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Fire regimes over a 1070-m elevational gradient, San Francisco Peaks/Dook'ooosłííd, Arizona, USA

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Abstract

Background Steep elevational gradients bring multiple forest types and fire regimes together in close proximity. The San Francisco Peaks/Dook'ooosłííd in northern Arizona rise to 3851 m elevation with slopes that span many of the major forest types of the southwestern US mountains. To reconstruct past fire regimes across this broad elevational gradient, we sampled fire-scarred trees across the south face of the Peaks, complementing previous research on forest structure, composition, and origin of aspen stands.

Results At the highest elevations, Rocky Mountain bristlecone pine forests had a mean fire interval (MFI) of 19.7 years prior to a modern fire exclusion period beginning after 1879. Other high-elevation (> 2800 m) mixed conifer forests had MFI = 5.7 years and low-elevation (< 2,800 m) pine forests had MFI = 4.0 years. After 1879, there were no large fires through the end of the twentieth century. Before 1879, fires occurred in the early to middle growing season, and fire event years were linked to climate across all elevations, with a stronger association to drought (i.e., the Palmer Drought Severity Index) than to El Niño-Southern Oscillation phase. Pulses of forest regeneration were associated with the fire regime, with the largest pulse occurring shortly after fire exclusion. In addition to fire exclusion, other factors such as post-fire sprouting and regeneration after tree harvesting likely contributed to the current dense forest structure on the Peaks.

Conclusions Following over a century of fire exclusion, fire activity has increased on the Peaks over the past two decades, with large recent fires of uncharacteristic severity raising concerns about tree mortality, erosion, flooding, and infrastructure damage in surrounding human communities. Past fire regimes provide useful insight into fire-climate-forest interactions and the conditions under which existing forest communities were well adapted, but adaptation to future conditions is likely to be challenging due to the rapid pace of projected environmental changes.

Keywords Dendrochronology, Fire scar, Climate, El Niño-Southern Oscillation, Fire exclusion, Wilderness, Indigenous, Sacred mountain

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Resumen

Antecedentes Los gradientes de elevación abruptos brindan múltiples tipos forestales y regímenes de fuego en zonas muy próximas entre sí. Los picos de San Francisco (Dook’o’ooslííd en idioma Navajo, o el pico que nunca se derriete), se eleva hasta los 3.851 m de altura, con pendientes que abarcan muchos de los mayores tipos forestales de las montañas del suroeste de los EEUU. Para reconstruir los regímenes de fuegos pasados a través de todo este gradiente de elevación, muestreamos cicatrices de fuego a lo largo de la cara sur de estos picos, complementando también investigaciones previas sobre la estructura forestal, composición, y origen de rodales de álamos.

Resultados A elevaciones altas, el pino bristlecone de las rocallosas tuvo un intervalo de fuego medio (MFI) de 19,7 años antes del período de exclusión moderno del fuego, que comenzó en 1879. Otros bosques mixtos de pinos que crecen a altas elevaciones (> 2.800 m) tuvieron un MFI= 5,7 años, y los bosques de pinos de elevaciones bajas un MFI=4,0 años. Después de 1879 no hubo grandes incendios hasta fines del siglo XX. Antes de 1879, los incendios ocurrían en la estación de crecimiento temprana y media, y años con eventos de fuego estaban ligados al clima en todo el gradiente de elevación, con una fuerte asociación a la sequía (i.e., el Índice de Severidad de Sequía de Palmer) más que a la fase de años Niño (ENSO). Los pulsos de regeneración del bosque estuvieron asociados con el régimen de fuegos, con el pulso más grande ocurrido muy poco después de la exclusión del fuego. Adicionalmente a la exclusión del fuego, otros factores como el rebrote post-fuego y la regeneración luego de las talas, también contribuyeron a la actual densa estructura forestal en esos picos.

Conclusiones Luego de un siglo de exclusión del fuego, la actividad de los incendios se ha incrementado en los picos de San Francisco en las dos últimas décadas, con grandes incendios recientes de severidad poco característica, creando preocupación sobre la mortalidad de árboles, erosión, inundaciones, y daños a la infraestructura en comunidades humanas aledañas. Los regímenes de fuegos pasados nos proveen de conocimientos sobre las interacciones entre el clima, los bosques y el fuego, y las condiciones bajo las cuales las comunidades forestales existentes estaban bien adaptadas, pero las adaptaciones a las condiciones futuras serán desafiantes dada la rápida marcha de los cambios ambientales proyectados.

Background

Mountain environments bring distinct climate zones, vegetation types, and disturbance regimes into close proximity. Gradients in elevation, slope, and aspect are central in driving vegetation dynamics in the dry interior region of southwestern North America, consisting of Arizona, New Mexico, and surrounding US and Mexican states, where most forests are restricted to high elevations. Fire regimes of southwestern forests have been widely studied, especially those of ponderosa pine forests (scientific names of species are given in Table 1). Fires recurred frequently in these forests with mean fire return intervals (MFI) typically < 10 years (Swetnam and Brown 2011). Surface fire regimes were interrupted across the region in the late nineteenth century except for a few sites distinguished by remoteness (Madany and West 1983; Rollins et al. 2001; Fulé et al. 2003b; Farris et al. 2010) or Indigenous management (Fulé et al. 2011; Stan et al. 2014). Higher-elevation mesic forests with aspen, spruce, and fir species, adapted to infrequent, stand-replacing fire regimes, make up < 5% of forests in the region (Shaw et al. 2018). Estimates of historical occurrence of severe fires, such as fire rotation values, are limited in the Southwest due to the small area and insufficient study of the spruce-fir-aspen forest type (Margolis et al. 2011), but several studies have described shifts toward an increasing

Table 1 Scientific and common names, and codes, for tree species in the study area. White fir *Abies concolor* (Gord. & Glend.) Hoopes. and Gambel oak *Quercus gambelii* Nutt. were occasionally present. The lower-elevation white pine is called *Pinus flexilis* (Mitton et al. 2000), consistent with Cocke et al. (2005), but there is *Pinus strobiformis* (southwestern white pine) and a hybrid zone between *P. flexilis* and *P. strobiformis* on the Peaks

Scientific name	Common name	Code
<i>Abies lasiocarpa</i> var <i>arizonica</i> (Hook.) Nutt	Corkbark fir	ABLA
<i>Pinus aristata</i> Engelm.	Rocky Mountain bristlecone pine	PIAR
<i>Picea engelmannii</i> Parry ex Engelm.	Engelmann spruce	PIEN
<i>Pinus flexilis</i> James	Limber pine	PIFL
<i>Pinus ponderosa</i> var. <i>scopulorum</i> P. & C. Lawson	Ponderosa pine	PIPO
<i>Populus tremuloides</i> Michx.	Quaking aspen	POTR
<i>Pseudotsuga menziesii</i> (Mirb.) Franco var. <i>glauca</i> (Beissn.) Franco	Douglas-fir	PSME

extent of stand-replacing fire at higher elevations since the onset of fire exclusion (Yocom-Kent et al. 2015).

Previous studies of fire occurrence in southwestern ponderosa forests have shown a strong association with shifts from El Niño to La Niña phases of the El Niño-Southern

Oscillation (ENSO) (Swetnam and Betancourt 1998). In the southwestern US, El Niño events are characteristically wet and La Niña events are dry, associated with drought conditions as measured by indices such as the Palmer Drought Severity Index (PDSI) (Wang and Kumar 2015). Swetnam and Betancourt (1998) hypothesized that herbaceous fuel growth in the wetter El Niño phase became susceptible to burning as ENSO switched to the drier La Niña phase. Historically, climatic drivers of fire activity also differed over elevational gradients. Higher-elevation forests were not typically limited by fuels and tended to burn in dry years irrespective of precipitation in prior years (Swetnam and Betancourt 1998).

Since fire exclusion in the late nineteenth century, fuels have become denser and more contiguous not only in low-elevation ponderosa pine/savanna systems but even in high-elevation spruce-fir and aspen forests, due to the effects of fire exclusion, post-harvest regeneration, and in some cases such as aspen or spruce-fir, recovery from severe fires (Fulé et al. 2003a). In combination with drought and warming climate (Williams et al. 2020), contemporary fires have increased in size and are burning a greater area at high severity (O'Connor et al. 2014; Singleton et al. 2019), raising concerns about long-term loss not only of individual species and communities (Adams and Kolb 2005) but also entire forest ecosystems (Noss et al. 2006; Parks et al. 2019).

