



RESEARCH

Open Access



Resource objective wildfires shifted forest structure and fuels toward pre-fire-exclusion conditions in a remote Arizona wilderness

John P. Roccaforte^{1*}, David W. Huffman¹, Kyle C. Rodman¹, Thomas A. Heinlein², Joseph E. Crouse¹ and Peter Z. Fulé³

Abstract

Background Large, severe fires are increasing throughout frequent-fire forests of the western United States due to warming climatic conditions, as well as legacies of early twentieth century land-use practices and anthropogenic fire exclusion. Resource objective (RO) wildfires—where naturally ignited wildfires are allowed to burn to accomplish management objectives—are increasingly accepted due to relatively low cost and flexibility on lands where mechanical treatments are not allowed (e.g., designated wilderness) or economically feasible. We previously implemented a field study across a portion of the Mount Trumbull Wilderness to identify differences between historical (ca. 1870) and contemporary (1999) forest structural conditions following 100+ years of fire exclusion. The study area subsequently experienced two RO wildfires (2012 and 2019), which presented an opportunity to (1) assess how closely post-wildfire (2023) conditions approximated historical forest conditions and (2) evaluate how RO fires influenced patterns of tree mortality and regeneration.

Results Reconstructed forest structure was made up of open stand conditions (density: 62 trees ha⁻¹; basal area: 9 m² ha⁻¹) with large ponderosa pines (quadratic mean diameter: 42 cm). By 1999, the site was dominated by closed-canopy stands with many small-diameter trees. In 2023, following the two RO wildfires, tree density, basal area, and canopy cover were significantly reduced (20–50%), and tree size significantly increased. Ponderosa pine regeneration density declined relative to pre-fire levels, whereas regeneration of sprouting hardwood species increased. About half of the old trees (i.e., pre-dating ca. 1870) that were alive in 1999 died by the end of the study period, likely due to effects of fire-caused injury and drought. High-severity patch sizes in each fire were relatively small (6.2–46.6 ha) and within the historical range of variability for southwestern ponderosa pine ecosystems. The 2012 fire reduced remotely sensed fire severity in 2019.

Conclusions Overall, RO fires shifted forest structure in a remote wilderness area toward open conditions that were historically present at the site, and likely reduced vulnerability to severe fire in the future. However, tree density remained six times higher than historical levels, and managers should consider allowing future RO wildfires to burn within the wilderness to further reduce tree density and accomplish ecological restoration goals.

Keywords Burn severity, Frequent-fire forests, Managed fire, Historical range of variability, Pine-oak forests, *Pinus ponderosa*, Reference conditions, Regeneration

*Correspondence:

John P. Roccaforte

john.roccaforte@nau.edu

Full list of author information is available at the end of the article

Resumen

Antecedentes Los incendios de gran magnitud y severidad se están incrementando a través de bosques en los que el fuego ha sido frecuente en el oeste de los EEUU, debido al calentamiento del clima y también a los legados de prácticas del manejo de tierras de principios del siglo XX y a la exclusión antrópica de fuegos naturales. El recurso objetivo de incendios (RO) –en los cuales los incendios iniciados de manera natural se dejan propagar de manera de conseguir objetivos de manejo–, se están aceptando cada vez más debido a su bajo costo y flexibilidad en aquellas tierras en las cuales los tratamientos mecánicos no son permitidos (e. g. áreas designadas como protegidas), o por ser económicamente inviables. Implementamos previamente un estudio de campo a través de una porción del área natural de Monte Trumbull para identificar las diferencias entre condiciones estructurales del bosque tanto históricas (ca. 1870) como contemporáneas (1999), luego de 100+ años de exclusión de incendios. A posteriori, el área de estudios experimentó dos incendios RO (2012 y 2019), que presentaron la oportunidad de: (1) determinar cuán cerca las condiciones post fuego (2023) se aproximaron a las condiciones históricas del bosque, y (2) evaluar cómo los incendios RO influenciaban los patrones de mortalidad y regeneración de los árboles.

Resultados La reconstrucción de la estructura forestal fue iniciada desde condiciones de rodales abiertos (densidad de 62 árboles ha^{-1} ; área basal: 9 $\text{m}^2 \text{ha}^{-1}$), con grandes árboles de pino ponderosa (diámetro cuadrático medio: 42 cm). Para 1999, el sitio estaba dominado por rodales con doseles cerrados con muchos árboles de diámetros muy pequeños. En 2023, luego de los dos incendios RO, la densidad de árboles, el área basal, y el dosel estaban significativamente reducidos (20–50%), y el tamaño de los árboles se había incrementado significativamente. La densidad de la regeneración de pino ponderosa disminuyó en relación a niveles pre fuego, mientras que se incrementó la regeneración de especies arbóreas rebrotantes de madera dura. Alrededor de la mitad de los árboles más longevos (i. e. pre-fechados ca 1870) que estaban vivos en 1999, murieron al final del período de estudio, probablemente debido a los efectos de heridas ajenas al fuego y sequías. Los parches de alta severidad de cada incendio fueron relativamente pequeños (6,2 a 46,6 ha) y dentro del rango histórico de variabilidad del fuego (HRV) para ecosistemas de pino ponderosa del suroeste. El incendio de 2012 redujo la severidad del fuego de 2019 de acuerdo a mediciones realizadas con sensores remotos.

Conclusiones De manera general, los incendios RO cambiaron la estructura del bosque, en un área natural remota, hacia condiciones más abiertas que estuvieron presentes históricamente en ese sitio, y que probablemente reduzcan la vulnerabilidad a fuegos severos en el futuro. Sin embargo, la densidad de árboles se mantuvo seis veces más alta que los niveles históricos, por lo que los gestores deberían considerar la posibilidad de permitir que futuros incendios RO quemen dentro de las áreas naturales y reduzcan aún más la densidad de árboles y de esa manera poder alcanzar metas de restauración ecológica.

Background

Large, severe fires are increasing throughout frequent-fire forests of the western United States due to warming climatic conditions, the legacies of early twentieth century land-use practices, and anthropogenic fire exclusion (Covington et al. 1994; Hessburg et al. 2019; Hagmann et al. 2021). Dense forests, extreme fire behavior, and widespread impacts to human communities and infrastructure have given rise to broad collaborative efforts aimed at restoring more fire-resistant forest conditions and reintroducing characteristic fire regimes over landscape-scale project areas (Schultz et al. 2012). With human-induced warming climatic conditions expected to drive further increases in fire activity into the future (Coop et al. 2022; Parks et al. 2025), broad-scale management actions including fuel reduction and ecological restoration treatments are now progressing at an accelerated pace (Prichard et al. 2021; US Forest Service

2022). However, only a portion of many forest landscapes can be treated using traditional means (e.g., mechanical tree removal, followed by prescribed fire), and other management approaches (e.g., fire-only treatments) are increasingly being applied to address these problems (Young et al. 2020). Understanding the effectiveness of such treatments to meet forest management objectives is therefore a key research need.

