



Historical and current fire regimes in ponderosa pine forests at Zion National Park, Utah: Restoration of pattern and process after a century of fire exclusion



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ABSTRACT

Plant community resilience in frequent-fire forests of western North America has been compromised by over a century of fire exclusion. However, there are several western National Parks where fires have been reintroduced over the past several decades, including Zion National Park in southwest Utah. Here we reconstruct historical components of fire regimes in ponderosa pine dominated forests at Zion and compare these to recent fire frequency, current plant community and fuels structure, and potential fire behavior. Historical fires burned every 9–10 years on average up until 1879, when fires ceased contemporaneous with introduction of Euro-American livestock grazing and timber harvest in upland forests. Abundant tree regeneration occurred after fire exclusion, with tree density averaging 45 trees ha⁻¹ in reconstructed 1880 forests versus 106 trees ha⁻¹ today. Intervals between recent (since 1988) wildfires and prescribed fires in these same stands ranged from 7 to 13 years, similar to historical fire timing. Depending on whether plots had burned from zero to three times in recent fires, we found significant differences in canopy base heights (increased), duff and litter depths (decreased), and percent cover of grass and forbs (increased), but not tree density, tree basal area, shrub height, shrub cover, or woody fuels. Combined effects of recent fires on overstory and understory structure resulted in a significant difference in likelihood of crown fire occurrence, declining from a mean of 58% in plots with no fire since 1879 to 13% in plots with three fires since 1988. Significant effects were generally seen after two or three fires, suggesting it is the reintroduction of the fire regime and not just individual fire events that restore resiliency. Overall, effects of recent fires are building on the latent resiliency of ponderosa pine forests at Zion National Park, although questions remain about extent and future dynamics of oak and manzanita shrubfields that occupy similar environmental settings, along with a general lack of ponderosa pine regeneration across all plots.

1. Introduction

Fire exclusion and timber harvest since the late 1800s have affected changes in plant and animal community structure, landscape patterns, and ecosystem processes in ponderosa pine (*Pinus ponderosa*) and similar dry conifer forests across western North America (Covington and Moore, 1994; Fulé et al., 2004; Reynolds et al., 2013; Hessburg et al., 2015; Addington et al., 2018). These changes have led to a loss of ecosystem resiliency in many areas as a result of recent, uncharacteristically severe wildfires burning through younger and denser forests than what existed historically (Chambers et al., 2016; Stephens et al., 2016; Stevens-Rumann et al., 2017). Recognition of these changes has led to widespread efforts at ecological restoration in recent decades, often through combinations of mechanical treatments to restore characteristic stand structures followed by prescribed fires (Reynolds et al., 2013; Hessburg et al., 2015; Addington et al., 2018). However, mechanical treatments and prescribed fires are relatively expensive to implement, and the pace and scale of restoration efforts in

ponderosa pine and related forests have lagged far behind what many managers and ecologists consider is needed (e.g., Ryan et al., 2013; Stephens et al., 2016).

Furthermore, in national parks or wilderness areas mechanical treatments are not feasible and fire alone must serve as the only viable restoration alternative (Collins and Stephens, 2007; Larson et al., 2013). Several national parks in the western United States have had policies in place for several decades that use fire alone to achieve resource management objectives, especially in their wilderness areas (USDI-NPS, 1968; van Wagtenonk, 1991; Collins and Stephens, 2007; Waring et al., 2016). These policies largely grew out of the 1963 Leopold Report (Leopold et al., 1963) that argued fire was a critical ecological factor affecting sustainable wildlife habitats, plant community composition and structure, and landscape patterns. Findings from the Leopold Report have been confirmed by overwhelming historical, ecological, and experimental evidence since the report was produced. National Park Service policies include use of both prescribed fires and lightning-ignited wildfires that are allowed to burn under moderate

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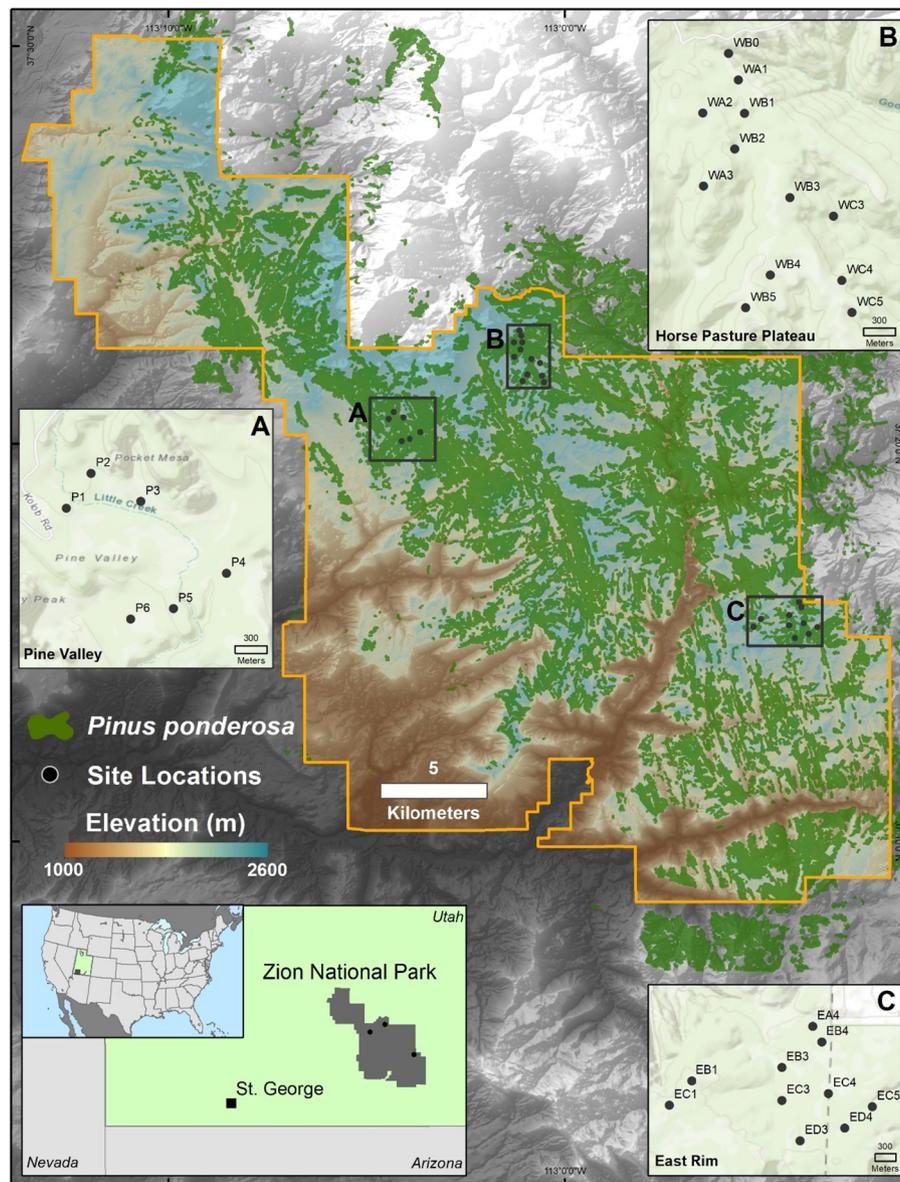


Fig. 1. Location of plots (black circles) in three study areas (inserts) at Zion National Park (yellow outline). Top right: Horse Pasture Plateau; left: Pine Valley; bottom right: East Rim. Green areas on larger map are ponderosa pine dominated areas (from Cogan et al., 2004). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.).

weather conditions (USDI-NPS, 1968; USDI-NPS, 1990). Restoration of fire as an ecological process takes advantage of the “latent resiliency” (Larson et al., 2013) that many of these ecosystems retain in order to achieve desired ecological outcomes. For example, tree recruitment during the period of fire exclusion has often resulted in uncharacteristically dense stands and changes in landscape pattern, especially loss of meadows and openings (Reynolds et al., 2013; Addington et al., 2018). However, low- to moderate-severity prescribed fires and managed wildfires have been shown to reduce stand-level surface and canopy fuel loadings and create more heterogeneous landscapes through mortality of small- to medium-sized (ca. < 1 to 100 ha) stands of trees (Collins and Stephens, 2007; Larson et al., 2013; Parks et al., 2014; Waring et al., 2016; Huffman et al., 2017; Walker et al., 2018). Areas of tree mortality after a century or longer of fire exclusion may be larger than what occurred historically (e.g., Lyderson et al., 2014), but overall spatial effects at stand and landscape scales can begin to return forests to a semblance of historically resilient conditions. Moderate-intensity wildfires can be especially effective in meeting management objectives in reducing stand-level tree densities

and fuel loads (Fulé et al., 2012; Walker et al., 2018; Huffman et al., 2018).