Climate and fire play important roles regulating forest regeneration. Frequent surface fires limit regeneration, except in “safe sites” such as rocky areas or ashbeds (White 1985) while larger patches of severe fire can provide open sites for sexual regeneration of slow-growing mesic mixed conifer species (Tepley and Veblen 2015) or clonal aspen regeneration (Margolis et al. 2011). Humid climatic periods (pluvials) favor sexual regeneration at decadal (Brown and Wu 2005) to annual or even seasonal scales (Savage et al. 1996; Davis et al. 2019). The widespread exclusion of fire from most North American forests in the late nineteenth century (Hagmann et al. 2021) was widely associated with a pulse of regeneration and persistent forest densification due in large part to the elimination of the thinning effect of fire.

Data on historical fire regime patterns and their relationship to climate and forest dynamics are therefore fundamental to understanding the drivers of current change and developing adaptive management strategies (Prichard et al. 2021). Fire regimes have been studied at many temporal and spatial scales from sedimentary analysis (Marlon et al. 2009) to remote sensing (Balch et al. 2020), as well as through fire-adaptive traits (Keeley and Pausas 2022). While these various approaches have unique advantages, tree-ring reconstructions of past fires

through dating of fire scars on surviving trees provide multi-century, annually resolved fire records that can be compared with data on climate, ages of tree cohorts, other disturbances, and human activities at a range of scales (Margolis et al. 2022).

Among the most emblematic and scientifically notable mountains of the Southwest are the San Francisco Peaks, or Dook'o'oolííd (“Gleaming Summit”) in the Diné (Navajo) language. The mountain is sacred to the Diné and numerous other Native American tribes and has many Indigenous names that reflect the cold, high environment: Nuvatukya'ovi (“Mountaintop Home of Snow,” Hopi), Wii Hagnbaja (“Snowy Mountain,” Hualapai), and others (Waring 2018). The elevational banding of vegetation types on the Peaks also inspired Merriam (1890) to describe life zones that equated elevation with latitude. Subsequent research over the elevational gradient by Forest Service scientists in the early twentieth century provided foundational information on the interaction of climate, soils, and vegetation types (Crouse et al. 2008). The Kachina Peaks Wilderness, 7536 ha, was designated in 1984 but much of the area below about 2700 m was managed with timber harvesting previously (Waring 2018). In recent decades, large areas of the Peaks were burned by the Schultz (2010), Museum (2019), Pipeline (2022), and Tunnel (2022) fires. These fires were uncontrollable wind- and drought-driven events leading to substantial soil erosion, loss of life, and costly ongoing damage to property and infrastructure (Neary et al. 2012; Porter et al. 2021; Edgeley and Colavito 2022). Even though substantial areas of these fires were not classified as “severe” (<https://www.mtbs.gov/>), tree mortality often reached 100% even in “moderate”-severity patches (Neary et al. 2012).

Surface fire regimes were characterized in the lower-elevation mixed conifer forests (2300–2700 m) on the west and east sides of the Peaks by Heinlein et al. (2005). Localized severe fires, ending in 1879, led to the establishment of a belt of even-aged aspen stands with mean elevation of 2954 m (Margolis et al. 2011). In the early 2000s, Cocke et al. (2005) established a systematic grid of 135 plots covering the south face of the Peaks, measuring forest composition, structure, regeneration, and fuels. Compared to dendrochronological reconstruction on the same plots in the late nineteenth century when the fire regime was interrupted, forests became much more dense and mesic species have encroached to lower elevations by the early twenty-first century (Cocke et al. 2005). These changes in fuel quantity and quality, coupled with steady warming of northern Arizona since the 1980s (Mueller et al. 2020), are linked to a recent rise in large, severe wildfires on the Peaks resulting in vegetation mortality, erosion,

flooding, invasive species increase, and high social disruption (Dodge et al. 2008; Neary et al. 2012; Edgeley and Colavito 2022).

Knowledge of past fire regimes and patterns of establishment is helpful to assess the conditions that could support forest communities on the Peaks in a changing environment. We initiated a study to reconstruct fire regimes from fire-scarred trees on the same elevational gradient studied by Cocke et al. (2005). We carried out a robust collection of samples from fire-scarred trees within and around the forest plot grid. Our goals were to quantify the characteristics of the fire regimes over the elevational gradient, assess the climate drivers of fire, and link fire and climate patterns to regeneration by species over the past several centuries. We developed several hypotheses based on the literature described above:

H₁: We expected MFI to become longer as elevation increased.

H₂: We expected the position of the fire injury within the growing ring to appear at earlier positions as elevation increased, due to the later seasonal development of xylem cells at cooler, higher elevations.

H₃: We expected fire occurrence to be less tightly linked to La Niña-El Niño phase shifts as elevation increased.

H₄: We expected fire occurrence to be more tightly linked to drought (PDSI) in the fire year as elevation increased.

H₅: We expected tree regeneration patterns to be increasingly pulsed and increasingly dominated by relatively mesic, fire-susceptible species after the interruption of the fire regime.

Methods

Study area

The study area comprised 1920 ha on the south face of the San Francisco Peaks in the Coconino National Forest, which reach the highest elevation in Arizona (3852 m) and support regionally rare forest ecosystems including the only populations of Rocky Mountain bristlecone pine in the state and unique alpine tundra environments. The study area lies almost entirely within the Kachina Peaks Wilderness Area. Weather data for the Flagstaff Region were summarized from 1900 to 2020 from the Flagstaff Weather Forecast Office, 2100-m elevation, located approximately 16 km south of the study area (<https://www.weather.gov/wrh/Climate?wfo=fgz>, accessed 13 Jan 2022). Average January temperatures were -1.8°C and average July temperatures were 18.9°C . Mean annual precipitation

was 53.34 cm. Weather stations were established by Pearson (1920a), Pearson (1920b) in 1916 and measured weekly from 1917 to 1920 along the elevational gradient from ponderosa pine (≈ 2290 m), Douglas-fir (≈ 2700 m), Engelmann spruce (≈ 3200 m), to treeline (above 3500 m). Mean annual temperatures ranged from 5.8°C in ponderosa pine, 5.4°C in Douglas-fir, 3.1°C in Engelmann spruce, and -0.05°C in treeline zones (Pearson 1920a; Crouse et al. 2008).

Field methods

We sampled 113 fire-scarred trees (living trees, snags, fallen trees, and stumps) on the south face of the Peaks from approximately 2400- to 3600-m elevation in 2000, 2001, and 2003, overlapping the network of forest plots installed on the same landscape by Cocke et al. (2005) (Fig. 1). Tree cores were collected in the same period on the forest plot network as described in detail in Cocke et al. (2005). Due to the Wilderness designation of almost all the study area, access was via hiking and only hand tools were used in sampling. We walked transects that crisscrossed the study area. We collected fire-scarred samples on or adjacent to transects throughout the study area up to timberline, except for the footprint of the 2001 Leroux Fire in the south-central portion, due to the hazard of working in the burned area. However, there were likely few old trees in that area because the Leroux fire burned mostly in a montane grassland, Freidlein Prairie, which had become dense with young ponderosa pines since the late 1800s (Cocke et al. 2005; Dodge et al. 2008).

We targeted trees that appeared to have the longest period of scar recording and the greatest number of visible scars, an efficient and accurate approach to fire history reconstruction (Farris et al. 2013). A small partial cross-section was removed with hand saws, augers, and chisels from one side of the fire-scarred wound on each sample tree (Fig. 1), a method that minimizes damage to the trees (Heyerdahl and McKay 2017).

