Forest Ecology and Management 571 (2024) 122226

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Trees have similar growth responses to first-entry fires and reburns following long-term fire exclusion

Kevin G. Willson [a,*], Ellis Q. Margolis [b], Matthew D. Hurteau [a]

[a] *Biology Department, University of New Mexico, Albuquerque, NM, United States*
[b] *US Geological Survey, New Mexico Landscapes Field Station, Santa Fe, NM, United States*

ARTICLE INFO

Keywords:
Aridity
Dendroecology
Fire effects
Ponderosa pine
Reburn
Managed wildfire
Tree growth

ABSTRACT

Managing fire ignitions for resource benefit decreases fuel loads and reduces the risk of high-severity fire in fire-suppressed dry conifer forests. However, the reintroduction of low-severity wildfire can injure trees, which may decrease their growth after fire. Post-fire growth responses could change from first-entry fires to reburns, as first-entry fires reduce fuel loads and the vulnerability among trees to fire effects, which may result in trees sustaining less damage during reburns. To determine whether trees had growth responses that varied from first-entry fires to reburns, we cored 87 ponderosa pine trees in the Gila Wilderness, New Mexico, USA that experienced 3–5 fires between 1950 and 2012 following long-term fire-exclusion and 67 unburned control trees from the Gila and Apache-Sitgreaves National Forests. We assessed tree growth response to fire by comparing tree-ring growth among burned and unburned trees from two years before to two years after fires. We compared growth between burned and unburned trees using a bootstrapping procedure to calculate annual median tree-ring width index values with 95 % confidence intervals. We compared post-fire growth after first-entry fires and reburns following long-term fire-exclusion. Burned trees had similar growth responses following first-entry fires and reburns, with lower growth during the fire year through two years post-fire compared to unburned controls. Burned tree growth returned to expected rates following these immediate post-fire growth reductions. Interestingly, trees had lower growth during the year before and the year of reburns compared to the first-entry fire, reflecting greater aridity before reburns. Greater aridity may have contributed to larger-than-expected growth reductions following reburns, which could explain similar growth responses to first-entry fires and reburns. Our results indicate that trees had consistent short-term growth responses to low-severity fires following long-term fire-exclusion. As trees retained vigor after multiple fires, managing fires for resource benefit is an effective approach to reduce the likelihood of high-severity fire without long-term negative effects on tree growth.

1. Introduction

Long-term fire-exclusion and increasing aridity has increased the intensity and severity of fires that burn dry conifer forests in the southwestern United States (Hagmann et al., 2021; Kreider et al., 2024; Mueller et al., 2020; Singleton et al., 2019). In response, land management agencies have implemented forest treatments, such as prescribed fire and thinning, to reduce fuel loads and the likelihood of high-severity wildfire. However, these treatments are difficult to apply in remote areas at the pace and scale necessary to prevent widespread forest loss (McDowell et al., 2016; North et al., 2012). Managing fire ignitions for resource benefit (hereafter 'managed wildfire') is a cost-effective strategy to reduce fuel loads over large areas of remote forest (Hunter et al., 2007; Cleaves et al., 1999; Holden et al., 2007), but can cause short-term reductions in growth among surviving trees (Keeling and Sala, 2012). As tree-ring growth often relates to tree vigor and portends mortality after drought and fire (Camarero et al., 2015; van Mantgem et al., 2018), assessing tree growth response to managed wildfire is necessary to determine its viability as a management tool for restoring dry conifer forests under increasingly hot and dry conditions (Gonzalez et al., 2018).

Fire can cause reductions in growth by inflicting damage to the crown, bole, and root structures of surviving trees. Increases in fire intensity often increase fire severity, with greater rates of needle scorch, necrosis in the cambium, cavitation in the xylem, and loss of roots near the soil surface (Hood et al., 2008; Keeley, 2009; Michaletz and Johnson, 2007; Michaletz et al., 2012; Wagner, 1973). As damage to the tree

* Corresponding author.
E-mail address: kevin.willson383@gmail.com (K.G. Willson).

https://doi.org/10.1016/j.foreco.2024.122226
Received 23 May 2024; Received in revised form 14 August 2024; Accepted 16 August 2024
Available online 28 August 2024

increases, it generally causes larger reductions in photosynthetic capacity, resource acquisition, and transport of water, nutrients, and sugars within the tree, which can lower the rate of growth (González-Rosales and Rodríguez-Trejo, 2004; Hood et al., 2018; O'Brien et al., 2010). Dry conifer forests contain tree species, such as ponderosa pine (*Pinus ponderosa*), that are adapted to tolerate frequent, low-intensity fires (Stevens et al., 2020). However, a century of fire-exclusion has resulted in greater stem density, more surface fuels, and trees with crowns and roots that are closer to the flaming front during fire, increasing their vulnerability to fire (Brown et al., 2019; Moore et al., 2004; Swezy and Agee, 1991; Westlind and Kerns, 2017). If greater fuel loads and tree and forest structural attributes increase fire intensity beyond thresholds trees are adapted to tolerate, the first-entry fire following long-term fire-exclusion may inflict more damage and cause larger growth reductions among trees than expected in fire-maintained forests.

The damage from fire fades over time as surviving trees grow new needles and roots, return to pre-fire growth rates, and drop needles and branches, reinitiating the process of fuel accumulation (Bär et al., 2019; Wenderott et al., 2022; Westlind and Kerns, 2017). In dry conifer forests, maintaining fuels conditions that cause fires to burn at low-intensity and severity requires regular fire (Buma et al., 2020; Parks et al., 2014b; Rodman et al., 2023). While reburns may also reduce tree growth, reductions in fuel loads, fire intensity, and the susceptibility among trees to fire effects should lead to smaller post-fire growth reductions compared to the first-entry fire (Brown et al., 2019; Coppoletta et al., 2016; Holden et al., 2007). However, increases in aridity can reduce tree vigor and increase the proportion of live and dead fuels available to burn by reducing fuel moisture, which may cause greater-than-expected growth reductions after fire relative to the amount of fuel on the landscape (Goodwin et al., 2021; Kolb et al., 2019; van Mantgem et al., 2018; Williams et al., 2019). As an increasing proportion of fires burn forests under drier conditions (Abatzoglou et al., 2021), tree growth following fire may deviate from expected patterns, creating uncertainty about changes in tree response from first-entry fires to reburns after long-term fire-exclusion.

Given the uncertainty of tree growth response to fires following long-term fire-exclusion, we asked: How does the first low-to-moderate-severity fire after long-term fire-exclusion affect tree growth relative to reburns at the same location? We hypothesized that the first-entry fire following fire-exclusion would cause a larger decrease in tree-ring growth during the year of and the year after fire compared to reburns because reburns would burn lower fuel loads, have lower fire intensity, and cause less damage to the tree. While fire weather at the time a particular location burns can influence fire effects at that location, our analysis is a multidecade retrospective, which limited data availability for potential predictor variables.

