# A multi-century perspective on contemporary fire deficit and anomalous fire severity in the southwestern United States

**Emma McClure**

emma_mcclure@nps.gov

National Park Service

**Jonathan Coop**

Western Colorado University

**Christopher Guiterman**

NOAA's National Centers for Environmental Information

**Ellis Margolis**

U.S. Geological Survey Fort Collins Science Center

**Sean Parks**

Aldo Leopold Wilderness Research Institute, Rocky Mountain Research Station, US Forest Service

https://orcid.org/0000-0002-2982-5255

**Article**

**Keywords:** dry conifer forests, fire regime departure, forest management, North American tree-ring fire-scar network, ponderosa pine, prescribed burn, wildfire

**Posted Date:** February 26th, 2024

**DOI:** https://doi.org/10.21203/rs.3.rs-3931790/v1



**Additional Declarations:** There is **NO** Competing Interest.

# Main Manuscript for

# A multi-century perspective on contemporary fire deficit and anomalous fire severity in the southwestern United States

Emma J. McClure[1,2*], Jonathan D. Coop[1], Christopher H. Guiterman[3,4], Ellis Q. Margolis[5], Sean A. Parks[6]

[1]Western Colorado University, Gunnison, CO 81231

[2]National Park Service, Redwood National and State Parks, Orick, CA 95555

[3]Cooperative Institute for Research in Environmental Sciences, CU Boulder, Boulder, CO 80309

[4]NOAA's National Centers for Environmental Information, Boulder, CO 80309

[5]United States Geological Survey, Fort Collins Science Center, New Mexico Landscapes Field Station, Santa Fe, NM 87508

[6]USDA Forest Service, Rocky Mountain Research Station, Aldo Leopold Wilderness Research Institute, Missoula, MT 59812

*Corresponding author.

**Email:** emma_mcclure@nps.gov

**Author Contributions:** Conceptualization: J.D.C, C.H.G., E.Q.M., E. J. Mc., and S.A.P. Methodology: J.D.C, C.H.G., E.Q.M., E. J. Mc., and S.A.P. Software: J.D.C, C.H.G., E.Q.M., E. J. Mc., and S.A.P. Formal analysis: C.H.G., E. J. Mc., and S.A.P. Visualization: C.H.G. and E. J. Mc. Writing—original draft: E. J. Mc. Writing—review and editing: J.D.C, C.H.G., E.Q.M., E. J. Mc., and S.A.P. Data curation: J.D.C, C.H.G., E.Q.M., E. J. Mc., and S.A.P.

**Competing Interest Statement:** The authors have no competing interests to disclose.

**Classification:** Biological Sciences; Ecology

**Keywords:** dry conifer forests, fire regime departure, forest management, North American tree-ring fire-scar network, ponderosa pine, prescribed burn, wildfire

**This PDF file includes:**

Main Text
Figures 1 to 4
Table 1

## Abstract

Wildfires in the southwestern United States are increasingly frequent and severe, but whether these trends are beyond historical norms remains contested. We combine dendroecological records, satellite-derived burn severity, and field measures of tree mortality to compare historical (1700-1880) and contemporary (1985-2020) fire regimes at 406 tree-ring fire-scar sites in Arizona and New Mexico. Contemporary fire frequency is over 80% lower than historical levels. Before 1880, fires averaged 0.87 fires/decade (every 11.4 years), followed by over a century without fire at most sites. Since 1985, the same sites averaged only 0.17 fires/decade (every 58.8 years).

Fire severity, however, has increased. At 42% of our sites, centuries-old fire-scarred trees recently experienced anonymously lethal fire severity. Suppressed wildfires tended to burn more severely than prescribed burns and managed “fire use” wildfires. These findings support the expanded use of low-severity prescribed and managed fire to restore ecosystem function and resilience to southwestern dry-conifer forests.

**Main Text**

# Introduction

Changing fire regimes pose mounting challenges to both natural and human systems. Intensifying wildfire activity associated with climate change is exerting increasing pressure on a wide range of forest ecosystems globally[1,2,3]. Where fire activity exceeds the range of conditions for which species are adapted, fires can drive local population extirpations and trigger ecosystem shifts[4]. Increasing fire ***frequency*** can significantly alter population vital rates, in particular when the time between successive fires is not sufficient for recovering tree species to achieve reproductive maturity[5,6]. Increasing fire ***severity*** can drive anomalous tree mortality, even for the most fire-tolerant species and individuals (e.g., giant sequoia [*Sequoiadendron giganteum*])[7], reducing propagule availability in extensive and homogenous high-severity patches and impeding recovery[8,9]. However, reduction or elimination of fire activity—often imparted by human fire exclusion—can also result in major ecological changes. For example, the loss of fire due to elimination of cultural burning practices, land use changes such as grazing that limit fire spread, and fire suppression has increased woody plants, altered species composition, and homogenized forest communities across North America and elsewhere[10,11,12]. Accordingly, understanding the extent and direction of contemporary fire regime departures from historical norms can provide critical insight into patterns of ecosystem changes and vulnerabilities, and inform conservation and management interventions.

Dry conifer forests in the American Southwest dominated by ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) are particularly vulnerable to the combined effects of altered fire regimes and climate change. A large body of evidence demonstrates that some proportion of these forests was historically characterized by frequent, low- to moderate-severity fires associated with prolonged dry seasons, profuse grassy fuels, and abundant ignitions from lightning and Indigenous land stewardship[13,14,15,16]. These fire regimes were disrupted by Euro-American colonization in the late 19th C, producing an enduring fire deficit[17] and initiating fuel build ups[18]. However, recent decades have been marked by a rapid return of fire to many dry southwestern forest landscapes. Increased fuel availability and continuity, combined with increasingly long fire seasons and dry fuels[19,20] and abundant human ignitions[21], have dramatically escalated wildfire activity including area of high-severity fire[22,23,24]. Compounded by warming and drying post-fire conditions, extensive high-severity burns are constraining regeneration by wind-dispersed conifers[8,25,26,27,28,29] and in some areas catalyzing persistent conversion to various non-forest vegetation types[30,31].

Although these processes are generally well understood, the extent to which contemporary fire regimes differ from historical norms in dry southwestern forests remains the subject of continued debate, leading to divergent interpretations with important implications for forest management[32,33,34,35]. One viewpoint has relied on data from General Land Office (GLO) surveys to reconstruct late 19th century stand structure based upon tree age-size relationships and forest density extrapolations, which were then interpreted to reconstruct fire severity[36]. These authors suggest that high-severity fire may have in fact been common in these systems long before the prominent ecological changes of the last century, and thus modern severe wildfires fall within historical norms[35]. If severe contemporary wildfires in fact represent historically normal events, then management efforts intended to restore dry forest ecosystem function, including thinning of small-diameter trees, fuels reduction treatments, and low-severity prescribed burn, would appear to be misguided. The other viewpoint holds that modern, high-severity wildfires drastically exceed historical fire regime norms[33]. This view is based upon large increases in tree density coincident with fire exclusion over the last century, documented through comparisons between historical reconstructions and modern measurements of forest structure[37,10,38], a paucity of documentary and scientific evidence of high-severity fire occurring in dry conifer forests in the 18th and 19th centuries[39,33], and abundant evidence of frequent, low-severity fires in the historical tree-ring fire-scar record[40]. Under this view, management efforts to reduce tree densities and fire severity are in fact essential to sustaining dry forest systems, particularly as climate warming increases exposure to severe fire-induced loss. Accordingly, an improved quantification of the differences between historical and contemporary fire frequency and severity would be useful in assessing forest vulnerabilities and setting management priorities[41], particularly as severe fire-driven vegetation conversions appear to be increasing over recent decades.

Two distinct lines of evidence—dendroecology and remote sensing—are widely used to characterize patterns of historical and contemporary fire activity, respectively. Tree-ring analyses of fire-scarred trees have been used as a primary means of characterizing low-severity historical fire regimes[42,43]. Where low-severity fires injure but do not kill trees, they leave behind scars preserved in growth rings that can be dated to the year and often the season of fire occurrence. While such records cannot be developed from areas where severe fires kill trees and consume the preceding record, comparisons between fire-scar records and documentary evidence demonstrate that fire scars provide direct evidence of low-severity fire regimes at sites and across landscapes[44,45]. In dry conifer forests of the southwestern US, hundreds of sites, incorporating thousands of fire-scarred trees, recorded frequent, low- to moderate-severity fire continuously for centuries prior to European settlement[16,40]. In contrast, contemporary fire regimes are measured primarily via satellite observations (particularly over the Landsat period of record, 1985-present) and on-the-ground field data collection[46,47]. Field-verified, satellite-derived metrics covering the full range of burn severity have facilitated analyses of recent burning trends in the modern era[48,49,24]. However, these two types of data—tree-ring fire-scar records and satellite observations—have not yet been brought together to compare attributes of historical and contemporary fire regimes. While fire-scar sites provide only a limited perspective on the area of historical high-severity fire, they stand as witness to centuries of recurrent low-severity fires, and where they have recently burned, provide a direct opportunity to assess changes in frequency and severity during the contemporary period.