Laboratory methods

In the laboratory, samples were glued to backing material and cut flat with a table saw. We sanded them with progressively finer grits of sandpaper from P120 to P400 in a wood shop until the cells were clearly visible under magnification. Portions with very small rings were further sanded by hand using $40\ \mu\text{m}$, $30\ \mu\text{m}$, and $15\ \mu\text{m}$ sandpaper (3 MTM). We visually crossdated the tree rings and measured ring widths using a sliding stage (Velmex). We used the COFECHA software (Holmes 1983) for quality control of crossdating. We identified fire scar dates and intra-annual position of the wounds following standard designations (Baisan and Swetnam 1990): dormant season; early-, mid-, or late-earlywood; and

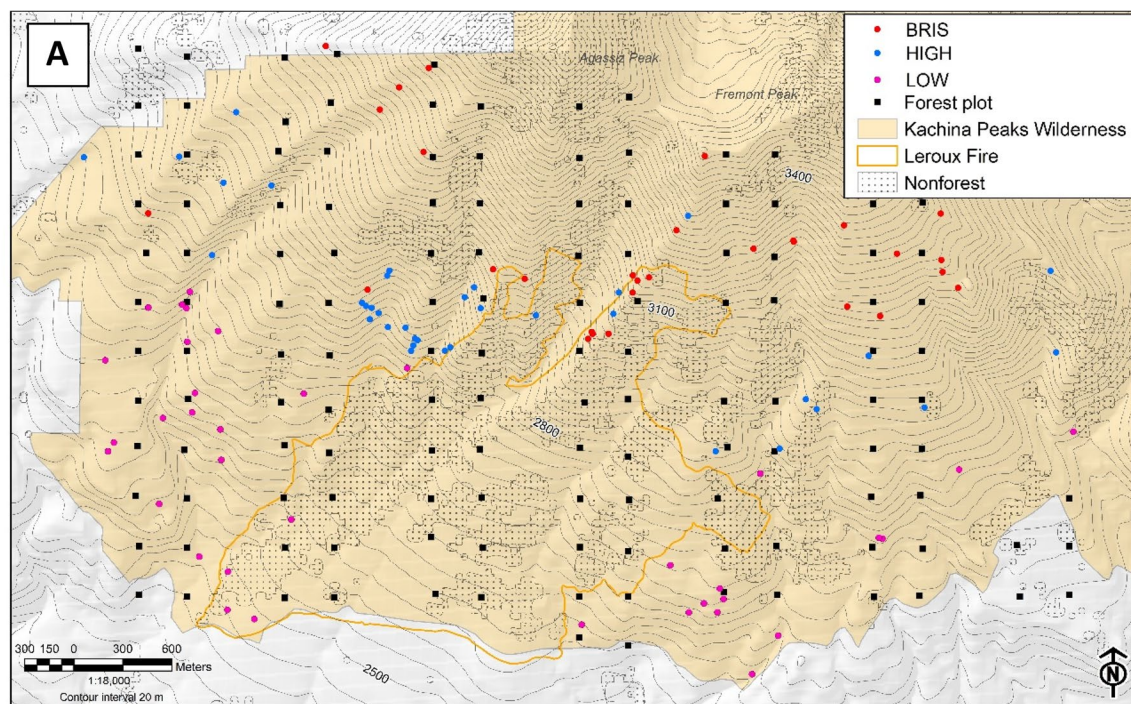


Fig. 1 Study area on the south face of the San Francisco Peaks/Dook'o'oslíí (A). High-elevation bristlecone pine, aspen, montane/subalpine grassland (B). Samples were collected with non-mechanized hand tools in the Kachina Peaks Wilderness (C, D)

latewood. Dormant-season scars were assigned to the spring season, prior to ring growth, following the standard approach in the southwestern US. Bristlecone pines presented challenges due to high ring compression in injured areas and the “strip-bark morphology” (Schauer et al. 2001) of the species. Injuries that could not be unambiguously assigned to fire were not included in the data set.

We divided the samples into three groups based on the predominant forest types on the Peaks (Cocke et al. 2005): fire-scarred trees below 2800 m in elevation (LOW, mostly ponderosa pine), above 2800 m (HIGH, mixed conifer and aspen), and a separate category for Rocky Mountain bristlecone pines only (BRIS). We assembled the fire-scar data for each group into FHX2-format files and analyzed fire intervals distributions, seasonality, and fire-climate links with Fire History Analysis and Exploration System (FHAES) software v. 2.0.2

(Brewer et al. 2015) and the R package burnR (Malevich et al. 2015). We began the analysis period at the first fire date with at least two scarred trees and at least 10% of recording trees. We measured the similarity of fire occurrence between groups and all groups combined using the Jaccard similarity index which ranges from 0 (complete dissimilarity) to 1 (complete similarity). We analyzed fire interval distributions with parametric statistics and also fit a Weibull function to express fire interval statistics in probabilistic terms. All fire interval analyses were based on fire dates with at least two fire scars and two recording trees per group, as well as a regional analysis of all groups combined. In addition to calculating fire interval analyses for the full data period per group, we also carried out the same calculations for common period for all three groups, 1715–1879.

We downloaded 2022 vegetation coverages from the Landfire program (<https://landfire.gov/>, accessed 7 June

2023) to estimate the area of each forest group sampled within the overall 1920-ha study area. While vegetation on the mountain is generally arranged in horizontal elevational bands, there is also considerable variety introduced by topographic features such as deep arroyos (Fig. 1). We calculated the areas corresponding to the BRIS, HIGH, and LOW groups. The BRIS group includes forests classified as spruce-fir by Landfire. Our field observations indicated that the Landfire classification may have overestimated this type because while Engelmann spruce and subalpine fir were present, these species predominated only in shaded, steep arroyos. Rocky Mountain bristlecone trees that were sampled for fire scars were well distributed across the upper mountain (Fig. 1) and Cocke et al. (2005) reported over twice as many plots dominated by bristlecone (23) as by spruce or fir (10). Therefore, we considered it appropriate to aggregate the areas classified by Landfire as bristlecone pine and spruce-fir.

We developed interpolated maps of fire occurrence using the Kernel Density tool in ArcGIS v. 10.8 (ESRI 2020, ArcGIS Desktop: Release 10.8. Redlands, CA: Environmental Systems Research Institute), with maps representing the density of fire presence records in a given year and location. We tested alternative approaches to mapping fire patterns using Thiessen polygons and inverse distance weighting (Hessl et al. 2007). However, our samples were not collected in a spatially systematic way as done in some studies (Farris et al. 2010; Whitehair et al. 2018), so the interpolated maps were influenced by the differential density of sampling, which would have led to unrealistic representations of fire patterns. We focused on presence data, rather than presence/absence, because fire absence records are sometimes “false” absences (Stephens et al. 2010) and kernel density estimation focuses on presence records, which are inherently more reliable.

Climate analyses

We assessed fire-climate relationships using superposed epoch analysis (SEA) in the R package burnR (Malevich et al. 2015). We created fire event files listing all fire dates and those scarring 25% or more of recording trees within each group. The 25% filter required events to have at least two recording trees, and at least one tree recording a fire event. We used two climate data sets: (1) reconstructed Summer Palmer Drought Severity Index (PDSI) for North American gridpoint 88, closest to Flagstaff, AZ (Cook 2008), and (2) reconstructed El Niño-Southern Oscillation (ENSO) values using the Niño 3 dataset (Cook 2000).

To capture potential lagged effects of climate on fire activity (e.g., ENSO switching), we used SEA to compare the climate variable, PDSI or ENSO, in fire event years as

well as a window of four preceding and three succeeding years, a time frame commonly used in such analyses to help identify switches from dry to wet periods such as those associated with El Niño-Southern Oscillation (Stephens et al. 2003). Statistical significance ($\alpha=0.05$) was estimated from the ranking of the observed data in comparison to 1000 bootstrapped samples of each climate variable.

Graphically comparing fire occurrence, climate and tree establishment events

To assess the relationships among large fire years, tree establishment events, and interannual/interdecadal climate variability, we also graphically compared tree age data from field plots and fire records from fire-scarred trees with the ENSO and PDSI reconstructions. First, we developed tree age distributions (densities $\text{ha}^{-1} \text{decade}^{-1}$) for each forest group used in fire history reconstructions (i.e., BRIS, HIGH, LOW) from age data in sampled trees ($n=1,742$). Because over 80% of increment cores were dated within 10 years of pith, we used 10-year bins in tree age distributions following Cocke et al. (2005). Next, we identified tree establishment events using a modified version of CharAnalysis (Higuera et al. 2010), adapted for use with tree ring data (Tepley and Veblen 2015; Andrus et al. 2018). Here, we used CharAnalysis to account for potential inconsistencies in historical time series (e.g., loss of evidence) and to identify periods with greater tree establishment than surrounding decades. CharAnalysis uses several parameters that control the sensitivity of event detection and the amount of temporal smoothing; we used a running median filter of 60 years for detrending, a statistical threshold of the 90th percentile in Gaussian mixture models, and a minimum number of 5 trees ha^{-1} to identify pulses of establishment across all species, though only minor differences were observed with changes in parameters. Additional descriptions of CharAnalysis and its use with tree age data are provided in Tepley and Veblen (2015). To summarize important years of fire activity, we identified fire years that were observed on at least 25% of recording trees within each forest group. However, because individual fires in BRIS were often recorded on a small number of trees, we also identified 10% fire years in this group. Lastly, we presented interannual and interdecadal trends in climate using reconstructions of ENSO and PDSI.

Results

We successfully crossdated samples from 102 fire-scarred trees covering a 1070-m elevational gradient (Table 2). The trees are identified by forest group (BRIS, HIGH, LOW) in Fig. 1. Though most trees tended to have southerly aspects and relatively large diameters (average

55–64 cm), across forest groups, the average slopes from which BRIS and HIGH trees were collected was twice that of the LOW samples (Table 2). About two-thirds of trees were dead in the HIGH and LOW categories; 56% of the BRIS samples were from living trees (Table 2).