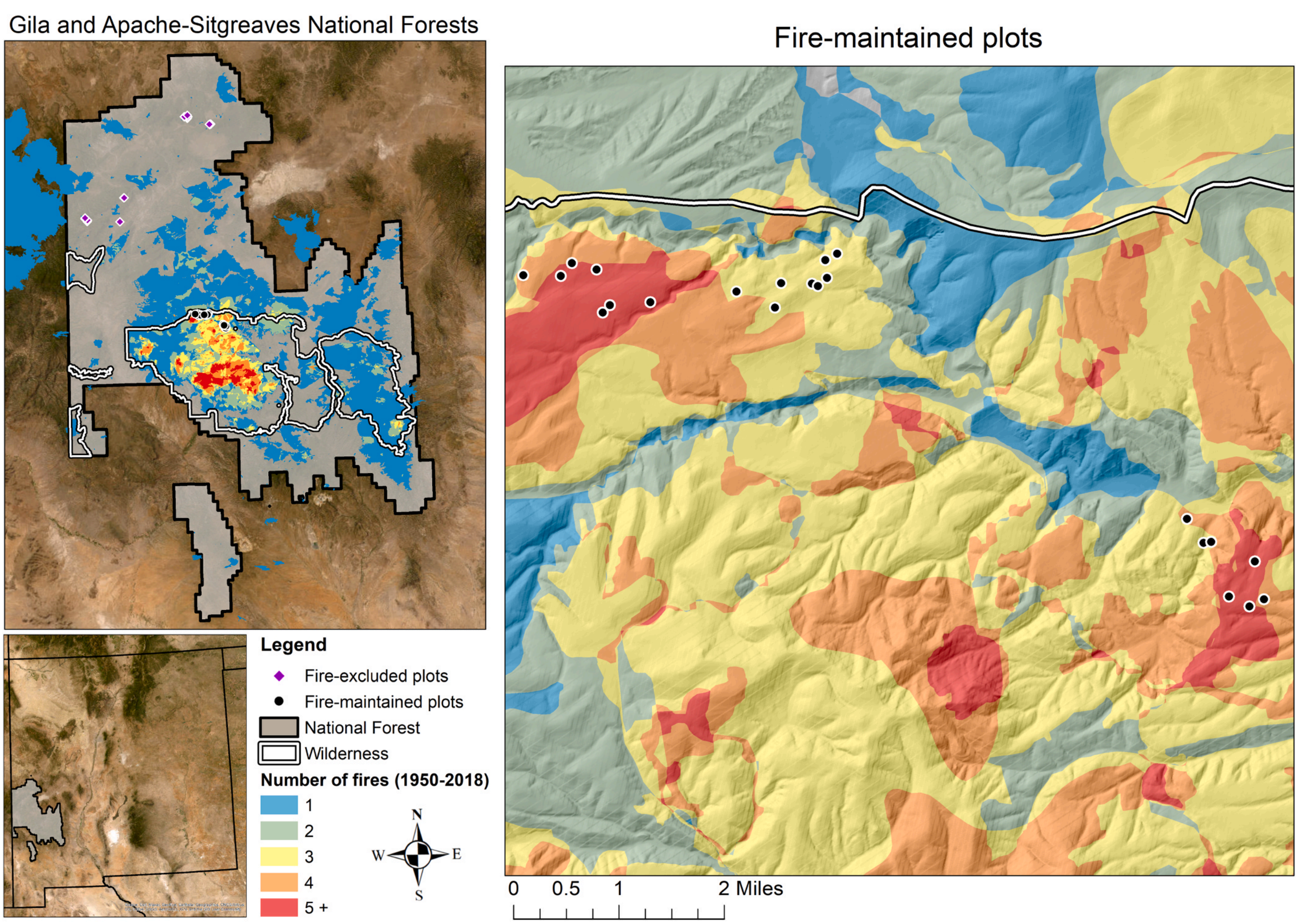

Fig. 1. Plot locations across the Gila Wilderness and Gila and Apache-Sitgreaves National Forests, New Mexico, USA. The lower left panel shows the location of the Gila within New Mexico. The upper left panel shows the sampling locations of fire-maintained and fire-excluded plots and the number of fires that have burned locations in the region. The right panel shows the number of fires that have burned sampling locations in fire-maintained forests. Credits for the basemap include: Esri, Maxar, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community.

2. Materials and methods

2.1. Study sites

We collected data from ponderosa pine forests in the Gila and Apache-Sitgreaves National Forests, located in west-central New Mexico (Fig. 1). Ponderosa pine forests in the region historically burned at low to moderate severity approximately every 5–20 years (Swetnam and Baisan, 1996), resulting in stands with low tree density (48–123 trees ha^{-1}; Garrett and Soulen, 1999; Moore et al., 2004; Roccaforte et al., 2015; Ryan, 2002). Around the turn of the 20th century, widespread overgrazing and active fire suppression increased fire-return intervals by more than 120 years in some areas (Swetnam and Baisan, 1996), which increased the rate of ponderosa pine seedling survival and recruitment into the overstory (Covington and Moore, 1994). Increased seedling survival and recruitment caused stand density to increase by 5–13-fold over the past century, resulting in greater fuel loads on the landscape (Holden et al., 2007; Moore et al., 2004). With the establishment of a wildland fire use policy in 1975, which allowed managers to manage lightning ignitions for resource benefit in wilderness (van Wagtendonk, 2007), wildfires have burned throughout most of the Gila Wilderness, with some locations experiencing surface fires at frequencies that are comparable to historical rates (Parks et al., 2023, Fig. 1). However, forests outside of designated wilderness have experienced few fires since the early 1900s, with many areas remaining fire-excluded.

Vegetation in the Gila ranges from desert grasslands at lower elevations to ponderosa pine forests at mid elevations and subalpine conifer forests at upper elevations (Keane et al., 2000). Climate in the study area is semi-arid, with a mean annual temperature of 12 °C and mean annual precipitation of 385 mm between 1990 and 2020 (https://www.ncdc.noaa.gov/cdo-web/datatools/normals, Gila Hot Springs station, NM; 1706.9 m). Precipitation is bimodal, with snow common in the winter and rain in the summer monsoonal period (Sheppard et al., 2002). Soils in the region are a mixture of Ustalf and Ustoll suborders (NRCS, 2022).

2.2. Data collection

We established plots to collect growth and stand structure data across dry conifer forests that contained ponderosa pine and experienced no fire or multiple surface fires since the establishment of fire suppression policies. We identified forests containing ponderosa pine using the LANDFIRE dataset to determine potential areas for data collection (Rollins, 2009). Within these forests, we identified areas that either have not experienced fire since 1909 or burned multiple times at low-to-moderate-severity since 1950 using three sources collectively referred to as fire atlas data, including: 1) burn severity data from the Monitoring Trends in Burn Severity dataset (MTBS; Eidenshink et al., 2007), 2) corrected fire boundary data from Parks et al. (2015), and 3) historical fire boundary records from Rollins et al. (2002). Within these areas, we ground-truthed and sampled locations in the Gila Wilderness that experienced at least three surface fires between 1950 and 2019 (hereafter fire-maintained forests) and places with similar elevation, slope, and aspect on the Gila and Apache-Sitgreaves National Forests that had no evidence of fire since 1909 (hereafter fire-excluded forests; Fig. 1). Prior to establishing plots, we ground-truthed fire-maintained areas to confirm that they had burned by visually surveying for char, reduced woody debris, differences in stand structure, and increased canopy base height compared to fire-excluded forests (Brown et al., 2019; Holden et al., 2007; North et al., 2009).

We established 22 plots in fire-maintained forest and 20 plots in fire-excluded forests to collect stand structure data, from which we identified dominant and co-dominant candidate trees to collect growth data. To quantify stand structure, we used a 0.20 ha circular plot to measure trees >50 cm diameter at breast height (dbh), a nested 0.10 ha plot to sample trees 15–50 cm dbh, and a nested 0.02 ha plot to sample trees 5–14.9 cm dbh. Within each plot, we recorded dbh and species of all individuals, delineating all overstory ponderosa pine trees (dbh > 20 cm) as candidates for increment coring. We randomly selected 3–7 candidate trees in each plot to core using a 5.15 mm increment borer. We sampled two cores from each of 87 trees in fire-maintained plots and 86 trees in fire-excluded plots. Field data collection occurred between 2020 and 2022.

We followed standard dendrochronological methods to dry, mount, and sand increment cores in preparation for crossdating and growth measurements (Stokes and Smiley, 1968). We scanned sanded cores using an Epson 12000XL scanner at high resolution to date and measure tree rings. We measured annual tree ring-widths to 0.001 mm using WinDENDRO (Regent Instruments, 2018). We statistically cross-dated cores between 1900 and 2018 using COFECHA (Holmes, 1983). Cross-dated series had intra-site correlation values that averaged 0.76 ± 0.08 standard deviations and inter-site correlation values that averaged 0.74 ± 0.07 standard deviations. Once cross-dated, we averaged annual tree ring-widths between the two cores collected from each tree and detrended the resultant values using the modified negative exponential detrending method, which removes age-related growth trends (Cook and Kairiukstis, 1990). This calculation provided tree-ring width index (RWI) values that represented standardized annual growth among trees.

2.3. Data analysis

To determine if tree growth varied following first-entry fires and reburns, we compared growth among trees during the years during the year of and the two years before and after each fire. We delineated years that trees experienced fire by overlaying the footprints of known fires from the fire atlas data on plot locations in fire-maintained forests. We defined the year a tree experienced a first-entry fire as the year of the earliest fire with a footprint that intersected the plot containing the tree. We defined years that a tree experienced reburns using the years of all fire footprints that intersected the plot after the first-entry fire. As plots within fire-maintained forests were located up to 13 km from one another (Fig. 1), sampled trees had a variety of fire histories, with trees experiencing their first-entry fire in one of five fires between 1950 and 1997 and two-to-four reburns from a subset of seven fires between 1985 and 2012 (Table 1).

