DIVERGENCE BETWEEN HISTORICAL AND CONTEMPORARY FIRE REGIMES AT TREE-RING FIRE HISTORY SITES IN DRY CONIFER FORESTS OF THE SOUTHWESTERN UNITED STATES

by

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This work is dedicated to Thomas Girard McClure, forever my biggest fan.

Divergence between historical and contemporary fire regimes at tree-ring fire history sites in dry conifer forests of the southwestern United States

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Abstract

Changing fire regimes raise concerns about increasing vulnerability of dry conifer forests in the southwestern United States. However, the extent to which contemporary fire regimes may have diverged from historical patterns and processes remains the subject of considerable uncertainty, and consequently, active scientific debate. At issue is the historical role and extent of high severity fire. Here, we contrast the frequency and severity of historical (1700-1880) and contemporary (1985-2020) fires across a network of 408 tree-ring fire-scar sites in Arizona and New Mexico. We combine dendroecological records, satellite-derived metrics of burn severity, and field measures of tree mortality. Historically, low- to moderate-severity fires burned at these sites with a mean fire return interval (MFI) of 16.9 (± 10.7) years. These fire regimes ended by

1880 at most sites, initiating a long fire-free period. Over the satellite record (1985-2020) nearly half of the sites did not experience fire, underscoring a still-growing fire deficit in large portions of these landscapes. Of the sites analyzed, 26.7% burned once while 24.3% burned two or more times. In first-entry contemporary fires, 42.4% of sites burned at severity more likely than not to kill trees (CBI>1.61), anomalous for sites where trees historically survived fire for centuries. Contemporary burn severity was linked to climatic variables including maximum temperature and vapor pressure deficit, but not to historical MFI or the length of time between historical and contemporary fire events. We did not directly evaluate pre-fire fuel conditions as a driver of burn severity, but suspect variability therein was overcome by long fire-free intervals across the region. These results underscore the consequences of climate change on the severity of forest fires, but also suggest that management interventions can achieve promising outcomes given the right climatic opportunities.

Keywords

Fire regimes, tree-ring dating, fire history, wildfire, forest management, ponderosa pine, dry conifer forests, southwestern United States, North American tree-ring fire-scar network.

Introduction

Anthropogenic climate change and changes in land use can weaken forest ecosystem resilience in the face of disturbance such as drought, flooding, and wildfire (Johnstone et al., 2016; Millar et al., 2007). Understanding resilience mechanisms – persistence, recovery, and reorganization – can help land managers adjust to changing ecological conditions and mitigate the effects of disturbances in an uncertain future (Falk et al., 2022). Wildfire constitutes one such disturbance type with significant potential to drive changes in ecosystem states.

In the western United States, fire activity is on the rise due to a range of compounding influences, many of which are likely to accelerate in the coming decades. Beyond anthropogenic climate change (Abatzoglou & Williams, 2016; Higuera & Abatzoglou, 2021; Westerling et al., 2006), a century of fire suppression (Schoennagel et al., 2004) has followed cultural severance of Indigenous burning practices (Kimmerer & Lake, 2001; Lake et al., 2017) and intensive Euro-American livestock grazing (Swetnam & Baisan, 1996). Increased fire activity has been demonstrated both in terms of total area burned and area burned in high severity (Parks & Abatzoglou, 2020), and in some areas now exceeds paleoecological norms (Higuera et al., 2021). These changes prompt concern for the persistence and recovery of forested landscapes experiencing increasing tree mortality (Cansler et al., 2020; van Mantgem et al., 2009), decreasing tree regeneration (Davis et al., 2019; Rodman et al., 2020; Stevens-Rumann et al., 2018), and consequently, growing vulnerability to enduring ecological shifts from forested to non-forested systems (Coop et al., 2020; Walker et al., 2018).

To respond to these challenges by implementing effective, adaptive land management strategies, managers seek to understand specific ecological conditions at various spatial and temporal scales (Walker et al., 2004). Understanding the historical disturbance regimes to which species are adapted and through which ecosystems are sustained can form a crucial starting point in land management, even when conditions are changing (Swetnam et al., 1999). Notable differences have been found in fire-adapted forests of the western US between fire regimes (for example, decreased fire frequency in concert with increased severity and extent) before and after Euro-American colonization and industrialization (Lavorel et al., 2006).

Dry conifer forests in the American Southwest composed of ponderosa pine (*Pinus ponderosa*), Douglas-fir (*Pseudotsuga menziesii*), and other species including Arizona pine (*Pinus arizonica*), and Southwestern white pine (*Pinus strobiformis*), and white fir (*Abies concolor*) were generally characterized by frequent, low- to moderate-severity fire prior to 1880 (Fulé et al., 1997; Swetnam & Baisan, 1996). These historical fire regimes were disrupted by overgrazing and other land use changes following the colonization of the region and violent removal of Indigenous peoples (Allen et al., 2002). In many cases, both anthropogenic and lightning-caused fires were thus completely halted in the 19th and 20th centuries, resulting in a fire deficit (Marlon et al., 2012). Cessation of frequent surface fire has led to subsequent canopy densification and increases in coarse woody debris and ladder fuels, in the form of shrubs and young trees in the under- and mid-story (Allen et al., 2002). Many recent fires in these forest types are burning severely (Singleton et al., 2019), potentially overwhelming the ecological resilience of dominant, wind-dispersed conifers (Chambers et al., 2016; Haffey et al., 2018; Savage & Mast, 2005). Southwestern dry conifer forests are therefore at high risk of fire-driven conversion from forest to non-forest conditions (Guiterman et al., 2018), positioning them as high-priority systems for both further research and management if the goal is persistence of these forest types.

Although these processes are well understood, the extent to which fire regimes now differ from historical norms in dry conifer forests of the southwest is still subject to uncertainty and scientific debate (Fulé et al., 2014; Williams & Baker, 2012). One viewpoint relies on data from General Land Office (GLO) surveys from 1880 (Williams & Baker, 2011); stand structure reconstructions based upon age-size relationships, and associated interpretations of fire severity. They suggest that mixed- and high-severity fire may have been prevalent in these systems long before the prominent ecological changes of the 20th century (Williams & Baker, 2012). The other, much more widely supported, argument contends that modern wildfires have dramatically differed in severity from historical norms (Fulé et al., 2014). This is based upon wide disparities in forest structure between historical reconstructions – using many lines of evidence – and modern measurements (Bowman et al., 2013; Swetnam & Brown, 2010), and a lack of documentary and scientific evidence of high-severity fire occurring in these systems in the 18th and 19th centuries (Cooper, 1960; Fulé et al., 2014).

Dendroecological research, including tree-ring analyses and cross-dating of fire-scarred trees, is a primary means of understanding historical fire regimes (Harley et al., 2018; Dieterich and Swetnam, 1984). Fire scars occur on trees when the heat load at the base of a tree is sufficient to injure but not kill the tree, which continues to grow around the wound resulting in a scar that can be dated to the year and often the season of burning. Comparison with fire atlases has shown that fire scars can accurately reconstruct many components of dry conifer forest fire regimes (Collins & Stephens, 2007; Farris et al., 2010). To facilitate regional analyses of historical fire activity, researchers have compiled an extensive, international array of tree-ring fire history data to create the North American tree-ring fire-scar network (NAFSN; Margolis, Guiterman, et al., *in press,* Falk et al., 2011). Many contributions to this network were conducted in dry conifer forests of Arizona and New Mexico, the area of interest for this study.

Where tree-ring records inform historical fire patterns, recent satellite observations form the basis of contemporary fire regime characterization (Eidenshink et al., 2007; Key, 2006). Composite Burn Index (CBI; Key & Benson, 2006), a field protocol for validating satellite-derived burn severity one year after fire, can now be modeled based solely on satellite data (Parks et al., 2019). This method gives a more ecologically intuitive and widely comparable metric than others like the delta Normalized Burn Ratio (dNBR; Key, 2006) and its derivatives. Satellite-derived metrics have facilitated analysis of burn severity trends in the modern era

(1984-present) in the western United States (Dillon et al., 2011; Mueller et al., 2020; Singleton et al., 2019).

The purpose of our research is to compare historical tree-ring reconstructed fire regimes to contemporary satellite-derived fire regimes. Fundamentally, we ask how historical and contemporary fire regimes differ at sites where trees survived low- and moderate-severity fire for centuries. At these fire history sites, our objectives were to 1) quantify and compare historical and contemporary fire frequency, 2) quantify and compare historical and contemporary fire severity, and 3) assess impacts upon contemporary fire frequency and severity of historical fire regime characteristics, recent climate, and management strategy.