Fire history charts (Fig. 2) illustrated that trees in the HIGH and LOW categories had the highest fire frequency and shortest chronosequences (i.e., shorter recording timespan). In contrast, BRIS samples had fewer fire scars and were typically older, with 47% dating before 1600 (Fig. 2). Fire regimes were interrupted abruptly after 1879 across the entire study area, with only one tree recording a fire in the twentieth century. The oldest fire scars were found in 1492 on a living Rocky Mountain bristlecone cone at 3188-m elevation and a limber pine snag at 2957 m. Examples of spatial patterns of fire as represented by interpolated maps (Fig. 3) show two fire years with widespread fire evidence (1715, 1879) versus restricted fire evidence (1773, 1834).

Fire interval analysis supported H₁: MFI was shortest (4.0 years) in the LOW group, increasing to 5.7 years in HIGH and more than tripling to 19.6 years in the BRIS category (Table 3). Starting dates for analysis were 1506 in BRIS, 1715 in HIGH, and 1684 in LOW. Many fire interval statistics were similar between HIGH and LOW, but increased in the BRIS group. However, the point MFI statistic, the average interval per individual sample, was 15.1 years in LOW but more than double that value in HIGH and BRIS (Table 3). When fire interval statistics for all three groups are compared over the common period, 1715–1879, there was a minor change in the LOW group; the MFI declined by ½ year or 12.5% compared to the full period (3.5 years in the 1684–1879 period, 4.0 years in the 1715–1879 period) (Table 3). The BRIS group had a larger decline of 9.5 years or 48% (19.6 years in the 1506–1879 period, 10.1 years in the 1715–1879 period) (Table 3).

The Jaccard Index of similarity in fire occurrence showed the greatest similarity between LOW and HIGH (0.429 on a 0–1 scale), with the lowest similarity between BRIS and LOW (0.255) (Table 4). The Jaccard Index can be interpreted as the degree to which fires occurred in the same year in these groups, meaning that LOW and HIGH shared nearly half of their fire dates and BRIS shared about a quarter of the fire dates of LOW.

The position of the fire injury within the growing ring differed between groups consistent with H₂: the modal injuries in the LOW group were in the middle earlywood, whereas the modes of the HIGH and BRIS injury distributions were in the dormant position (Fig. 4). The BRIS scars had the most dominant seasonal aggregation, with 61% of injuries in the dormant position. Overall, 92.5% of fire scars occurred in the earlier part of the growing season (dormant through middle earlywood) and only 7.5% occurred in late earlywood or latewood.

Climate and fire assessed with superposed epoch analysis were closely coupled on the Peaks, as has been demonstrated throughout the Southwest (Swetnam and Brown 2011), but relationships had differential results with respect to H₃ and H₄. La Niña events, the negative phase of ENSO, were significant drivers of fire years at all elevations when all fire events were considered (Fig. 5) but not when the analysis was limited to fire years in which 25% or more of samples were scarred (Fig. 6). In contrast, drought, as represented by PDSI, was associated with fire years at all elevations, contrary to H₄. Fire years in all categories consistently displayed negative PDSI, both for all fire years and for those in which 25% or more of recording samples were scarred (Figs. 5 and 6). Significant shifts from wet to dry phases were not observed for ENSO but did occur for PDSI at lower elevations (Figs. 5 and 6).

Tree establishment on the sampling plot grid of Cocker et al. (2005) was associated with both historical fire-free periods and contemporary fire regime interruption (i.e., 1879), consistent with H₅ (Fig. 7, Figure S1). Pulses of establishment in the LOW group occurred in the 1880s, 1900s, and 1940s, all following the onset of fire regime interruption. In contrast, we identified pulses of establishment in HIGH (1780 and 1820 s) and BRIS (1690 and 1740 s) prior to fire regime interruption during relatively fire quiescent periods. However, HIGH (1890 and 1970 s) and BRIS (1870s, 1900s, and 1940s) also showed strong pulses of establishment during contemporary fire regime interruption. Shifts in species composition following fire exclusion were most evident at HIGH, where aspen regeneration essentially ceased by 1900 and Rocky Mountain bristlecone pine, Engelmann spruce, and corkbark fir regeneration became proportionately more dominant (Fig. 7).

Table 2 Crossdated fire-scarred samples divided into three groups. Species codes are given in Table 1

Group	N	Elevation, m (min–max)	Area sampled (ha)	Average slope (%)	Average aspect (°)	Species	Average diameter (cm)	Living samples (%)
Bristlecone pine (BRIS)	30	2934–3567	534	43.9%	199.2°	PIAR	55.2 cm	56%
Other trees above 2800 m (HIGH)	36	2805–3062	692	42.2%	197.1°	PIFL, PIPO, PSME, PIEN	60.8 cm	31%
Other trees below 2800 m (LOW)	36	2497–2775	689	21.3%	188.9°	PIPO, PIFL	63.8 cm	31%

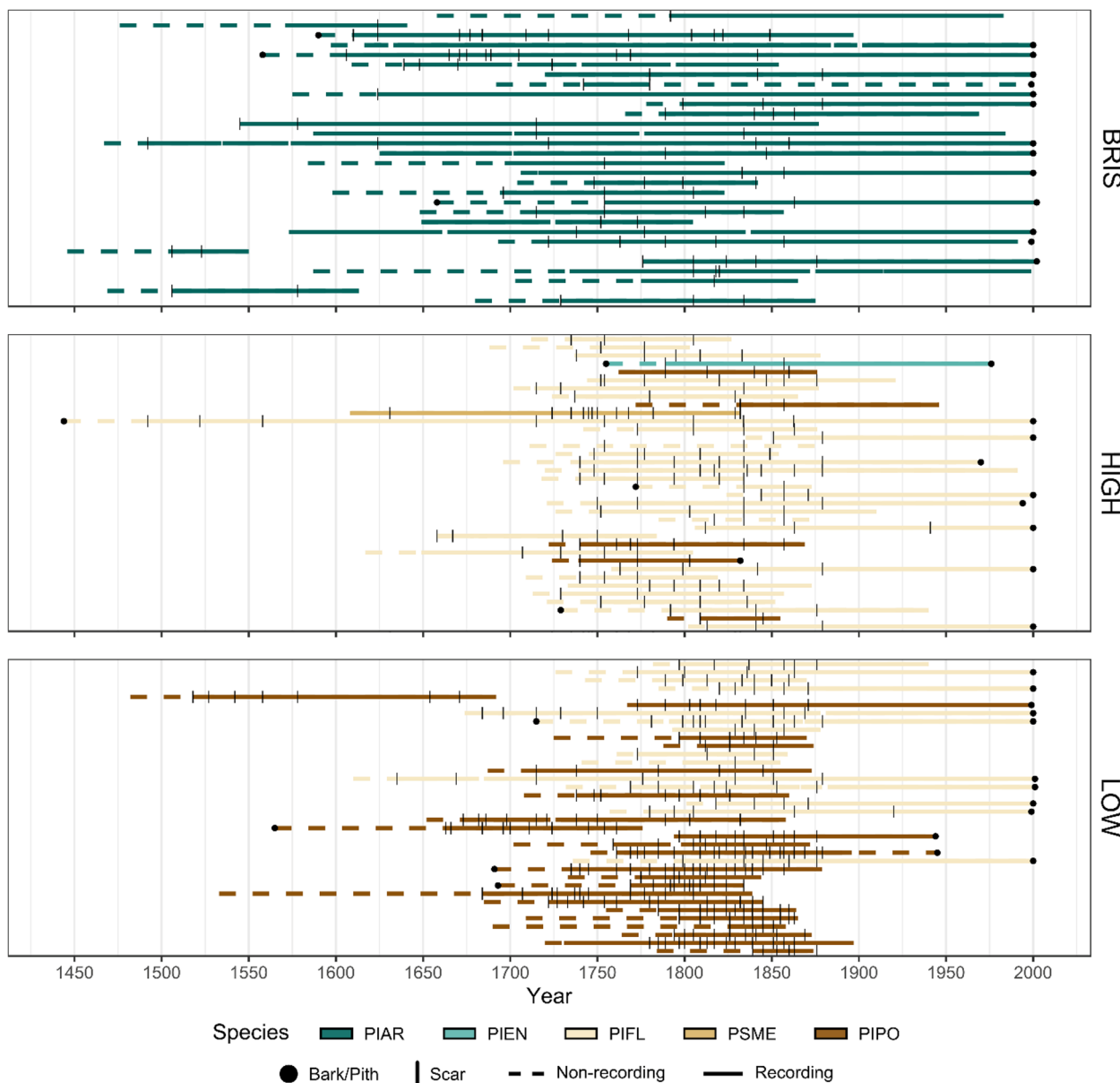


Fig. 2 Fire history charts for BRIS, HIGH, and LOW. Species codes for sampled trees are given in Table 1

Discussion

We reconstructed evidence of fires on the San Francisco Peaks/Dook’o’ooshtíid from the oldest detected date in the fifteenth century (1492 CE) until the start of the twenty-first century. The study crossed a 1070-m elevational gradient from ponderosa pine forests (LOW) through aspen and mixed conifer (HIGH) to Rocky Mountain bristlecone pine forests (BRIS) at the highest elevations. Fire regimes in the LOW and HIGH were linked to climatic patterns, and fire dates were well connected between these two groups, generally

consistent with findings in these forest types in western North America. Rocky Mountain bristlecone pine is more sparsely distributed in the West; the Peaks hold an isolated population at the southwesternmost extent of the species and the present study is the first documentation of its fire history in Arizona (Coop and Schoettle 2011).