The variability in fire histories suggested that trees experienced a variety of climatic conditions surrounding fires. In dry conifer forests, climate influences annual tree growth, fire characteristics, and fire effects, as greater aridity, often quantified by higher vapor pressure deficit (VPD), results in lower growth rates, increased heat release during fires, and greater fire severity (Goodwin et al., 2021; Kusnierczyk and Ettl, 2002; Wasserman and Mueller, 2023; Williams et al., 2013). We used fire severity to measure and account for variability in the amount of damage stands sustained to photosynthetic structures during fire (Parks et al., 2014a), which can influence tree growth within the stand after fire (Sparks et al., 2023). To quantify variability in fire severity within and between fires, we used the relativized burn ratio (RBR) calculated by Parks et al. (2015), which calculated RBR values at 30 m resolution for all fires beginning in 1984. To quantify variability in aridity during the

Table 1
The number of plots and sampled trees burned in the Gila study area, New Mexico, USA, in different fire-year sequences.

First-entry fire year	Reburn years	Number of plots	Number of trees
1950	1993, 2002, 2007, 2012	3	16
1950	1993, 2002, 2012	1	4
1950	1997, 2002, 2012	1	7
1979	1985, 1993, 2006, 2012	6	18
1985	1993, 2006, 2012	1	3
1993	2006, 2012	8	27
1997	2002, 2012	2	12

study period, we calculated annual mean maximum and 3-year rolling means from VPD from 1940 to 2019 using interpolated monthly maximum VPD (PRISM, 2023).

Because VPD varied across the study period (Fig. 2), we used a subset of trees from fire-excluded forests to control for the effect of climate while assessing changes in growth during the year of and years after fire. To select control trees, we ran an interseries correlation analysis from 1948 to 2018 between individual growth series of trees from fire-excluded forests and a master growth chronology of the 87 trees from fire-maintained forests. We retained 67 of the control trees from fire-excluded forests that had interseries correlation values > 0.65. To determine if the 67 trees represented reasonable controls, we compared their growth response to monthly climate (Meko et al., 2011; Zang and Biondi, 2015) and checked their expressed population signal (EPS) with trees from fire-maintained forests (Wigley et al., 1984). The 67 control trees had similar correlations to precipitation and temperature (Supplemental figure S1) and an EPS value of 0.99 with trees from fire-maintained forests, indicating that they represented reasonable controls for expected growth responses to climate.

To test if fire altered tree growth while controlling for climate, we compared annual growth between burned and control trees during the five-year period surrounding fires. We compared annual growth using median annual RWI values with 95 % confidence intervals, which we calculated using a cluster bootstrap procedure. We used a cluster bootstrap procedure to account for known dependencies in tree growth that may have caused autocorrelation within the five-year windows surrounding specified fire years (Field and Welsh, 2007; Kohyama et al., 2005). Specified fire years included every year a burned tree experienced a fire and all fire years among unburned controls. The five-year windows included the two years before, the year of, and the two years after a fire. We limited windows to the five years surrounding an event to maintain independence while calculating pre- and post-fire growth rates, as some areas burned within five growing seasons of the previous fire (Table 1).

To perform the cluster bootstrap procedure, we grouped sets of RWI values surrounding an event by first-entry fire or reburn and standardized time by the lag year relative to fire. We sampled five sets of RWI values by group with replacement to calculate a mean annual RWI value for burned and control trees over the five-year period surrounding first-entry fires and reburns. We repeated this sampling procedure 1000 times to obtain median and 95 % confidence intervals of RWI values by lag year using the bootstraps() function in the 'rsample' package (Silge et al., 2022). To test if tree growth significantly changed after first-entry fires or reburns, we compared median and 95 % confidence intervals of annual RWI between burned and control trees surrounding the first-entry fire and reburns. To test for differences in tree growth response between first-entry fires and reburns, we calculated the median and 95 % confidence interval of bootstrapped differences in RWI between burned and control trees for first-entry fires and reburns and compared the resulting values.

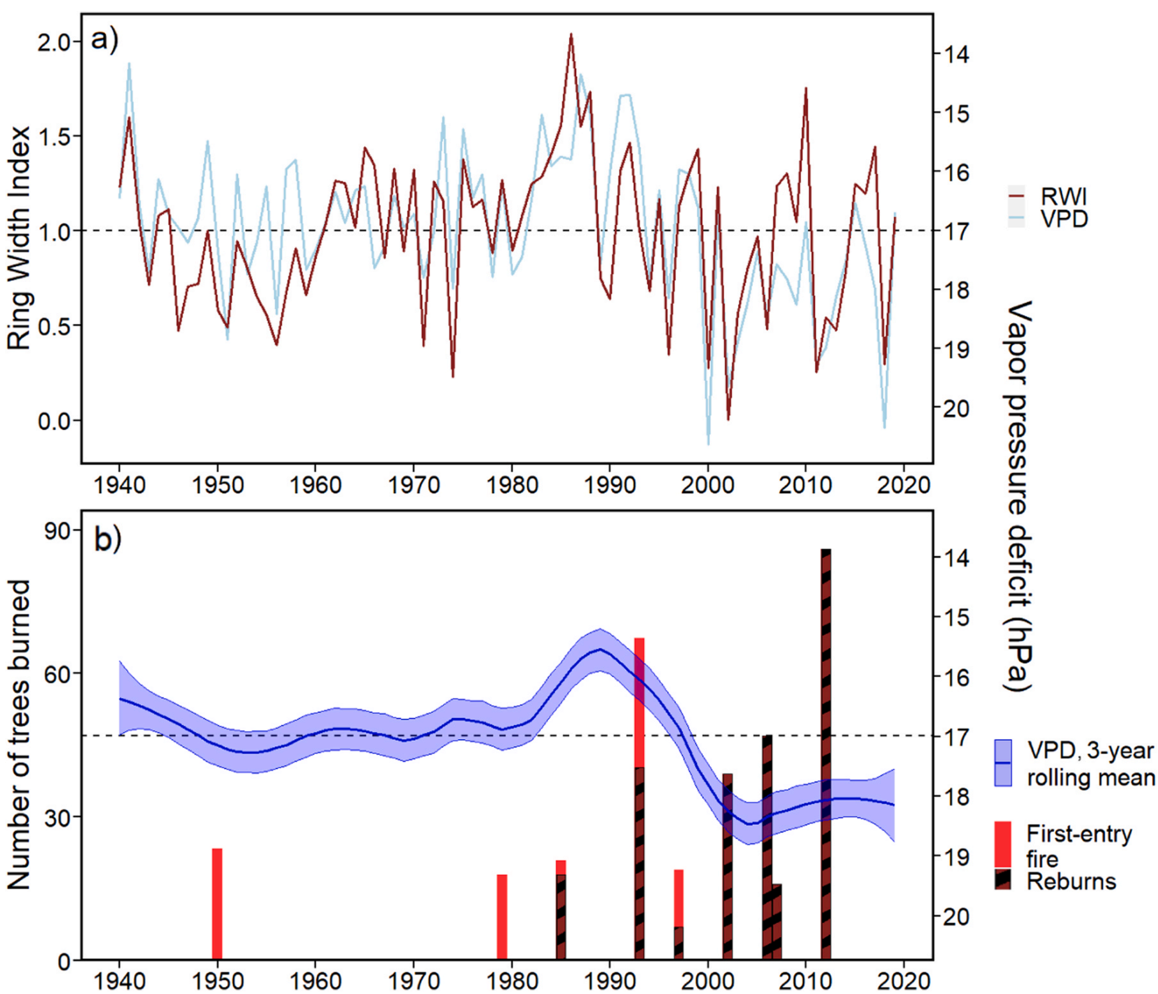

Fig. 2. Vapor pressure deficit (VPD), tree-ring width index (RWI) of burn trees, and the number of sampled trees that experienced fire by year from 1940 to 2019 in the Gila study area, New Mexico, USA. Panel a plots annual VPD and a chronology of RWI from 1940 to 2019. Panel b shows the number trees burned in first-entry fires and reburns (bars) and the 3-year rolling means for annual VPD ± one standard deviation (blue line and shading). The black dashed line represents average annual VPD (17 hPa) over the 80-year period. We calculated the growth chronology using the chron() function from the dplR package in R (Bunn, 2010).