The purpose of our study is to bring together tree-ring records, satellite imagery, and field measures to ask: ***at sites where tree-ring records demonstrate that trees historically survived frequent low-severity fire for centuries, are contemporary fires burning differently?*** Specifically, we (1) quantify and contrast historical (1700-1880) and contemporary (1985-2020) fire frequency at 406 tree-ring fire-scar fire history sites (Fig. 1) to assess whether recent fires are occurring at frequencies that approach historical levels. Next, we (2) quantify and contrast historical and contemporary fire severity using satellite-measured burn severity and field data on tree mortality to derive a binary metric of severity: unlikely vs. likely mature tree mortality. Because historical fires left many surviving fire-scarred trees over decades and centuries prior to 1880, with some trees surviving and recording as many as 41 fires over 180 years, it follows that those fires were characterized by predominantly low probability of tree mortality. We compare observed and satellite-measured fire-related tree mortality at these sites to inferred historical norms. Finally, given the strong rationale for reducing fire severity to sustain southwestern forests under climate warming[25], we were also interested in assessing the extent to which varying contemporary fire management strategies (prescribed burn, managed wildfire, and full suppression) might be associated with different fire outcomes. Taken together, our findings are intended to bear directly on the appropriateness, or lack thereof, of prescribed burning and managed wildfire to promote low-severity burning of southwestern dry forest ecosystems.

# Results

## *Historical and contemporary fire frequency*

Trends in multidecadal fire activity across the full period of record (1700-2020) show a pronounced decline from historically frequent to near-zero fire for much of the 20th century, with recent increases yet to equal historical fire occurrence at regional and landscape scales (Fig. 2). Based on averages of fires per decade for all fire history sites, the mean site fire frequency was 0.87 fires per decade from 1700 to 1880. This rate dropped to <0.1 fires per decade between 1880-1985 (equivalent to >100-year fire interval) and rose to an average of 0.17 fires per decade since 2000, though with considerable regional variation (Fig. 2). A comparison of mean fires per decade between the historical and contemporary periods indicates that 14.2% of sites have returned to or exceeded historical frequency since 1985, and 85.8% of sites are burning less frequently than historically.

During the historical period, across all sites, 5150 individual trees recorded an average of 6.6 fires per tree (maximum 41; Fig. S3), with 1521 trees recording at least nine fires. Sites averaged 14.2 fires (maximum 78; Fig. S3), with 102 sites recording at least 17 historical fires. During the 36-year contemporary period (1985-2020), 206 sites (50.7%) burned at least one time; 99 sites (24.2%) burned twice or more within that time frame, including 17 sites burning three or more times. One site in the Gila Mountains in New Mexico burned in five fires since 1985 and one site in the Rincon Mountains burned seven times. Contemporary fire occurrence varied by geographic area of fire history sites – generally higher in our field-sampled mountain ranges than elsewhere in the study area – with 55% of sites in the Jemez Mountains and 100% of sites burned in both the Kaibab Plateau and Chiricahua Mountains (Table 1). We were able to classify the type of fire and thus its management strategy for 89 (of 102) contemporary fires. Of those, 56 (62.9%) were suppressed wildfires (burning 243 fire history sites, including reburns), whereas 24 (27.0%) were wildland fire use fires (burning 51 sites) and 9 (10.1%) were prescribed burns (burning 15 sites).

## *Historical and contemporary fire severity*

We conducted field surveys at 74 distinct fire history sites burned in contemporary fires, in which we were able to locate at least one fire-scar sampled tree, stump, or log at 25 sites (33.8%); we located multiple sampled trees at four sites, all in the Rincon Mountains (see Fig. S1 for images of fire-scarred stumps and trees). We found a range of fire effects at the field plots. Those with high burn severity as measured by CBI generally showed high tree mortality (Fig. 3). Overall, 14.9% of plots were devoid of live trees of any size, and 31.1% had no live overstory trees. These areas were often characterized by high dead and down fuel loads, especially in larger size classes, from the fire-killed forest. We observed that modern fires at many of sites had in fact completely consumed fire scarred material sampled by earlier researchers. At the low end of the severity gradient, plots contained intact tree canopies of mixed size- and age-classes, often displaying relatively light fuel loads, particularly on the Kaibab Plateau and in the Rincon Mountains, where the use of prescribed burn has been most abundant.

To compare contemporary fire severity with the historical, low-severity fires that scarred but did not kill fire-recording trees, we developed a logistic regression model predicting tree mortality from contemporary satellite-measured CBI. Our model (Fig. S2) identified the 0.5 probability of overstory tree mortality (mortality more likely than not) at a modeled CBI threshold of 1.61. This value aligns closely with the 1.73 CBI threshold of 0.5 probability of ponderosa pine tree mortality in the Forest Inventory and Analysis data (FIA)[50] from Arizona and New Mexico identified by Woolman et al.[51].

Applying our 1.61 CBI threshold, 42.4% of the 206 sites that burned in the contemporary period had first-entry fires in which overstory tree mortality was more likely than not.

We assessed associations between fire management strategy and contemporary burn severity, finding that 14.8% of sites in wildland fire use, 15.4% in prescribed burns, and 53.4% of sites in suppressed wildfires burned at severity levels exceeding 1.61 CBI (Fig. 4). Fire history sites burned in wildland fire use and prescribed burns burned significantly less severely than sites burned in suppressed wildfires (-0.9 and -0.8 units of AIC, respectively, linear mixed-effects model $p<0.001$, N = 302). Proportion of sites burning above the threshold for likely tree mortality varied by geographic area, from 0% on the Kaibab Plateau to 73.1% in the Santa Catalina Mountains (Table 1). In contrast with the first contemporary fire, 35.7% of sites burned by second-entry fires and 0% of sites burned by third-entry fires exceeded the likely mortality threshold (these sites included areas that sustained live trees but also treeless areas where canopy trees had been entirely killed by the first contemporary fire).

# Discussion

In this study, we brought together two large-scale, long-term datasets to quantify fire regime changes over the last three centuries in dry conifer forests of the southwestern US. At hundreds of tree-ring fire history sites, fire regimes were historically dominated by frequent and low-severity fires that effectively ended in the late 1800s. While the collapse of this fire regime and resulting changes to forests is well documented[13,14,16], the extent to which recent increases in fire activity might fall within historical norms has not been rigorously analyzed. Here, we show that contemporary patterns of burning in dry conifer forests bear little resemblance to historical fire regimes in two important ways. First, despite rapid increases in fire activity observed over the last several decades[52,20], fires are still burning far less frequently now than they were historically. Second, where stands of individual trees historically survived many fires over centuries, recent fires are anomalously lethal to trees.

Although recent climate-driven increases in fire activity[19] are evident across our study area, our findings highlight that fire is still very infrequent relative to historical norms. Based on ten-year moving averages of fires per decade, fire since 1985 is over 80% less common as it was historically, with fires burning at a rate of 0.17 per decade in the 21st century vs. 0.87 per decade over most of the 18th and 19th centuries. Many studies have demonstrated major decreases in modern fire frequency relative to historical ranges of variability in fire-adapted forests of the western US[53,54]. The causes of the decline in fire, including the removal of fine fuels following the onset of livestock grazing, cessation of Indigenous burning, and later direct fire suppression, are well understood[55,56]. Historical fire regimes across our study sites effectively ended in the late 19th century. Contemporary fire occurrence is approaching historical norms in some areas, with 15% of sites burning as frequently or more frequently than historically (though burn severity at these sites may be more severe, as described below). Notably, the return of fire occurred relatively early in the Rincon Mountains in southern Arizona, where progressive fire management was initiated in the 1970s[57]. Still, most sites are burning far less often than historically: as of 2020, half of our fire history sites had yet to burn in the contemporary period, attesting to a still growing fire deficit[17,58].