Methods

Study area

This study comprises tree-ring fire history sites and satellite-inferred burn severity data from Arizona and New Mexico, USA (Fig. 1). We chose to focus on these southwestern states because of the particularly robust, extensive, and accessible tree-ring records there, as collated in the North American tree-ring fire-scar network (NAFSN; Margolis, Guiterman, et al. *in press*). 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 the region receives up to 50% of its annual precipitation (Sheppard et al., 2002). The fire history sites examined in this study range from 1552 to 3105 meters in elevation and are located in forest stands with abundant fire-scarred trees, including ponderosa pine, Douglas-fir, Arizona pine, and Southwestern white pine. These sites can also contain white fir, Engelmann spruce (*Picea engelmannii*), piñon pine

(*Pinus edulis*), juniper (*Juniperus deppeana, osteosperma,* and *scopulorum*), and oak (*Quercus* spp., especially *Q. gambelii*).

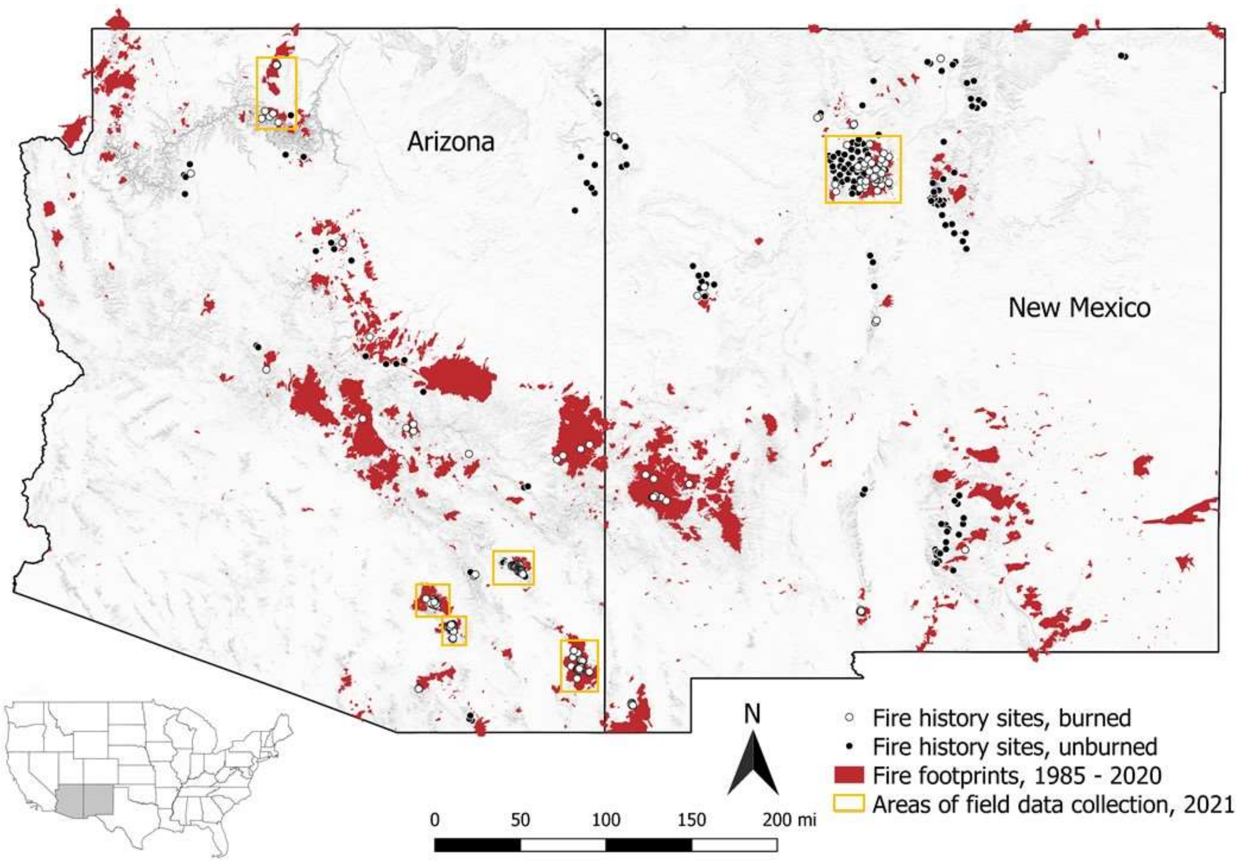

Figure 1. Study area map showing locations of all 600 fire history sites included in the North American tree-ring fire-scar network (NAFSN) and contemporary fire history (1985-2020).

Historical fire records

As of 2021, the NAFSN database incorporated 2,562 fire-scar study sites, including 600 in Arizona and New Mexico (https://doi.org/10.5066/P9PT90QX). More than half of the sites (54.3%) burned in the 1985-2020 timeframe for which suitable satellite data were available (Fig. 1). Of these 600 sites, site-level data in the fire history exchange format (.fhx) with sufficient sample sizes (at least three trees, recording at least four fires between 1700-1880) were available for 408 sites (208 burned and 200 unburned in the contemporary period). We focused on these 408 sites for the purposes of this study, hereafter referred to as fire history sites, and created site composites and ran interval analyses for each. Fire chronologies were constructed based on a filter of 10% of trees recording fire with a minimum of two trees recording and two trees scarred per fire. These were then used to calculate site-specific mean fire return interval (MFI), median fire return interval (med_FI), range of fire years, and date of last historical fire.

Contemporary fire records

The Monitoring Trends in Burn Severity program (MTBS) generates burn severity data from satellite imagery, making it available nationally to managers, policymakers, and researchers. We intersected fire perimeters from MTBS (https://www.mtbs.gov/direct-download), including all wildfires and prescribed burns over 1,000 acres (404.7 ha), occurring from 1985 to 2019 in Arizona and New Mexico, with fire history site locations (Fig. 1). To capture fire events with acreage less than 1,000 and/or occurring in 2020, we accessed fire perimeter data from the National Interagency Fire Center (NIFC; https://data-nifc.opendata.arcgis.com/) and local fire atlases for some of the National Parks and National Forests in the study area.

We identified 102 contemporary fires at 208 (of 408) fire history sites. Next, we generated burn severity grids for each fire, representing modeled Composite Burn Index (CBI). We chose to use modeled CBI as opposed to delta Normalized Burn Ratio (dNBR, see Eidenshink et al., 2007), due to the comparability across fires, sites, and years inherent in the standardized scale of 0 to 3 (see Parks et al., 2019). Woolman et al. (*in review*) found that modeled CBI was positively correlated with dNBR and that the former better predicted overstory ponderosa pine tree mortality. These raster datasets were produced in Google Earth Engine using code developed and distributed by Parks et al. (2019), after which point, we extracted CBI values at each site.

Field sampling

To quantify tree mortality and other effects of recent fires, we visited and installed field sampling plots at a subset of fire history sites. These field plots were located across gradients of both contemporary burn severity and fire management histories. Data collection was focused on six key mountain ranges (see Table 1) in which extensive networks of fire history sites were established prior to subsequent wildfires in the past ten years (2011-2020). During the contemporary period of 1985-2020, these sites ranged from one to seven times burned overall. These areas included the Rincon (Farris et al., 2010), Santa Catalina (Baisan et al., 1998), Pinaleño (O'Connor et al., 2014), and Chiricahua (Kaib, 1998; Kaib, 2001; Kaib et al., 1996; Minor, 2017; Morino et al., 2000; Secklecki et al., 1996) Mountains and the Kaibab Plateau (Fulé, Crouse, et al., 2003; Fulé et al., 2003) in Arizona, and the Jemez Mountains in New Mexico (Allen, 1989; Allen et al., 2008; Dewar et al., 2021; Margolis & Malevich, 2016). The Rincon Mountains sites are mostly located in Saguaro National Park (NP), in protected wilderness, while about half of those on the Kaibab Plateau are in proposed wilderness in Grand Canyon NP. In the Jemez Mountains, sites are located in Bandelier NP, Valles Caldera National Park and Preserve, and the Santa Fe National Forest.

At each fire history site, we established a 10-m radius circular 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. 2), whereas 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 within a site, but at least 20 m apart, we installed a plot at each tree, leading to 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. We recorded an overall count of trees both live and dead, standing and down, and tallied seedlings by species in a 5-m subplot, centered at the plot center. Finally, we used a modified version of the CBI field protocol (Key & Benson, 2006) to assess fire effects on trees and heavy fuels of the most recent fire. CBI plot measurement involves rating the degree of mortality and/or consumption on various strata from soils and fine fuels, grasses and other herbaceous plants, shrubs, up to tree canopies. We restricted our focus to durable strata, as some sites burned up to ten years prior to data collection. As with typical collection of one-year post-fire CBI data, for sites burned more than once in the contemporary period, attributing fire effects to the most recent fire required careful judgment of, for example, the age of charred materials. Further qualitative description of site conditions, along with photographic documentation (Fig. 2, 3, 4), completed our characterization of these sites.