The cluster bootstrap analysis suggested that annual growth rates differed between first-entry fires and reburns during the year before and year of fire. These differences in growth may have indicated differences in aridity before first-entry fires and reburns that can influence fire characteristics and tree growth response after fire (Goodwin et al., 2021; Kolb et al., 2019; Kusnierczyk and Ettl, 2002; van Mantgem et al., 2018). To compare growth rates between first-entry fires and reburns, we used t-tests to assess differences in RWI values by lag year. We included burned and control trees to test for differences in growth during the two years before fire and control trees for the year of and year after fire. The analysis showed that growth rates did not differ between first-entry fires and reburns during the growing season two years before or the year after fire ($p > 0.1$). However, all trees grew at slower rates during the year before reburns and control trees grew at slower rates during the year of reburns.

To determine if aridity and fire severity caused tree growth to vary during periods surrounding a fire, we developed a linear mixed model with a first-order autoregressive covariance structure. We assessed growth response to aridity and severity during years surrounding fires, using annual RWI as the response variable, annual mean maximum VPD and RBR as predictor variables, and a random effect that included a variable denoting plot ID nested by year. We gathered RBR values for all fires each tree experienced by overlaying plot locations on RBR raster data. We assigned RBR values to plots for each fire by identifying the RBR pixel that overlapped with plot center. We distributed plot RBR values to trees within the plot for the year of and the two years after every fire. We assigned trees an RBR value of zero for all other years and excluded data predating 1984 from the analysis, before which fire severity data did not exist (Parks et al., 2015).We checked the normality of model residuals using a Shapiro-Wilk test and assessed all other assumptions graphically. We transformed the response variable with a square root transformation and included the growth of 73 trees from fire-maintained forests to meet model assumptions. All statistical analyses were performed in R version 4.3.2 (R Core Team, 2020).

3. Results

Burned trees had similar growth reductions after first-entry fires and reburns. Burned and control trees had comparable growth rates during the years before fire (Fig. 3, Supplemental figures S2–8). Burned trees had lower growth than control trees during the year of fire (Figs. 4a and 4b), as RWI increased among control trees ($\Delta RWI_{first\text{-}entry}$: 11.08 %; ΔRWI_{reburn}: 8.96 %) but decreased among burned trees ($\Delta RWI_{first\text{-}entry}$: −24.90 %; ΔRWI_{reburn}: −19.93 %). The difference in growth between burned and control trees peaked during the year after fire ($\Delta RWI_{first\text{-}entry}$: −0.55, ΔRWI_{reburn}: −0.42; Fig. 4c). Burned tree growth remained significantly lower than control trees immediately after fire, but growth between the two groups converged within two years after fire. The difference in growth between burned and control trees was similar before, during, and after the fire (Fig. 4c), indicating similar growth responses among burned trees after first-entry fires and reburns.

Aridity increased after the year 2000 and fire severity did not change, yet burned trees had similar growth responses after first-entry fires and reburns (Fig. 2b). While all first-entry fires occurred prior to 2000, the majority of reburns occurred after 2000 and had VPD values that were 5 % greater (17.34 hPa ± 0.60 standard error, SE) during the year before fire compared to the first-entry fire (16.53 hPa ± 0.67 SE). The difference increased to 8 % during the year of fire ($VPD_{first\text{-}entry}$: 16.30 hPa ± 0.31 SE, VPD_{reburn}: 17.55 hPa ± 0.63 SE). Higher aridity was associated with lower growth during the year before ($p_{control} < 0.001$; $p_{burned} < 0.001$) and year of reburns ($p_{control} < 0.001$) compared to first-entry fires, as increases in VPD reduced RWI values (Table 2; Fig. 2a). Despite greater pre-fire aridity, fires burned plots at similar low severities before and after 2000 ($RBR_{pre\text{-}2000} = 35.10 \pm 7.82$ SE; $RBR_{post\text{-}2000} = 28.62 \pm 12.75$ SE, $p = 0.71$, Parks et al., 2015) regardless of the event ($RBR_{first\text{-}entry} = 49.64 \pm 10.64$ SE; $RBR_{reburn} = 27.95 \pm 9.62$ SE, $p = 0.37$) and trees had similar reductions in growth after each fire (Fig. 4). The recent increases in aridity and reductions in pre-fire growth may have contributed to the similarities in post-fire growth response following first-entry fires and reburns.

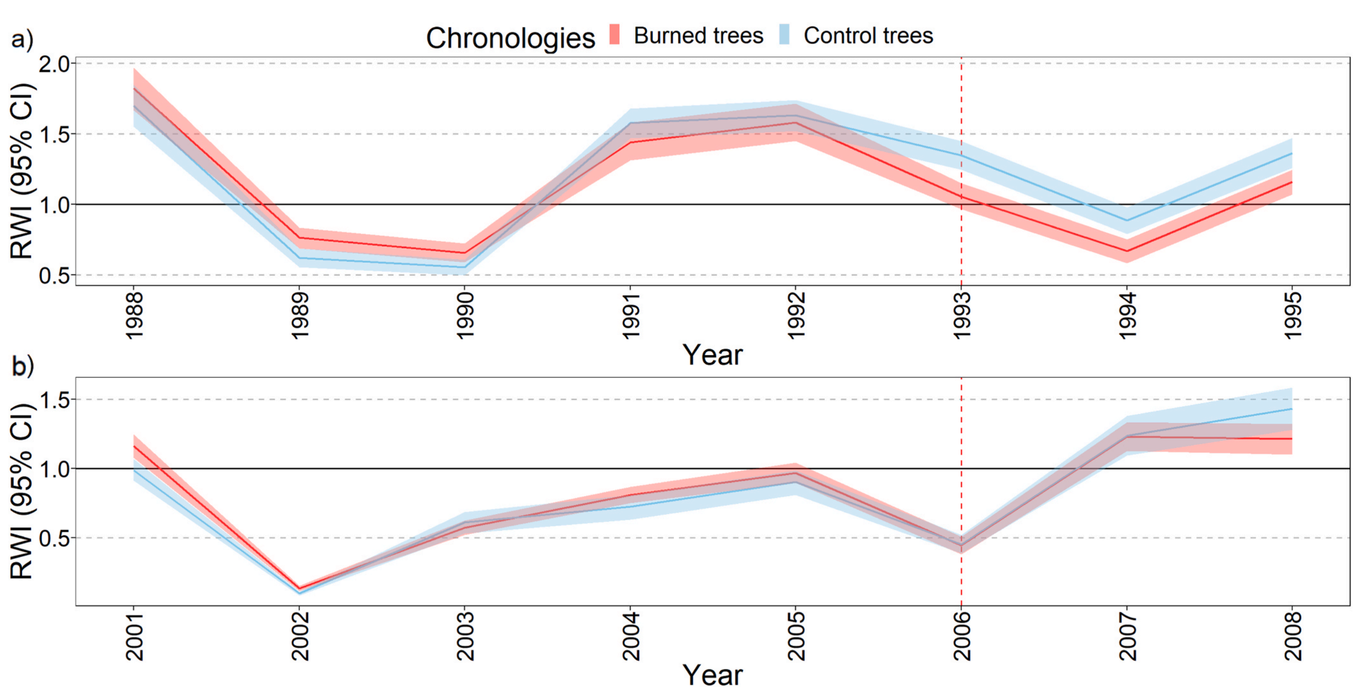

Fig. 3. Growth chronologies of burned (red) and control trees (blue) with 95 % confidence intervals (CI, shading) during the period surrounding fires that occurred in 1993 ($n_{burned} = 68$, $n_{control} = 67$; panel a) and 2006 ($n_{burned} = 48$, $n_{control} = 67$; panel b) in the Gila study area, New Mexico, USA. The solid black line represents average growth (RWI = 1) across all trees from fire-maintained and fire-excluded forests used in the study. Growth chronologies and 95 % confidence intervals were calculated using tree-ring width index values (RWI) and the chron.ci() function from the dplR package in R (Bunn, 2010).