At fire history sites where fire has returned, contemporary burn severity is substantially higher than that recorded in the tree ring record. Of tree-ring fire-scar sites that burned, nearly half (42%) experienced fire effects that are more likely than not to be lethal to mature trees. While such events may have occurred occasionally at our sites over the historical period, they cannot have been as common as they are now. Fire history sites averaged 14.2 fires historically and individual trees recorded on average 6.6 fires. If these fires burned at moderate to high severities (CBI > 1.61), trees would have a 50% probability of mortality in each fire, and the chance that a tree at such a site survives more than six fires is less than 1% ($0.5^{6.6} = 0.01$). And yet regionally, more than 5000 sampled trees survived to record the historical fires. Our empirical assessment that contemporary severity is much higher than historical norms is consistent with the interpretations of many previous studies[33,34], and adds to a growing body of evidence that contemporary fires in southwestern forests have recently become more severe, not just in recent decades, but also in sharp contrast to events over recent centuries or longer[59,60,61,23,62].

Sites burned in suppressed wildfires exhibited higher severity than those burned in other fire types, likely related to the conditions under which those different fire types were burning, with prescribed and managed fires occurring under less extreme fire weather. Both prescribed burn and wildfires managed for resource benefit have been shown to moderate the severity of subsequent wildfires and sustain dry forest ecosystem function[63,64,65,66,67]. Second-entry fires also burned less severely than the first contemporary wildfire in our study, in accord with the findings of prior research[68,65].

### *Conclusions and management implications*

Our findings add to a growing body of evidence that contemporary fire regimes in southwestern dry conifer forests are anomalously infrequent and severe, supporting management interventions that restore the historical role of fire. These findings are in direct contrast to the assertion that burning patterns today are within the range of variability that occurred prior to Euro-American colonization[35,36], an assertion that has been largely invalidated due to methodological inaccuracies and unsupported logical inferences[44,45,69,33,70,34,61,71]. By design, our study is not intended to assess the historical prevalence of high-severity fire in these systems, but our results clearly demonstrate that hundreds of fire history sites that burned frequently at low severity are now burning far less often, and far more severely, than they have for centuries.

The clear management implications of our findings are the need to 1) **promote low-severity fire**, and 2) **avert high-severity fire** in settings where fire regimes have shifted. In ecosystems with markedly altered structure, function, and disturbance regimes, intentional and informed management strategies are essential[41]. Our findings support the increased use of prescribed burn and managed wildfire, burning under moderate climatic and weather conditions, to restore the historical role of fire. Where the risks of severe fire are particularly high (e.g., in the wildland urban interface or watersheds that provide critical surface water supplies), antecedent thinning and fuels reduction treatments may be essential prior to the restoration of low-severity fire[18,72]. Abundant previous research has demonstrated how prescribed burn and managed wildfire can achieve stated objectives[73], emulate historical forest structure[74], and increase forest resilience to future fire[68]. Unfortunately, prescribed fire in our study area and timeframe was even less prevalent than managed wildfire, and apart from the southeastern US, the application of prescribed burning has been decreasing in the past two decades[75]. Restoration of Indigenous fire stewardship and integration of diverse stakeholder collaboration offer two promising paths forward to expanding the ecological and social benefits of fire in sustaining ecosystem values[76,41,72,77]. However, regardless of management intensity, in the presence of warmer, drier, and more flammable future conditions, current approaches may be inadequate to maintain structure and function of dry forest ecosystems in the southwestern US[78], highlighting the need to develop, test, and apply novel management strategies and tactics.

## Materials and Methods

### *Study area*

Our southwestern US study area comprises the states of Arizona and New Mexico (Fig. 1). We focused on these states because of the recent increase in high-severity fire in this region[24,22] and the availability of extensive tree-ring fire-scar records available through the North American tree-ring fire-scar network v1.1 (NAFSN)[40]. The climate of the study area is semi-arid, with bimodal precipitation peaking in winter (December to February) and during the summer monsoons (July to September), when portions of the region can receive > 50% of their annual precipitation[79]. Fire history sites used in this study range from 1552 to 3105 m elevation. Dry forests are dominated by ponderosa pine and Douglas-fir and may also contain Arizona pine (*Pinus arizonica*), Southwestern white pine (*Pinus strobiformis*), white fir (*Abies concolor*), quaking aspen (*Populus tremuloides*), and rarely, Engelmann spruce (*Picea engelmannii),* piñon pine (*Pinus edulis* and *P. monophylla*), juniper (*Juniperus deppeana, J. osteosperma,* and *J. scopulorum*), and oak (*Quercus spp*., especially *Q. gambelii*).

### *Historical fire records*

At the time of our analysis (2021), the NAFSN contained 2,562 fire-scar sites, including 600 in Arizona and New Mexico (https://doi.org/10.5066/P9PT90QX). Of the 600 sites, tree-level fire-scar data with sufficient sample size (at least three trees, recording at least four fires between 1700-1880) were available for 406 sites, which we refer to hereafter as our fire history sites (Fig. 1; Table S1). To generate a single composite time series of fire occurrence at each site, we used the burnr package (v. 0.6.1)[80] in the R statistical platform (v. 4.1.3)[81], applying filters for minimum number of trees recording (two), minimum number of trees scarred (two), and proportion of trees scarred (0.10).

### *Contemporary fire records*

We obtained fire perimeters from the Monitoring Trends in Burn Severity (MTBS) program[46,82], which includes all fires over 1,000 acres (405 ha) in our southwestern study area occurring from 1985 to 2019. To capture fires with acreage less than 1,000 and/or occurring in 2020, we acquired fire perimeters from the National Interagency Fire Center (NIFC)[83]. We intersected these fires with fire history site locations (Fig. 1), identifying 102 contemporary fires intersecting 206 of 406 fire history sites. Fires ranged in size from 25 to over 538,000 acres.

For each fire, we generated a gridded burn severity map, represented as modeled Composite Burn Index (CBI), using Google Earth Engine[84] and code developed and distributed by Parks et al.[85]. CBI, which scales from 0 to 3, was developed as a field protocol for validating satellite-derived burn severity one year after fire[86] (Key & Benson, 2006), and can be modeled from satellite data[85]. We used modeled CBI as opposed to delta Normalized Burn Ratio (dNBR)[46,47], to improve comparability across fires, sites, and years[85]. Satellite-measured CBI better predicts overstory ponderosa pine tree mortality in the southwestern US than dNBR[51]. Subsequently, we extracted CBI values at each fire history site for each overlapping recent burn.

### *Field sampling*

To quantify tree mortality from contemporary fires, we sampled fire effects at a subset of fire history sites across a gradient of contemporary burn severity and fire management strategies. Data collection focused on six key geographic areas (Fig. 1, Table 1) where networks of fire history sites had been established prior to wildfires occurring over the past ten years (2011-2020): the Jemez[87,88,89,90], Rincon[45], Santa Catalina[91,92], Pinaleño[61], and Chiricahua Mountains[93,94,95,96,97,98], and the Kaibab Plateau[99]. In the Jemez Mountains, sites are in Bandelier National Monument (including the Bandelier Wilderness), the Valles Caldera National Preserve, and the Santa Fe National Forest. The Rincon Mountain sites are mostly located in the Saguaro Wilderness (within Saguaro National Park), and about half of the sites on the Kaibab Plateau are in proposed wilderness in Grand Canyon National Park; the remaining sites sampled in Arizona are within the Coronado and Kaibab National Forests.

We relocated each fire history site and established a 10-m radius plot. If we found a tree or stump sampled in the original fire history data collection, the plot was centered at its location (see Fig. S1); if a sampled tree was not located (typically due to high severity fire effects), we centered the plot at the coordinates provided by the original researcher. Where we found multiple sampled trees or stumps at least 20 m apart, we installed a plot at each, for a total of 91 plots at 74 distinct fire history sites. For all trees in the field plots, we recorded tree diameter at breast height (dbh), species, and status (live or dead). We measured diameter and assigned species for downed logs and recorded an overall count of trees both live and dead, standing and down. A qualitative description of site conditions and photographic documentation (Fig. 3; Fig. S1) completed our site characterization.

### *Data analysis*

Our first objective was to compare historical and contemporary fire frequency. Although some fire history sites recorded fires in the 1400s and earlier, a consistent record across all sites was not interpretable until around 1700. To generate a continuous time series of fire occurrence from both the site composites generated from site-level fire history data and contemporary fire dates from MTBS and NIFC, we cut off the dendroecological record at the year 1984, from which point (1985-2020) we appended the modern, satellite-derived fire record. Because the contemporary period was only 35 years and included relatively few fires (and even fewer reburns necessary to produce a fire intervals) compared with 180 years for the historical period (1700-1880), a direct comparison of mean fire return interval at individual fire history sites was infeasible. Instead, we calculated 10-year moving averages of fires at each site from 1700 through 2020, which we used to examine fire frequency trends across the entire time frame of our study.