Limitations on the scope of this study included the difficulty of access and the requirement that sampling be done with hand tools due to the Wilderness designation of the Peaks. Despite a substantial investment of field

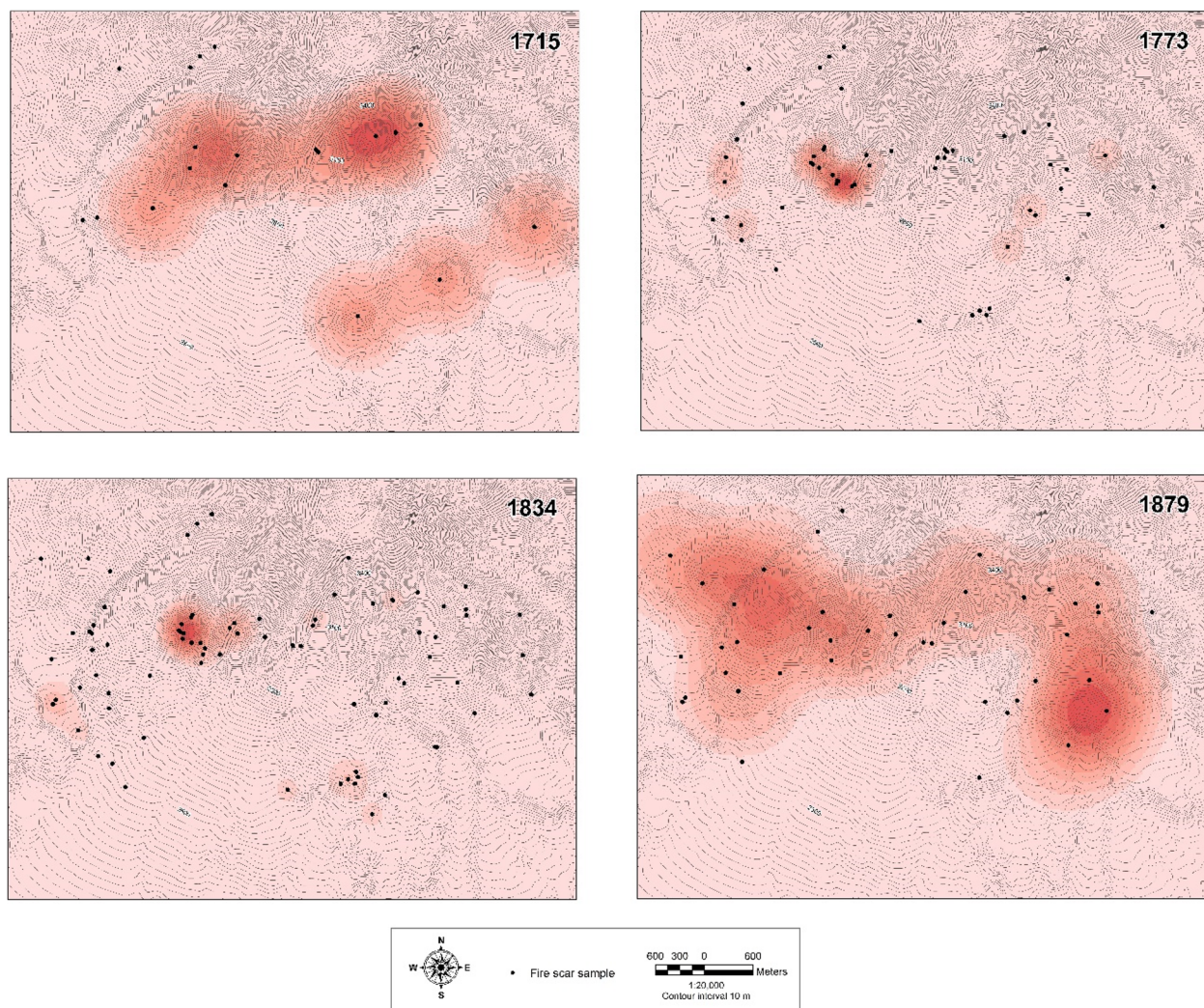


Fig. 3 Spatial pattern of four representative fire years. Black dots show locations of fire-scarred sample trees that were alive at the time. Red color indicates kriged fire locations using the Kernel Density function

effort over multiple seasons, only 102 dateable samples were obtained in the 1920-ha study area (average of one sample per 19 ha). However, this sampling rate is similar to other landscape-scale studies in the Southwest, e.g., one sample per 8 ha (Huffman et al. 2015) to one sample per 25 ha (Whitehair et al. 2018), and more dense than other landscape-scale analyses, e.g., one sample per 108 ha (Merschel et al. 2018). The Peaks samples were also spatially clustered, as discussed above, limiting the consistency of interpolated maps. Despite the limitations, our results are relatively robust because sampling sizes were similar (30 trees in BRIS, 36 in HIGH and LOW; Table 2) and areas represented by sampling were similar (range 534–689 ha, maximum difference 22%; Table 2). Compared during the common overlapping period of analysis of all three forest groups, 1715–1879, there was a minor difference in MFI for the LOW group but the BRIS

group declined by nearly half. In the subsequent discussion, we primarily refer to fire regime statistics for the full periods of analysis as shown in Table 3 because they provide the most complete picture of the data.

Findings of frequent surface fire and increased (longer) MFI with increasing elevation, H_1 , were within the range of previous reports in the Southwest (Swetnam and Baisan 1996). Fire regime studies over elevation gradients at the Grand Canyon, about 80 km north of the Peaks, found an MFI value (10% scarred) of 4.0–7.4 years in ponderosa pine and 8.7 years in mixed conifer (Fulé et al. 2003a). At Mt Graham, another sky island located about 325 km southeast of the Peaks, ponderosa pine MFI was 4.8 years and mixed conifer values were slightly lower, 3.4–4.8 years (O'Connor et al. 2014). The MFI in the bristlecone pine on the Peaks (19.6 years, Table 3) is the first report of Rocky Mountain bristlecone fire regime in Arizona, but Brown

Table 3 Fire return interval analyses by group. All composite analyses are based on a minimum of three recording samples and at least two fire scars. Statistics for BRIS and LOW groups are reported for the full analysis periods as well as the common period, 1715–1879, shared by all groups. NA, not applicable

Composite parameters	BRIS		HIGH		LOW	
Dates of analysis	1506–1879	1715–1879	1715–1879	1684–1879	1715–1879	1715–1879
Total number of intervals	19	15	29	49	47	47
Mean fire interval (yr)	19.6	10.9	5.7	4.0	3.5	3.5
Median fire interval (yr)	12.0	9.0	4.0	3.0	3.0	3.0
Standard deviation (yr)	19.6	8.5	3.7	3.3	2.2	2.2
Fire frequency (yr ⁻¹)	0.05	0.09	0.18	0.25	0.3	0.3
Coefficient of variation	1.0	0.8	0.7	0.8	0.6	0.6
Skewness	1.4	1.2	1.0	2.6	1.4	1.4
Kurtosis	1.4	1.5	-0.9	8.8	2.5	2.5
Minimum fire interval (yr)	1	1	2	1	1	1
Maximum fire interval (yr)	72	32	14	19	11	11
Weibull mean (yr)	19.6	10.9	5.7	4.0	3.5	3.5
Weibull median (yr)	13.8	9.0	5.1	3.4	3.2	3.2
Individual-sample parameter						
Point mean fire interval (yr)	36.9 ^a	NA	32.9	15.1	NA	NA