Ecological restoration “is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed” (Society for Ecological Restoration 2004). Most active restoration projects in western US forests manipulate overstory structure and composition to set the ecosystem on a trajectory toward recovering more characteristic levels of resilience to natural disturbances. For example, resilience to wildfire—where forests may recover structural and compositional characteristics and avoid crossing thresholds into a different ecological

state (Stephens et al. 2005)—is an intended goal of many restoration efforts, particularly in frequent-fire forests (e.g., those dominated by ponderosa pine; *Pinus ponderosa* Lawson & C. Lawson var. *brachyptera* [Engelm.] Lemmon) (Schoennagel and Nelson 2011; Stephens et al. 2021). In the southwestern US, structural reference conditions are commonly used to inform the development of ecological restoration prescriptions for ponderosa pine forests, degraded by unmanaged livestock grazing in the early twentieth century and decades of subsequent fire exclusion (Covington and Moore 1994; Fulé et al. 2002; Waltz et al. 2003; Sánchez Meador et al. 2010). Modern reference sites with intact fire regimes can provide valuable information when developing restoration treatments for degraded areas, yet such sites are very uncommon and may not be representative of all forests (White and Walker 1997; Stephens and Fulé 2005). Instead, historical reference conditions that describe ecological attributes of the system prior to degradation are often used to assess the degree of decline as well as to develop restoration treatment strategies and evaluate restoration success (Moore et al. 1999; Egan and Howell 2001; Roccaforte et al. 2010). Historical conditions can be derived using dendroecological reconstruction techniques that make use of contemporary field measurements of tree size, age, and condition (Fulé et al. 1997; Huffman et al. 2001; Bakker et al. 2008; Sánchez Meador et al. 2010). Thus, site-specific reference conditions, derived from reconstructions, provide an ideal method to evaluate restoration treatment success in southwestern forests.

The use of naturally ignited wildfires (or targeted areas within them) is an increasingly accepted approach for reducing hazardous fuels and meeting restoration objectives (North et al. 2021). Terminology has evolved over time (Van Wagtenonk 2007) with more recent agreement on the term “Resource Objective” (hereafter “RO”) to describe this wildfire management strategy (Huffman et al. 2020). RO wildfires are a form of active management—managers actively and continuously monitor fire behavior, use strategic interventions to allow the fire to burn within predetermined burning prescriptions and boundaries, and respond in a flexible and dynamic manner until the fire stops burning or is reclassified for full suppression (Thompson et al. 2022). This approach is particularly valuable on lands such as designated wilderness areas where mechanical treatments and/or prescribed fire are neither feasible nor acceptable (Young et al. 2020; Iniguez et al. 2022). Since the passage of the 1964 Wilderness Act (“Wilderness Act 16 U.S. Code § 1131”), management policy in many designated wilderness areas has focused on maintaining “untrammeled” conditions, which may preclude the use of mechanical treatments and many other active management

approaches (Naficy et al. 2016). However, wilderness designation is a contemporary administrative overlay onto landscapes with complex land use histories. Though often less impacted than other ecosystems, many forests within wilderness areas have been affected by past land use practices such as fire suppression (Miller 2012; Kreider et al. 2023). In wilderness areas, RO wildfires are an acceptable and plausible restoration treatment option, although many stakeholders in the wildland fire community have recently advocated for allowing more active approaches, including using human-ignited fire, to expedite restoration efforts (Boerigter et al. 2024).

Studies on the effects of RO wildfires have been somewhat limited; however, a recent uptick in research in this discipline has occurred in the past two decades. Very little research has evaluated the ability of RO wildfires to accomplish restoration objectives, such as by comparing post-fire landscapes to the historical range of variability (HRV) or site-specific reference conditions (Huffman et al. 2020). Past research on RO fires has evaluated changes in forest structure (Collins et al. 2011; Huffman et al. 2017, 2018; Higgins et al. 2015), overall fire characteristics (Collins et al. 2007; Miller et al. 2012), and resultant landscape patterns (Steel et al. 2018; Donager et al. 2022). It is now well understood that wildfires can have positive management outcomes, such as reducing the severity of subsequent fires (Rodman et al. 2023; Davis et al. 2024), including in wilderness (Parks et al. 2014). As is common with many studies examining wildland fire effects, pre-fire field data are often unavailable (van Mantgem et al. 2001), which can make it challenging to evaluate the effects of RO fire at restoring structural and compositional characteristics of forested ecosystems.

In 1999, we installed 20 permanent sampling plots across a portion of the Mount Trumbull Wilderness to determine historical and contemporary forest structural conditions and inform potential ecological restoration options. Using fire scar patterns, Heinlein et al. (1999) found that the historical frequent-fire regime at the site ceased in 1870, so forest conditions at the site in 1999 reflected more than 100 years of fire exclusion. In 2023, we remeasured the plots to examine fire effects and spatial patterns of severity and predicted overstory survival following two RO wildfires that burned in 2012 and 2019. In this study, we sought to understand the effectiveness of these RO fires for meeting restoration objectives based on common goals for southwestern ponderosa pine forests and site-specific historical reference conditions. We aimed to answer the following questions: (1) How did pre-fire (1999) forest structure and fuels differ from reconstructed historical (ca. 1870) conditions? (2) How did the 2012 and 2019 RO wildfires affect structure and fuels? (3) How closely did post-wildfire (2023) conditions