Our second objective was to compare historical and contemporary fire severity. To compare fire effects derived from two distinct types of evidence (tree-ring fire scars and satellites), we classified contemporary severity as a binary categorical variable related to mature tree mortality vs. survival, as follows: 1) **unlikely tree mortality**; consistent with a tree surviving to record fire scars, and 2) **likely tree mortality**, wherein trees are killed and thus would not record fire. To develop this classification, we generated a logistic regression model to identify the threshold above which probability of overstory tree mortality exceeds 0.5—in other words, trees are more likely than not to be killed. Using this 0.5 probability of mortality threshold avoids biasing our contemporary severity classification toward either a more conservative or liberal interpretation of severity. Our implicit assumption is that, for living trees at fire history sites to have recorded multiple short-interval fires without being killed, those fires must have predominantly burned below this threshold. To generate our 0.5 probability of mortality threshold (Fig. S2), we used site-specific modeled CBI to predict field-measured overstory (dbh ≥ 12.7 cm) tree mortality from 834 trees at 87 plots after filtering for size and our assessment of whether, if dead, they were killed by the most recent fire. Our use of trees ≥ 12.7-cm dbh to model the relationship between burn severity and tree survival is consistent with prior studies[51], but also generally supported by the historical ages at which trees recorded fire scars (and thus demonstrate survival through historical fires), as follows. The median number of years between the pith ring (center) of the tree and the first fire scar across our data (n = 2215 trees with pith) was 47 years, which corresponds well with 35 years reported by Brown et al.[100] and 52 years reported by Yocom and Fulé[101]. Given that most fire scar samples are collected at <30 cm above the ground, the years between pith and first fire scar may underestimate tree age by up to 5 years[102,90]; accordingly, we estimate the median tree age at first fire scar in our study to range between 47 and 52 years. Established age-size relationships for ponderosa pine in Arizona (age = $10.4 * \mathrm{dbh(cm)}^{0.66}$)[103], the most frequently-sampled tree species, yield size estimates of 10-12 cm dbh for 47-52-year old ponderosa pine trees. Thus, we expect that historical fires were generally not lethal for trees ≥ 12.7-cm dbh for these trees to have recorded multiple fires over 5-10 decades beginning at ages between 47 and 52 years. By species, trees included in this analysis were 29.1% ponderosa pine, 27.2% Douglas fir, 14.7% Southwestern white pine, 12.1% white fir, 9.1% Arizona pine, 4.9% quaking aspen, 1.9% Engelmann spruce, and less than 1% of other species. To assess the robustness of our results to the tree dbh cutoff used in this model, we also examined relationships between tree survival and CBI using larger cutoff values of 25.4 and 38.1 cm. We present both the satellite-derived modeled CBI values of contemporary fires as well as the number of fires occurring above or below the tree mortality threshold.

To assess associations between contemporary burn severity and management strategy, we classified fires by management type (suppressed wildfire, wildland fire use, and prescribed burn). This could not be determined for 13 fires (12.7%). For the wildland fire use category, we included wildfires listed as “resource benefit,” “prescribed natural fire,” or “managed for multiple uses,” depending on source and date as these terms were variously applied over the study period to refer to the same type of fire. We tested for differences in burn severity (CBI) between sites burned in suppressed wildfire vs. wildland fire use and prescribed burn using a linear mixed-effects model with fire management strategy as a fixed effect and fire identity as a random effect. All analyses and data visualization were performed in R (v. 4.1.3)[81].

# Acknowledgments

The authors would like to thank Melanie Armstrong, Ashley Woolman, Chloe Beaupré, and Petar Simic for assistance in project conceptualization and manuscript review. Thanks also to Mason Kunkel for excellent assistance in field data collection. For providing additional data and aiding with field research logistics we would like to thank Craig Allen; Peter Fulé at Northern Arizona University; Calvin Farris with the National Park Service, Pacific West Region; Bob Parmenter at Valles Caldera National Park and Preserve; Ronda Newton, Li Brannfors, and the Fire Effects crew at Grand Canyon National Park; Perry Grissom and Adam Springer at Saguaro National Park; and JoAnn Blalack at Chiricahua National Monument. We wish to express our gratitude to all contributors to the North American tree-ring fire-scar network[40] and to the United States Forest Service for funding this research. This research was supported in part by the USDA Forest Service, Rocky Mountain Research Station, Aldo Leopold Wilderness Research Institute. Additional support provided by the USGS Climate R&D program. The findings and conclusions in this publication are those of the authors and should not be construed to represent any official USDA or U.S. Government determination or policy.
CHG would like to acknowledge the following funding sources: Ecological Restoration Institute at Northern Arizona University via a grant (Award 21-DG- 11030000-019) from the U.S. Forest Service. NOAA cooperative agreements NA17OAR4320101 and NA22OAR4320151.

# References

1. A. Burrell, E. Kukavskaya, R. Baxter, Q. Sun, K. Barrett. Post-fire Recruitment Failure as a Driver of Forest to Non-forest Ecosystem Shifts in Boreal Regions. In J. G. Canadell & R. B. Jackson (Eds.), Ecosystem Collapse and Climate Change (pp. 69–100). Springer International Publishing (2021).
2. J. D. Coop, et al. Wildfire-driven forest conversion in western North American landscapes. *BioScience*, 70(8), 659–673 (2020).
3. A. Karavani et al. Fire-induced deforestation in drought-prone Mediterranean forests: Drivers and unknowns from leaves to communities. *Ecol. Monogr.*, 88(2), 141–169 (2018).
4. D. A. Falk et al. Mechanisms of forest resilience. *For. Ecol. Manag.*, 512, 120129 (2022).
5. N. J. Enright et al. Interval squeeze: Altered fire regimes and demographic responses interact to threaten woody species persistence as climate changes. *Frontiers in Ecology and the Environment,* 13(5), 265–272 (2015).
6. M. G. Turner, K. H. Braziunas, W. D. Hansen, B. J. Harvey. Short-interval severe fire erodes the resilience of subalpine lodgepole pine forests. *Proc. Natl Acad. Sci.*(2019).

7. K. L. Shive, A. Wuenschel, L. J. Hardlund, S. Morris, M. D. Meyer, S. M. Hood. Ancient trees and modern wildfires: Declining resilience to wildfire in the highly fire-adapted giant sequoia. *For. Ecol. Manag.*, 511, 120110 (2022).
8. M. E. Chambers, P. J. Fornwalt, S. L. Malone, M. A. Battaglia. Patterns of conifer regeneration following high severity wildfire in ponderosa pine – dominated forests of the Colorado Front Range. *For. Ecol. Manag.*, 378, 57–67 (2016).
9. B. J. Harvey, D. C. Donato, M. G. Turner. High and dry: Post-fire tree seedling establishment in subalpine forests decreases with post-fire drought and large stand-replacing burn patches. *Global Ecology and Biogeography*, 25(6), 655–669 (2016).
10. W. W. Covington, M. M. Moore. Southwestern Ponderosa Forest Structure: Changes Since Euro-American Settlement. *Journal of Forestry*, 92(1), 39–47 (1994).
11. D. Li, D. Waller. Drivers of observed biotic homogenization in pine barrens of central Wisconsin. *Ecology*, 96(4), 1030–1041 (2015).
12. Mariani, M. et al. Disruption of cultural burning promotes shrub encroachment and unprecedented wildfires. *Frontiers in Ecology and the Environment*, 20(5), 292–300 (2022).
13. C. D. Allen. Lots of lightning and plenty of people: An ecological history of fire in the upland Southwest (Vol. 143). Island Press, Washington, DC (2002).
14. P. Z. Fulé, W. W. Covington, M. M. Moore. Determining reference conditions for ecosystem management of Southwestern ponderosa pine forests. *Ecol. Appl.,* 7(3), 895–908. JSTOR (1997).
15. C. I. Roos, N. C. Laluk, W. Reitze, O. K. Davis. Stratigraphic evidence for culturally variable Indigenous fire regimes in ponderosa pine forests of the Mogollon Rim area, east-central Arizona*. Quaternary Research*, 113, 69–86 (2023).
16. T. W. Swetnam, C. H. Baisan. Historical fire regime patterns in the Southwestern United States since AD 1700. Fire Effects in Southwestern Forests: Proceedings of the Second La Mesa Fire Symposium, Los Alamos, New Mexico, March 29-31, 1994.- Fort Collins, Colo. (USA): USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, [1996]., 11–32 (1996).
17. Marlon, J. R. et al. Long-term perspective on wildfires in the western USA. *Proc. Natl Acad. Sci.*, 109(9), E535–E543 (2012).
18. C. D. Allen, M. Savage, D. A. Falk, K. F. Suckling, T. W. Swetnam, T. Schulke, P. B. Stacey, P. Morgan, M. Hoffman, J. T. Klingel. Ecological restoration of Southwestern ponderosa pine ecosystems: A broad perspective. *Ecol. Appl.*, 12(5), 1418–1433 (2002).
19. J. T. Abatzoglou, A. P. Williams. Impact of anthropogenic climate change on wildfire across western US forests. *Proc. Natl Acad. Sci.*, 113(42), 11770–11775 (2016).
20. Westerling, A. L., H. G. Hidalgo, D. R. Cayan, T. W. Swetnam. Warming and earlier spring increase western U.S. forest wildfire activity. *Science*, 313(5789), 940–943 (2006).
21. J. K. Balch, B. A. Bradley, J. T. Abatzoglou, R. C. Nagy, E. J. Fusco, A. L. Mahood. Human-started wildfires expand the fire niche across the United States. *Proc. Natl Acad. Sci.,* 114(11), 2946–2951 (2017).
22. S. A. Parks, J. T. Abatzoglou. Warmer and drier fire seasons contribute to increases in area burned at high severity in western US forests from 1985 to 2017. *Geophys. Res. Lett.*, 47(22), e2020GL089858 (2020).
23. S. A. Parks, L. M. Holsinger, K. Blankenship, G. K. Dillon, S. A. Goeking, R. Swaty. Contemporary wildfires are more severe compared to the historical reference period in western US dry conifer forests. *For. Ecol. Manag.*, 544, 121232 (2023).
24. M. P. Singleton, A. E. Thode, A. J. Sánchez Meador, J. M. Iniguez. Increasing trends in high-severity fire in the southwestern USA from 1984 to 2015. *For. Ecol. Manag.*, 433, 709–719 (2019).
25. Davis, K. T., et al. (2023). Reduced fire severity offers near-term buffer to climate-driven declines in conifer resilience across the western United States. *Proc. Natl Acad. Sci.*, 120(11), e2208120120.