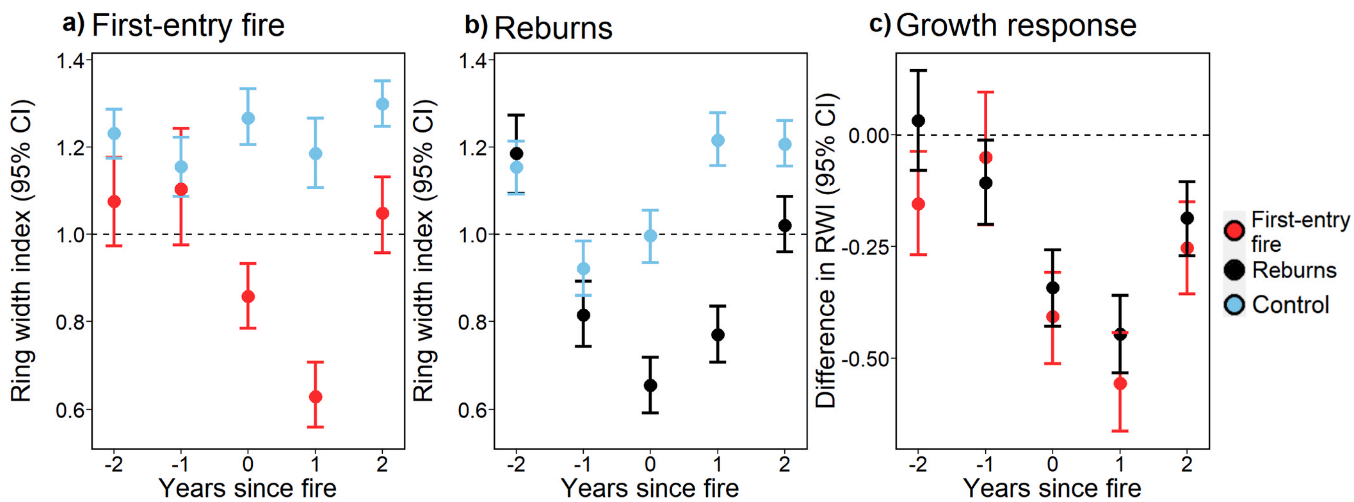

Fig. 4. Annual tree-ring width index (RWI) and differences in RWI between burned (n = 87) and control trees (n = 67) normalized to years since fire in the Gila study area, New Mexico, USA. Panel a represents annual growth rates among burned and control trees surrounding first-entry fires. Panel b represents annual growth rates among burned and control trees surrounding reburns. Panel c compares tree growth response after first-entry fires and reburns, calculated by taking the bootstrapped difference in RWI between burned and control trees. CI = confidence interval.

Table 2
Fixed effects on estimates of tree-ring width index for ponderosa pine trees in the Gila study area, New Mexico, USA.

Parameters	Estimates	SE
(Intercept)	3.3745*	0.1023
Vapor pressure deficit	−0.1383*	0.0058
Relativized burn ratio	−0.0014*	0.0002
Observations	2529	
Marginal R^2/conditional R^2	0.4093/0.6340	

* represents $p < 0.001$.

4. Discussion

Managed wildfire can reduce fuel loads and decrease the risk of dry conifer forests burning at high-severity following long-term fire-exclusion, but these fires can also damage trees (Holden et al., 2007; Rodman et al., 2023). We hypothesized that trees would have larger growth reductions after the first-entry fire than reburns, as greater fuel loads and vulnerability to fire effects would increase the amount of damage trees sustained during the first-entry fire. However, after controlling for climate, both first-entry fires and reburns caused similar reductions in growth during the year of and immediately following the fire.

Tree growth reductions did not vary from first-entry fires to reburns as expected, meaning trees had consistent responses to the damage they sustained from low-severity fires after long-term fire-exclusion. Consistent growth responses to fires with similar effects indicated that, despite recognized differences in fuel loads among fire-excluded and fire-maintained forests (Chamberlain et al., 2023; Westlind and Kerns, 2017), first-entry fires and reburns may have inflicted similar amounts of damage to surviving trees. Low fire severity can damage the leaves of overstory trees, causing needle loss and a reduction in photosynthetic capacity (Bär et al., 2019; Parks et al., 2014a; Wagner, 1973). Consistent severities among fires indicated that stands experienced similar rates of needle scorch during first-entry fires and reburns. Post-fire reductions in photosynthetic capacity often decreases basal growth (González-Rosales and Rodríguez-Trejo, 2004), suggesting that the negative relationship between fire severity and post-fire growth rates may have resulted from needle loss and reductions in photosynthetic capacity among burned trees.

The first-entry fire may have inflicted less damage than expected by burning forests under wetter conditions than reburns. First-entry fires occurred during years characterized by greater tree growth and lower vapor pressure deficit compared to reburns, signaling lower atmospheric water demand that often corresponds to greater fuel moisture (Abatzoglou et al., 2021; Biederman et al., 2016). Greater fuel moisture reduces fuel consumption and fire intensity (Dickman et al., 2023; Jolly, 2007; Rothermel, 1972), which coincides with lower fire severity (Keeley, 2009). Although several of our first-entry fires pre-date the start of the fire severity satellite record in 1984, similar VPD values suggest that fuel moisture was probably similar and all of the first-entry fires in our data set were more likely than not to have burned at similar severities. Greater pre-fire aridity and lower fuel moisture may have increased fuel consumption and fire intensity during reburns to cause fire effects that resembled those of first-entry fires. If so, changing climatic conditions and fuel moisture levels may have caused forests to burn at consistent severities over time that resulted in similar growth responses after first-entry fires and reburns.

Similarities in post-fire growth reductions following first-entry fires and reburns could also be a function of climatic conditions prior to and during the fire events. Climatic conditions were more arid prior to and during reburns as compared to the first-entry fires (Fig. 2), which may have increased tree water stress and reduced photosynthetic output among surviving trees (Biederman et al., 2016; Brodribb et al., 2020). Growth reductions triggered by aridity can cause trees to have lower growth for years after aridity subsides (Peltier et al., 2022). This climate memory may have predisposed trees to lower post-fire growth following reburns, as drought-stressed trees can have larger growth reductions after fire (Li et al., 2023; Sparks et al., 2018). If greater aridity increases tree growth response to the damage fires inflict, projected increases in aridity may cause larger post-fire growth reductions among trees in southwestern forests (Gonzalez et al., 2018). As larger post-fire growth reductions coincide with an increased likelihood of post-fire mortality (van Mantgem et al., 2018), future research should assess the interactive effect of managed wildfire, climate change, and climate memory on tree vigor.

While our results show that trees had similar growth responses to the damage they sustained during first-entry fires and reburns, fire also alters tree water use, canopy architecture, and stand structure that can vary independently from tree growth reductions after fire. Trees can sustain damage to xylem during fire that does not reduce photosynthetic capacity, resulting in post-fire water use that is decoupled from growth rates (Renninger et al., 2013). Fire-maintained forests have trees with greater canopy base height and stands with more spatial heterogeneity than fire-excluded forests, which increases the resilience of forest structure to future fires (Brown et al., 2019; Chamberlain et al., 2023). However, tree growth response may not reflect greater structural resilience if increases in pre-fire aridity cause trees to sustain consistent growth reductions after fire. Therefore, future studies should also assess changes in post-fire water use efficiency, canopy base height, and spatial heterogeneity of horizontal forest structure to determine if other physiological, structural, and ecological responses differ from first-entry fires to reburns.

Fire is a management tool that reduces fuel loads at a consistent short-term cost to forest productivity. While prescribed fire is a popular method for returning low-severity fire to fire-adapted systems, the accompanying legal requirements and restrictions, coupled with logistical difficulties, limit its application in remote areas (North et al., 2012). Managing natural ignitions for resource benefit can overcome many of these limitations (USDA/USDI, 2005), providing an alternative mechanism for managers to reintroduce low-to-moderate-severity fire in remote forests. Our results demonstrate that managed wildfires cause multiyear reductions in tree growth, which resembles tree growth response to prescribed fire (Peterson et al., 1994; Wenderott et al., 2022). Unlike most prescribed fire programs where resources limit burn frequency, though, natural ignitions often burn forests at frequencies that approximate historical fire regimes, resulting in forests with 50–70 % fewer trees than untreated stands (Cleaves et al., 1999; Holden et al., 2007; Parks et al., 2023). As the reduction in stand density meets thinning recommendations for southwestern ponderosa pine forests (McCauley et al., 2022; Reynolds et al., 2013), managing natural ignitions for resource benefit can meet treatment goals in remote dry conifer forests with acceptable effects on tree growth.

Although managed wildfires temporarily reduce forest productivity, the short-term cost may act as an investment in the long-term retention of dry conifer forests. Dry conifer forests have increasingly burned at high-severity across the southwest because of high fuel loads and recent increases in aridity (Singleton et al., 2019). Higher aridity has concurrently reduced conifer regeneration in high-severity patches, resulting in ecosystem conversions to non-forested landscapes (Coop et al., 2020). Managed wildfires decrease the risk of high-severity fire and ecosystem conversion by reducing fuel loads in remote forests that are otherwise inaccessible for treatment (Rodman et al., 2023). As managed wildfires can burn frequently over large areas (Parks et al., 2023), managing natural ignitions for resource benefit reduces the risk of widespread forest loss. Preventing forest loss often represents an important management objective in southwestern forests, meaning the increased likelihood of long-term forest retention can compensate for short-term growth reductions after managed wildfires.