^a Also 5 samples with only 1 fire scar, for which the mean cannot be calculated

Table 4 Jaccard similarity of fire occurrence (minimum 2 trees scarred/group) across the three groups. The Jaccard Index ranges from 0 (complete dissimilarity) to 1 (complete similarity). Empty cells at the bottom left of the table are duplicates of the cells symmetrically located across the central diagonal

Group	BRIS	HIGH	LOW
Bristlecone pine (BRIS)	1.000	0.316	0.255
Other trees above 2800 m (HIGH)		1.000	0.429
Other trees below 2800 m (LOW)			1.000

and Schoettle (2008) reported an intermittent surface regime in a Colorado bristlecone pine stand. Two Rocky Mountain bristlecone stands in central Colorado had MFI values ranging of 19 and 56 years, with 13 and 12 fire-scarred trees sampled, respectively (Donnegan et al. 2001). The only data from a latitude comparable to the Peaks in the Southwest was a collection of 19 trees in the Sangre de Cristo mountains of northern New Mexico; the MFI value was 59 years (Coop and Schoettle 2011). In sum, the fire frequency for Rocky Mountain bristlecone pine on the

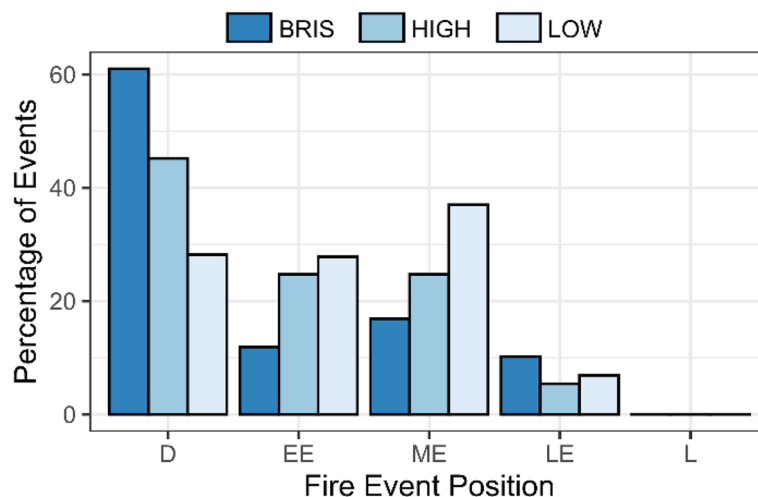


Fig. 4 Seasonal distribution of fire events as inferred from position of the fire-caused injury within the growing ring in the three forest groups: Bristlecone pine (BRIS), other species above 2,800-m elevation (HIGH), and other species below 2,800-m elevation (LOW). Positions are dormant (D), early-earlywood (EE), middle-earlywood (ME), late-earlywood (LE), and latewood (L)

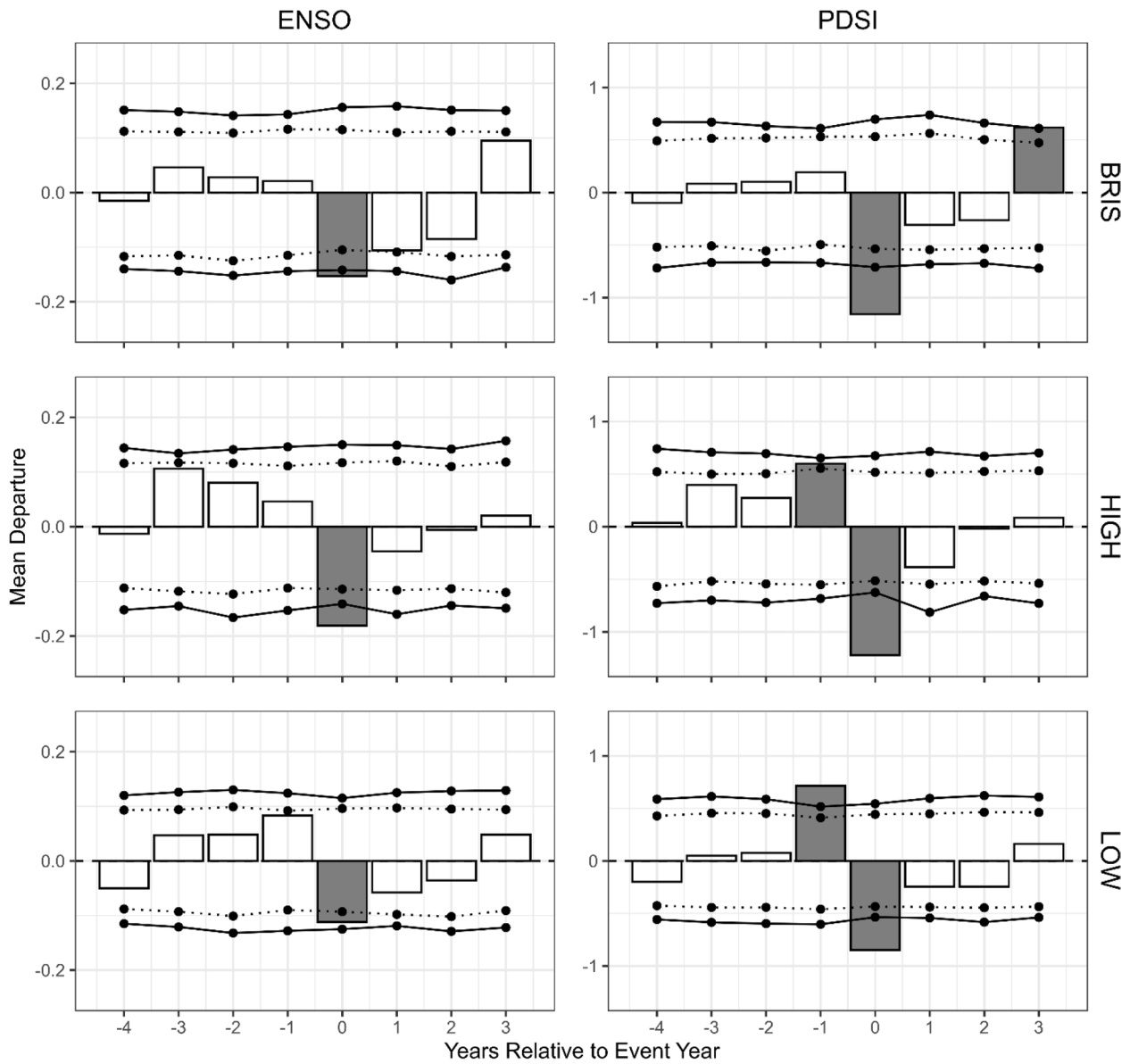


Fig. 5 Superposed epoch analysis (SEA) results using all fire years. ENSO is the reconstructed El Niño-Southern Oscillation index and PDSI is the reconstructed summer Palmer Drought Severity Index. These indices are described in the text. Lines identify statistical thresholds ($\alpha=0.05$, dashed; $\alpha=0.01$, solid) derived from 1000 bootstrapped samples of each climate variable, used to evaluate significance of relationships between climate variables and fire event years

Peaks is within the range of values found in previous studies, but falls at the low end of the MFI range. The relatively frequent fire regime may be related to the steep southerly aspect of the study area, which would have supported dry conditions, coupled with abundant fires coming from lower elevations and high lightning activity near the mountaintop. Taking the shorter common period of analysis of all three groups, the BRIS group MFI declined to 10.1 years, which would make it the most frequent for Rocky Mountain bristlecone pine (Coop and Schoettle 2011).

The formation of scars at earlier positions in the growing ring with increased elevation, H_2 , was consistent with the relatively stable pattern of the monsoon tending to begin in midsummer (Routson et al. 2022). With the dry, windy conditions most favorable for fire spread tending to occur in the early half of the summer and lightning associated with the onset of storms (Carlaw et al. 2017), trees at lower elevation will have had a longer period of growth at any given date than trees at higher elevations (Bunn et al. 2018; Knowles et al. 2020).