26. K. C. Rodman et al. A changing climate is snuffing out post-fire recovery in montane forests. *Global Ecology and Biogeography*, 1–14 (2020).
27. C. Haffey, T. D. Sisk, C. D. Allen, A. E. Thode, E. Q. Margolis. Limits to ponderosa pine regeneration following large high-severity forest fires in the United States southwest. *Fire Ecology*, 14(1), 143–163 (2018).
28. M. Savage, J. N. Mast. How resilient are southwestern ponderosa pine forests after crown fires? Canadian Journal of Forest Research, 35(4), 967–977 (2005).
29. C. S. Stevens-Rumann et al. Evidence for declining forest resilience to wildfires under climate change. *Ecology Letters*, 21(2), 243–252 (2018).
30. J. D. Coop. Postfire futures in southwestern forests: Climate and landscape influences on trajectories of recovery and conversion. *Ecol. Appl.*, 33(1), e2725 (2023).
31. C. H. Guiterman et al. Vegetation type conversion in the US Southwest: Frontline observations and management responses. *Fire Ecology*, 18(1), 6 (2022).
32. W. L. Baker, C. T. Hanson, M. A. Williams, D. A. DellaSala. Countering Omitted Evidence of Variable Historical Forests and Fire Regime in Western USA Dry Forests: The Low-Severity-Fire Model Rejected. *Fire*, 6(4), 4 (2023).
33. P. Z. Fulé et al. Unsupported inferences of high-severity fire in historical dry forests of the western United States: Response to Williams and Baker. *Global Ecology and Biogeography*, 23(7), 825–830 (2014).
34. R. K. Hagmann et al. Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests. *Ecol. Appl.* (2021).
35. M.A. Williams, W.L. Baker. Spatially extensive reconstructions show variable-severity fire and heterogeneous structure in historical western United States dry forests. *Global Ecology and Biogeography*, 21(10), 1042–1052 (2012).
36. M.A. Williams, W.L. Baker. Testing the accuracy of new methods for reconstructing historical structure of forest landscapes using GLO survey data. *Ecol. Monogr.*, 81, 63–88 (2011).
37. D. M. J. S. Bowman, R. J. W. Brienen, E. Gloor, O. L. Phillips, L. D. Prior. Detecting trends in tree growth: Not so simple. *Trends in Plant Science*, 18(1), 11–17 (2013).
38. T. L. Swetnam, P. M. Brown. Comparing selected fire regime condition class (FRCC) and LANDFIRE vegetation model results with tree-ring data. *Int. J. Wildland Fire*, 19(1), 1 (2010).
39. C. F. Cooper. Changes in Vegetation, Structure, and Growth of Southwestern Pine Forests since White Settlement. Ecol. Monogr., 30(2), 129–164 (1960).
40. E. Q. Margolis et al. The North American tree-ring fire-scar network. *Ecosphere*, 13(7), e4159 (2022).
41. P. F. Hessburg, Prichard, S. J., R. K. Hagmann, N. A. Povak, F. K. Lake, Wildfire and climate change adaptation of western North American forests: A case for intentional management. *Ecol. Appl.* (2021).
42. G. L. Harley et al. Advancing dendrochronological studies of fire in the United States. *Fire*, 1(1), 11 (2018).
43. J. H. Dieterich, T. W. Swetnam. Dendrochronology of a fire scarred ponderosa pine. *For Sci* 30:238–247 (1984).
44. B. M. Collins, S. L. Stephens. Fire scarring patterns in Sierra Nevada wilderness areas burned by multiple Wildland fire use fires. *Fire Ecology*, 3(2), 53–67 (2007a).
45. C. A. Farris, C. H. Baisan, D. A. Falk, S. R. Yool, T. W. Swetnam. Spatial and temporal corroboration of a fire-scar-based fire history in a frequently burned ponderosa pine forest. *Ecol. Appl.*, 20(6), 1598–1614 (2010).
46. J. Eidenshink, B. Schwind, K. Brewer, Z.-L. Zhu, B. Quayle, S. Howard. A project for monitoring trends in burn severity. *Fire Ecology*, 3(1), 3–21 (2007).
47. C. H. Key. Ecological and sampling constraints on defining landscape fire severity. *Fire Ecology*, 2(2), 34–59 (2006).
48. G. K. Dillon et al. Both topography and climate affected forest and woodland burn severity in two regions of the western US, 1984 to 2006. *Ecosphere*, 2(12), art130 (2011).

49. S. E. Mueller, A. E. Thode, E. Q. Margolis, L. L. Yocom, J. D. Young, J. M. Iniguez. Climate relationships with increasing wildfire in the southwestern US from 1984 to 2015. *For. Ecol. Manag.*, 460, 1–14 (2020).
50. S. W. Woudenberg, B. L. Conkling, B. M. O'Connell, E. B. LaPoint, J. A. Turner, K. L. Waddell. The Forest Inventory and Analysis Database: Database description and users manual version 4.0 for Phase 2 (RMRS-GTR-245; p. RMRS-GTR-245). U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. https://doi.org/10.2737/RMRS-GTR-245 (2010).
51. A. M. Woolman, J. D. Coop, J. D. Shaw, J. DeMarco. Extent of recent fire-induced losses of ponderosa pine forests of Arizona and New Mexico, USA. *For. Ecol. Manag.,* 520, 120381 (2022).
52. P. E. Dennison, S. C. Brewer, J. D. Arnold, M. A. Moritz. Large wildfire trends in the western United States, 1984–2011. *Geophys. Res. Lett.*, 41(8), 2928-2933 (2014).
53. D. A. Falk et al. Multi-scale controls of historical forest-fire regimes: New insights from fire-scar networks. *Frontiers in Ecology and the Environment*, 9(8), 446–454 (2011).
54. P. F. Hessburg et al. Climate, environment, and disturbance history govern resilience of western North American forests. *Frontiers in Ecology and Evolution*, 7, 239 (2019).
55. C. H. Guiterman, E. Q. Margolis, C. H. Baisan, D. A. Falk, C. D. Allen, T. W. Swetnam. Spatiotemporal variability of human–fire interactions on the Navajo Nation. *Ecosphere* 10(11): e02932 (2019).
56. T. W. Swetnam, J. Farella, C. I. Roos, M. J. Liebmann, D. A. Falk, C. D. Allen. Multiscale perspectives of fire, climate and humans in western North America and the Jemez Mountains, USA. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 371(1696), 20150168 (2016).
57. J. W. Van Wagtendonk. The history and evolution of wildland fire use. *Fire Ecology*, 3(2), 3-17 (2007).
58. S. A. Parks, C. Miller, M.-A. Parisien, L. M. Holsinger, S. Z. Dobrowski, J. T. Abatzoglou. Wildland fire deficit and surplus in the western United States, 1984–2012. *Ecosphere*, 6(12), art275 (2015).
59. E. R. Bigio, T. W. Swetnam, C. H. Baisan. Local-scale and regional climate controls on historical fire regimes in the San Juan Mountains, Colorado. *Forest Ecology & Management,* 360 pp. 311-322 (2016).
60. C. H. Guiterman, E. Q. Margolis, C. D. Allen, D. A. Falk, T. W. Swetnam, Long-term persistence and fire resilience of oak shrubfields in dry conifer forests of northern New Mexico. *Ecosystems*, 21, 943-959 (2018).
61. C. D. O'Connor, D. A. Falk, A. M. Lynch, T. W. Swetnam. Fire severity, size, and climate associations diverge from historical precedent along an ecological gradient in the Pinaleño Mountains, Arizona, USA. *For. Ecol. Manag.*, 329, 264–278 (2014).
62. J. N. Williams, H. D. Safford, N. Enstice, Z. L. Steel, and A. K. Paulson. "High-Severity Burned Area and Proportion Exceed Historic Conditions in Sierra Nevada, California, and Adjacent Ranges." *Ecosphere* 14(1): e4397. (2023).
63. B. M. Collins, S. L. Stephens. Managing natural wildfires in Sierra Nevada wilderness areas. *Frontiers in Ecology and the Environment*, 5(10), 523–527 (2007b).
64. J. D. Coop, S. A. Parks, S. R. McClernan, L. M. Holsinger. Influences of prior wildfires on vegetation response to subsequent fire in a reburned Southwestern landscape. *Ecol. Appl.*, 26(2), 346–354. JSTOR (2016).
65. S. A. Parks, C. Miller, C. R. Nelson, Z. A. Holden. Previous fires moderate burn severity of subsequent wildland fires in two large western US wilderness areas. *Ecosystems*, 17(1), 29–42 (2014).
66. P. M. Fernandes, H. S. Botelho. A review of prescribed burning effectiveness in fire hazard reduction. *Int. J. Wildland Fire*, 12(2), 117 (2003).
67. J. Pollet, P. N. Omi. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. *Int. J. Wildland Fire*, 11(1), 1 (2002).