CRediT authorship contribution statement

Kevin G. Willson: Writing – review & editing, Writing – original draft, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Matthew D. Hurteau:** Writing – review & editing, Visualization, Validation, Supervision, Resources, Methodology, Investigation, Formal analysis, Conceptualization. **Ellis Q. Margolis:** Writing – review & editing, Validation, Supervision, Formal analysis.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

I have shared the link to my data/code at the 'Attach File' step

Trees have similar growth responses to first-entry fires and reburns following long-term fire exclusion (Original data) (Dryad)

Acknowledgements

We acknowledge and appreciate the assistance of Emily Schmitt and Kurt Willson for their help with data collection. Comments from A. Wion improved the manuscript. This work was supported by the Joint Fire Science Program under Project JFSP 21-1-01-31. This work was also supported by the Agriculture and Food Research Initiative - Education and Workforce Development program grant no. 2022-67011-36462/ project accession no. 1028071 from the USDA National Institute of Food and Agriculture. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.foreco.2024.122226.

References

Abatzoglou, J.T., Battisti, D.S., Williams, A.P., Hansen, W.D., Harvey, B.J., Kolden, C.A., 2021. Projected increases in western US forest fire despite growing fuel constraints. Commun. Earth Environ. 2, 1–8. https://doi.org/10.1038/s43247-021-00299-0.

Bär, A., Michaletz, S.T., Mayr, S., 2019. Fire effects on tree physiology. New Phytol. 223, 1728–1741. https://doi.org/10.1111/nph.15871.

Biederman, J.A., Scott, R.L., Goulden, M.L., Vargas, R., Litvak, M.E., Kolb, T.E., Yepez, E. A., Oechel, W.C., Blanken, P.D., Bell, T.W., Garatuza-Payan, J., Maurer, G.E., Dore, S., Burns, S.P., 2016. Terrestrial carbon balance in a drier world: the effects of water availability in southwestern North America. Glob. Change Biol. 22, 1867–1879. https://doi.org/10.1111/gcb.13222.

Brodribb, T.J., Powers, J., Cochard, H., Choat, B., 2020. Hanging by a thread? Forests and drought. Science 368, 261–266. https://doi.org/10.1126/science.aat7631.

Brown, P.M., Gentry, C., Yao, Q., 2019. Historical and current fire regimes in ponderosa pine forests at Zion National Park, Utah: restoration of pattern and process after a century of fire exclusion. For. Ecol. Manag. 445, 1–12. https://doi.org/10.1016/j.foreco.2019.04.058.

Buma, B., Weiss, S., Hayes, K., Lucash, M., 2020. Wildland fire reburning trends across the US West suggest only short-term negative feedback and differing climatic effects. Environ. Res. Lett. 15, 034026 https://doi.org/10.1088/1748-9326/ab6c70.

Bunn, A.G., 2010. Statistical and visual crossdating in R using the dplR library. Dendrochronologia 28, 251–258. https://doi.org/10.1016/j.dendro.2009.12.001.

Camarero, J.J., Gazol, A., Sangüesa-Barreda, G., Oliva, J., Vicente-Serrano, S.M., 2015. To die or not to die: early warnings of tree dieback in response to a severe drought. J. Ecol. 103, 44–57. https://doi.org/10.1111/1365-2745.12295.

Chamberlain, C.P., Cova, G.R., Cansler, C.A., North, M.P., Meyer, M.D., Jeronimo, S.M. A., Kane, V.R., 2023. Consistently heterogeneous structures observed at multiple spatial scales across fire-intact reference sites. For. Ecol. Manag. 550, 121478 https://doi.org/10.1016/j.foreco.2023.121478.

Cleaves, D.A., Haines, T.K., Martínez, J., 1999. Prescribed Burning Costs: Trends and Influences in the National Forest System (No. Gen. Tech. Rep. PSW-GTR-173), Proc. of the Symposium on fire economics, planning, and policy: Bottom lines. U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station, Albany, CA.

Cook, E.R., Kairiukstis, L.A., 1990. Methods of Dendrochronology: Applications in the Environmental Sciences. Springer. ISBN-13: 978-0-7923-0586-6.

Coop, J.D., Parks, S.A., Stevens-Rumann, C.S., Crausbay, S.D., Higuera, P.E., Hurteau, M. D., Tepley, A., Whitman, E., Assal, T., Collins, B.M., Davis, K.T., Dobrowski, S., Falk, D.A., Fornwalt, P.J., Fulé, P.Z., Harvey, B.J., Kane, V.R., Littlefield, C.E., Margolis, E.Q., North, M., Parisien, M.-A., Prichard, S., Rodman, K.C., 2020. Wildfire-Driven Forest Conversion in Western North American Landscapes. Biosci. biaa061. https://doi.org/10.1093/biosci/biaa061.

Coppoletta, M., Merriam, K.E., Collins, B.M., 2016. Post-fire vegetation and fuel development influences fire severity patterns in reburns. Ecol. Appl. 26, 686–699. https://doi.org/10.1890/15-0225.

Covington, W.W., Moore, M.M., 1994. Post settlement changes in natural fire regimes and forest structure: ecological restoration of old-growth ponderosa pine forests. J. Sustain. For. 2, 153–181.

Dickman, L.T., Jonko, A.K., Linn, R.R., Altintas, I., Atchley, A.L., Bär, A., Collins, A.D., Dupuy, J.-L., Gallagher, M.R., Hiers, J.K., Hoffman, C.M., Hood, S.M., Hurteau, M.D., Jolly, W.M., Josephson, A., Loudermilk, E.L., Ma, W., Michaletz, S.T., Nolan, R.H., O'Brien, J.J., Parsons, R.A., Partelli-Feltrin, R., Pimont, F., Resco de Dios, V., Restaino, J., Robbins, Z.J., Sartor, K.A., Schultz-Fellenz, E., Serbin, S.P., Sevanto, S., Shuman, J.K., Sieg, C.H., Skowronski, N.S., Weise, D.R., Wright, M., Xu, C., Yebra, M., Younes, N., 2023. Integrating plant physiology into simulation of fire behavior and effects. New Phytol. 238, 952–970. https://doi.org/10.1111/nph.18770.

Eidenshink, J., Schwind, B., Brewer, K., Zhu, Z.-L., Quayle, B., Howard, S., 2007. A Project for Monitoring Trends in Burn Severity. Fire Ecol. 3, 3–21. https://doi.org/10.4996/fireecology.0301003.

Field, C.A., Welsh, A.H., 2007. Bootstrapping Clustered Data. J. R. Stat. Soc. Ser. B: Stat. Methodol. 69, 369–390. https://doi.org/10.1111/j.1467-9868.2007.00593.x.

Garrett, L.D., Soulen, M.H., 1999. Changes in Character and Structure of Apache/ Sitgreaves Forest Ecology: 1850-1990. In Proceedings of the Fourth Biennial Conference of Research on the Colorado Plateau, eds van Riper, C., III, and Stuart, M. A., 25-59.

Gonzalez, P., Garfin, G.M., Breshears, D.D., Brooks, K.M., Brown, H.E., Elias, E.H., et al., 2018. In: Reidmiller, D.R., Avery, C.W., Easterling, D.R., Kunkel, K.E., Lewis, K.L.M., Maycock, T.K., Stewart, B.C. (Eds.), Southwest. In Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II. U.S. Global Change Research Program, Washington, DC, USA, pp. 1101–1184. https://doi.org/10.7930/NCA4.2018.CH25.

González-Rosales, A., Rodríguez-Trejo, D.A., 2004. Efecto del chamuscado de copa en el crecimiento en diámetro de Pinus hartwegii Lindl. en el Distrito Federal, México. Agrociencia, 38, 537-544.

Goodwin, M.J., Zald, H.S.J., North, M.P., Hurteau, M.D., 2021. Climate-driven tree mortality and fuel aridity increase wildfire's potential heat flux. Geophys. Res. Lett. 48, e2021GL094954 https://doi.org/10.1029/2021GL094954.