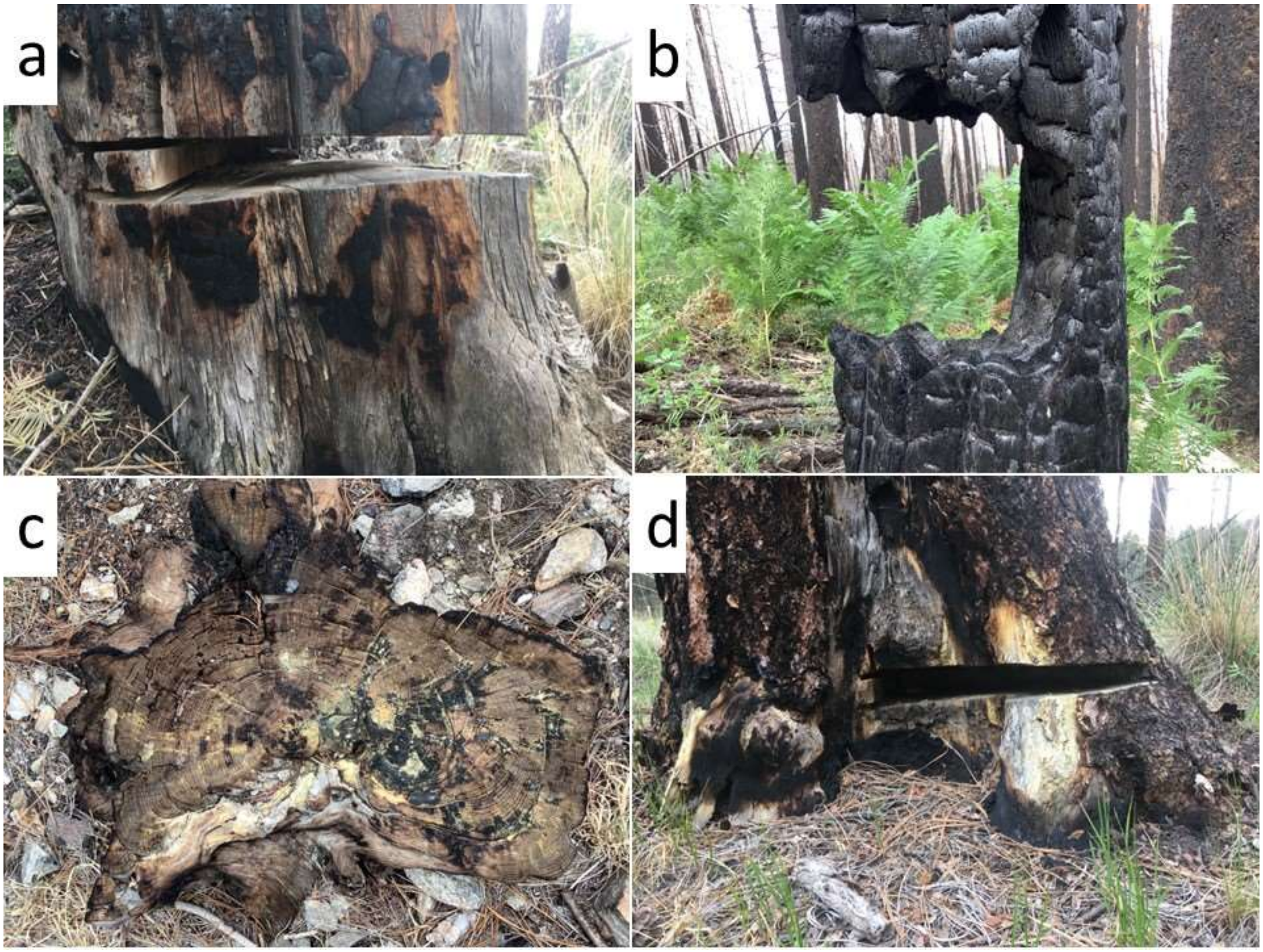

Figure 2. Fire-scarred trees sampled for tree-ring fire history reconstructions and relocated to assess the effects of contemporary fires since sampling. The stump in (a), at Redondo Border in the Jemez Mountains, the tree in (b) in the Pinaleño Mountains, and the stump in (c), on Mica Mountain in the Rincon Mountains, were most likely dead when originally sampled, while the tree in (d), at Valle Jaramillo in the Jemez Mountains, was living when sampled and subsequently survived 2013's Thompson Ridge fire; it remained alive on June 5, 2021. Photo credit: E. McClure, S. Parks, & Mason Kunkel.

Data analysis

Because the contemporary period was only 35 years for this study, compared with 180 years for the historical period (1700-1880), a direct comparison between historical and contemporary mean fire return interval was not feasible. Using the site composites generated from site-level fire history data, and contemporary fire dates from MTBS, NIFC, and local fire atlases, we calculated ten-year moving averages of fires per year from 1700 through 2020 to examine fire frequency trends across the entire time frame of our study (Fig. 6). Although some fire history sites recorded fires in the 1400s and earlier, a consistent record across all sites is interpretable beginning in the 18th century.

To facilitate comparison of fire effects between historical and contemporary fires, we classified the latter into binary categories of severity relating to mature trees, as follows: 1) low probability of overstory tree mortality (P(mort); consistent with a tree surviving to record fire scars), and 2) high P(mort) (wherein trees do not record fire). To develop this classification, we generated logistic regression models to identify thresholds above which probability of overstory tree mortality exceeds 50%. In other words, the likelihood of tree mortality exceeds that of tree survival. We set this at 50% because the fire history sites recorded anywhere from 4 to 92 fires (mean = 16.7) historically. Thus, even if a site burned only four times at severity levels exceeding 50%, the trees in that site would have a 6.25% (0.5^4 = 0.0625) chance of surviving all four fires.

For the first of these thresholds, we used an analysis of tree mortality data from the United States Forest Service's Forest Inventory and Analysis program (FIA; Woudenberg et al., 2010), alongside satellite-derived burn severity in modeled CBI for fires which subsequently burned FIA plots in ponderosa pine forests (Woolman et al., *in review*). Woolman et al. assessed the relationship between field-recorded tree mortality and contemporary burn severity, as well as vegetation composition before and after fire, including post-fire regeneration. We used their data to calculate a tree mortality threshold of 1.73 CBI. Our second logistic regression model used modeled CBI data and overstory (dbh ≥ 12.7 cm) trees measured in our field plots (Fig. 5).

We used linear models and model selection processes (dredge function) to evaluate the drivers of burn severity of the first and second contemporary fires, in modeled CBI. We tested an array of potential predictors including characteristics of historical fire regimes, fire season averages of climate metrics, and fire type. All variables were scaled to between 0 and 1.

Historical fire regime characteristics considered as predictor variables again included historical (1700-1880) mean fire return interval (MFI); median fire return interval (median FI); last fire date documented in tree-ring record; fire-free interval (FFI), calculated as the difference between the first contemporary fire year, or 2020 for unburned sites and the last fire year recorded in the tree-ring record; and disruption metrics calculated as the log-transformed ratios of MFI to FFI and median FI to FFI. For climate metrics, we used the methodology employed in Parks & Abatzoglou (2020) to average monthly means for the three months leading up to onset of fire season, in this case April - June. These metrics included maximum temperature (T_{max}), precipitation, vapor pressure deficit (VPD), climate water deficit (CWD), actual evapotranspiration (AET), and soil moisture, from TerraClimate (Abatzoglou et al., 2018) and PRISM (Daly et al., 2008).

We classified fires by management type (Suppressed Wildfire, Wildland Fire Use, and Prescribed Fire), although this was unknown for 13 fires (12.7%). For the Wildland Fire Use category, we included wildfires listed as “resource benefit,” “type 49,” or “managed for multiple uses,” depending on source and date as these terms were applied differently over the course of the study period. We then combined Prescribed Fire and Wildland Fire Use to create a binary predictor variable via categorizing fires by whether they were suppressed wildfires or not.

We were also interested in the predictors of contemporary fire frequency, measured simply as the number of fires occurring between 1985 and 2020. We tested the above historical fire regime characteristics as predictor variables, using generalized linear models with the negative binomial family.

Further, for our field plots we used linear models to assess effects of the above predictor variables upon field measurements including field-measured CBI, overstory tree mortality, and conifer seedling and sapling densities.