68. L. B. Harris, S. A. Drury, A. H. Taylor. Strong legacy effects of prior burn severity on forest resilience to a high-severity fire. *Ecosystems* (2020).
69. C. A. Farris, C. H. Baisan, D. A. Falk, M. L. Van Horne, P. Z. Fulé, T. W. Swetnam. A comparison of targeted and systematic fire-scar sampling for estimating historical fire frequency in south-western ponderosa pine forests. *Int. J. Wildland Fire*, 22(8), 1021 (2013).
70. P. Z. Fulé, J. E. Crouse, T. A. Heinlein, M. M. Moore, W. W. Covington, G. Verkamp. Mixed-severity fire regime in a high-elevation forest of Grand Canyon, Arizona, USA. *Landscape Ecology*, 18, 465–486 (2003a).
71. M. L. Van Horne, P. Z. Fulé. Comparing methods of reconstructing fire history using fire scars in a southwestern United States ponderosa pine forest. *Canadian Journal of Forest Research*, 36(4), 855–867 (2006).
72. S. J. Prichard et al. Adapting western North American forests to climate change and wildfires: Ten common questions. *Ecol. Appl.* (2021).
73. D. W. Huffman, A. J. Sánchez Meador, M. T. Stoddard, J. E. Crouse, J. P. Roccaforte. Efficacy of resource objective wildfires for restoration of ponderosa pine (Pinus ponderosa) forests in northern Arizona. *For. Ecol. Manag.*, 389, 395–403 (2017).
74. B. M. Collins, R. Everett, D. L. Fry, J. M. Lydersen, S. L. Stephens. Impacts of different land management histories on forest change. *Ecol. Appl.*, 27(8), 2475–2486. JSTOR (2017).
75. C. A Kolden. We're not doing enough prescribed burn in the western United States to mitigate wildfire risk. *Fire*, 2(2), 30 (2019).
76. C. Adlam, D. Almendariz, R. W. Goode, D. J. Martinez, B. R. Middleton. Keepers of the Flame: Supporting the Revitalization of Indigenous Cultural Burning. *Society & Natural Resources*, 1–16 (2021).
77. C. I. Roos et al. Indigenous fire management and cross-scale fire-climate relationships in the Southwest United States from 1500 to 1900 CE. *Science Advances*, 8(49), eabq3221 (2022).
78. R. Loehman, W. Flatley, L. M. Holsinger, A. E. Thode. Can land management buffer impacts of climate changes and altered fire regimes on ecosystems of the Southwestern United States? *Forests*, 9(4), 32 (2018).
79. P. R. Sheppard, A. C. Comrie, G. D. Packin, K. Angersbach, M. K. Hughes. The climate of the US Southwest. *Climate Research*, 20 (2002).
80. S. B. Malevich, C. H. Guiterman, E. Q. Margolis. burnr: Fire history analysis and graphics in R. *Dendrochronologia*, 49, 9–15 (2018).
81. R Core Team. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/ (2022).
82. Monitoring Trends in Burn Severity (MTBS). Burned Area Boundaries Dataset Direct Download. Accessed Feb. 28, 2021. https://www.mtbs.gov/direct-download (2021).
83. National Interagency Fire Center (NIFC). NIFC Open Data Site, Historic Wildland Fires Data Download. Accessed Feb. 28, 2021. https://data-nifc.opendata.arcgis.com/ (2021).
84. N. Gorelick, M. Hancher, M. Dixon, S. Ilyushchenko, D. Thau, R. Moore. Google Earth Engine: Planetary-scale geospatial analysis for everyone. *Remote Sens. Environ.*, 202, 18-27 (2017).
85. S. A. Parks et al. Giving ecological meaning to satellite-derived fire severity metrics across North American forests. *Remote Sensing*, 11(14), 1735 (2019).
86. C. H. Key, N. C. Benson. Landscape Assessment (LA). In: Lutes, Duncan C.; Keane, Robert E.; Caratti, John F.; Key, Carl H.; Benson, Nathan C.; Sutherland, Steve; Gangi, Larry J. 2006. FIREMON: Fire effects monitoring and inventory system. Gen. Tech. Rep. RMRS-GTR-164-CD. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. (pp. 1–55) (2006).
87. C. D. Allen. Changes in the landscape of the Jemez Mountains, New Mexico [Doctoral dissertation]. University of California at Berkeley (1989).

88. Allen, C. D., Anderson, R. S., Jass, R. B., Toney, J. L., & Baisan, C. H. (2008). Paired charcoal and tree-ring records of high-frequency Holocene fire from two New Mexico bog sites. *Int. J. of Wildland Fire*, 17(1), 115. https://doi.org/10.1071/WF07165
89. J. J. Dewar, D. A. Falk, T. W. Swetnam, C. H. Baisan., C. D. Allen, R. R. Parmenter, E. Q. Margolis, E. J. Taylor. Valleys of fire: Historical fire regimes of forest-grassland ecotones across the montane landscape of the Valles Caldera National Preserve, New Mexico, USA (2021).
90. Margolis, E. Q., & Malevich, S. B. (2016). Historical dominance of low-severity fire in dry and wet mixed-conifer forest habitats of the endangered terrestrial Jemez Mountains salamander (*Plethodon neomexicanus*). *Forest Ecol. and Management*, *375*, 12-26.
91. C. H. Baisan, K. A. Morino, H. D. Grissino-Mayer. Fire history in ponderosa pine and mixed conifer forests of the Catalina Mountains [Final Report to the USDA Forest Service]. https://www.ltrr.arizona.edu/webhome/cbaisan/Class/CatRpt6.pdf (1998).
92. J. M. Iniguez, T. W. Swetnam, C. H. Baisan. Spatially and Temporally Variable Fire Regime on Rincon Peak, Arizona, USA. *Fire Ecology* 5, 3–21 (2009).
93. J. M. Kaib. Fire history in riparian canyon pine-oak forests and the intervening desert grasslands of the Southwestern borderlands: A dendroecological, historical, and cultural inquiry. 235 (1998).
94. J. M. Kaib. Fire history reconstructions in the Mogollon province ponderosa pine forests of the Tonto National Forest, central Arizona. 7 (2001).
95. J. M. Kaib, C. H. Baisan., H. D. Grissino-Mayer, T. W. Swetnam. Fire History in the Gallery Pine-Oak Forests and Adjacent Grasslands of the Chiricahua Mountains of Arizona, in: Ffolliott, P.F., DeBano, L.F., Maker Jr., M.B., Gottfried, G.J., Solis-Garza, G., Edminster, C.B., Neary, D.G., Allen, L.S., Hamre, R.H. (Eds.), Effects of Fire on Madrean Province Ecosystems: A Symposium Proceedings. USDA Forest Service Gen. Tech. Rep. RM-GTR-289, pp. 253–264 (1996).
96. J. J. Minor. Anthropogenic influences on fire regimes and post-fire ecological communities in an Arizona Sky Island [Doctoral dissertation]. University of Arizona (2017).
97. K. A. Morino, C. H. Baisan., T. W. Swetnam. An examination of fire along an elevation gradient: Historical fire regimes in the Chiricahua Mountains, Arizona and in mixed-conifer forest (2000).
98. M. T. Secklecki, H. D. Grissino-Mayer, T. W. Swetnam. Fire history and the possible role of Apache-set fires in the Chiricahua Mountains of southeastern Arizona. Effects of Fire on Madrean Province Ecosystems : A Symposium Proceedings.- Fort Collins, CO (USA): U.S. Dept. of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, 1996, 238–246 (1996).
99. P. Z. Fulé, T. A. Heinlein, W. W. Covington, M. M. Moore. Assessing fire regimes on Grand Canyon landscapes with fire-scar and fire-record data. *Int. J. Wildland Fire*, 12(2), 129 (2003b).
100. Brown, P. M., Wienk, C. L., & Symstad, A. J. (2008). Fire and forest history at Mount Rushmore. *Ecol. Appl.*, *18*(8), 1984-1999.
101. Yocom Kent, L. L., & Fulé, P. Z. (2015). Do rules of thumb measure up? Characteristics of fire-scarred trees and samples. *Tree-Ring Research*, *71*(2), 78-82.
102. Margolis, E. Q. (2014). Fire regime shift linked to increased forest density in a piñon–juniper savanna landscape. *Int. J. Wildland Fire*, *23*(2), 234-245.
103. Burch, B. D., & Sánchez Meador, A. J. (2018). Comparison of forest age estimators using k-tree, fixed-radius, and variable-radius plot sampling. *Canadian Journal of Forest Research*, *48*(8), 942-951.