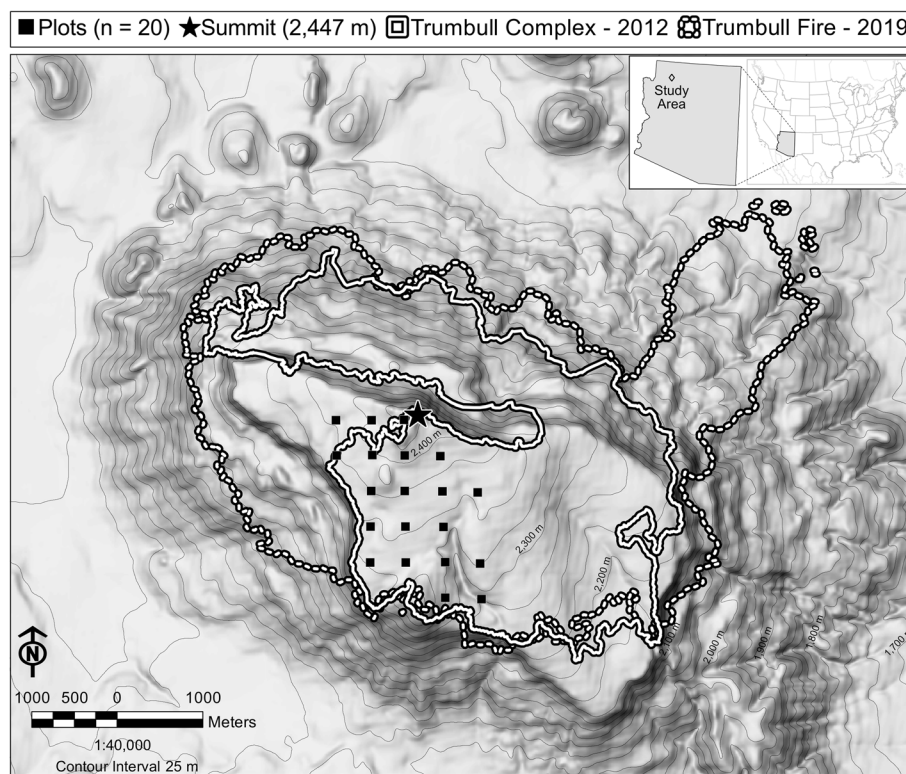


Fig. 1 Site map showing field plot locations ($n=20$) and fire perimeters of the 2012 Trumbull Complex and the 2019 Trumbull Fire that occurred in the Mount Trumbull Wilderness Area in northern Arizona, USA

approximate historical conditions? (4) What were the landscape patterns of burn severity in these RO fires, and how are they likely to influence subsequent fire severity and tree regeneration?

Methods

Study site

Our study site (36.4038° N, -113.1388° W) was located in the Mount Trumbull Wilderness (designated 1984) within the boundary of Grand Canyon-Parashant National Monument in northwestern Arizona, USA (Fig. 1). Forests in the study area (elevation 2280–2447 m) are primarily composed of ponderosa pine and Gambel oak (*Quercus gambelii* Nutt.) in the overstory while Utah Juniper (*Juniperus osteosperma* (Torr.) Little) and twoneedle pinyon (*Pinus edulis* Engelm.) occur in minor amounts. Hardwoods such as New Mexico locust (*Robinia neomexicana* A. Gray) and Gambel oak are common in lower canopy strata. Soils are primarily derived from basaltic parent material. From 1991 to 2020, mean annual temperature at our site was 8.8°C , and total annual precipitation was 459 mm (PRISM Climate Group 2023). Climate at Mount Trumbull is continental, with strong diurnal and interannual fluctuations in temperature—average July maximum temperature is

27.1°C and January minimum temperature is -4.9°C . Precipitation is bimodal, with most falling during the summer monsoon season (i.e., 36.9% from July to September) or in the winter season. An extended drought occurred from 1994 to 2021 throughout northwestern Arizona (Williams et al. 2022).

Mount Trumbull has a layered and complex history of human land use and management of forests. Evidence of Native American presence dates to at least the early Archaic period, beginning about 8000 years before present (Altschul and Fairley 1989). Relatively open forest structural conditions were maintained by a frequent surface fire regime linked to lightning and cultural burning (Roos et al. 2022) prior to Euro-American settlement in 1870 (Heinlein et al. 1999). Between 1596 and 1870, the fire return interval (Weibull Median Probability Interval, WMPI) averaged 9.5 years for fires scarring 25% or more of sampled trees (Heinlein et al. 1999). Though fire regimes were disrupted in ca. 1870, several fires occurred at the study site after this date, including an 1886 fire that may have been widespread, fires in 1888 and 1957 that each scarred individual trees, and a wildfire in 1989 that burned over the northern portion of the study site. There was no evidence of recent tree cutting nor livestock grazing, and

current management follows the 1964 Wilderness Act guidelines to maintain “a natural ecological landscape essentially free from human-induced contrast” (Bureau of Land Management 1990).

Resource objective wildfires

Two wildfires burned within the Wilderness Area after the initiation of the study (Table S1). Both fires were monitored by Bureau of Land Management (BLM) staff officers and managed to accomplish resources objectives, which included restoration of forest structure and reduction of hazardous fuels. In late July 2012, lightning ignited three fires: the Mt. Trumbull Fire, the Coyote Fire, and the Pueblo Fire. The three fires later merged into one complex fire (“Trumbull Complex”) and burned 648 ha. In August 2019, another lightning-ignited fire (“Trumbull Fire”) burned 1184 ha, encompassing most of the 2012 Trumbull Complex perimeter. Due to its remoteness and wilderness status, the study site had not been mechanically treated.

Field methods

In 1999, we installed 20 permanent plots on a 300-m grid covering approximately 180 ha across the higher elevations of Mount Trumbull. Plot design and sampling protocols were modified from the National Park Service’s Fire Monitoring plot protocol (USDI National Park Service 2003). Plot area was 0.1 ha (20 × 50 m), with the 50-m sides oriented parallel with aspect. We remeasured all plots in 2006 (sampled only overstory and pole tree conditions and diameters) and 2023 (sampled all components of plots).

Overstory trees (≥ 15 cm diameter at breast height, DBH) were measured on the entire plot (1000 m²) and pole-sized trees (2.5–14.9 cm DBH) were measured on one-quarter of the plot (250 m²); all trees were individually tagged to track demographic data through time. Tree attributes measured included species, DBH, condition class (Thomas et al. 1979), and crown base height. Tree regeneration (< 2.5 cm DBH) was tallied by species, condition, and height class in a 50-m² subplot. Ponderosa pine trees were considered potentially pre-fire-exclusion if DBH was ≥ 37.5 cm or if bark was yellowed (White 1985). Trees of other species were considered potentially pre-fire-exclusion if DBH ≥ 17 cm DBH (Barger and Ffolliott 1972). We collected tree cores using increment borers at 40 cm above ground level for all potentially pre-fire-exclusion trees and for a random 10% sample of all other live trees ≥ 2.5 cm to determine past size, as described below. Canopy cover measured by vertical projection (Ganey and Block 1994) was recorded at 3-m

intervals along the two 50-m sidelines of each plot for a total of 32 points per plot.

Reconstruction and laboratory methods

We reconstructed historical forest structure (tree density, DBH distribution, and basal area [BA] by species) using dendroecological methods described in detail by Fulé et al. (1997). In the laboratory, increment cores were mounted and surfaced using progressively finer grit sandpaper until the ring boundaries were clearly visible under a dissecting microscope (Speer 2010). Annual ring dates were verified against other sampled trees using COFECHA software (Holmes 1983). We reconstructed DBH for all living trees (see Fulé et al. 1997), in short, by subtracting the radial growth since 1870 measured on increment cores and estimated death date of dead trees based on tree condition class using diameter-dependent snag decomposition rates (Thomas et al. 1979; Rogers et al. 1984). Diameters were adjusted for bark thickness using species-specific allometric equations from other sites in the Southwest. We performed a sensitivity analysis by using the 25th, 50th, and 75th percentile of tree decomposition on estimates of death date and 1870 structure. Less than $\pm 1\%$ change in reconstructed density or BA occurred during this analysis, so the 50th percentile reconstruction was used in this study.