Analyses were performed in R (v. 4.1.3; R Core Team, 2022), including the following packages: tidyverse (v. 1.3.1; Wickham et al., 2019), burnr (v. 0.6.1; Malevich et al., 2018), glmmTMB (v. 1.1.3; Brooks et al., 2017), MUMin (v. 1.46.0; Barton, 2009), reshape2 (v. 1.4.4; Wickham, 2007), viridis (v. 0.6.2; Garnier et al., 2021), and RColorBrewer (v. 1.1.3; Neuwirth, 2014).

Results

Fire frequency

a. Historical

The historical mean fire return interval for all fire history sites with sufficient sample size was 16.9 (± 10.7) years, with an overall median fire return interval of 14.4 (± 10.4) years. Site-specific MFI ranged from 2.6 to 91.3 years, while MFI averaged within field-sampled geographic areas ranged from 7.5 years on the Kaibab Plateau to 18.7 years in the Rincon Mountains (Table 1). The last historical fire date varied from 1875 in the Pinaleño Mountains to 1920 both on the Kaibab Plateau and in the Rincon Mountains.

b. Contemporary

Of 408 fire history sites with adequate sample sizes, 51.0% (208 sites) burned between 1985-2020. 24.2% (99 sites) burned twice or more within that time frame, including 4.2% (17 sites) burning three or more times. Contemporary fire occurrence varied by geographic area of fire history sites, with 55% of sites in the Jemez Mountains and 100% of sites in both the Kaibab Plateau and Chiricahua Mountains burning.

In all, 102 fires burned fire history sites in the contemporary period, 89 of which we were able to classify by fire type. Of those, 56 (62.9%) were suppressed wildfires (burned 243 fire history sites, including reburns), whereas 24 (27.0%) were Wildland Fire Use fires (burned 51 sites) and 9 (10.1%) were prescribed burns (burned 15 sites). Proportion of fires suppressed versus controlled varied by geographic area, for example with 14.3% of fires on the Kaibab Plateau were wildfires managed under a full-suppression strategy as opposed to 100% in both the Pinaleño and Santa Catalina Mountains (Table 1).

c. Comparing historical and contemporary

From calculating ten-year moving averages for all fire history sites across the timeframe of 1700-2020, we found that from 1700 to 1880, the mean site-specific fire frequency was 0.08 fires per year, which translates to a fire occurring every 12.5 years on average (Fig. 3). This rate dropped to 0.01 between 1880-1985 (100-year fire interval), rising again to 0.02 fires per year for 1985-2020, equivalent to sites burning every 50 years on average. Comparing mean fires per year over these timeframes, 14.2% of sites have returned to the same or higher level of fire frequency by 2020, leaving 85.8% with less frequent fire than historically (1700-1880) seen.

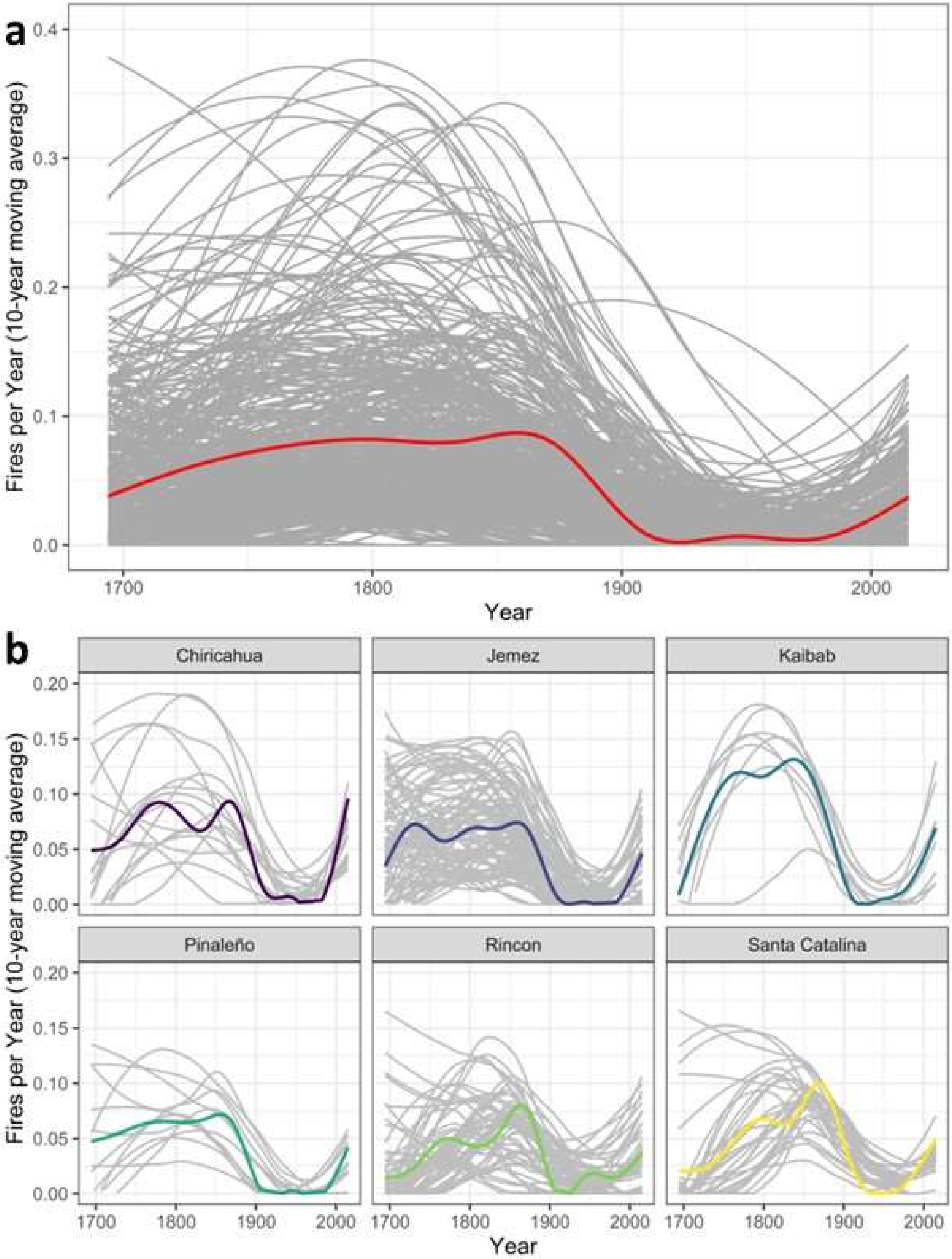

Figure 3. Smoothed ten-year moving average of fires per year, for all fire history sites (a; n = 408) and split out by geographic areas targeted for field sampling (b; n = 228). Grey lines represent individual sites while colored lines show averages.

Field sampling

Of the 74 distinct fire history sites we visited, we were able to relocate a tree, stump, or log (see Fig. 2) at 25 sites (33.8%). We relocated multiple sampled trees at four sites, all in the Rincon Mountains. We found a range of fire effects at the field plots. Those with high satellite-derived burn severity generally showed high tree mortality (Fig. 4). Overall, 14.9% of plots were devoid of live trees and 31.1% had no live overstory trees. These areas were often characterized by high fuel loads, especially in larger size classes. Regeneration was mixed across field plots; we recorded no live saplings in 56.8% of all plots, no seedlings in 52.7%, and no conifer seedlings in 58.1%. At the other end of the severity gradient (Fig. 5), many 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.

