009; Margolis and Swetnam, 2013). Changes to the structure and species composition of dry mixed-conifer forest during the long fire-free interval (O'Connor, 2013) provide further support for the idea that former dry mixed-conifer forests now share structural and species characteristics with more mesic forest types that historically experienced mixed to high-severity fire regimes.

Climate conditions associated with recent spreading fires suggests that the extreme drought conditions that enabled fire spread into mesic forest types historically are no longer necessary as a result of an extended period of fuel accumulation in surrounding dry forests. Moderate drought conditions during the 1996 and 2004 fires were more similar to conditions during 10 larger landscape scale fires that burned around the perimeter of mesic forests than the severe drought during the1685 fire that burned into mesic forests. Multi-year drought conditions more severe than those in 2004 occurred four times in the reconstructed period prior to 1995, but did not result in fire spread into the spruce-fir forest.

The 2004 fire was unique in the fire record as the only spreading fire to occur during El Niño conditions. Warmer than average winter and spring temperatures that accelerated snow melt (Koprowski et al., 2013), and abnormally high wind speeds in May and June (Desert Research Institute, 2013) contributed to more extreme fire weather conditions than would typically occur during El Niño years with above-average snow accumulation. Swetnam and Betancourt (1990a,b) noted a weakening of the relationship between the Southern Oscillation and area burned after the mid-1960s, and additional weakening through the mid-1980s, potentially indicating a reduction in ENSO strength that is altering ENSO-fire relationships in the southwest in a way that may be amplified by warmer winter conditions. Fire-climate associations are known to fluctuate depending on decadal and longer scale patterns of temperature and moisture (Grissino-Mayer and Swetnam, 2000). The later part of the 20th century may represent such a change. Warming winter temperatures coupled with a shift from positive to negative Pacific Decadal Oscillation (PDO) in the late 1990s led to enhanced La Niña and dampened El Niño conditions, contributing to persistent drought across the American southwest (Hoerling and Kumar, 2003; Crimmins, 2011). Evidence of a weakening fire-ENSO association is further supported by a recent switch from La Niña-to El Niño-associated fires in northern Mexico (Yocom et al., 2010). These fires were also associated with anomalously warm temperatures during the shift in PDO. It would be worth exploring the relationship between ocean–atmosphere teleconnections and other large southwestern fires over the past several decades of warming temperatures to determine if fires during El Niño conditions are becoming more common as mean winter temperatures continue to increase.

6.2. Changes to fire severity

Prior to 1880, high-severity fire was rare, accounting for less than 10% of burned area under all but the most extreme climate conditions. Fires in 1996 and 2004 were associated with drought and temperature conditions that were not exceptional in the historical record, but the proportion of area burned at high-severity was two to four times that of previous landscape-scale fires, even with extensive efforts to contain and suppress fire spread. The only fire in the reconstructed period with severity similar to the 2004 fire took place in 1685 during the most extreme drought conditions in 400 years.

The ecological effects of recent fires appear to be more severe than those of any fire in the reconstructed period. Although sampling resolution limited our ability to reconstruct patchiness within the high-severity fire extent of the 1685 fire, substantial growth suppression of spruce and corkbark fir seedlings establishing after 1765 indicate closed canopy conditions within 65 years of the fire. Although no Engelmann spruce or corkbark fir predating 1685 were found to confirm mixed fire severity, rapid post-fire seedling establishment suggests that seed sources were available. Mature spruce and corkbark fir that survived the 1685 fire would most likely have died over the more than 300 years of subsequent spruce beetle (*Dendroctonus rufipennis* Kirby) and western balsam bark beetle (*Dryocoetes confusus* Swaine) outbreaks recorded in this forest (O'Connor, 2013). Within the perimeter of the 2004 fire, surviving trees in high-severity burn patches are isolated to the outer margins, small groups of Douglas-fir along ridges, and Engelmann spruce and corkbark fir in and around perennial springs. Much of the high-severity burned area of the 2004 fire remains barren with extremely limited seedling establishment eight years after the event (Fig. 7). In addition, where recent fires spread into dry and mesic forests, severity was higher than in any recorded fires during the period of reconstruction. Fire scar evidence of the 1685 fire was well recorded along the high-severity burn perimeter, whereas fire scars recorded within or near the perimeter of the 2004 fire were rare. High tree mortality in the 2004 fire eliminated the potential to form fire scars over much of the burned area, although for surviving trees, the extended fire-free interval would have increased tree size and bark thickness, potentially reducing the probability of scar formation on those trees (Gill, 1974; but see Baker and Dugan, 2013).