In our reconstruction methods, we assumed that remnant evidence of trees (i.e., snags, logs, stumps, stump holes) extant in 1870 was intact and correctly identified by field crews in 1999 during the pre-treatment inventory. The probability that this occurred was relatively high given the absence of fire since 1870 combined with the semiarid environment limiting the decomposition of conifer wood (Fulé et al. 1997; Mast et al. 1999; Waltz et al. 2003) and because field crews were trained to identify the presence and species of pre-fire-exclusion structures. Although dead structures (logs, snags, etc.) are important ecological attributes of historical stand conditions, these structures are difficult to precisely quantify; thus, in this analysis we focused on live trees. Analysis of these reconstruction field techniques in similar environments and forest type were reliable within $\pm 10\%$ for live tree density over approximately 90 years (Moore et al. 2004).

To analyze differences in canopy fuel load (CFL; kg m⁻²), we used allometric relationships relating tree component (e.g., crown foliage) biomass and DBH. For ponderosa pine crown biomass, we used equations given in Fulé et al. (2001). For pinyon and juniper biomass, we used equations described by Grier et al. (1992), and for oak biomass we used those of Clary and Tiedemann (1986). Mean canopy base height (CBH) was calculated

as the average of the crown base height of trees of all species on each plot.

To test for differences between time periods (1999 vs. 2023; 1870 vs. 2023), we used one-sided Wilcoxon signed rank tests for tree density, basal area (BA), quadratic mean diameter (QMD), canopy cover, CFL, and regeneration density. We hypothesized that tree density, BA, and CFL would increase from 1870 to 2023, whereas QMD would decline. However, we also hypothesized that fires would drive declines in density, BA, canopy cover, and CFL from 1999 to 2023, whereas QMD would increase. Finally, we hypothesized that hardwood (i.e., oak and locust) regeneration might increase from 1999 to 2023 due to fire-triggered resprouting, whereas conifer regeneration would decline. We used an α of 0.05 for all analyses.

Landscape-scale patterns of burn severity

To describe patterns of burn severity in the 2012 Trumbull Complex and the 2019 Trumbull Fire, we developed 30-m maps of the predicted composite burn index (CBI) in Google Earth Engine (Gorelick et al. 2017), following the modeling approach of Parks et al. (2019). CBI, ranging from 0 (unburned) to 3 (highest severity), is an indicator of burn severity that combines fire effects on soil and multiple vegetation strata (Key and Benson 2006). CBI predictions were based on a Random Forest model (Breiman 2001) using field-derived CBI data ($n=263$ fires throughout North America) and Landsat-derived spectral variables, latitude, and average climate, followed by bias correction (Parks et al. 2019). CBI predictions corresponded well with field data in Arizona (cross-validated $R^2=0.82$; Parks et al. 2019), and with recent (i.e., 2006–2023) basal area change in our field plots ($R^2=0.57$). Because CBI is intended to be used in forested areas, we restricted predictions to 30-m pixels with $>10\%$ tree canopy cover in the year before each fire (RAP v.3.0; Allred et al. 2021).

We summarized the average and range of CBI within each fire, as well as the percentages of each fire that burned at low ($\text{CBI} < 1.25$), moderate ($1.25 < \text{CBI} \leq 2.25$), and high ($\text{CBI} \geq 2.25$) fire severity (Miller and Thode 2007). We also described the size of the largest contiguous high-severity patch using 8-neighbor contiguity (Turner and Gardner 2015). As a metric of potential impacts to tree regeneration, we summarized the area within each fire that burned at high severity and was far from available seed sources—i.e., >60 m from low- or moderate-severity patch edges that had pre-fire forest cover (Chambers et al. 2016). Finally, we compared predicted CBI values from the

2019 fire, inside and outside of areas that previously burned in 2012, to understand how the 2012 fire may have reduced fire severity in 2019.

Results

Historical changes

Reconstructed forest structure was dominated by large ponderosa pines at the time of fire exclusion (i.e., 1870). Mean total (i.e., all species combined) tree density was 62.3 trees ha^{-1} , mean BA was 8.8 $\text{m}^2 \text{ha}^{-1}$, and QMD was 41.9 cm in 1870 (Table 1). Ponderosa pine made up more than 98% of tree density and BA prior to fire exclusion. The pre-fire-exclusion diameter distribution was unimodal with fewer than 10 trees ha^{-1} below 20 cm DBH and 80% of trees between 20 and 60 cm DBH (Fig. 2a).

Table 1 Tree density (trees/ha) and basal area (m^2/ha) (live trees ≥ 2.5 cm DBH) at the Mount Trumbull Wilderness study site, Arizona, USA, in 1870 (reconstructed), 1999 (pre-fire), and 2023 (post-fire)

Year	Total	PIPO	QUGA	JUOS	PIED
<i>Density (trees/ha)</i>					
1870	62.3 (39.3)	61.3 (39.2)	0.5 (2.2)	0.5 (2.2)	0
	0–160.4	0–160.4	0–10.0	0–10.1	
1999	1223.3 (804.7)	1041.7 (764.9)	175.0 (368.6)	0.5 (2.2)	6.1 (19.9)
	261.6–2902.1	180.9–2902.1	0–1550.1	0–10.0	0–81.6
2023	606.4 (390.3) † §	567.2 (392.7) † §	36.6 (79.3) † §	0.5 (2.2)	2.0 (9.1)
	0–1607.8	0–1607.8	0–326.3	0–10.0	0–40.8
<i>Basal area (m^2/ha)</i>					
1870	8.8 (6.4)	8.8 (6.5)	0.003 (0.01)	0.05 (0.2)	0
	0–21.8	0–21.8	0–0.06	0–0.9	
1999	34.9 (7.9)	34.0 (8.4)	0.9 (1.9)	0.03 (0.1)	0.01 (0.05)
	22.8–49.5	20.3–49.5	0–8.36	0–0.5	0–0.2
2023	27.8 (10.0) † §	27.5 (10.0) † §	0.3 (0.6) † §	0.03 (0.1)	0.02 (0.07)
	0–48.2	0–48.2	0–2.6	0–0.6	0–0.3
<i>Quadratic mean diameter (cm)</i>					
1870	41.9 (7.9)	41.6 (7.6)	N/A	N/A	N/A
	26.2–53.5	26.2–54.6			
1999	21.7 (6.1)	23.5 (6.9)	8.9 (3.2)	N/A	5.0 (1.4)
	13.1–35.8	13.1–39.8	5.3–14.9		4.0–6.0
2023	26.3 (6.6) † §	27.5 (7.6) † §	9.4 (2.6) †	N/A	N/A
	17.1–43.8	17.1–44.8	6.9–13.2		