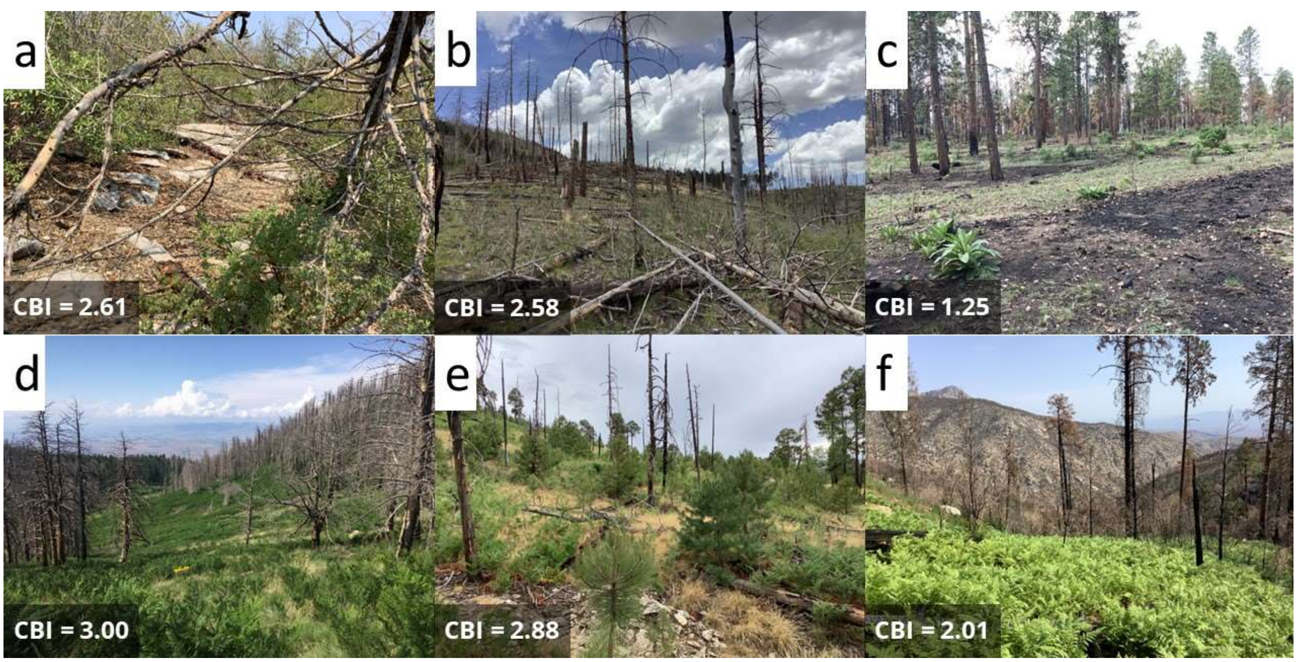

Figure 4. Plot photographs from fire history sites in relatively high-severity burned portions of six field-sampled geographic areas: Chiricahua (a), Jemez (b), Kaibab (c), Pinaleño (d), Rincon (e), and Santa Catalina (f) Mountains. The satellite-derived burned severity rating for each of these sites, in modeled Composite Burn Index (CBI), which scales from 0 to 3, are given in the bottom left corner. Photo credit: E. McClure, S. Parks, & Mason Kunkel.

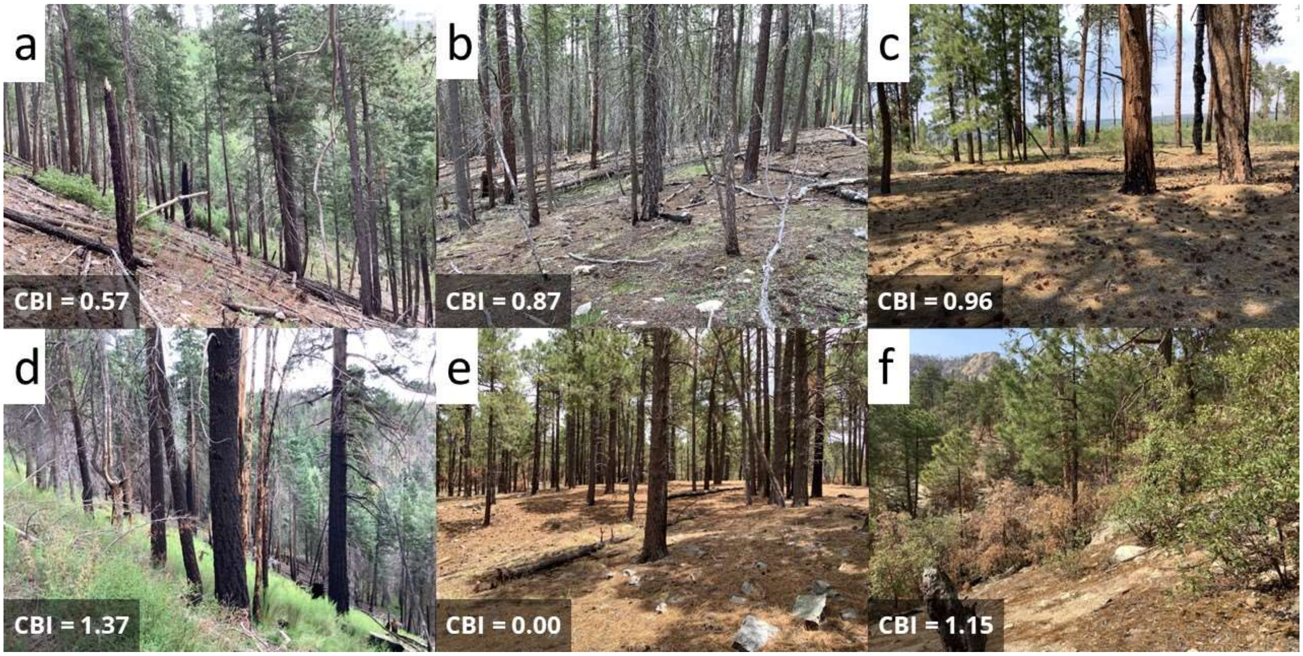

Figure 5. Plot photographs from fire history sites in relatively low-severity burned portions of six field-sampled geographic areas: Chiricahua (a), Jemez (b), Kaibab (c), Pinaleño (d), Rincon (e), and Santa Catalina (f) Mountains. The satellite-derived burned severity rating for each of these sites, in modeled Composite Burn Index (CBI) which scales from 0 to 3, are given in the bottom left corner. Photo credit: E. McClure, S. Parks, & Mason Kunkel.

Table 1. Fire history characteristics averaged for each geographic area where field data collection was performed, with standard deviations in parentheses. Contemporary fires, averaged by area, occurred between 1985 and 2020. Fire-free interval calculated as the difference between first contemporary fire, or 2020 if unburned, and last fire recorded in tree-ring record. Percent burned refers to the proportion of sites which burned at least once in the contemporary era, while percent high P(mort) (probability of mortality) refers to, of those that burned, the proportion of sites which burned above a tree mortality threshold of 1.61 Composite Burn Index (CBI) in their initial contemporary fire. Percent suppressed is the percentage of fires which were suppressed wildfires, as opposed to prescribed burns or wildland fire use fires.

Geographic area	N sites	N field plots	MFI (yrs)	Median FI (yrs)	Last fire	Fire-free interval (yrs)	% sites burned	% sites high P(mort)	Ownership	N fires	% suppressed
Chiricahua	17	5	12.3 (±6.2)	11.4 (±6.8)	1897 (±40.9)	108.3 (±44.9)	100.0	35.3	USFS/NPS	5	80.0
Jemez	100	25	18.0 (±10.0)	15.8 (±10.6)	1889 (±33.8)	123.4 (±37.5)	55.0	48.1	USFS/NPS	19	52.6
Kaibab	8	10	7.5 (±2.8)	6.9 (±3.5)	1920 (±58.2)	84.0 (±70.0)	100.0	0.0	USFS/NPS	14	14.3
Pinaleño	13	13	16.6 (±6.7)	15.9 (±8.8)	1875 (±16.3)	137.8 (±19.9)	84.6	63.6	USFS	3	100.0
Rincon	60	25	18.7 (±9.8)	15.8 (±10.2)	1920 (±58.1)	83.9 (±64.6)	70.0	29.3	NPS	18	33.3
Santa Catalina	30	13	15.5 (±7.7)	13.5 (±8.0)	1913 (±34.5)	90.4 (±38.5)	96.7	73.1	USFS	5	100.0
All	408	91	16.9 (±10.7)	14.4 (±10.4)	1895 (±47.1)	116.3 (±52.6)	50.9	42.4	NA	102	54.9

Fire severity

To generate a severity threshold in modeled CBI with which to compare the 1.73 CBI threshold calculated from FIA data (Woolman et al., *in review*), we used a logistic regression model with data from our field plots. Analyses used 834 overstory (dbh ≥ 12.7 cm) trees at 87 plots after filtering for size and our assessment of whether, if dead, they were killed by the most recent fire. By species, the trees 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 the following: Rocky Mountain maple (*Acer glabrum*), Arizona madrone (*Arbutus arizonica*), and alligator (*Juniperus deppeana*) and Utah juniper (*J. osteosperma*). Our model (Fig. 6) yielded a modeled CBI threshold of 1.61, which is a conservative threshold as it indicates 50% probability of overstory mortality - in other words, the threshold above which the probability of tree mortality exceeds that of survival.