High tree mortality in recent fires is likely related to a combination of several factors. Increased stem densities and a shift toward fire-intolerant species promoted crown-fire in mixed-conifer forest surrounding the spruce-fir zone. Extended drought and warm winter and summer temperatures increased tree drought stress and cured accumulated standing and downed fuels (van Mantgem et al., 2013). The use of chemical accelerants during aggressive back-burning of fire lines consumed living trees and snags that had withstood centuries of past fires (Frye, 1996; USDA Forest Service, 2004). In the case of the 2004 fire, a preceding decade of compounded insect disturbances resulted in an abundance of standing snags with retained fine fuels (O'Connor et al., 2013).

6.3. Management implications

Differences in fire severity and spread before and after 1880 are attributable to changes to forest structure and fuels as a result of management activities, active fire suppression efforts, decadal to multi-decadal climate variability, and a series of recent high-severity insect outbreaks. Recent fires have had mixed effects on the risk of future high-severity fires. In parts of the mesic mixed-conifer and spruce-fir forests, fire risk has been reduced by the consumption of a significant proportion of available fuels. However, over the majority of the area, fire suppression efforts preserved heavy surface fuels and dense canopy structure. Continuing efforts to suppress fires maintains these heavy fuel loads and heightens the risk of future high-severity fires already occurring in mesic and dry mixed-conifer forests in many areas of the western United States (Stephens et al., 2013).

Fig. 7. A high-severity burn patch eight years after the 2004 Nuttall Complex Fire. Photo: C.D. O'Connor. June 12, 2012.

In current spruce-fir and mesic mixed-conifer forests, recovery of the historical species composition in high-severity fire patches remains uncertain. Distance to seed sources and degradation of soil substrates hinder the establishment of seedlings, and warming, drying conditions may make parts of the former spruce-fir forest no longer suitable for the species assemblage (Notaro et al., 2012; Falk, 2013). Efforts to re-establish spruce-fir seedlings through planting may be warranted to discourage encroachment by shrubs and other lower elevation species (Stephens et al., 2013). Although there is little precedent for fire-induced conversion of spruce-fir to mixed-conifer forest, intense wildfire following fire-suppression in nearby pine-oak communities has been shown to alter the long-term trajectory of post-fire species composition and structure for at least the next several decades to centuries (Barton, 2002).

In dry forests historically adapted to frequent fire, initial thinning of the understory and selective removal of encroaching non-fire-adapted species, followed by reintroduction of fire provides the best opportunity for restoring fire resilience to a system expected to experience more frequent fire under warming, drying conditions in the American Southwest (USDA Forest Service, 2010; Hurteau et al., 2014). Breaking fuel continuity of dry forests and allowing frequent low and mixed-severity fires to burn would restore the buffer that once limited fire spread from dry to wetter forests above them. Returning heterogeneity of forest structure and fire frequency would limit the threat of fire to the astrophysical infrastructure located in the spruce-fir forest and preserve habitat for the majority of threatened and endangered species that are dependent on dense cover in mesic forests for protection from predation and abundant food sources (Sanderson and Koprowski, 2009).

7. Conclusions

Large high-severity fire patches in the Pinaleño Mountains were rare historically and limited to only the most mesic, productive sites prior to fire exclusion. Interruption of spreading fires in the late 1800s led to changes in forest structure and species composition (O'Connor, 2013) that altered fire behavior and fire-climate associations. Recent shifts in decadal to multidecadal climate modes coupled with warming temperatures may have accelerated changes to fire–climate relationships by lengthening the fire season and diminishing the effects of winter precipitation on reduced fire spread. Conditions during recent fires suggest that the drought thresholds necessary for fire to spread across multiple forest types have been reduced as a result of fuel accumulation and positive feedbacks from warming temperatures. Similar relationships have been identified in other western forests where the trend of increasing winter and spring temperatures is contributing to earlier snow melt and longer spring fire-weather conditions (Pederson et al., 2011). Over the past four years, record-setting fires in New Mexico, Arizona, and California are in line with the predictions of more frequent and larger fires in a warming, drying southwest (Attiwill and Binkley, 2013). The return of landscape-scale fires coincident with substantial fuel accumulations due to a reduction of fire frequency and tree stress is increasing the severity in at least some areas that were historically characterized by low-to-moderate fire (Miller et al., 2012; Mallek et al., 2013; van Mantgem et al., 2013).

Without restoration of forest structure and fire, inland forests of the western States are likely to continue to burn with increasing fire size and severity, with increased risk to human interests and sensitive species adapted to specific site conditions. Efforts to restore historical forest structure and fire regimes have largely been successful where attempted (Mast et al., 1999; Fulé et al., 2001, 2004; Hurteau and North, 2008). Identifying and prioritizing landscapes suitable for restoration like the Pinaleño Mountains, where they are likely to produce the greatest benefit to natural and human interests, will be vital to successful landscape management in the future.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.foreco.2014.06.032.

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