Statistics presented are the mean (standard deviation) and below that number minimum–maximum. Tree density, BA, and QMD were tested for differences between years using Wilcoxon signed rank tests

PIPO *Pinus ponderosa*, QUGA *Quercus gambelii*, JUOS *Juniperus osteosperma*, PIED *Pinus edulis*

Within each variable, † indicates significantly different means ($p < 0.05$) between 1999 and 2023; § indicates significantly different means ($p < 0.05$) between 1870 and 2023. $n=20$

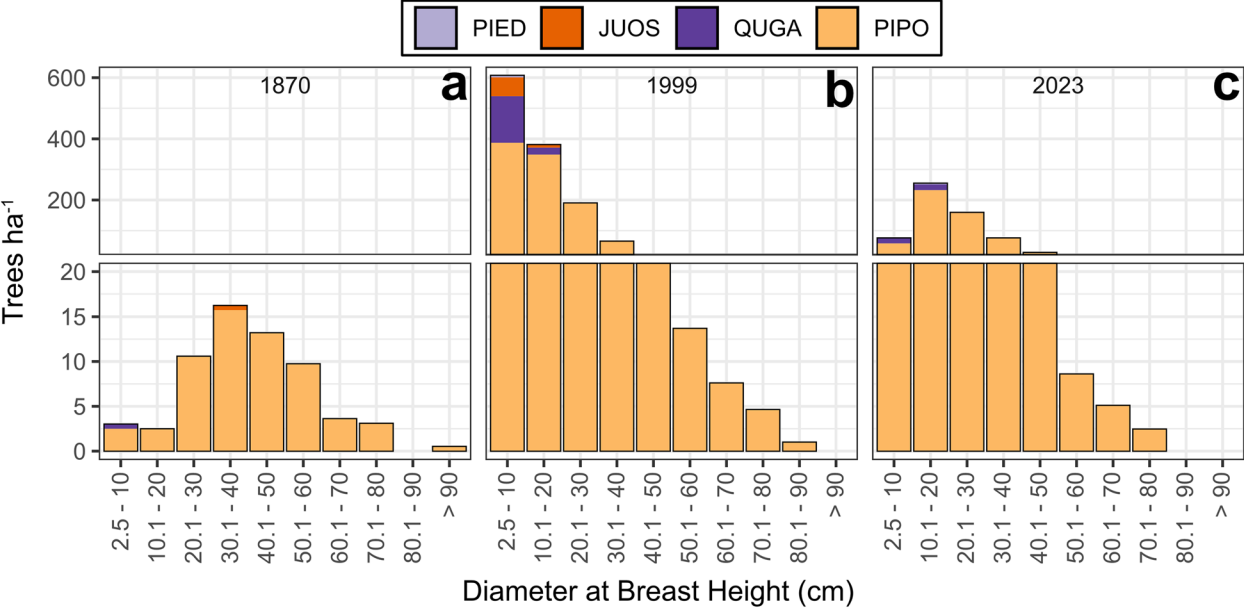


Fig. 2 DBH distributions at the Mount Trumbull Wilderness study site, Arizona, USA, in **a** 1870 (reconstructed), **b** 1999 (pre-fire), and **c** 2023 (post-fire)

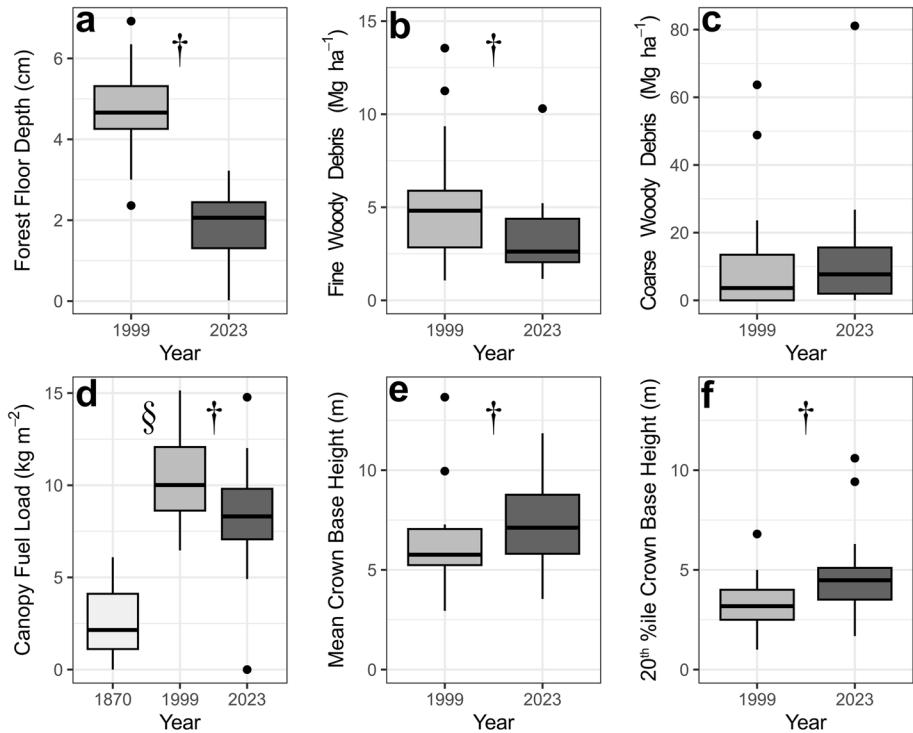


Fig. 3 Surface (**a–c**) and canopy (**d–f**) fuels at the Mount Trumbull Wilderness study site, Arizona, USA. Horizontal lines in each box indicate the median for each variable; black dots indicate outliers identified as $> 1.5 \times$ the interquartile range (IQR) outside the IQR. Within each attribute, † indicates significantly different means ($p < 0.05$) between 1999 and 2023; § indicates significantly different means ($p < 0.05$) between 1870 and 2023 (Canopy Fuel Load was the only attribute tested). $n = 20$ plots in **a–d** and 19 plots in **e, f**

Mean CFL was 0.25 kg m^{-2} in 1870 (Fig. 3d). By 1999, the study site was dominated by closed-canopy stands comprised of relatively small-diameter trees. Mean total tree density increased by 1864% to $1223.3 \text{ trees ha}^{-1}$, BA increased by 297% to $34.9 \text{ m}^2 \text{ ha}^{-1}$, and QMD decreased by 48% to 21.7 cm between 1870 and 1999. Ponderosa pine dominated the study site; however, Gambel oak was more prevalent than in 1870 comprising 14% of tree density and 3% of BA. Pinyon was absent and juniper made up a minor fraction of the total tree density in 1870 (Fig. 2). By 1999, the diameter distribution had shifted with 75% of trees below 20 cm DBH (Fig. 2b). CFL increased by 312% to 1.0 kg m^{-2} by 1999 (Fig. 3d). Mean CBH was 6.3 m (Fig. 3e) in 1999.