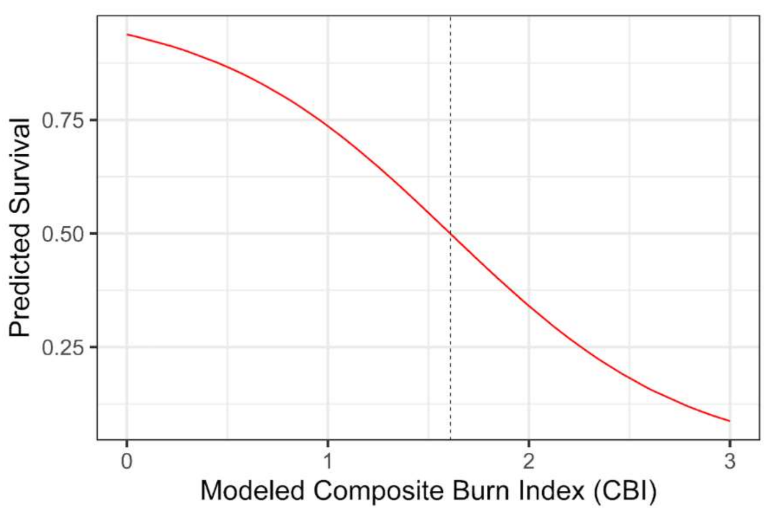

Figure 6. Predicted overstory tree (dbh ≥ 12.7 cm) survival as a function of satellite-derived Composite Burn Index (CBI) of most recent fire. Dashed line represents the 1.61 modeled CBI threshold calculated from the field-measured overstory tree mortality.

Applying this threshold, of the 208 burned sites with tree-ring fire history and contemporary burn severity data available, 42.4% burned at high mortality probability severity in first-entry contemporary fires. We broke this down by management strategy, finding that 14.8% of sites in wildland fire use, 15.4% in prescribed fires, and 53.4% of sites in suppressed wildfires burned at severity levels exceeding 1.61 CBI (Figure 7). Proportion of sites burning above that threshold varied by geographic area, from 0% of the Kaibab Plateau sites to 73.1% in the Santa Catalina Mountains (Table 1). Compared with 42.4% of sites burning at severity levels exceeding the 1.61 modeled CBI threshold in the first contemporary fire, 35.7 and 0% of sites experienced fire more likely than not to kill overstory trees in the second and third fires, respectively.

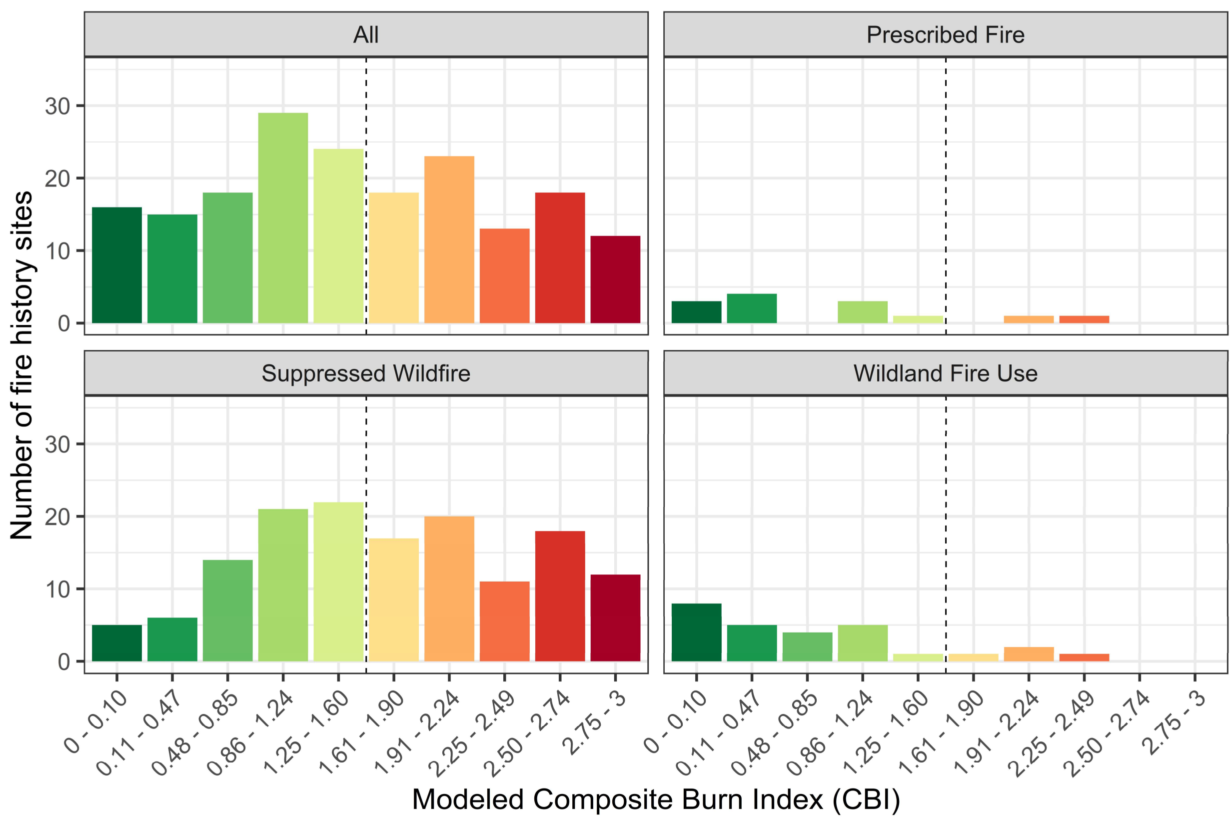

Figure 7. Burn severity, given in modeled Composite Burn Index (CBI), for fire history sites in their initial fire event 1985-2020, in all fire types, prescribed fires, suppressed wildfires, and in wildland fire use fires. Sites were filtered by our ability to assign fire type and CBI value, so n = 186 for all, 13 for prescribed fire, 146 for suppressed wildfire, and 27 for wildland fire use. Divisions correspond to CBI severity classes subdivided by thirds, wherein unburned areas display values ranging from 0-0.10, low severity 0.11-1.24, moderate severity 1.25-2.24, and high severity 2.25-3. Dashed line represents a CBI value of 1.61, corresponding to the threshold above which the probability of overstory tree mortality exceeds 50% (high P(mort)).

Assessing drivers of contemporary fire frequency

We built a series of individual negative binomial models, using all sites, both burned and unburned in the contemporary period (n = 408). No correlations were found between number of fires and MFI or median FI. We found that the last fire year as documented in the tree-ring data was positively correlated ($P < 0.001$) with the number of fires occurring between 1985 and 2020, as were the disruption metrics we calculated as log-transformed ratios between mean ($P < 0.001$) and median fire return interval with fire-free interval ($P < 0.01$). FFI itself was negatively correlated ($P < 0.001$) with the number of contemporary fires. The best fit model for predicting the number of contemporary fires, identified using model selection, contained only FFI (Table 2).

Table 2. Results, given as model coefficients, from best fit linear models predicting contemporary fire frequency, in number of fires 1985-2020. N = 408. Asterisks indicate significance as follows: *** $P < 0.001$, ** $P < 0.01$, * $P < 0.05$, **.** $P < 0.1$.

Predictor variable	Field CBI
Fire-free interval (FFI; yrs)	-4.06 ***
Model intercept	-0.78 ***

Assessing drivers of contemporary fire severity

No relationships ($P < 0.05$) were found between contemporary burn severity of initial fire and the historical fire regime characteristics examined, including mean and median fire return intervals, last historical fire year, and fire-free interval. When tested individually, statistically significant ($P < 0.05$) predictor variables for severity of the first contemporary fire included precipitation (–), CWD (+), VPD (+), AET (–), and fire type (+, wherein suppressed wildfire correlated with higher severity). In predicting burn severity of the second contemporary fire, precipitation (–), T_{max} (+), VPD (+), fire type (+), and severity of first contemporary fire (+) were significant factors. A model selection process indicated a best linear model for severity of initial contemporary fire consisting of maximum temperature, VPD, and fire type as predictor variables, while the best model predicting severity of the second fire included severity of first fire, VPD, last historical fire date, FFI, and the log-transformed ratio of MFI and FFI (Table 3).

Table 3. Results, given as model coefficients, from best fit linear models predicting modeled Composite Burn Index (CBI) of first and second contemporary fire. Climate variables calculated for time of corresponding fire. N = 182 sites included for CBI of first fire, n = 67 for CBI of second fire. Asterisks indicate significance as follows: *** $P < 0.001$, ** $P < 0.01$, * $P < 0.05$, **.** $P < 0.1$.

Predictor variable	CBI, first fire	CBI, second fire
Max temperature (T_{max})	-0.76 **	NA
Vapor pressure deficit (VPD)	0.64 **	0.44 *
Last historical fire date	NA	-5.74 *
Fire-free interval (FFI; yrs)	NA	-5.72 *
log(MFI:FFI)	NA	-0.35 **.**
Fire type (binary)	0.30 ***	NA
CBI, first fire	---	0.24 *
Model intercept	0.34 ***	5.63

We tested for influences upon the following field-derived response variables: field-measured CBI, overstory tree mortality, conifer seedling density and conifer sapling density, using modeled CBI, historical fire regime characteristics, slope, and aspect as predictor variables. For the purposes of these models, we filtered out any plots which burned more than once, as attributing fire effects of the most recent fire was difficult (n = 42). All variables were scaled to between 0 and 1. No relationships ($P < 0.05$) were found for MFI, median FI, slope, or aspect on any of the response variables when tested individually. None of the tested predictors showed a relationship ($P < 0.05$) to conifer seedling or sapling density.