Zion National Park in southwest Utah has had an active fire management program since 1985, utilizing both prescribed fires and managed wildfires in its ponderosa pine forests (USDI-NPS 1985, 2005; Bastian, 2001; Waring et al., 2016). Ponderosa pine forest is found mainly in upland plateau locations in the park, with ponderosa pine-associated communities estimated to cover ~20% of the total park area (Cogan et al., 2004). Much of the upland ponderosa pine forest has burned either in prescribed fires or wildfires in recent decades, with some locations burning up to three times since 1988. A factor in the establishment of the park’s fire management program in 1985 was a fire history study conducted in ponderosa pine forests on the west side of the park on Horse Pasture Plateau and two isolated mesa tops in the early 1980s (Madany, 1981; also reported in Madany et al., 1982; Madany and West, 1980, 1983). These studies reported mean fire intervals (MFI) on the plateau from 1751 to 1882 that ranged from 4 to 8 years. These values are similar to what other studies have found in ponderosa pine forests in northern Arizona (e.g., Hunter et al., 2007)

and elsewhere in southern Utah (Heyerdahl et al., 2011). However, the Madany study only sampled living scarred trees and did not use dendrochronological crossdating methods, which provide absolute dates for tree rings and fire scars recorded in them. Absolute dates for fire scars permits fire dates to be compared across landscapes and to compare with climate reconstructions of droughts or climate teleconnections such as the El Niño-Southern Oscillation (ENSO; e.g., Brown et al., 2008a). For example, fire scars on eight of the Madany samples were crossdated using dendrochronological methods to test the accuracy of fire dates estimated from ring counts (Madany et al., 1982). Agreement between the two methods was poor, with only 26% of 39 fire dates determined to be the same. Perhaps more critically for Zion, crossdating also allows for sampling and analysis of remnant (dead) material for which an outside date is unknown. Large portions of accessible ponderosa pine forests and woodlands at Zion were harvested for timber beginning the late 1800s and continuing into the early twentieth century (Markoff et al., 2009). A number of fire-scarred ponderosa pine stumps were noted during surveys for an assessment of historical ranges of variability (HRV) in fire regimes and community conditions in the park (Brown et al., 2014). Ponderosa pine stumps and other dead wood are highly susceptible to loss from burning but otherwise heartwood can persist several decades after tree death. At the time of the HRV assessment in 2014, a proposal was made to sample these trees before they are lost in future prescribed fires or wildfires, which formed the basis for this study.

The objectives of this study are to use dendrochronological methods to reconstruct fire and forest histories in ponderosa pine forests at Zion National Park and to compare current forest and fuel structure and potential fire behavior relative to the number of times stands have burned in recent decades. We use a systematic sampling design to reconstruct tree recruitment and fire timing, frequency, and seasonality over the past five + centuries in 28 plots across three study areas. We also sampled understory cover, overstory structure, and fuel conditions in each plot to use in fire behavior modeling with the program CFIS (Crown Fire Initiation and Spread; Cruz et al., 2004). Our goal with this analysis is to estimate likelihood of crown fire behavior in individual stands based on current forest conditions relative to the number of times each plot has burned in recent decades. We use these results to infer whether resiliency has increased in Zion National Park ponderosa pine forests compared to its historical fire regime and stand conditions, and the implications of these findings to future fire management here and in similar ecosystems across the region.

2. Methods

2.1. Study area

Zion National Park is located at southwest corner of the Colorado Plateau and eastern edge of the Great Basin in southwestern Utah (Fig. 1). The park landscape is characterized by deep and narrow canyons, steep cliffs, broad upland plateau areas, and isolated mesas, pinnacles, and rock formations. The most spectacular and well-known topographic feature is Zion Canyon, the deep inner gorge of the North Fork of the Virgin River that cuts 700–900 m down through surrounding uplands of the Markagunt Plateau. The park is 59,900 ha in area and ranges in elevation from 1117 to 2660 m. A total of 53,667 ha (~90% of the park) is managed as wilderness (USDI-NPS, 2005).

Intensive Euro-American settlement began with small communities on the North Fork of the Virgin River in the early 1860s. Springdale, at the southwestern edge of the park in the Virgin River valley, was settled beginning in 1862, although permanent occupation did not occur until 1869 (Woodbury, 1944; Crawford, 1986; Markoff et al., 2009). Initial settlement was concentrated near riparian areas, with river diversions for agriculture along the floodplains and dryland farming elsewhere in the valleys (Steen-Adams, 2002). Livestock grazing occurred in both lowland and upland locations, at times to the point of heavy impact on

rangelands (Woodbury, 1933; Steen-Adams, 2002). Logging also had a major impact in upland ponderosa pine forests through the early 1900s (Markoff et al., 2009; Brown et al., 2014). The spectacular scenery of the main canyon led to the establishment of Mukuntuweap National Monument in 1909 (Steen-Adams, 2002; USDI-NPS, 2005). Mukuntuweap is the aboriginal name of the canyon formed by the North Fork of the Virgin River. The monument was enlarged in 1918 to include more upland plateau areas, primarily to mitigate over-grazing of surrounding rangelands and related flooding impacts in the main canyon (Steen-Adams, 2002). The monument was renamed to Zion National Park in 1919.

Fire suppression was the default form of fire management through most of the twentieth century. A 1985 fire management plan established guidelines for the use of both lightning ignited wildfires and prescribed fires in the park (USDI-NPS, 1985). These guidelines emphasized the utility of prescribed and natural fires for re-establishment of functioning ecosystems across the park, especially in fire-adapted upland communities. A second goal of prescribed fires is protection of park infrastructure through fuel reduction treatments, particularly in the main canyon corridor.

2.2. Historical fire regimes and tree age structure

We established 28 plots in three study areas to reconstruct historical tree age and stand structure, fire timing from fire-scarred trees, and to examine fire-climate relationships (Fig. 1). Plot locations were randomly selected in areas dominated by ponderosa pine that had burned from zero to three times in recent prescribed and wildland fires since 1988. We did not distinguish between fire type for this study because of the relatively small numbers of plots and their frequencies of burning, although there can be large differences in fire behavior between prescribed fires and wildfires that affect variations in post-fire forest and fuel structure (e.g., Nesmith et al., 2011). Gambel oak (*Quercus gambelii*) is a common understory shrub in much of the ponderosa pine forest, and also forms large, dense, continuous shrubfields on both Horse Pasture Plateau and East Rim study areas. Greenleaf manzanita (*Arctostaphylos patula*) is another common understory shrub, generally occurring on drier sites. Bracken fern (*Pteridium aquilinum*) is a dense understory plant in localized areas of the Pine Valley study area (Cogan et al., 2004). Rocky Mountain juniper (*Juniperus scopulorum*), Utah juniper (*J. osteosperma*), two-needle piñon (*Pinus edulis*), and one-needle piñon (*P. monophylla*) are other tree species found occasionally with ponderosa pine. Plot elevations ranged from 1972 to 2256 m.

Plot centers were located in the field using a handheld global positioning system unit. We used an *n*-tree density-adapted sampling approach to select the nearest ca. 30 trees (range 21–32) within 30 m to plot centers to characterize tree demography in each plot (Brown and Wu, 2005; Brown et al., 2008b; Heyerdahl et al., 2011). Trees selected for sampling included all living trees ≥ 20 cm diameter at breast height (DBH) and any remnant tree (stump, snag, or log). Increment cores from living trees and cross sections from remnants were collected from ~10 cm height above ground level. Cores had to be no more than a field-estimated 5 years from pith to minimize pith offset when determining recruitment dates (Brown et al., 2019). Tree species were recorded, DBH was measured on living trees, and diameters at 10 cm height were measured on both living and remnant trees. We measured total tree height and height to first live branch (hereafter: crown base height, CBH) on living trees, and distance and azimuth to all plot trees for later stem mapping. We classified remnant trees as to whether bark, sapwood, or only heartwood was present. In addition to plot trees, we also collected cross sections from recently killed Gambel oak ($n = 30$) and greenleaf manzanita ($n = 8$) stems from 12 plots to assess potential ages of shrubs in relation to the plot fire-scar record.