Fire effects

By 2023 (following RO wildfires in 2012 and 2019), total tree density was significantly reduced by 50% (compared with 1999) to $606.4 \text{ trees ha}^{-1}$ ($V=210$; $p<0.001$) (Table 1). Similarly, total BA was reduced by 20% to $27.8 \text{ m}^2 \text{ ha}^{-1}$ ($V=184$; $p<0.001$). Canopy cover significantly declined from 66.7% to 43.1% ($V=207$; $p<0.001$). Fires also drove increases in average tree size from 1999 to 2023, where total QMD significantly increased by 21% to 26.3 cm ($V=9$; $p<0.001$). Similarly, the post-fire diameter distribution shifted toward the historical distribution with reductions in tree density primarily occurring in the smallest DBH classes (i.e., trees $<30 \text{ cm DBH}$) (Fig. 2c). Forest floor (i.e., litter and duff) depth was significantly reduced by 60% to 1.9 cm ($V=210$; $p<0.001$) and fine woody debris was significantly reduced by 36% to 3.4 Mg ha^{-1} ($V=153$; $p=0.038$) (Fig. 3). CFL was significantly reduced by 19% to 0.8 kg m^{-2} ($V=0$; $p<0.001$), mean CBH ($V=46$; $p=0.025$) increased significantly to 7.4 m, and 20th percentile CBH ($V=17$; $p<0.001$) increased significantly to 4.7 m by 2023. Though the fires altered forest structure and fuel loading, total density ($V=1$; $p<0.001$) and total BA ($V=9$; $p<0.001$) remained significantly higher in 2023 relative to 1870 values, and total QMD was significantly lower ($V=171$; $p<0.001$). CFL was significantly higher in 2023 compared to historical conditions in 1870 ($V=7$; $p<0.001$), increasing by 228%.

Fires also drove changes in regeneration and the abundance of old trees. While regeneration patterns were variable, overall trends indicated decreased ponderosa pine regeneration and moderate increases in sprouting hardwood species (oak and locust) between 1999 and 2023 (Table S2). Ponderosa pine regeneration decreased by 35% from 172.1 to $111.1 \text{ stems ha}^{-1}$ following the RO fires, though decreases were not significant ($V=39$; $p=0.131$). Pine regeneration was generally sparse and was only present in 40% and 15% of plots in 1999 and

2023, respectively. Total hardwood regeneration differed by 15% from 3933.4 (1999) to 4530.5 stems ha^{-1} (2023), but this change was not significant ($V=65$; $p=0.449$). Hardwoods were located across a greater amount of the landscape than pine, both before and after fire; 65% and 80% of plots had hardwood regeneration in 1999 and 2023, respectively.

The majority of pre-fire-exclusion trees alive in 1999 died by 2023 and, within this group, those that died tended to be older than those that survived. Total old-tree (i.e., pre-dating ca. 1870; all were ponderosa pine) mortality between 1999 and 2023 was 59% across the study area. Mortality of old trees was 23% between 1999 ($35.0 \text{ trees ha}^{-1}$) and 2006 ($26.8 \text{ trees ha}^{-1}$) and 47% between 2006 and 2023 ($14.1 \text{ trees ha}^{-1}$). The average center date for pre-fire-exclusion ponderosa pines that died between 1999 and 2023 was 1706; trees that died before 2006 had an average center date of 1691, and those that died after 2006 had an average center date of 1718. Pre-fire-exclusion trees that were still alive in 2023 had an average center date of 1759.

Burn severity

Satellite-derived patterns of burn severity were variable throughout the two RO fires, though most field plots burned at low severity (i.e., $\text{CBI}<1.25$) in each event (Fig. 4). On average, the 2012 Trumbull Complex burned more severely (mean [2.5th–97.5th percentiles] $\text{CBI}=0.71$ [0–1.67]) (Fig. 4a) than the 2019 Trumbull Fire ($\text{CBI}=0.60$ [0–2.68]) (Fig. 4b). However, the 2012 fire also had a lower percentage of high severity area (1.1% vs. 8.1%) and smaller high-severity patch size (6.2 ha vs. 46.6 ha) than the 2019 fire. Less than 1 ha (2012) and 12.9 ha (2019) burned at high severity and were beyond typical ponderosa pine dispersal distances (i.e., $>60 \text{ m}$; Chambers et al. 2016) from live seed sources. The 2012 fire appeared to influence fire severity in the 2019 event (Fig. 4d). Fire severity was much lower in portions of the 2019 fire that burned seven years prior (mean [2.5th–97.5th percentiles] $\text{CBI}=0.36$ [0–1.43]) when compared to places without recent fire (0.90 [0–2.79]).

Discussion

In this study, we leveraged a pre-established monitoring network with detailed reconstructions of historical reference conditions to understand the outcomes of two overlapping RO wildfires in a remote wilderness area in northern Arizona, USA. Wilderness areas are a critical element of broader conservation networks, facilitating species migration and in situ adaptation in a rapidly changing world (Belote et al. 2017; Aycrigg et al. 2022). Many of these areas have been designated as wilderness recently (e.g., <50 years), but often show signs of