Last historical fire was negatively correlated with field CBI ($P < 0.01$), as was the log-transformed ratio of MFI and FFI ($P < 0.05$), and the log-transformed ratio of med_FI and FFI ($P < 0.05$). FFI positively predicted field CBI ($P < 0.01$), as did modeled CBI ($P < 0.001$) and fire type ($P < 0.001$). Last historical fire was negatively correlated with percent overstory mortality ($P < 0.01$). FFI was positively correlated with percent overstory mortality ($P < 0.01$), as was the modeled CBI ($P < 0.001$), field CBI ($P < 0.001$), and fire type ($P < 0.05$).

We ran model selection processes for each response variable with significant relationships found (Table 3). For predicting field CBI, the best fit model included modeled CBI and fire type (df = 39, $P < 0.001$). For overstory tree mortality, the predictors in the best fit model were field CBI, modeled CBI, last historical fire, FFI, and the log-transformed ratio of MFI and FFI (df = 36, $P < 0.001$).

Table 4. Results, given as model coefficients, from best fit linear models predicting field-measured variables; severity in modified Composite Burn Index (CBI) and percent overstory mortality. N = 42. Asterisks indicate significance as follows: *** $P < 0.001$, ** $P < 0.01$, * $P < 0.05$, **.** $P < 0.1$.

Predictor variable	Field CBI	Overstory tree mortality (%)
Modeled CBI	0.40 **	0.45 **
Fire type (binary)	0.20 **.**	NA

Fire-free interval (FFI; yrs)	NA	-1.54 .
Last historical fire date	NA	-2.35 **
Log(MFI:FFI)	NA	1.00 ***
Field CBI	—	0.96 ***
Model intercept	0.10	1.26

Discussion

Here, we brought together two large-scale, long-term datasets to quantify the extent of fire regime change in dry conifer forests in Arizona and New Mexico. Following a widely documented fire deficit (Marlon et al., 2012), recent years have brought increased occurrence of both high-severity and large-scale wildfires (Mueller et al., 2020; Parks & Abatzoglou, 2020). We found that contemporary fires (1985-2020) are burning less often and more severely than historical fires (1700-1880) in these systems. Historical fire regimes were dominated by low- and moderate-severity, high-frequency fire that shaped the composition and function of dry conifer forest ecosystems. Such a dramatic departure in recent years has significant implications for ecosystem resilience (Falk et al., 2022). In contrast to our expectations, specific characteristics of historical fire regimes did not influence contemporary burn severity. Instead, and demonstrating important opportunities for managing these forest types, fire-season climate and type of fire were the key predictors of burn severity, of those factors analyzed here.

Fire frequency

Our study confirms a decrease in overall fire frequency within these dry conifer forests. Based on our calculations using ten-year moving averages of fires per year, the overall average fire frequency has decreased by 57.0%, with fires burning once per 12.5 years on average in the 18th

and 19th centuries, compared with once per 100.0 years in the 20th, and once per 50.0 years in the 21st. Further, 49.0% of fire history sites have not burned in the 35-year contemporary period examined. Other studies have demonstrated the divergence of contemporary fire frequency from historical ranges of variability in fire-adapted forests of the western US (Falk et al., 2011; Hessburg et al., 2019; Lavorel et al., 2006) and specifically in southwestern dry conifer forests dominated by ponderosa and other pine species (Dewar et al., 2021; Swetnam et al., 2016).

Across the 408 fire history sites, the number of contemporary fires was positively predicted by the date of the last historical fire and the log-transformed ratio of MFI and FFI. The number of fires was negatively correlated with FFI. These significant relationships may be purely mathematical, wherein sites with a shorter fire-free interval, or later last historical fire date, had less time to burn more than once in the contemporary period. Mean and median fire return intervals did not influence contemporary fire frequency at those sites, which runs counter to our hypotheses wherein sites adapted to more frequent fire would experience higher rates of burning.

Fire severity

To make comparison between contemporary burn severity and historical fire effects possible, we calculated threshold values of modeled CBI, below which we expect all historical fires recorded in the tree-ring fire-scar record would fall if their satellite-derived burn severity data could be analyzed. Of the two thresholds we generated from logistic regression models, we applied the one derived from our own field data (1.61) rather than from FIA data (1.73; Woolman et al., *in review*). Despite using a smaller sample of trees to build this model, we deemed it more applicable to our study due to the broader suite of species and ecosystems included, as the FIA data were restricted to ponderosa pine trees. Both thresholds fell in the middle of the moderate severity classification per CBI field protocol, corresponding to CBI values of 1.25 – 2.24 (Key & Benson, 2006). This moderate threshold was chosen given that repeat fires at these severity levels would lead to steep declines in the survival rate of overstory trees.

As calculated using our CBI threshold value of 1.61, of those 51.0% of fire history sites which burned in the contemporary period, 42.4% did so at anomalously high severity when compared with historical norms. This finding bolsters other research which showed increasing burn severity over recent decades in southwestern forests (Dillon et al., 2011; Mueller et al., 2020; Parks & Abatzoglou, 2020; Singleton et al., 2019). Recent studies comparing contemporary burn severity and frequency to historical patterns based on LANDFIRE Biophysical Settings (available at www.landfire.gov) have found significant fire deficits in low- to moderate-severity fire particularly in low- to mid-elevation forests in the Sierra Nevada (Mallek et al., 2013), and dry conifer forests in the Pacific Northwest (Haugo et al., 2019), in other systems historically characterized by low- to moderate-severity, high-frequency fire. We demonstrated that a similar departure in low- to moderate-severity fire can be seen in dry conifer forests in the southwest, situating these trends in a broader context by cross-referencing satellite and tree-ring data.

The most significant factors predicting burn severity of the initial contemporary fire were fire season mean climate factors, especially precipitation, vapor pressure deficit (VPD), climate water deficit (CWD), and actual evapotranspiration (AET; see Table 2). The influence of VPD, due to its consequent impact on fuel aridity, have been well documented (Higuera & Abatzoglou, 2021; Marlon et al., 2012; Parks & Abatzoglou, 2020). Further, atmospheric drying, measured in VPD, is predicted to increase in the century ahead (Ficklin & Novick, 2017).

Our study found a correlation between fire type and severity of initial contemporary fire, wherein sites burned in suppressed wildfires exhibited higher severity than those burned in other fire types. This is consistent with prior research on the impacts of management strategy on severity, with controlled fires, either natural starts managed for resource benefit or prescribed burns, tending to result in lower-severity fire effects than wildfires managed under a full-suppression strategy. The ability of prescribed fire to moderate severity of subsequent wildfires has been well documented in general (Fernandes & Botelho, 2003) and in ponderosa pine forests (Pollet & Omi, 2002). Wildfires managed for resource benefit have been shown to operate similarly in these systems (Collins & Stephens, 2007a; Coop et al., 2016; Parks et al., 2014). Indeed, in our study we found that sites burned in managed wildfires showed slightly lower severity on average than those burned in prescribed fires.

Although in general second-entry fires burned less severely, severity of first contemporary fire was positively correlated with severity of the second. We suspect this relationship stems from the presence, in the second fire, of higher loading of dead fuels in the form of standing dead trees and downed logs resulting from the initial fire, along with higher shrub cover in some cases (Coop et al., 2016; Harris et al., 2020; Parks et al., 2014).

Finally, in assessing factors which impacted burn severity among our complete sample of 208 burned (1985-2020) fire history sites, we found that no historical fire regime characteristics had an effect, which surprised us. We hypothesized that sites with lower MFI and longer fire-free interval (FFI) would show higher severity in their initial contemporary fire, consistent with our understanding of the impacts of such fire regime departure (Schoennagel et al., 2004). Along with the larger influence of fire-season climate factors, particularly as more extreme weather conditions are seen (Ficklin & Novick, 2017; Higuera & Abatzoglou, 2021), this result could be explained by the extent of departure from historical trends. Research has shown that the effect of prior fire to mitigate fire spread persists for only about 6 years in forest types common in the southwestern US (Parks et al., 2015). Although this figure addresses spread, not severity as such (which is likely closer to 20 years), 75.5% of the fire history sites in our study had an FFI exceeding 100 years, with a maximum of 338. Thus, it is possible that we did not detect a significant effect from fire-free interval because the length of time elapsed far exceeded those 6-20 year thresholds.