We searched for and sampled all fire-scarred remnant trees located within ~100 m of each plot center. We also opportunistically sampled fire-scarred remnant trees throughout the three study areas as

encountered. This includes an earlier effort on Horse Pasture Plateau and the East Rim in which data from 22 fire-scarred remnant trees were collected. No living fire-scarred trees were sampled. Sample collection from fire-scarred remnant trees involved removal of partial cross sections with a chainsaw. We were able to obtain a minimum tool use permit for use of a chainsaw in Zion National Park wilderness areas, in contrast to the usual requirement of hand tools only. Removal of cross section samples using a chainsaw is a more effective and efficient method of extracting fire history information from remnant wood. The high cutting speed of a chainsaw allows cross sections to be cut from older, often highly decayed, wood that would otherwise be difficult or impossible to sample using hand saws. Use of plunge cuts with a chainsaw also permits smaller partial cross sections (in contrast to full rounds) to be extracted from just the catface area with a minimum of wood removed. This minimizes visual and structural impacts from sampling. Sampling with a chainsaw also increases the chances of obtaining the most complete record of fire from each tree since pieces of often highly decayed wood are less likely to be ripped off and lost as may occur during hand cutting.

Cores were glued to wooden core mounts and cross sections were stabilized and glued to backing boards as needed before sanding. Wood surfaces were prepared using hand planers, belt sanders, and hand sanding to 400 grit sandpaper to the point we were able to see cell structure in tree rings. Crossdating was done with both skeleton plot and visual crossdating methods using a master tree-ring chronology for the park developed for this project (extending from 1180 to 2016 CE). Overlaid concentric circles of varying circumference were used to estimate distance and number of rings to pith for cores and cross sections that did not intersect pith but contained visible inside ring curvature (Brown et al., 2019). After crossdating of ring series was completed for fire-scarred samples, we assigned dates for fire scars. We also assigned seasonal positions for fire scars based on scar location within an annual ring boundary. Scar positions assigned were: early-earlywood: within first 1/3 of earlywood band; middle-earlywood: second 1/3 of earlywood; late-earlywood: last 1/3 of earlywood; latewood: within the latewood band; dormant: between 2 rings; or unknown, due to narrowness of ring or quality of scar. Time spans of individual trees and fire-scar dates were compiled into plot-level fire-demography data sets (Brown et al., 2008b) using program FHX2 (Grissino-Mayer, 2001).

We used superposed epoch analysis (SEA) to compare average annual climate conditions during fire years recorded on ≥ 2 trees at each site to examine fire-climate relationships (Brown et al., 2008a). Two reconstructed climate parameters were tested in SEA, Palmer drought severity indices (PDSI; North American Drought Atlas Version 2, grid point 87; Cook et al., 2004) and the Southern Oscillation Index (SOI; Stahle et al., 1998), an index of the El Niño-Southern Oscillation (ENSO). Significant climate departures were defined as those exceeding 99% confidence intervals determined by bootstrapping in SEA (1000 trials; Grissino-Mayer, 2001; Brown et al., 2008a).

2.3. Current stand structure and fire behavior

We measured surface fuels, ground cover, and overstory tree density and basal area in each plot to examine effects of varying numbers of recent fires (0 to 3). We used the fire behavior modeling program CFIS (Crown Fire Initiation and Spread; Cruz et al., 2004) to estimate the probability of crown fire initiation in each plot given current fuel and canopy conditions. CFIS is a probabilistic fire behavior model used to estimate likelihood of crown fire initiation and rate of spread under varying wind speeds, canopy fuel heights, fine fuel moistures, and surface fuel consumption rates. Advantages to use of CFIS include evaluation against independent observations from crown fire experiments (Cruz et al., 2005) and, especially for our purposes, the use of fuel stratum gap (FSG) for modeling (Tinkham et al., 2016). FSG is the distance between mean height of surface fuels and mean canopy base height (CBH). Shrubs, especially Gambel oak and greenleaf manzanita,

are important understory components of upland forests at Zion National Park (Cogan et al., 2004) and a major factor in fire behavior at both stand and landscape scales. We ran CFIS for all plots using 4% fine fuel moisture and 10 km hr⁻¹ open wind speed.

Plot tree density and basal area were determined from 10 (ft² ac⁻¹) basal area factor variable radius prism plots centered on tree demography plots. These data along with mean tree heights from age structure plots were used to calculate canopy bulk density for each plot using the Canopy-Fuel Characteristics Calculator (Alexander and Cruz, 2010). We used measured values from the plots to estimate CBH. We established a 30 m long transect oriented N-S from plot center to measure understory cover and fuels. Fuel amounts (1-hour, 10-hour, and 100-hour timelag dead fuels plus grass and forbs) were measured and averaged from four 1 m² quadrats evenly spaced along the transect using photoload fuel loading estimation methods (Keane and Dickinson, 2007). Litter and duff depths were measured in each corner of the quadrats and averaged for the plot. Ground cover was tallied every 50 cm along the transect, with shrub species and heights recorded as encountered. Large woody fuels (1000-hour timelag) were tallied and measured as encountered on the transect. Trees < 20 cm DBH and seedlings (< BH tall) were tallied in a 2 m × 30 m belt transect centered on the main transect.

Overstory, understory, and CFIS results from each plot were grouped by number of times the plots had burned and tested for significant differences using analysis of variance (ANOVA). For significant effects identified by ANOVA, post-hoc pairwise comparisons were performed with Tukey honest significant difference (HSD). Significance for all tests was $\alpha = 0.05$.

Finally, we calculated density and basal areas of trees present in 1880 in six plots that had not burned in recent fires to compare to those same values measured in the current forest to assess changes in stand structure due to fire exclusion. Tree density was based on numbers of trees present in the plot in 1880 scaled to a per hectare basis. Tree basal areas were determined from tree diameters measured or estimated at the point when trees were alive in 1880 (following methods in Brown et al., 2008b). We did not calculate these values for plots that have burned in recent fires because of both likely missing historical evidence and changes in recent structure due to tree mortality during fires since 1988.

3. Results

3.1. Historical fire regimes and tree age structure

Tree-ring data were collected from 815 trees from 28 plots in the three study areas. The majority of these were ponderosa pine with 42 juniper, 32 Gambel oak, and 3 piñon trees also collected. A total of 780 trees (96%) were able to be crossdated; many of the remainder were ponderosa pine remnants too decayed or that contained too few rings to have confidence in crossdating. A total of 22% of the trees sampled were remnants, with 58% of these stumps. We also were not able to crossdate many of the Gambel oak (all either snags or logs), mainly because of very tight outer rings and a general lack of crossdating with the ponderosa pine. We also were not able to crossdate any of the Greenleaf manzanita samples owing to extreme ring suppression; ring counts on these were on the order of 80 to > 120 rings, in stems with 10 cm height basal diameters < 4–8 cm.