Hagmann, R.K., Hessburg, P.F., Prichard, S.J., Povak, N.A., Brown, P.M., Fulé, P.Z., Keane, R.E., Knapp, E.E., Lydersen, J.M., Metlen, K.L., Reilly, M.J., Sánchez Meador, A.J., Stephens, S.L., Stevens, J.T., Taylor, A.H., Yocom, L.L., Battaglia, M.A., Churchill, D.J., Daniels, L.D., Falk, D.A., Henson, P., Johnston, J.D., Krawchuk, M.A., Levine, C.R., Meigs, G.W., Merschel, A.G., North, M.P., Safford, H.D., Swetnam, T. W., Waltz, A.E.M., 2021. Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests. Ecol. Appl. 31, e02431 https://doi.org/10.1002/eap.2431.

Holden, Z.A., Morgan, P., Crimmins, M.A., Steinhorst, R.K., Smith, A.M.S., 2007. Fire season precipitation variability influences fire extent and severity in a large southwestern wilderness area, United States: GILA CLIMATE SEVERITY. Geophys. Res. Lett. 34 https://doi.org/10.1029/2007GL030804.

Holmes, R.L., 1983. Computer-assisted quality control in tree-ring dating and measurement.

Hood, S.M., Cluck, D.R., Smith, S.L., Ryan, K.C., 2008. Using bark char codes to predict post-fire cambium mortality. Fire Ecol. 4, 57–73. https://doi.org/10.4996/fireecology.0401057.

Hood, S.M., Varner, J.M., van Mantgem, P., Cansler, C.A., 2018. Fire and tree death: understanding and improving modeling of fire-induced tree mortality. Environ. Res. Lett. 13, 113004 https://doi.org/10.1088/1748-9326/aae934.

Hunter, M.E., Shepperd, W.D., Lentile, L.B., Lundquist, J.E., Andreu, M.G., 2007. A Comprehensive Guide to Fuels Treatment: Practices for Ponderosa Pine in the Black Hills, Colorado Front Range, and Southwest 101.

Jolly, W.M., 2007. Sensitivity of a surface fire spread model and associated fire behaviour fuel models to changes in live fuel moisture. Int. J. Wildland Fire 16, 503. https://doi.org/10.1071/WF06077.

Keane, R.E., Mincemoyer, S.A., Schmidt, K.M., Long, D.G., Garner, J.L., 2000. Mapping vegetation and fuels for fire management on the Gila National Forest Complex, New Mexico (No. RMRS-GTR-46). U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ft. Collins, CO. https://doi.org/10.2737/RMRS-GTR-46.

Keeley, J.E., 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. Int. J. Wildland Fire 18, 116. https://doi.org/10.1071/WF07049.

Keeling, E.G., Sala, A., 2012. Changing growth response to wildfire in old-growth ponderosa pine trees in montane forests of north central Idaho. Glob. Change Biol. 18, 1117–1126. https://doi.org/10.1111/j.1365-2486.2011.02574.x.

Kohyama, T., Kubo, T., Macklin, E., 2005. Effect of temporal autocorrelation on apparent growth rate variation in forest tree census data and an alternative distribution function of tree growth rate. Ecol. Res. 20, 11–15.

Kolb, T., Keefover-Ring, K., Burr, S.J., Hofstetter, R., Gaylord, M., Raffa, K.F., 2019. Drought-Mediated Changes in Tree Physiological Processes Weaken Tree Defenses to Bark Beetle Attack. J. Chem. Ecol. 45, 888–900. https://doi.org/10.1007/s10886-019-01105-0.

Kreider, M.R., Higuera, P.E., Parks, S.A., Rice, W.L., White, N., Larson, A.J., 2024. Fire suppression makes wildfires more severe and accentuates impacts of climate change and fuel accumulation. Nat. Commun. 15, 2412. https://doi.org/10.1038/s41467-024-46702-0.

Kusnierczyk, E., Ettl, G.J., 2002. Growth response of ponderosa pine (Pinus ponderosa) to climate in the eastern Cascade Mountains, Washington, USA: Implications for climatic change. Ecoscience, 9, 544–551.

Li, H., Speer, J.H., Thapa, I., 2023. Quantitative analysis of multiple forest disturbances and their compound influences on tree growth in the western U.S. Dendrochronologia 79, 126084. https://doi.org/10.1016/j.dendro.2023.126084.

McCauley, L.A., Bradford, John, B., Robles, M.D., Shriver, R.K., Woolley, T.J., Andrews, C.A., 2022. Landscape-scale forest restoration decreases vulnerability to drought mortality under climate change in southwest USA ponderosa forest. For. Ecol. Manag. 509, 120088 https://doi.org/10.1016/j.foreco.2022.120088.

McDowell, N.G., Williams, A.P., Xu, C., Pockman, W.T., Dickman, L.T., Sevanto, S., Pangle, R., Limousin, J., Plaut, J., Mackay, D.S., Ogee, J., Domec, J.C., Allen, C.D., Fisher, R.A., Jiang, X., Muss, J.D., Breshears, D.D., Rauscher, S.A., Koven, C., 2016. Multi-scale predictions of massive conifer mortality due to chronic temperature rise. Nat. Clim. Change 6, 295–300. https://doi.org/10.1038/nclimate2873.

Meko, D.M., Touchan, R., Anchukaitis, K.J., 2011. Seascorr: A MATLAB program for identifying the seasonal climate signal in an annual tree-ring time series. Comput. Geosci. 37, 1234–1241. https://doi.org/10.1016/j.cageo.2011.01.013.

Michaletz, S.T., Johnson, E.A., 2007. How forest fires kill trees: a review of the fundamental biophysical processes. Scand. J. For. Res. 22, 500–515. https://doi.org/10.1080/02827580701803544.

Michaletz, S.T., Johnson, E.A., Tyree, M.T., 2012. Moving beyond the cambium necrosis hypothesis of post-fire tree mortality: cavitation and deformation of xylem in forest fires. New Phytol. 194, 254–263. https://doi.org/10.1111/j.1469-8137.2011.04021.x.

Moore, M.M., Huffman, D.W., Fulé, P.Z., Covington, W.W., Crouse, J.E., 2004. Comparison of Historical and Contemporary Forest Structure and Composition on Permanent Plots in Southwestern Ponderosa Pine Forests. For. Sci. 50, 162–176. https://doi.org/10.1093/forestscience/50.2.162.

Mueller, S.E., Thode, A.E., Margolis, E.Q., Yocom, L.L., Young, J.D., Iniguez, J.M., 2020. Climate relationships with increasing wildfire in the southwestern US from 1984 to 2015. For. Ecol. Manag. 460, 117861 https://doi.org/10.1016/j.foreco.2019.117861.

North, M., Collins, B.M., Stephens, S., 2012. Using fire to increase the scale, benefits, and future maintenance of fuels treatments. J. For. 110, 392–401. https://doi.org/10.5849/jof.12-021.

North, M., Hurteau, M., Innes, J., 2009. Fire suppression and fuels treatment effects on mixed-conifer carbon stocks and emissions. Ecol. Appl. 19, 1385–1396. https://doi.org/10.1890/08-1173.1.

NRCS. 2022. "Distribution Maps of Dominant Soil Orders." https://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/survey/class/maps/?cid=nrcs142p2_053589. [Accessed 14 June 2022].

O'Brien, J.J., Kevin Hiers, J., Mitchell, R.J., Varner, J.M., Mordecai, K., 2010. Acute physiological stress and mortality following fire in a long-unburned longleaf pine ecosystem. fire Ecol. 6, 1–12. https://doi.org/10.4996/fireecology.0602001.

Parks, S.A., Dillon, G.K., Miller, C., 2014a. A new metric for quantifying burn severity: the relativized burn ratio. Remote Sens. 6, 1827–1844. https://doi.org/10.3390/rs6031827.

Parks, S.A., Holsinger, L.M., Blankenship, K., Dillon, G.K., Goeking, S.A., Swaty, R., 2023. Contemporary wildfires are more severe compared to the historical reference period in western US dry conifer forests. For. Ecol. Manag. 544, 121232 https://doi.org/10.1016/j.foreco.2023.121232.

Parks, S.A., Holsinger, L.M., Miller, C., Nelson, C.R., 2015. Wildland fire as a self-regulating mechanism: the role of previous burns and weather in limiting fire progression. Ecol. Appl. 25, 1478–1492. https://doi.org/10.1890/14-1430.1.

Parks, S.A., Miller, C., Nelson, C.R., Holden, Z.A., 2014b. Previous fires moderate burn severity of subsequent wildland fires in two large Western US wilderness areas. Ecosystems 17, 29–42. https://doi.org/10.1007/s10021-013-9704-x.