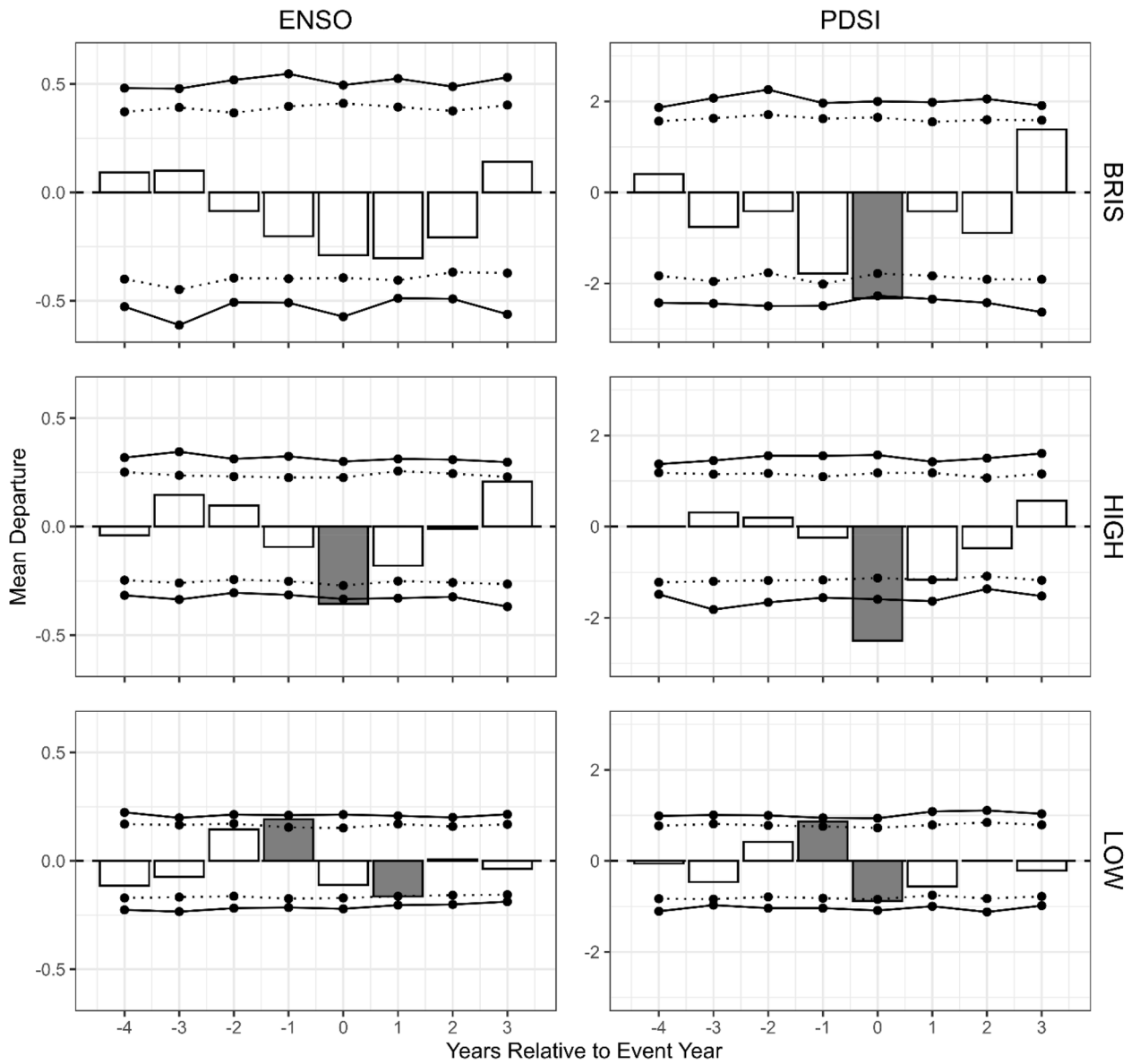


Fig. 6 Superposed epoch analysis (SEA) results using fire years in which 25% or more of recording samples were scarred. ENSO is the reconstructed El Niño-Southern Oscillation index and PDSI is the reconstructed summer Palmer Drought Severity Index. These indices are described in the text. Lines identify statistical thresholds ($\alpha=0.05$, dashed; $\alpha=0.01$, solid) derived from 1000 bootstrapped samples of each climate variable, used to evaluate significance of relationships between climate variables and fire event years

(See figure on next page.)

Fig. 7 Climate, fire, and tree center dates compared. Reconstructions of **a** ENSO and **b** Summer PDSI are given at annual (thin lines) and decadal (black lines, 10-year running mean) scales for comparison with fire history and tree establishment. Panels **c–e** compare relatively large fire events, scarring 10% or 25% of recording samples, with tree center dates by species. Tree center dates are at coring height, 40 cm. Species codes are given in Table 1. ENSO is the reconstructed El Niño-Southern Oscillation index and PDSI is the reconstructed summer Palmer Drought Severity Index. These indices are described in the text

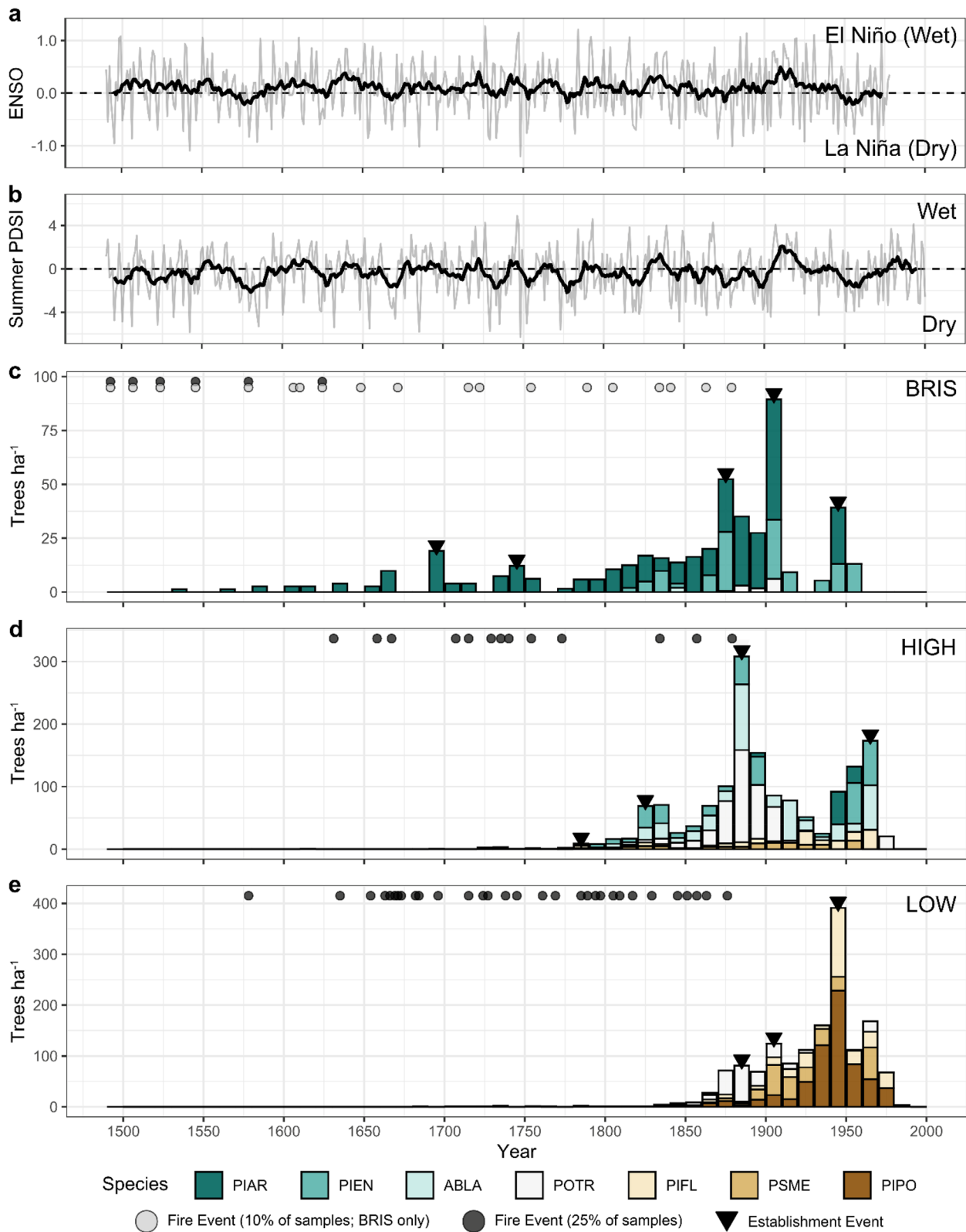


Fig. 7 (See legend on previous page.)