Fires were historically frequent for several centuries before the 20th century (Fig. 2, Table 1). Mean and median intervals between fires recorded on ≥ 2 trees from the early 1600s to the last widespread historical fire date in 1879 across Horse Pasture Plateau and East Rim study areas were 9 to 10 years with similar ranges of intervals (Table 1). We did not find many old trees with fire scars in Pine Valley, and only two historical fire dates were recorded on ≥ 2 trees in this study area. We therefore did not have enough data to calculate fire interval statistics. Intervals between recent fires (since 1988) in all three study

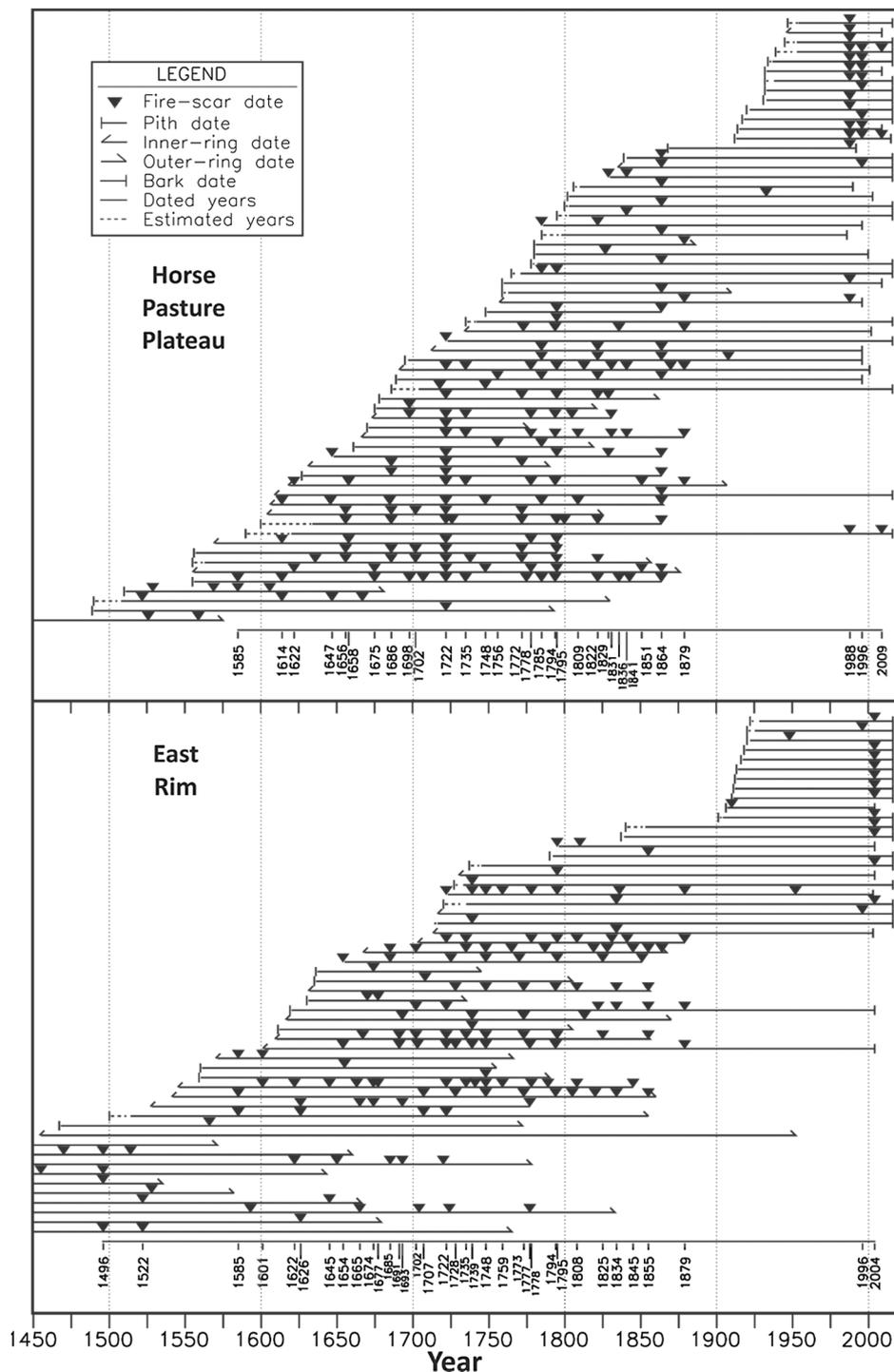


Fig. 2. Fire chronologies for Horse Pasture Plateau (top) and East Rim (bottom) study areas. Time spans of individual trees are indicated by horizontal lines with dates of fire scars indicated by inverted triangles. Dates at the bottom of each panel are those recorded on ≥ 2 trees in each study area.

areas are within ranges of historical intervals found on Horse Pasture Plateau and East Rim (Table 1). Fire scars were predominately recorded later in the growing seasons, with 79% of scars in which seasonality could be assessed recorded as either late-earlywood, latewood, or dormant (dated to the previous year).

Historical fires occurred during years of significantly low PDSI in both Horse Pasture Plateau and East Rim study areas (Fig. 3A, C). In addition, two years prior to fire years were significantly wetter than average. This wet/dry pattern was present in SOI in Horse Pasture Plateau fire years (Fig. 3B) but not in East Rim fire years (Fig. 3D).

Of the plot trees collected for age data, 631 contained pith or for

which a pith could be estimated (Fig. 4). A total of 73.9% of these trees either contained pith or were ≤ 5 rings from pith for pith estimation, and 91.4% were ≤ 10 rings from pith (see also Brown et al., 2019, for a similar result). The oldest tree sampled (a log) had a pith date of 1180, while the oldest living trees we sampled had pith dates in the early 1500s (1525 on Horse Pasture Plateau and 1530 on East Rim). Approximately half (56%) of the juniper and piñon trees had established after 1879 but we did find several older juniper trees, including a Rocky Mountain juniper log with a pith date of 1370, and the oldest living juniper with a pith date of 1738.

Table 1

Fire interval statistics (in years) for study areas at Zion National Park. Historical intervals are those between fires recorded on ≥ 2 trees (dates in Fig. 2). Historical interval statistics for Pine Valley were not calculated because of probable missing records (see text). Recent fire dates and intervals between them are shown for comparison to historical intervals.

Study area	Historical intervals			Recent fires (intervals)
	Period of analysis	MFI \pm SD ¹	MeFI ²	
Horse Pasture Plateau	1647 to 1879	10 \pm 5	10	1 to 20
Pine Valley	—	—	—	1998, 2007/8 ³ (9–10)
East Rim	1622 to 1879	9 \pm 6	9	1 to 24

¹ Mean fire interval \pm standard deviation.

² Median fire interval.

³ Prescribed fires that occurred over two seasons.

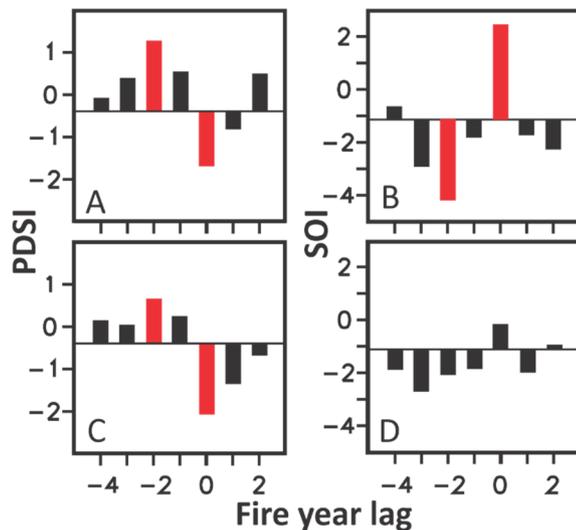


Fig. 3. Superposed epoch analysis (SEA) of PDSI (left panels) and SOI (right panels) anomalies during fire years recorded on ≥ 2 trees in Horse Pasture Plateau (top panels, A and B) and East Rim (bottom panels, C and D). Note that SOI is reversed relative to PDSI; high SOI are La Niña (dry) years and low are El Niño (wet) years. Fire year lag 0 on x-axis is the average climate anomaly for fire years, with antecedent conditions indicated by negative lags. Y-axis units are relative anomalies from mean of each index. Significant departures from means are in red based on bootstrapped confidence intervals in SEA ($P \leq 0.01$).

3.2. Current stand structure, fuels, and fire regimes

There were significant differences in canopy base height between plots depending on number of fires plots experienced in recent decades (Fig. 5D). CBH averaged from 2.7 m in plots without recent fire to 6.4 m in plots that experienced three fires. There were no differences between plots in tree basal areas, densities, or shrub heights (Fig. 5A, B, C). Basal area averaged 7.1–17.0 m² ha⁻¹, tree density averaged 62–117 trees ha⁻¹, and shrub height averaged 0.5–0.8 m across all plots depending on number of recent fires. We found extremely few seedlings (a total of eight, three of which were Rocky Mountain juniper) or trees < 20 cm DBH (a total of 15) across all plots.