In field data collection for this study, we were able to relocate sampled trees at just over a third of the 74 fire history sites attempted, which burned after original sampling. When examining effects of management legacies, we found differences in burn severity between field-sampled geographic areas along fire type and management agency lines. Areas burned initially in wildland fire use or prescribed fire as opposed to suppressed wildfire and administered by the National Park Service as opposed to the US Forest Service experienced lower rates of high mortality fire (Table 1).

Regeneration was generally low in our field data collection, with no seedlings recorded in 52.7% of plots, no saplings in 56.8%, and no live trees of any size in 14.9%. These findings add to a growing body of research which documents depressed tree regeneration in southwestern dry conifer forests (Coop et al., 2016; Davis et al., 2019; Rodman et al., 2020; Stevens-Rumann & Morgan, 2019), paired with high tree mortality in conifer forests more broadly (Cansler et al., 2020; Hood et al., 2018; van Mantgem et al., 2009; Woolman et al., *in review*). These dynamics can ultimately lead to lasting conversion from forested to non-forested ecological states (Coop et al., 2020), especially given that both decreases in regeneration (Savage et al., 2013) and increases in mortality of mature trees (McDowell & Allen, 2015) are likely to continue in the coming decades with increasing effects of warming and drought.

In modeling drivers of field-measured severity and overstory tree mortality, as expected we found that modeled CBI strongly positively predicted both field CBI and overstory tree mortality. However, we also found that FFI positively predicted both field CBI and percent overstory mortality, and last historical fire was negatively correlated with both responses, which supports our hypothesis that a longer FFI and earlier end date of historical regimes would lead to more severe fire effects, but runs counter to our findings in the broader sample. We found this incongruence perplexing. It is important to note that fire-free interval provides a proxy for fuel conditions at these sites, and an imperfect one at that. Land use impacts such as logging and grazing significantly alter fuel beds in these systems, and as such one possible explanation for the different model findings lies in the fact that most of our field sites were in areas protected from extractive industries, such as wilderness areas or National Parks. The lack of logging means that when fire was removed from these forests, fuels subsequently built up relatively unchecked (Swetnam et al., 1999).

Study limitations and suggestions for further research

A few caveats and potential areas of uncertainty were implicit in our study. There is some potential for a severity bias in site locations, as only areas having burned in lower severity provide a substantial tree-ring record. Comparisons between fire types and geographic areas were unfortunately subject to different sized samples, and some sites were excluded as we could not classify every site by fire type. Imperfections and inconsistencies in data on wildland fires via sources such as MTBS, NIFC, and local fire atlases inevitably resulted in some small, older contemporary fires (eg. in the late 1980s and early 1990s) being missed and thereby excluded from analysis (eg. see Farris et al., 2010). The classification of Wildland Fire Use was only applied for a portion of our study timeframe, so we assigned it to other wildfires known from fire atlases and other sources to have been managed for resource benefit as well. We did not incorporate any direct metrics of fuel conditions (including pre-fire composition, moisture, continuity) when assessing drivers of fire frequency or severity, as discussed above.

These findings are particular to dry conifer forests in Arizona and New Mexico, dominated by species of trees which readily record scars from low- to moderate-severity fire such as ponderosa pine, Southwestern white pine, and Douglas fir. The availability of the new NAFSN database provides ample opportunity to conduct similar analyses in different geographic areas across the continent, or indeed on a much broader scale (Margolis, Guiterman, et al., *in press*).

Conclusions and management implications

Overall, our findings add to a growing body of evidence for substantial differences between historical and contemporary fire regimes, in terms of frequency, severity, extent, and other facets such as spatial heterogeneity within fire footprints. Findings of several studies which assert that patterns today are not divergent from those seen prior to Euro-American colonization (Williams & Baker, 2011, 2012; and others) have been carefully, systematically invalidated based upon methodological inaccuracies and unsupported logical inferences (Collins & Stephens, 2007; Farris et al., 2010; Farris et al., 2013; Fulé et al., 2014; Fulé et al., 2003; Hagmann et al., 2021; O'Connor et al., 2014; Van Horne & Fulé, 2006; and others). While our methods did not attempt to capture instances or prevalence of high-severity fire prior to 1985, as did those of Williams & Baker, our findings affirm those of other studies which demonstrate significantly more high-severity fire specifically in areas with known patterns of frequent, low- to moderate-severity fire (Haire & McGarigal, 2010; Savage & Mast, 2005).

We do not know how representative the sites are of the landscapes overall, as some researchers may have covered a given area systematically, while some may have sampled opportunistically. We feel that analyzing data together from many different study designs is appropriate however, and indeed is a strength of this research, due to comparative studies within our study area showing that targeted and random sampling tends to generate similar fire frequency and area burned estimates (Farris et al., 2013; Van Horne & Fulé, 2006). Bringing data from different researchers and approaches thus can only serve to diminish bias in the overall sample.

In ecosystems with markedly altered structure, function, and disturbance regimes, intentional and informed management strategies are essential (Hessburg et al., 2021), particularly as disturbance frequency and severity is expected to accelerate in the coming decades (Higuera & Abatzoglou, 2021). Our findings show that areas not burned in suppressed wildfires, but rather in wildfires managed for resource benefit or prescribed fires, were more likely to experience fire with low probability of tree mortality. This aligns with other research, where moderate-severity wildfires managed for resource benefit have been shown to achieve stated objectives (Huffman et al., 2017), emulate historical forest structure (Collins et al., 2017), and increase forest resilience to future fire (Harris et al., 2020). Prescribed fire in our study area and timeframe was even less prevalent than the rare wildland fire use, and apart from the southeast, the application of prescribed burning in the US has been not only minimal, but decreasing in the past two decades (Kolden, 2019). We argue that increased application of both categories of controlled fire is warranted in our study area, along with other management tools for lightening loads of both live and dead fuels.

The fact that burn severity was predicted primarily by climatic factors, again with little to no impact from historical fire regime characteristics, bears both ominous and opportune implications for management. Future climate predictions suggest overall drops in ideal weather windows for either letting natural starts burn or starting prescribed fires. However, our findings empower managers to aggressively pursue those windows which do arise, without hindrance from constraints based on factors outside of immediate fuel and weather conditions. Studies have shown repeatedly that in the absence of extreme fire weather (Parks et al., 2015), fuel conditions are the primary factor influencing fire effects, in terms of connectivity (Archibald et al., 2012), amount (Parks et al., 2018), and moisture (Parks et al., 2014a).

Regardless of management intensity, in the presence of warmer and drier future conditions, current strategies appear to be inadequate to maintain structure and function of dry forest ecosystems in the southwest (Loehman et al., 2018), highlighting the need for development, testing, and adaptive application of novel approaches to fire management. Increased application of managed wildfire and prescribed burning is necessary to increase effective treatment but is especially difficult given that public perception and opinion in the western US are not favorable or forgiving when it comes to detrimental effects of controlled fire on human communities (Kolden, 2019). When facing these daunting challenges, we truly need input from as wide a range of perspectives as possible. As such, and paired with our ancestral legacy of imperialism and genocide, restoration of Indigenous fire stewardship and integration of diverse stakeholder collaboration is both just and pragmatic (Adlam et al., 2021; Hessburg et al., 2021; Lake et al., 2017; Prichard et al., 2021). Task forces and working groups in some states are already making strides to increase application of prescribed fire, especially with emphasis on Indigenous leadership in fire management (California Strategic Plan, 2022; https://fmtf.fire.ca.gov/).

To engage in effective land management, it is often important to strive for a deeper understanding of the ecological conditions which prevailed before Euro-American humans became the overwhelmingly dominant force upon forested systems (Walker et al., 2004). However, in conditions of such dramatically changing land use and climate, striving to replicate these conditions is often impossible (Aplet & Cole, 2010). In such cases, our finding that historical fire characteristics do not dictate the outcomes of contemporary fire lends further credence to a growing body of research which encourages managers to carefully consider facilitating ecosystem reorganization – or directing change – as opposed to managing for persistence or resisting change altogether (Aplet & Cole, 2010; Falk et al., 2022; Millar et al., 2007; Walker et al., 2018).

In the presence of mounting scientific evidence depicting a future characterized by significant climatic and ecological change, adaptive management informed by the best available research is crucial, especially in systems significantly departed from historical conditions (Agee & Skinner, 2005). The key findings of our study are 1) contemporary fire patterns differ markedly from those of historical regimes in southwestern dry conifer forests, exhibiting lower frequency and higher severity, and 2) contemporary burn severity was influenced primarily by fire-season climate variables, with little impact from historical fire regime factors. These underscore the prevailing influence of climate and weather, and thus the difficulty presented by managing forested landscapes into an increasingly hot and dry future. However, our conclusions also suggest opportunities for restoration and conservation where bold and collaborative approaches are applied.

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