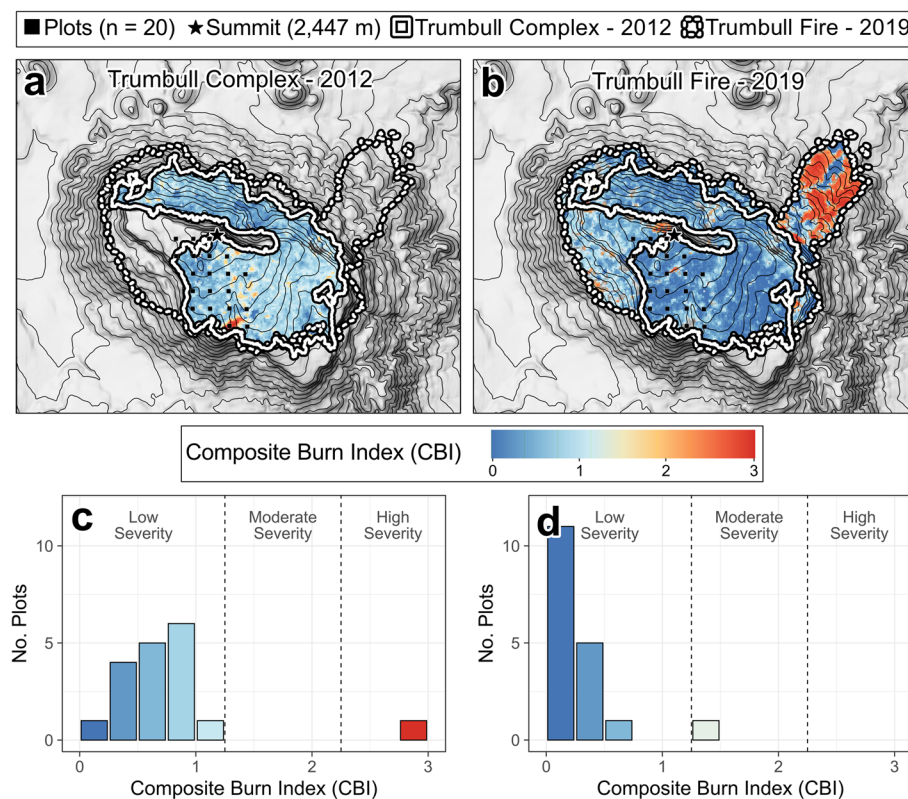


Fig. 4 Patterns of remotely sensed burn severity (i.e., predicted composite burn index; CBI; Parks et al. 2019) in the 2012 Trumbull Complex (a, c) and 2019 Trumbull Fire (b, d) in the Mount Trumbull Wilderness Area, Arizona, USA. Panels a, b show patterns of burn severity throughout each fire perimeter, restricted to 30-m pixels with at least 10% tree cover in the year before a fire (Allred et al. 2021). Panels c, d give remotely sensed burn severity at the locations of field plots ($n=20$), where dashed vertical lines are at common CBI thresholds separating low-, moderate-, and high-severity fire (Miller and Thode 2007)

ecological degradation due to the absence of fire for more than a century and may require proactive management strategies to increase resilience (Hagmann et al. 2021). While current wilderness policies constrain some active restoration approaches such as mechanical treatment, these areas also provide unique management opportunities due to their relative isolation and long distances from the wildland urban interface and other human infrastructure. For example, RO fires are more likely to be utilized as a management tool in and around wilderness areas (Iniguez et al. 2022), and they have been allowed to burn to modify forest structure and affect ecosystem function for decades in areas such as the Selway-Bitterroot Wilderness in Idaho/Montana and the Gila Wilderness in New Mexico (Van Wagtendonk 2007). Though both the research and application of RO wildfires have increased in recent years (Huffman et al. 2020; Young et al. 2020), little is known about their ability to accomplish site-specific forest restoration objectives. Fire is often referred to as a “blunt tool” when it comes to modifying and restoring forest conditions, but it also has the potential to treat vast areas (North et al. 2021) and may

limit some negative impacts more commonly associated with mechanical treatments such as soil compaction and introduction of invasive species (Sieg et al. 2003; Crawford et al. 2021); thus, the effectiveness of RO fires as a restoration tool is a critical management question. Overall, we found that forest structure shifted considerably from the timing of Euro-American settlement and widespread anthropogenic fire exclusion to 1999, but that two RO fires reduced forest densities, altered size structure, and influenced patterns of tree regeneration at our monitoring sites. For some objectives, such as conservation of old trees, these fires had undesirable effects. In general, however, the fires pushed forest conditions in the direction of HRV and likely reduced future fire severity.

Changes in forest structure and composition since fire exclusion

Prior to 1870, fires burned frequently across the study site and forest structure was relatively open and dominated by large trees. Reconstructed (1870) mean tree density and BA were at the low end of historical ranges for ponderosa pine and pine-oak sites (Reynolds et al.

2013; Wasserman et al. 2021). Reconstructed mean QMD was larger than those observed in 1999 or 2023 and similar to previously reported historical values, and diameter distributions resembled those previously reported at similar sites (Covington et al. 1997; Fulé et al. 1997; Moore et al. 2004; Roccaforte et al. 2010, 2015). The substantial increases in tree density and BA, and associated diameter distribution shifts between the onset of fire exclusion in 1870 until our pre-fire re-measurement in 1999 were not an uncommon occurrence in southwestern frequent-fire forests. Similar changes following fire exclusion were also reported across other frequent-fire forests in the western US (Harrod et al. 1999; Fulé et al. 2001; Waltz et al. 2003; Youngblood et al. 2004; Hagmann et al. 2021; Prichard et al. 2021).

Restoration effectiveness of RO fires

The significant changes in forest structure and fuels attributes following the 2012 and 2019 RO wildfires shifted pre-fire values toward the site-specific HRV, modified size distributions, and altered tree regeneration processes. Although post-fire (2023) values for density, BA, and canopy fuels were significantly higher, and QMD was significantly smaller than reconstructed (1870) values, RO wildfires reduced total tree density by 50% and total BA and CFL by 20% while increasing QMD by 4.6 cm DBH. Based on the historical tree density (62 trees ha⁻¹) at the site, the fires were more than 50% effective at attaining an HRV-based restoration objective for tree density. However, restoration targets are often set above HRV to allow a margin for unintended mortality (Covington et al. 1997). Further, the historical density may have been slightly higher (<10%) than our estimates due to potential uncertainties in our reconstruction model (Moore et al. 2004). Thus, it is likely that post-fire density may have been even closer to actual targets set by managers. Restoration targets at a nearby site were set 1.5–3 times greater than HRV, and post-treatment density remained more than three times greater than 1870 values (Roccaforte et al. 2010). The post-fire diameter distributions showed that the fires killed many small-diameter trees which likely reduced ladder fuels and future crown fire vulnerability (Roccaforte et al. 2008). Although the fires shifted forest structure toward historical conditions and likely reduced potential crown fire hazard, tree density at the site remained about 10 times greater than HRV. Sprouting hardwood species dominated tree regeneration before and after the two RO wildfires and expanded across the landscape by 2023. In contrast, ponderosa pine regeneration was sparse throughout the study and showed decreases in density and spatial extent after the two RO wildfires. Similar trends of increased hardwood and decreased pine regeneration have been reported in

this region following both mechanical treatments and prescribed fire as well as RO wildfires (Stoddard et al. 2020; Roccaforte et al. 2024), and forest conversion to shrubfields or grasslands is common following high-severity burns across the western US (Stevens-Ruman and Morgan 2019; Coop et al. 2020).