Understory cover varied significantly depending on number of recent fires for both litter and duff depths and grass and forb cover, but not for shrub cover or woody fuels (Fig. 6). Litter and duff depth averaged 4.5 cm for plots that had not experienced any recent fires, while plots that had burned from 1 to 3 times averaged 1.7–2.4 cm (Fig. 6B). Grass and forb cover was low for all plots, and averaged from 3 to 18% (Fig. 6D). Shrub cover averaged 12–27% across all plots and numbers of fires. Woody fuels averaged 4.06 Mg ha⁻¹ (range 0.25–18.21 Mg ha⁻¹) with no differences based on number of fires. Scattered shallow needle litter and bare ground or rock were the most common ground cover across all plots, with an average of 61% (range

39–90%; data not shown).

There was a significant difference in likelihood of crown fire occurrence from CFIS modeling based on the number of recent fires (Fig. 7). The probability of crown fire declined from an average of 58% in the plots that have not experienced fire since 1879, to 13% on average in plots that have experienced up to three fires in recent decades.

Finally, tree densities were significantly lower in the 1880 forest than in the current forest in the six plots that have not burned in recent fires, all on Horse Pasture Plateau (Fig. 8, left). Basal area tended to be lower in the 1880 forest but was not significant (Fig. 8, right). Tree density in the six plots that had not burned in recent fires averaged 45 trees ha⁻¹ in 1880 and 106 trees ha⁻¹ today, and basal areas averaged 12.2 m² ha⁻¹ in 1880 and 17.0 m² ha⁻¹ today. We were not able to reconstruct historical stand structure in all plots because of probable or, in several plots, known missing historical forest structure evidence in stands that had burned during recent fires. We attempted to collect cross section samples from all remnant trees, but in several recently burned stands remnant trees were too eroded to collect a usable sample. This evidence included charred roots or other fragmentary pieces of charred wood, often surrounding a ground depression indicating where a tree once stood.

4. Discussion

4.1. Fire and forest history

Fires were frequent occurrences in ponderosa pine forests at Zion National Park for several centuries before 1879 (Fig. 2, Table 1). Evidence suggests these were primarily surface fires due to the abundant fire scars recorded on trees and a lack of cohort structure in stands in relation to fire occurrences (*sensu* Brown et al., 2008b; Fig. 2). Fires ceased after 1879, likely as a result of heavy grazing that accompanied Euro-American settlement in the region (Woodbury, 1933; Steen-Adams, 2002). Livestock grazing, especially when lands were overgrazed (e.g., Woodbury, 1933), disrupted fine fuel continuity and led to a general cessation of spreading fires in ponderosa pine forests throughout western North America (e.g., Swetnam et al., 1999). Historical fires occurred during drought years but were typically preceded by wet conditions that would have increased grass and forb (fine fuel) amount and continuity (Fig. 3). Needle litter and dead woody fuels were likely not as critical as fine fuels for fire occurrence since there would have been less time for these fuels to build up between successive fires occurring every 9–10 years on average. A wet/dry coupling in climate forcing has often been found in historical fire climatology in many southwestern dry pine forests, where spreading fires were limited by grass and forb fine fuel continuity (Swetnam et al., 1999; Brown and Wu, 2005). Several other studies also have found ENSO to be a primary driver of this wet/dry oscillation in many southwest forests (Swetnam et al., 1999; Brown and Wu, 2005; Brown et al., 2008a). An interesting pattern in the Zion data is that ENSO

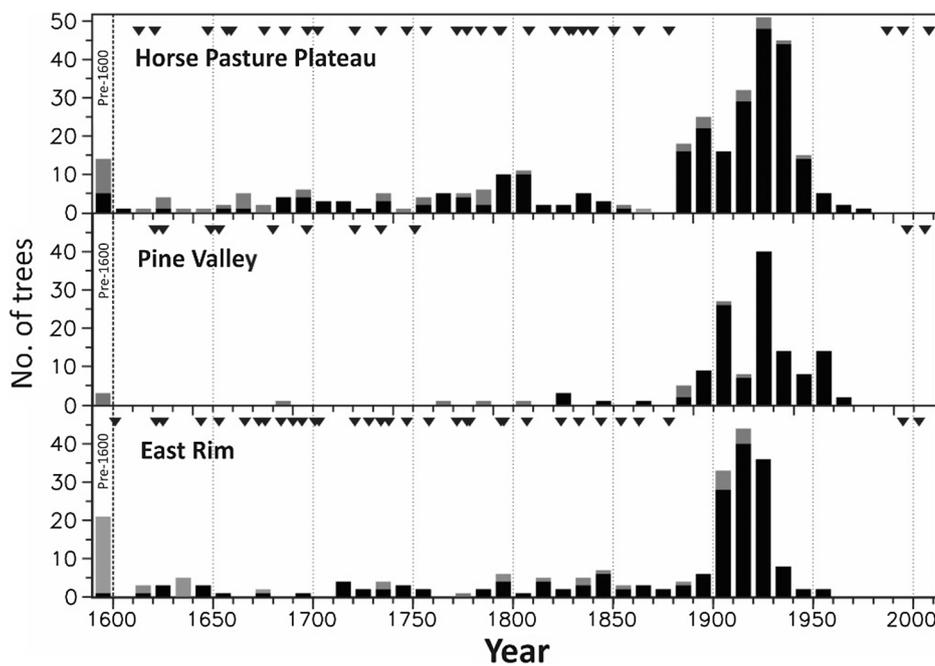


Fig. 4. Tree recruitment chronologies for three study areas at Zion National Park. Recruitment dates (histograms) are for sampled trees that either had pith or for which a pith date could be estimated with confidence. Fire dates (triangles) are those recorded on ≥ 2 trees (dates in Fig. 2), except for Pine Valley which are fire dates recorded on any tree. Grey bars are recruitment dates of remnant trees, black bars are recruitment dates of living trees. “Pre-1600” bars are numbers of trees with recruitment dates predating 1600.

appears to be a driver of these oscillations on the west side of the main canyon but not in the east (Fig. 3). Only nine dates between 1585 and 1879 were found on both sides of Zion Canyon (Fig. 2), which suggests varying climatic or other forcing factors for fire occurrence. Brown et al. (2008a) also found ENSO effects in some sites but not others across Utah. Southern and central Utah is the “pivot point” in ENSO effects on precipitation patterns in the Southwest (Brown et al., 2008a), and the historical Zion fire regime appears to follow a similar pattern across a much smaller scale.

Cessation of fires after 1879 was followed by abundant ponderosa pine regeneration in all plots, although a majority of regeneration was delayed by a few years into the early twentieth century on the East Rim (Fig. 4). Recruitment on Horse Pasture Plateau shows similar patterns to those found by Madany and West (1983) for the same area. Reconstructed 1880 tree density just before effects of fire exclusion in plots that have not had recent fires was 40% of current density (Fig. 8). Cohorts of post-fire exclusion tree recruitment are common throughout the range of ponderosa pine forests (e.g., Brown and Wu, 2005; Brown et al., 2008b; Reynolds et al., 2013; Addington et al., 2018). Frequent pre-settlement fires would have killed a majority of ponderosa pine seedlings and led to temporally and spatially patchy recruitment of overstory trees (White, 1985; Brown and Wu, 2005). Without fire-caused mortality after 1879, regeneration was unchecked and led to increased tree densities both at stand and landscape scales. A lag in regeneration timing after 1879 on the East Rim (Fig. 4) may have been the result of more intense grazing pressures in some areas relative to others (Madany and West, 1983). However, whether immediately or a few decades after 1879, all plots saw increased tree densities that continue to structure stands to the present.

A pattern of abundant post-1879 tree recruitment appears to have been especially prevalent in Pine Valley. We found very few trees predating 1879 anywhere across the study area (Fig. 4), although current stands contain tree densities similar to Horse Pasture Plateau and East Rim. We suspect much of this area was an open grassland or very sparse woodland prior to fire exclusion. We doubt that the lack of older trees was the result of a stand-replacement fire event because we did not find large numbers of older logs or snags, which would be expected to still be present if such an event had occurred within the past century or two. Rather, we suggest that an open landscape in Pine Valley was likely maintained by frequent surface fires. However, we found very

few fire-scar records because of the lack of older trees. The few fire scars that were dated from Pine Valley all matched fire dates found on nearby Horse Pasture Plateau. Both areas are within the same elevation range and there is relatively contiguous upland landscape between the two areas, thus it is probable that Pine Valley burned as often as Horse Pasture Plateau prior to 1879 (Table 1).