Peltier, D.M.P., Anderegg, W.R.L., Guo, J.S., Ogle, K., 2022. Contemporary tree growth shows altered climate memory. Ecol. Lett. 25, 2663–2674. https://doi.org/10.1111/ele.14130.

Peterson, D., Sackett, S., Robinson, L., Haase, S., 1994. The Effects of Repeated Prescribed Burning on Pinus ponderosa Growth. Int. J. Wildland Fire 4, 239. https://doi.org/10.1071/WF9940239.

PRISM Climate Group. 2023. PRISM climate data [online]. Available from http://www.prism.oregonstate.edu/ [accessed 17 April 2023].

R Core Team, 2020. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria (URL https://www.R-project.org/).

Regent Instruments. 2018. WinDENDRO 2019a: for tree-ring analysis, 19 June 2018, Canada.

Renninger, H.J., Clark, K.L., Skowronski, N., Schäfer, K.V.R., 2013. Effects of a prescribed fire on water use and photosynthetic capacity of pitch pines. Trees 27, 1115–1127. https://doi.org/10.1007/s00468-013-0861-5.

Reynolds, R.T., Sanchez Meador, A.J., Youtz, J.A., Nicolet, T., Matonis, M.S., Jackson, P. L., DeLorenzo, D.G., Graves, A.D., 2013. Restoring composition and structure in Southwestern frequent-fire forests: A science-based framework for improving ecosystem resiliency (No. RMRS-GTR-310). U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ft. Collins, CO. https://doi.org/10.2737/RMRS-GTR-310.

Roccaforte, J.P., Huffman, D.W., Fulé, P.Z., Covington, W.W., Chancellor, W.W., Stoddard, M.T., Crouse, J.E., 2015. Forest structure and fuels dynamics following ponderosa pine restoration treatments, White Mountains, Arizona, USA. For. Ecol. Manag. 337, 174–185. https://doi.org/10.1016/j.foreco.2014.11.001.

Rodman, K.C., Davis, K.T., Parks, S.A., Chapman, T.B., Coop, J.D., Iniguez, J.M., Roccaforte, J.P., Sánchez Meador, A.J., Springer, J.D., Stevens-Rumann, C.S., Stoddard, M.T., Waltz, A.E.M., Wasserman, T.N., 2023. Refuge-yeah or refuge-nah? Predicting locations of forest resistance and recruitment in a fiery world. Glob. Change Biol. gcb 16939. https://doi.org/10.1111/gcb.16939.

Rollins, M.G., 2009. LANDFIRE: a nationally consistent vegetation, wildland fire, and fuel assessment. Int. J. Wildland Fire 18, 235. https://doi.org/10.1071/WF08088.

Rollins, M.G., Morgan, P., Swetnam, T., 2002. Landscape-scale controls over 20th century fire occurrence in two large Rocky Mountain (USA) wilderness areas. Landsc. Ecol. 17, 20.

Rothermel, R.C., 1972. A mathematical model for predicting fire spread in wildland fuels. Intermt. For. Range Exp. Station For. Serv. US Dep. Agric. 115.

Ryan, K.C., 2002. Dynamic interactions between forest structure and fire behavior in boreal ecosystems. Silva Fenn. 36, 13–39.

Sheppard, P., Comrie, A., Packin, G., Angersbach, K., Hughes, M., 2002. The climate of the US Southwest. Clim. Res. 21, 219–238. https://doi.org/10.3354/cr021219.

Silge, J., Chow, F., Kuhn, M., Wickham, H., 2022. rsample: General Resampling Infrastructure. R package version 1.1.0. https://CRAN.R-project.org/package=rsample.

Singleton, M.P., Thode, A.E., Sánchez Meador, A.J., Iniguez, J.M., 2019. Increasing trends in high-severity fire in the southwestern USA from 1984 to 2015. For. Ecol. Manag. 433, 709–719. https://doi.org/10.1016/j.foreco.2018.11.039.

Sparks, A.M., Blanco, A.S., Wilson, D.R., Schwilk, D.W., Johnson, D.M., Adams, H.D., Bowman, D.M.J.S., Hardman, D.D., Smith, A.M.S., 2023. Fire intensity impacts on physiological performance and mortality in Pinus monticola and Pseudotsuga menziesii saplings: a dose–response analysis. Tree Physiol. 43, 1365–1382. https://doi.org/10.1093/treephys/tpad051.

Sparks, A.M., Talhelm, A.F., Feltrin, R.P., Smith, A.M.S., Johnson, D.M., Kolden, C.A., Boschetti, L., 2018. An experimental assessment of the impact of drought and fire on western larch injury, mortality and recovery. Int. J. Wildland Fire 27, 490. https://doi.org/10.1071/WF18044.

Stevens, J.T., Kling, M.M., Schwilk, D.W., Varner, J.M., Kane, J.M., 2020. Biogeography of fire regimes in western U.S. conifer forests: A trait-based approach. Glob. Ecol. Biogeogr. 29, 944–955. https://doi.org/10.1111/geb.13079.

Stokes, M.A., Smiley, T.L., 1968. An Introduction to Tree-ring Dating. University of Chicago Press, Chicago.

Swetnam, T.W., Baisan, C.H., 1996. Fire histories of montane forests in the Madrean Borderlands. 0277-5786, United States Department of Agriculture Forest Service General Technical Report RM (1996).

Swezy, D.M., Agee, J.K., 1991. Prescribed-fire effects on fine-root and tree mortality in old-growth ponderosa pine. Can. J. . Res. 21, 626–634. https://doi.org/10.1139/x91-086.

USDA/USDI. 2005. Wildland Fire Use, Implementation Procedures Reference Guide. Washington, D.C. 71 p.

van Mantgem, P.J., Falk, D.A., Williams, E.C., Das, A.J., Stephenson, N.L., 2018. Pre-fire drought and competition mediate post-fire conifer mortality in western U.S. National Parks. Ecol. Appl. 28, 1730–1739. https://doi.org/10.1002/eap.1778.

van Wagtendonk, J.W., 2007. The history and evolution of wildland fire use. fire Ecol. 3, 3–17. https://doi.org/10.4996/fireecology.0302003.

Wagner, C.E.V., 1973. Height of crown scorch in forest fires. Can. J. . Res 3, 373–378. https://doi.org/10.1139/x73-055.

Wasserman, T.N., Mueller, S.E., 2023. Climate influences on future fire severity: a synthesis of climate-fire interactions and impacts on fire regimes, high-severity fire, and forests in the western United States. Fire Ecol. 19, 43. https://doi.org/10.1186/s42408-023-00200-8.

Wenderott, Z., van Mantgem, P.J., Wright, M.C., Farris, C.A., Sherriff, R.L., 2022. Long-term effects of prescribed fire on large tree growth in mixed conifer forests at Lassen Volcanic National Park, California. For. Ecol. Manag. 517, 120260 https://doi.org/10.1016/j.foreco.2022.120260.

Westlind, D.J., Kerns, B.K., 2017. Long-term effects of burn season and frequency on ponderosa pine forest fuels and seedlings. Fire Ecol. 13, 42–61. https://doi.org/10.4996/fireecology.130304261.

Wigley, T.M.L., Briffa, K.R., Jones, P.D., 1984. On the average value of correlated time series, with applications in dendroclimatology and hydrometeorology. J. Appl. Meteorol. Climatol. 23, 201–213.

Williams, A.P., Abatzoglou, J.T., Gershunov, A., Guzman-Morales, J., Bishop, D.A., Balch, J.K., Lettenmaier, D.P., 2019. Observed Impacts of Anthropogenic Climate Change on Wildfire in California. Earth's Future 7, 892–910. https://doi.org/10.1029/2019EF001210.

Williams, A.P., Allen, C.D., Macalady, A.K., Griffin, D., Woodhouse, C.A., Meko, D.M., Swetnam, T.W., Rauscher, S.A., Seager, R., Grissino-Mayer, H.D., Dean, J.S., Cook, E. R., Gangodagamage, C., Cai, M., McDowell, N.G., 2013. Temperature as a potent driver of regional forest drought stress and tree mortality. Nat. Clim. Change 3, 292–297. https://doi.org/10.1038/nclimate1693.

Zang, C., Biondi, F., 2015. treeclim: an R package for the numerical calibration of proxy-climate relationships. Ecography 38, 431–436. https://doi.org/10.1111/ecog.01335.