## Figures and Tables

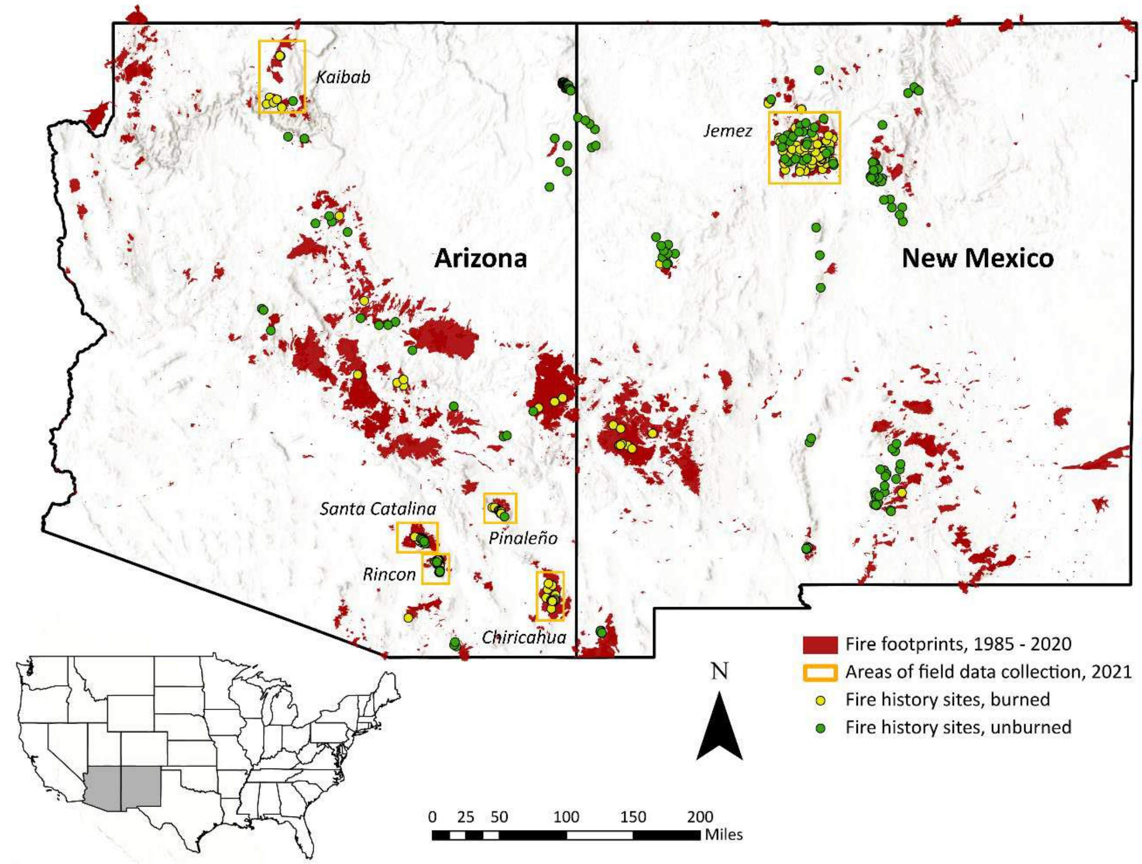


**Figure 1.** Study area map showing locations of 406 tree-ring fire history sites in the southwestern United States analyzed in this study and contemporary fire history (1985-2020).

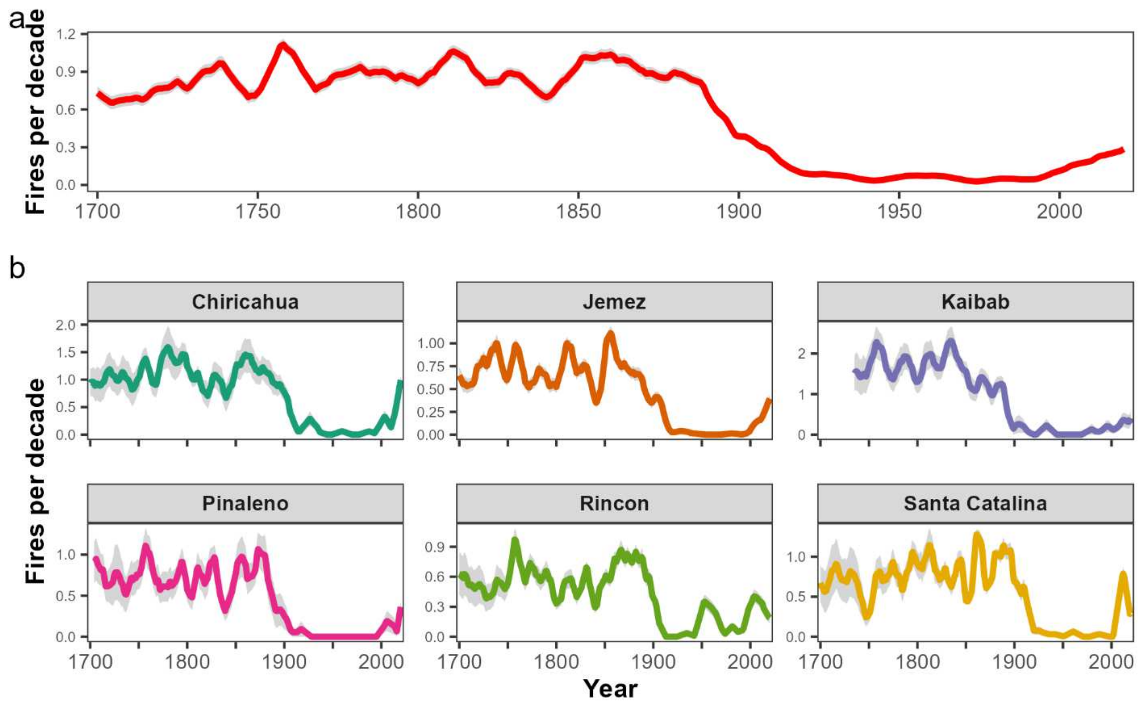


**Figure 2.** Smoothed ten-year moving average of fires per decade from 1700 to 2020 for a) all tree-ring fire history sites in the southwestern United States (n = 406) and b) geographic areas targeted for field sampling (n = 226 total sites). The time series were created by combining tree-ring fire scar records with contemporary fire perimeter data. Grey shading shows the standard error of mean fires per decade.