We found eight trees that had previously been sampled by Madany (1980) still living on Horse Pasture Plateau. Madany and West (1983) reported MFI on the plateau to be between 4 and 7 years prior to 1881, which is lower than the MFI of 10 years found by this study for the same area (Table 1). However, Madany’s data were not crossdated, which likely resulted in more fire dates than actually occurred. For example, Madany et al. (1982) list 1881, 1879, 1875, 1872, 1866, 1864, 1860, and 1856 as widespread fire dates for the last 30 years of the historical fire regime in their ring-count study. We found crossdated fire scars recorded on ≥ 2 trees for this same period only in 1879, 1870, 1864 and 1854 (Fig. 2). We found no scar dates on Horse Pasture Plateau after 1879 except for scars recorded on single trees in 1908 and 1933, until recent decades when fires started up again beginning in 1988.

4.2. Current fuels and forest structure

Restoration of fire into the Zion landscape has had effects on some overstory and understory components but not others (Figs. 5 and 6). As a general pattern, significant effects were evident only after two or three fires. Other studies that have looked at restoration of fire in frequent-fire ecosystems also have found that more than one fire is typically necessary to begin to meet restoration objectives (Larson et al., 2013; Waring et al., 2016; Walker et al., 2018). These studies also found restoration effectiveness may be evident in some metrics but not others (e.g., Huffman et al., 2017). For example, we did not see significant effects of repeat fires on overstory tree density or basal area in the plots we sampled (Fig. 5A, B), even though an objective outlined in the 2005 Zion revised Fire Management Plan (USDI-NPS, 2005) is reduction of overstory density by 30–60% (Waring et al., 2016). It is likely we did not see effects on tree density because of variability in biophysical locations of our plots. All the unburned plots we examined were in relatively dry stands on the north end of Horse Pasture Plateau, and are likely low density for bioclimatic reasons rather than disturbance history. On the other hand, trees in these plots had significantly lower

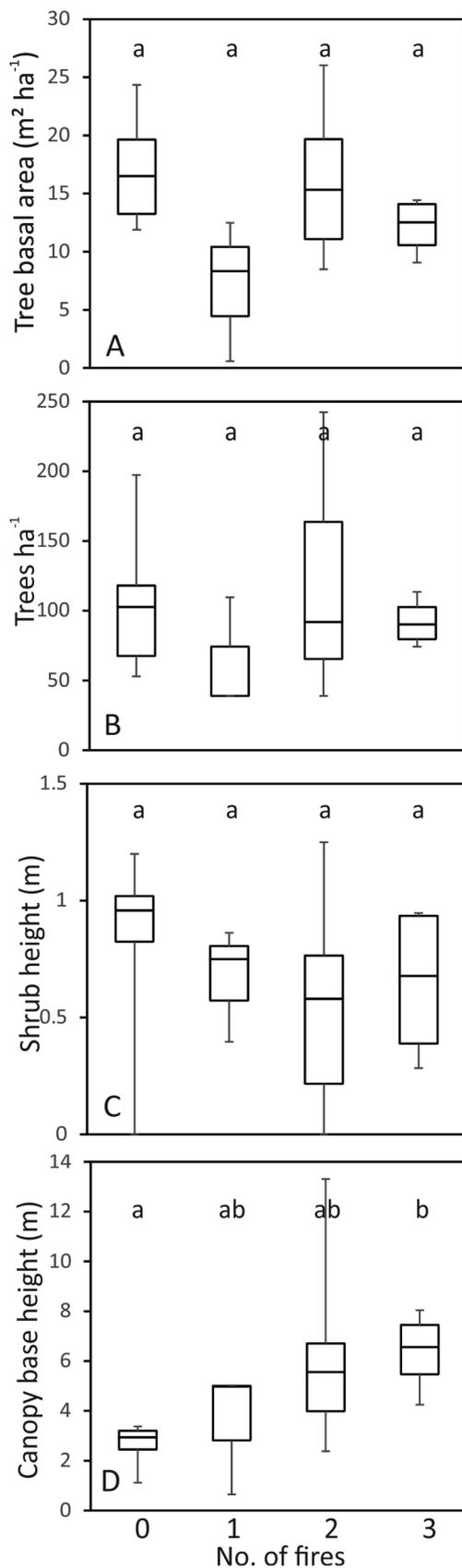


Fig. 5. Box plots of tree and shrub structure in plots that burned 0–3 times at Zion National Park: A) tree basal area; B) tree density; C) shrub heights; D) tree canopy base heights. Box plot lines represent ranges of values, boxes are 25th and 75th percentiles, and horizontal lines in each box are medians. Letters denote significant differences between number of fires in Tukey’s HSD test ($\alpha \leq 0.05$).

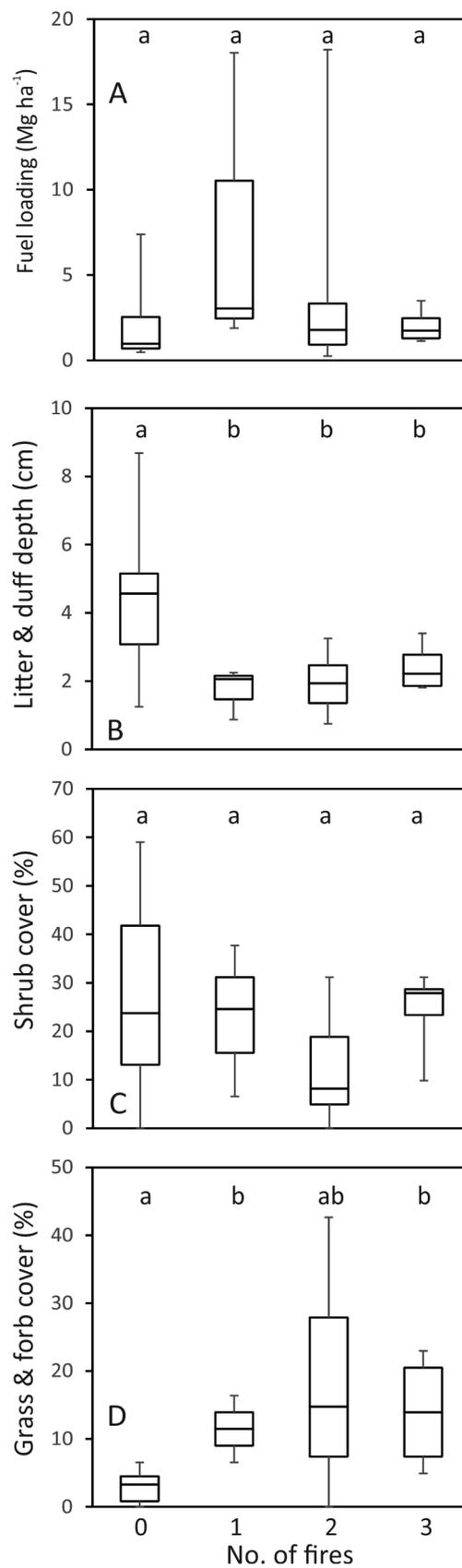


Fig. 6. . Box plots of understory structure in plots that burned 0–3 times at Zion National Park: A) fuel loading (1–1000 h timelag fuels); B) litter plus duff depths; C) shrub cover; D) grass and forb cover. Box plot lines represent ranges of values, boxes are 25th and 75th percentiles, and horizontal lines in each box are medians. Letters denote significant differences between number of fires in Tukey’s HSD test ($\alpha \leq 0.05$).