Climatic coupling of fire occurrence on the Peaks was less evident with the phase of ENSO (H_3) than directly with drought as measured by PDSI (H_4). Fire years tended to be in the significantly negative (La Niña) phase, but the relationship was nuanced, with a stronger relationship to all fires events than for the larger fires (25% scarred). In contrast, PDSI was significantly negative in all analyses, but with a greater relative importance in HIGH and BRIS groups, when compared to LOW. The average PDSI value in the fire year was slightly higher in BRIS than HIGH, but much lower (i.e., drier) in LOW. In sum, this implies that all fires and large fires were closely linked to significantly dry conditions. Switching from significantly wet pre-fire (-1) years to significantly dry fire (0) years was apparent for HIGH and LOW in the PDSI analysis. Similar patterns have been observed across the Southwest, with a likely explanation being the production of fine fuels, including abundant herbaceous biomass, additional needle fall, and after heavier snow fall, many small, broken branches in wet years, all providing a continuous fuelbed when conditions switch to drought (Swetnam and Betancourt 1998). As observed in other high mesic forests (Sibold and Veblen 2006), the link to significantly wet pre-fire years was weaker going up in elevation and was not observed in BRIS.

The role of Indigenous people in managing fire environments prior to colonization is of great interest (Swetnam et al. 2016) but it is often difficult to disentangle evidence of human-altered fire patterns such as increased, decreased, or seasonally or spatially distinct burning against a background of generally dry climate and abundant lightning in the Southwest (Allen 2002). Promising studies involve interpreting fire-scar-based reconstruction in the context of Indigenous knowledge (Swetnam et al. 2016), archeological and ethnobotanical evidence (Sullivan et al. 2015), and comparisons of changes in fire-climate relationships, such as those studied here, in times and places where there is evidence of the presence or absence of Indigenous groups (Guiterman et al. 2019; Roos et al. 2022). Our present results from the Peaks cannot be interpreted directly in terms of Indigenous fire management practices because much of this context is not available or beyond the scope of this study. However, adding additional fire regime data to the growing North American dataset (Margolis et al. 2022), particularly in unique or poorly understood areas such as the Peaks, is of value for improving future analyses aimed at this question.

We were able to assess the relationship between the fire regime and tree regeneration, H_5 , because of the forest structure data collected along the same elevational gradient (Cocke et al. 2005). At every elevation, a major pulse of tree establishment occurred after fire exclusion, with

some earlier cohorts tending to occur in fire-free periods (Fig. 7), but there were notable differences among the different elevational/species groups. The BRIS group included the oldest living trees, extending the age distribution about two centuries earlier than the HIGH and LOW groups. Commercially valuable ponderosa pine and Douglas-fir were logged prior to Wilderness designation, which is likely related to the lack of older trees. As described above, the surface fire regime in BRIS was much less frequent and less spatially connected across the landscape than that at lower elevations. The combination of relatively intermittent surface fires and long-lived trees is consistent with patterns reported in the same species in Colorado (Baker 1992; Brown and Schoettle 2008). Our data do not allow us to assess whether patches of severe fire may have occurred in between groups of surviving trees (Baker 1992), but this is a reasonable hypothesis given the lower occurrence of surface fire, allowing fuels to accumulate. We also encountered many dead mature bristlecone pines that were heavily charred (P.Z. Fulé, personal observation). Post-fire regeneration at the edges of severe burns in bristlecone pines in Colorado was shown to be a slow, multi-decadal process, with seed dispersal primarily driven by wind rather than animals (Coop and Schoettle 2009). This pattern is consistent with the steady bristlecone regeneration observed on the Peaks throughout the nineteenth century even prior to fire exclusion (Fig. 7). Spruce and fir regeneration had multiple cohorts with the peaks after fire exclusion in 1879, but older cohorts were evident in an earlier fire-free period a fire-free period from approximately 1770–1840 (Fig. 7), likely reflecting the importance of a pause in fires to allow vulnerable young spruce and fir trees to survive.

Regeneration in the HIGH forest group peaked in the late nineteenth century with aspen making up the plurality of trees, matching the study by Margolis et al. (2011) that attributed much of the initiation of the aspen band on the Peaks to the 1879 fire. The LOW group was dominated by ponderosa pine regeneration but with ample limber pine regeneration as well. While all three groups showed the vast majority of regeneration occurring after fire exclusion, there were distinct twentieth century peaks as well. Ponderosa pine regeneration peaked in the 1940s, later than the widely observed ponderosa regeneration pulse in the region following the 1919 seeding event associated with favorable climatic conditions (Savage et al. 1996; Brown and Wu 2005). It may be the case that many of the ponderosa pines on the Peaks did germinate in 1919 but varying times required for trees to grow to the 40-cm increment core sampling height (Cocke et al. 2005) blurred the decadal age distribution. The lower elevations of the Peaks were roaded and trees were

harvested in the twentieth century prior to Wilderness designation in 1984, so some fraction of the mid-twentieth century pulses of regeneration in LOW and HIGH may be responses to timber harvests, as documented in nearby ponderosa forests (Sánchez Meador et al. 2008).

Overall, the fire-regeneration comparison suggests that the elimination of fire after 1879 was responsible for much of the 70–460% increase in basal area in the study area between 1876 and 2000 (Cocke et al. 2005). Associated driving factors may have been natural regrowth of aspen following the late nineteenth century fires, domestic livestock grazing, the wet period in the early twentieth century that facilitated regeneration (Fig. 7) (Savage et al. 1996), and post-harvest regeneration. Harvest would have had a countervailing influence (i.e., reduction in basal area), as large, mature trees were typically removed (Schubert 1974), although harvest can lead to increases in stand density over time through the abundant establishment of small trees (Naficy et al. 2010).

Conclusions

The forests of the Peaks are in transition. Data from the present study and others on the Peaks show that these forests have experienced centuries of fires in a combination of surface and stand-replacing fire regimes until the exclusion of fires after 1879 (Heinlein et al. 2005; Margolis et al. 2011). Cocke et al. (2005) suggested that the combination of fire regime disruption, extensive regeneration and forest densification, downslope establishment of mesic species, and climate change was a recipe for catastrophic fire and flooding. Since then, the large fires of the early twenty-first century on the Peaks have burned mostly to the south or east of the present study area. When similar fires do burn through the study area, they have the potential to create vast losses of natural and cultural heritage, as well as multi-billion dollar flood destruction (Combrink et al. 2013) through the Río de Flag watershed. Recognizing the threat of severe wildfires in the wildland-urban interface and the watersheds on the Peaks above Flagstaff and surrounding communities, there is a three-decade-long series of collaborative projects aimed at ecological restoration (McCauley et al. 2019; Beeton et al. 2022). If fully implemented, these efforts may help to protect forests on the Peaks from severe fires entering from below. The Coconino National Forest's desired condition for the Kachina Peaks Wilderness states "Disturbances, including fire and flooding, are able to play their natural role within the wilderness area while accounting for public health and safety concerns outside of the wilderness area" (United States Department of Agriculture 2018a).

Management actions such as intentional burning or managing naturally ignited fires for resource benefits

have not been analyzed in detail for the Kachina Peaks Wilderness to date (United States Department of Agriculture 2018b). Simulation modeling of similar nearby forests under future climate and fire scenarios suggest that it is unclear whether attempts to restore fire regimes could successfully recreate historical or desirable ecosystem characteristics in a warming climate, given the contemporary fuel accumulation and ecological stresses associated with warming of high-elevation ecosystems (Flatley and Fulé 2016). However, future fires are inevitable and it would be useful to consider a range of possible approaches for adapting to living with them in ecological and social terms (Schoennagel et al. 2017). Data from the present study provide useful reference points for evaluating future scenarios. Numerous constraints exist to making changes in fire management approaches on the Peaks (Cocke et al. 2005) as well as Wilderness Areas in general (Miller and Aplet 2016; Schoennagel et al. 2017), but time is running short.

Supplementary Information

The online version contains supplementary material available at <https://doi.org/10.1186/s42408-023-00204-4>.

Additional file 1: Figure S1. Tree establishment dates organized by forest type. Note that the forest types in this figure differ from the forest groups (BRIS, HIGH, LOW) used for the present study. The forest types in the figure are based on dominant species encountered on plots as described in Cocke et al. (2005). Species codes are given in Table 1.

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Authors' contributions

Research was conceived by PF, WC, and MM and designed by PF, AC, TH, JC, JR, and MS. Data acquisition was done by PF, AC, TH, JC, JR, MS, and DN. Analysis and interpretation were done by PF, MPB, AC, and KR. PF drafted the manuscript and all authors reviewed the text and contributed edits.

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Availability of data and materials

The datasets generated and/or analyzed during the current study will be submitted to the International Multiproxy Paleofire Database (IMPD) repository, <https://www.ncei.noaa.gov/products/paleoclimatology/fire-history>, upon acceptance of the manuscript. A permanent web link will be provided prior to publication.

Declarations

Ethics approval and consent to participate

Not applicable.

Consent for publication

Not applicable.

Competing interests

The authors declare that they have no competing interests.

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