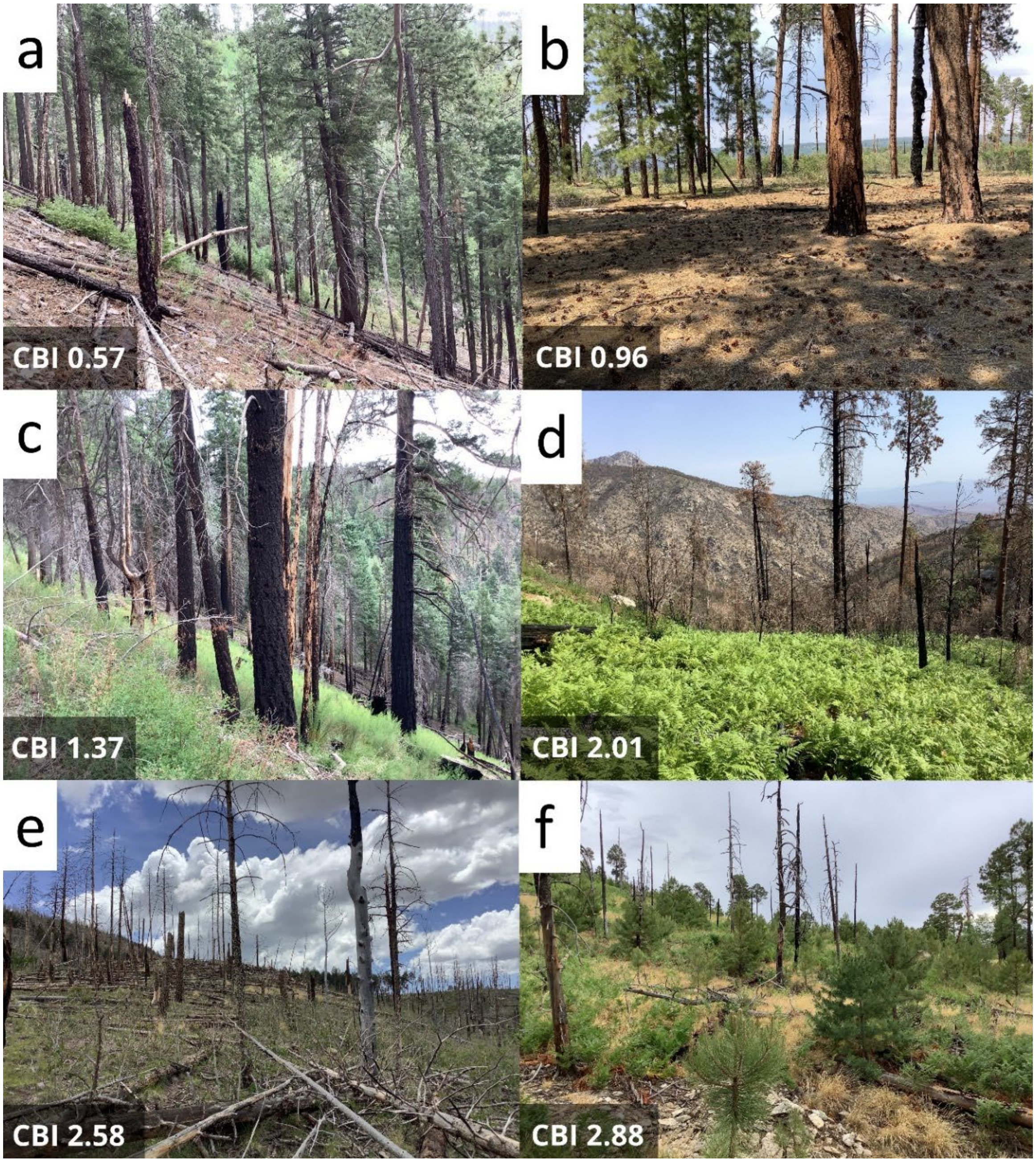


**Figure 3.** Plot photographs from fire history sites across the spectrum of burn severity in six field-sampled geographic areas: (a) Chiricahua, (b) Kaibab, (c) Pinaleño, (d) Santa Catalina, (e) Jemez, and (f) Rincon Mountains. The satellite-derived burn severity rating is given for each of these sites, in units of modeled Composite Burn Index (CBI), which scales from 0 (low-severity) to 3 (high-severity), are given in the bottom left corner. Photo credit: E. McClure, S. Parks, & M. Kunkel.

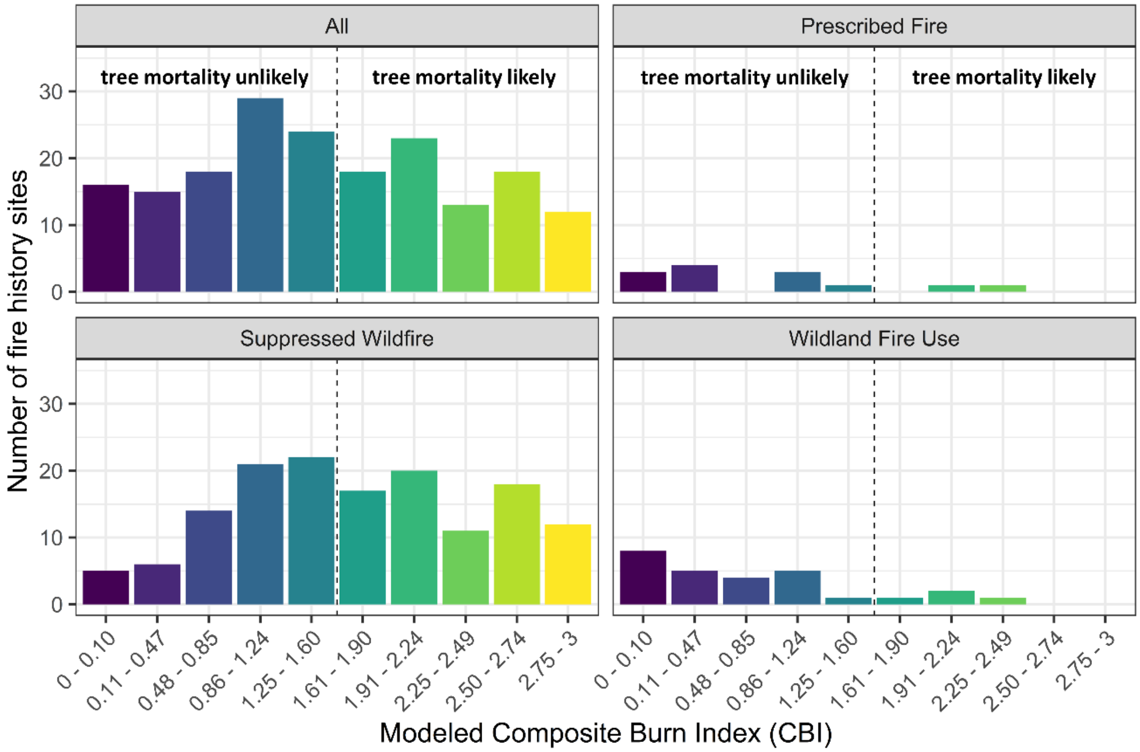


**Figure 4.** Burn severity, quantified by modeled Composite Burn Index (CBI), for the first fire between 1985 and 2020 at tree-ring fire history sites in the southwestern United States. Burn severity is reported for all fire types, prescribed burns, suppressed wildfires, and wildland fire use fires. Sites were filtered by our ability to assign fire type and CBI value: n = 186 for all, 13 for prescribed burn, 146 for suppressed wildfire, and 27 for wildland fire use. Divisions correspond to standard CBI severity classes (Key & Benson, 2006) subdivided by thirds. The dashed line represents a CBI value of 1.61, corresponding to the threshold above which the probability of overstory tree mortality exceeds 50% (mortality more likely than not).

**Table 1.** The number of tree-ring fire-scar sites, field plots, contemporary fires, and the percent of tree-ring fire-scar sites burned from 1985 - 2020 in the southwestern United States and multiple sub-regions**.**

| Geographic area | tree-ring fire scar sites | field plots | fires 1985-2020 | % tree-ring sites burned 1985-2020 |
|---|---|---|---|---|
| Chiricahua | 17 | 5 | 5 | 100.0 |
| Jemez | 100 | 25 | 19 | 55.0 |
| Kaibab | 8 | 10 | 14 | 100.0 |
| Pinaleño | 11 | 13 | 3 | 84.6 |
| Rincon | 60 | 25 | 18 | 70.0 |
| Santa Catalina | 30 | 13 | 5 | 96.7 |
| Other | 180 | 0 | 38 | 23.9 |
| All | 406 | 91 | 102 | 50.7 |

# Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- McClureetalTableS1.csv
- McClureetalSupplementalTablesandFigures.pdf