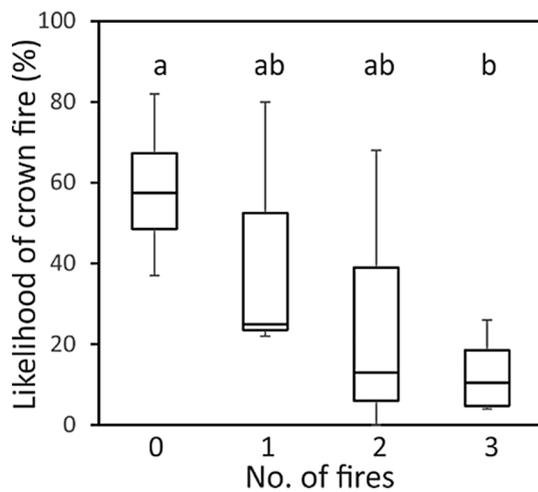


Fig. 7. Likelihood of crown fire occurrence from CFIS modeling of fire behavior based on crown and fuel structures in plots that burned 0–3 times at Zion National Park. Box plot lines represent ranges of values, boxes are 25th and 75th percentiles, and horizontal lines in each box are medians. Letters denote significant differences between number of fires in Tukey's HSD test ($\alpha \leq 0.05$).

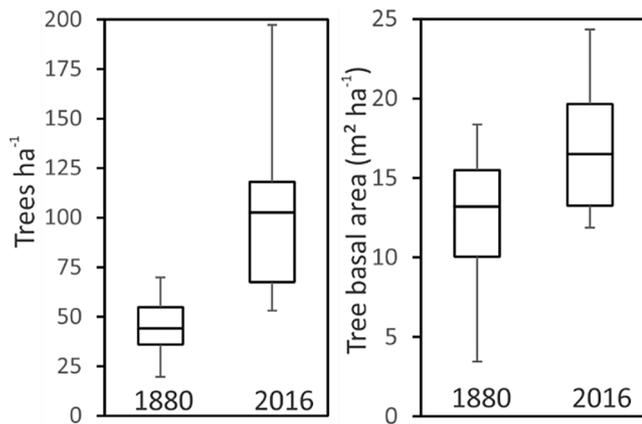


Fig. 8. Tree densities (left) and basal areas (right) reconstructed in 1880 and measured in 2016 for six unburned plots on Horse Pasture Plateau study area. Box plot lines represent ranges of values, boxes are 25th and 75th percentiles, and horizontal lines in each box are medians. Tree densities but not basal areas were significantly different between the two dates in a paired t -test ($\alpha \leq 0.05$).

canopy base heights (CBH) in comparison to trees in plots that have been burned three times (Fig. 5D). Many of the trees in these plots established post-1879 and have never seen fire, which if it does not kill the tree will tend to scorch lower branches and raise CBH. Tree densities and basal areas in the plots we examined fall within target metrics proposed by Huffman et al. (2017) for assessing efficacy of resource objective wildfires in ponderosa pine forests in northern Arizona. Interquartile ranges (IQR) proposed by Huffman et al. include tree densities of 56–138 trees ha⁻¹ (plots in this study ranged from 64 to 125 trees ha⁻¹) and basal areas of 9–17 m² ha⁻¹ (this study 11–19 m² ha⁻¹). In addition, Huffman et al. (2017) suggest an IRQ for CBH of 5–7 m; trees in plots in this study that experienced three fires had CBH averaging 4.3–8.0 m (Fig. 5D).

Shrub height is another metric that was not significantly different between plots (Fig. 5C), but one that again is not only affected by fire history but also bioclimatic conditions and, more importantly at Zion, varying shrub species. Several of our plots had a mix of Gambel oak and greenleaf manzanita as dominate understory shrubs. Greenleaf manzanita tends not to grow as tall as Gambel oak; the few apparently old stems (probably dating to post-1879 mortality) we sampled in

unburned plots were < 1 m tall. In contrast in plots that had burned in recent fires, dead Gambel oak stems were often up to twice as tall as current living stems, often > 2 m. This mix of shrub species resulted in a mix of shrub heights not dependent on how often the stand had recently burned (Fig. 5C).

We found significant differences in both litter and duff depths and grass and forb cover after only a single recent fire in plots (Fig. 6). Litter and duff depths were not great for any plot, but declined by over half after one fire relative to plots that had not burned. Grass and forb cover was generally low for all plots, and may be a legacy of heavy livestock grazing in the uplands (Woodbury, 1933; Steen-Adams, 2002). For example, Madany and West (1983) found an average of 5% grass and forb cover in their ponderosa pine plots on Horse Pasture Plateau in 1980 before recent fires, similar to our finding of 3% mean cover in plots that had not burned from this same area. They found considerably greater grass and forb cover on isolated mesas that were never grazed, with an average of 49%. As fire is reintroduced into our study areas with a history of often heavy grazing, grass and forb cover is beginning to recover, with means ranging from 12% to 18% in plots that have burned one to three times, with a maximum in a single plot on East Rim of 43% (Fig. 6D).

4.3. Management implications

Ponderosa pine forests at Zion National Park exhibit latent resiliency (Larson et al., 2013) that is accommodating resumption of frequent, low- to moderate-severity fires after the century-plus of fire exclusion since 1879. Stands that have burned up to three times since 1988 have reduced litter and duff depths (Fig. 6B) and, more importantly, increased canopy base heights (Fig. 5D), that together with other fuel and structural effects are contributing to a reduction in the likelihood of crown fire behavior (Fig. 8). These results agree with other studies that examined use of low- to moderate-severity fire alone as a restoration management strategy and found largely beneficial effects on ecosystem conditions (Collins and Stephens, 2007; Larson et al., 2013; Lydersen et al., 2014; Parks et al., 2014; Walker et al., 2018). Both this study and others have reached a similar conclusion that repeat fires are key to restoring resilience through their cumulative effects on community and fuel structure. It is the return of the fire regime and not just individual fires that confers resilience on frequent-fire adapted ecosystems. Fire affects community and landscape structure but is, in turn, driven by feedbacks from community and fuel conditions. This self-regulation of process and pattern was interrupted by fire exclusion, but there is enough “memory” (Peterson, 2002) in these ecosystems to be able to return to semblances of historical trajectories once frequent fires are reintroduced.

A central question for fire management at Zion National Park is the extent and prevalence of Gambel oak and other shrubfields on the current landscape, and the future trajectory of pine-oak woodland versus shrubfield mosaics under the reintroduced fire regime. Pine-oak woodlands and oak shrubfields represent two alternative stable states for landscapes not only at Zion but many areas around the Southwest (Abella, 2008; Guiterman et al., 2018; Fig. 9; S1, S2). Guiterman et al. (2018) found that these two ecosystems occur at similar climatic and topographic settings in the Jemez Mountains of New Mexico, with disturbance histories more important than biophysical factors in controlling their historical and recent landscape mosaic. It is likely the same situation at Zion. Our study targeted sampling only to areas of extant ponderosa pine forest, with Gambel oak a common understory shrub in many of the plots. In these stands under the historical fire regime, above-ground oak stems would likely have not grown very tall before being killed by the next fire burning through the stand. Many of the oak stems we were able to crossdate had pith dates in 1880 or 1881, having sprouted immediately after the last fire in 1879. Above-ground oak stems often post-date the last pre-settlement fire (e.g., Heyerdahl et al., 2011; Guiterman et al., 2018). Recent fire behavior in the stands

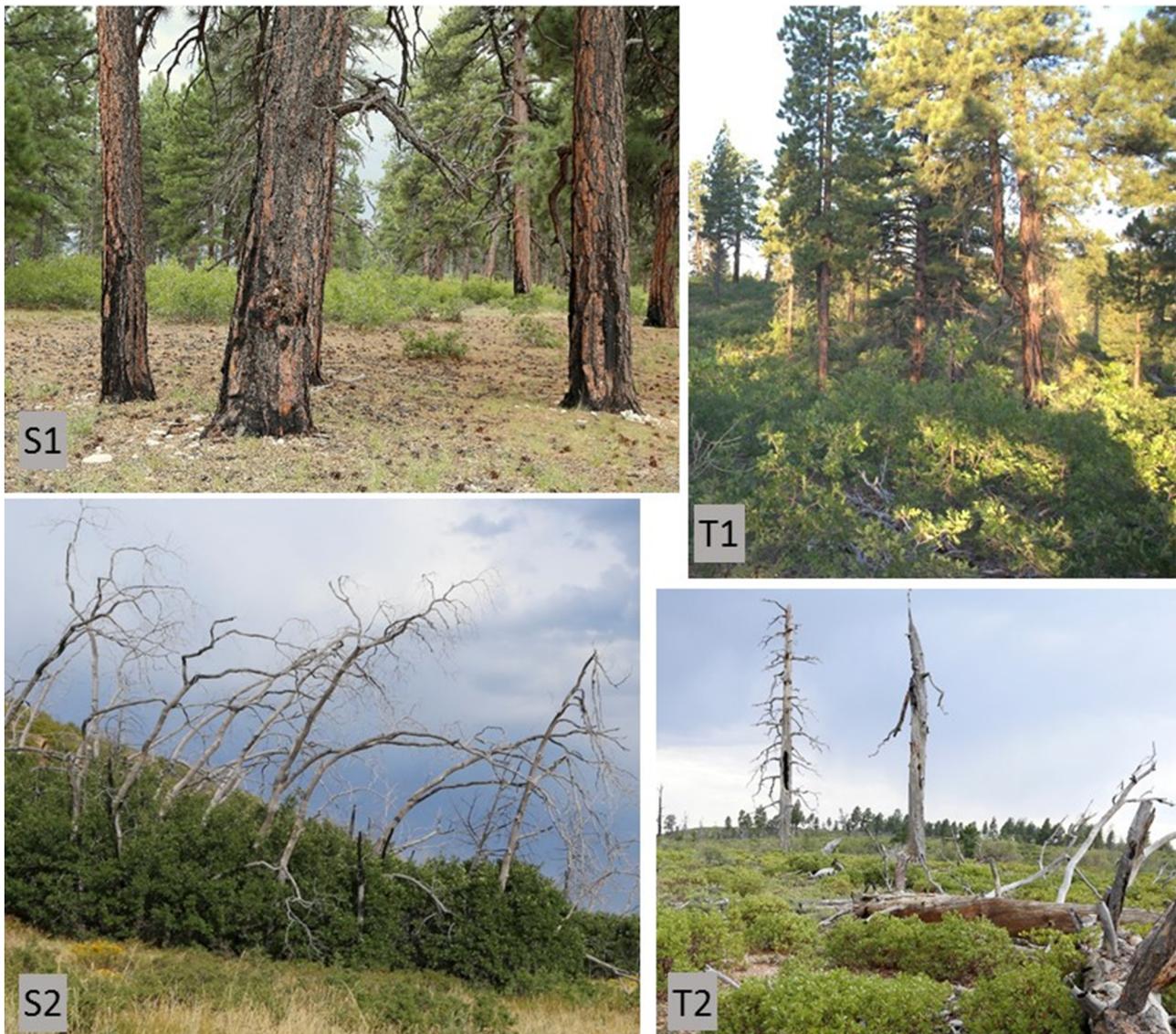


Fig. 9. Examples of stable (S1, S2) and transitional (T1, T2) forest and shrub structures in uplands at Zion National Park, all on Horse Pasture Plateau. S1: Pine-oak woodland, with overstory ponderosa pine and low (< 0.5 m) understory Gambel oak. Frequent, low-intensity fires historically maintained this structure, and are doing so again under the reintroduced fire regime. This stand burned in recent wildfires in 1988, 1996, and 2009. S2: Persistent Gambel oak shrubfield with large, older oak stems killed in recent high-severity fire and new oak stems sprouting back. Frequent to infrequent high-intensity fires maintain this structure through time. This stand also burned in recent wildfires but it is not known which fire or combination of fires killed the older stems. T1: Pine-oak woodland with understory of older (established immediately after 1879), tall (> 1 m) oak stems. This is a stand that has not burned in recent fires. Reintroduction of low-to moderate-intensity fire could reduce oak cover while maintaining the overstory pine, which would restore the stand to S1, or kill overstory pine in high intensity fire and shift to T2. T2: Recently killed ponderosa pine overstory surrounded by a shrub matrix, in this case a mix of Gambel oak and greenleaf manzanita. This area may transition to either S2 or, if ponderosa pine is able to reestablish, T1 or S1. This stand burned in recent fires but again we do not know which fire or combination of fires resulted in the pine mortality.

we sampled was of low to moderate enough intensity to have burned through the oak understorey yet left a majority of the ponderosa pine overstorey intact (e.g., Fig. 9; S1). Waring et al. (2016) found similar results after first and second entry burns at Zion. However, many of the pine-oak woodlands where we sampled are bordered by other areas where it is visually evident that pine overstorey was killed by high-intensity fire behavior during recent fires (e.g., Fig. 9; T2). This was especially the case on Horse Pasture Plateau, which saw wildfires in 1988, 1996, and 2009, and where large areas (ca. 10–100 ha) of what was apparently pine-oak woodland before these recent fires are today dominated by Gambel oak or greenleaf manzanita with no pine present.

A balance between these alternative stable states undoubtedly existed historically, and the question thus becomes one of relative scale of pine woodlands versus shrubfields across the historical landscape

compared to the mosaic present today. Guiterman et al. (2018) found in the Jemez Mountains of New Mexico locations of oak shrubfields and pine-oak forests have been relatively stable through much of the twentieth century. Recent fires at Zion - especially wildfires on Horse Pasture Plateau but also prescribed fires on the East Rim - contained a mix of fire behavior that resulted in not only increased resilience in stand and fuel structure in pine-dominated forest stands as found by this study (Fig. 9, S1), but also a shift in other areas to oak- or manzanita-dominated shrubfields that are likely more extensive than what was present historically (Fig. 9, T2). Whether pine can ever recover in these areas is a question for the future, especially if climate change leads to warmer and drier conditions in which pine seedlings have difficulty in re-establishing (e.g., Stevens-Rumann et al., 2017). We saw very few seedlings or small diameter (< 20 cm) trees anywhere in our

transects, which may support a hypothesis that recent ponderosa pine recruitment has been limited by unsuitable climate conditions. However, ponderosa pine regeneration tends to be episodic (e.g., [Savage et al., 1996](#); [Brown and Wu, 2005](#); [League and Veblen, 2006](#); [Feddema et al., 2013](#)) and it is possible that proper conditions for regeneration have not occurred in recent years. Continued fires will be needed to maintain and continue to promote greater resilience in the pine-shrub forested areas. Furthermore, it may be that frequent fire could be useful to reduce oak or manzanita cover enough that pine can get a foothold in areas where it has been lost, or in other areas that have been historically dominated by shrubfields. There are many areas on both Horse Pasture Plateau and East Rim with relatively small, isolated stands of pine-oak woodland or pine meadows in the shrubfield matrix that could possibly serve as “anchor points” for expansion of woodlands through targeted repeat application of higher intensity fires on short intervals (e.g., [Abella, 2008](#)). Once shrub density is reduced, it may be possible for ponderosa pine to reestablish in these areas, setting up conditions for low-intensity fires to burn through more expansive pine-shrub woodlands some day in the future.

At this point in time, the critical question for land management across the western US is if episodic low-intensity fires can ever again play their central, keystone role in structuring frequent fire-adapted ecosystems, especially across large landscapes ([North et al., 2015](#)). Numerous studies have shown that recent high-severity fires severely compromise the resilience of frequent-fire forests (e.g., [Chambers et al., 2016](#); [Stevens-Rumann et al., 2017](#)), while others just as clearly show that reintroduction of low to moderate intensity fires – especially repeat fires burning at historical fire frequencies – can and do take advantage of latent resiliency to begin to return these forests to longer-term, sustainable ecological trajectories (e.g., [Collins and Stephens, 2007](#); [Larson et al., 2013](#); [Lydersen et al., 2014](#); [Parks et al., 2014](#); [Walker et al., 2018](#); this study). Wildfires will inevitably occur in these ecosystems in the future, and therefore the main question managers and society must ask is what kind of fire it will be. There may be large social, economic, and ecological costs to allowing fire to once again burn as the keystone process in fire-adapted ecosystems, yet there can be even greater costs to not letting it do so. Efforts to reestablish characteristic fire regimes in fire-adapted communities that have been initiated and promoted by National Park Service policies and personnel should be encouraged, expanded, and supported by additional research as much as possible. Managed wildfires also are a relatively cost-effective means to increase the scale and pace of fire restoration throughout frequent-fire forests wherever they are located (e.g., [North et al., 2015](#)). Continued and expanded use of prescribed fires and managed wildfires will return this central process to fire-adapted ecosystems to enhance their resilience to not only wildfires but also climate change and other disturbances going into the future.

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