Fire can also have unintended consequences that push some forests away from desired future conditions—such as by triggering mortality of old and valuable trees, increasing heavy surface fuel loading, and creating high-severity patches that limit future tree regeneration—though we found only limited evidence for such consequences in the Mount Trumbull Wilderness. Mortality of old trees is of particular concern because these trees provide structural and genetic diversity to the ecosystem and take centuries to replace (Moore et al. 1999; Lindenmayer and Franklin 2002). It should be noted that there is presently limited information on amounts of coarse wood in forests prior to fire exclusion and mortality of large, old trees serves to replenish this component in forests where it is lacking due to past management and disturbance (Ganey et al. 2015). Old-tree mortality observed at the study site between 1999 and 2023 was high (i.e., 59%), likely due to the combined effects of drought and fire. For example, extreme drought stress in the early 2000s led to below-average radial growth rates in the Mount Trumbull area in many individual years (Roccaforte et al. 2024; Rodman et al. 2025). Drought-stressed trees are more susceptible to fire-caused mortality, even under similar levels of fire behavior and injury (van Mantgem et al. 2013; Cansler et al. 2024). Thus, we infer that while these RO fires primarily burned at low to moderate severity, they contributed to elevated levels of large-tree mortality due to lingering physiological stress from drought. However, the mortality at this site may be unique; old-tree mortality was <35% following RO wildfire at other sites in the western US (Keane et al. 2006; Stoddard et al. 2020). It should be noted that the RO fire at the pine-oak site (19% old-tree mortality) reported by Stoddard et al. (2020) was not a first-entry fire in a dense forest but instead burned across an area that had a somewhat intact frequent-fire regime (i.e., at least three fires since 1879) and therefore had relatively low tree density. In contrast, the RO fire at the mixed-conifer site (32% total old-tree mortality, ~20% ponderosa pine old-tree mortality) had not burned since 1879 and was comprised of dense stands like those at our study site (Stoddard et al. 2020).

In some cases, fires that trigger mortality can increase heavy surface fuels and coarse woody debris, which may contribute to high-severity reburns (e.g., Coppoletta et al. 2016; Lydersen et al. 2019), but we did not find evidence for this effect in our study area. For example, coarse

woody debris increased only modestly after the two RO fires (Fig. 3c), whereas fine fuels and canopy fuels generally declined. The maximum high-severity patch size in both fires at our study site (6.2 and 46.6 ha) fell within the HRV (0.1–100 ha) reported for southwestern frequent-fire forests (Iniguez et al. 2009; Yocom-Kent et al. 2015; Guiterman et al. 2017). Finally, based on established relationships between post-fire tree regeneration and distance to seed source in ponderosa pine (Chambers et al. 2016), we found that high-severity patch sizes were unlikely to meaningfully influence longer-term regeneration patterns at the landscape scale.

The future role of RO wildfire in the Mount Trumbull Wilderness

Across the Mount Trumbull Wilderness landscape, the use of RO fires is providing a foundation for the future use of beneficial fire. Indeed, within the extent of the larger, 2019 fire, we found that remotely sensed fire severity (i.e., the predicted CBI) was 60% lower in areas that burned in 2012 (mean CBI=0.36) than in areas without recent fire (mean CBI=0.9). These findings align with research throughout the western US suggesting that fire can be a self-limiting process, reducing the severity of subsequent fire for 10 years or more (Buma et al. 2020; Rodman et al. 2023; Davis et al. 2024). Thus, this landscape, which experienced a history of frequent, low-severity fire, is likely to benefit from continued maintenance burns within the HRV. Fuel availability will limit fire severity and undesired outcomes with burn intervals that are similar to the 7-year period between the RO fires studied here.

Conclusions

In this study, we capitalized on an array of long-term monitoring plots to test the effectiveness of two subsequent and overlapping wildfires for restoring forest structure and reducing hazardous fuels within a remote wilderness in northern Arizona. The long-term data provided a unique opportunity to assess forest structure changes over the 24-year period and evaluate RO wildfire effectiveness for restoring structure by using site-specific, reconstructed reference conditions. Our data showed that the wildfires shifted forest structural conditions toward the more open conditions indicated in dendroecological reconstructions and likely reduced vulnerability to future high-severity crown fires. Although the fires created more open conditions, mean tree density remained well above the historical reference levels. Additional tree reduction will be required to more closely meet management objectives based on restoration of the HRV. Frequent-fire forests in many designated wilderness areas across the western US show the effects of

prolonged fire exclusion and need restoration, but policy constraints appear to limit management options. Managers striving to maintain untrammelled character in wilderness areas may be reluctant to implement more active restoration approaches including intentional prescribed burning (Boerigter et al. 2024). Thus, RO fires are currently the primary strategy to reduce fuels and shift forest structure to meet restoration objectives. Managers relying on RO fires to increase forest resiliency to drought and mitigate severe wildfire should understand the limitations of this strategy, particularly in terms of treatment effectiveness, lack of precision, and temporal uncertainties (Huffman et al. 2018). We found that, even after two RO wildfires, forests in the Mount Trumbull Wilderness still reflect decades of fire exclusion, conditions which preceded its formal designation as a protected area. The period of time and number of additional low-intensity fires required to restore more ecologically functional conditions at this site are unknown, although a combined strategy of managing natural ignitions for restoration objectives as well as planned prescribed burning may be more effective and expedient than sole reliance on RO wildfires.

Abbreviations

RO	Resource objective
HRV	Historical range of variability
DBH	Diameter at breast height
CFL	Canopy fuel load
BA	Basal area
QMD	Quadratic mean diameter
CBI	Composite burn index
CBH	Canopy base height

Supplementary Information

The online version contains supplementary material available at <https://doi.org/10.1186/s42408-025-00415-x>.

Supplementary Material 1: Table S1. Fire, weather, and fuel moisture information for two resource objective (RO) wildfires that burned within the Mount Trumbull Wilderness, Arizona, USA. *Note: Weather and fuel moisture content (FMC) values were derived from FireFamily Plus (Bradshaw and McCormick 2000) outputs using the Mt. Logan Remote Automated Weather Station (RAWS). Bradshaw, L. and E. McCormick. 2000. FireFamily Plus user's guide, version 2.0. RMRS-GTR-67WWW. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, Utah. https://www.fs.usda.gov/rm/pubs/rmrs_gtr067.pdf.*

Supplementary Material 2: Table S2. Regeneration density (stems/ha) at the Mount Trumbull Wilderness, Arizona, USA, in 1999 (pre-fire) and 2023 (post-fire). Statistics presented are the mean (standard deviation) and below that number minimum–maximum. Total regeneration densities for each height class and total (all height classes combined) were tested for differences between years using Wilcoxon signed rank tests. Within each height class, † indicates significantly different means ($p < 0.05$) between 1999 and 2023. $n = 20$; PIPO: *Pinus ponderosa*; QUGA: *Quercus gambelii*; JUOS: *Juniperus osteosperma*; PIED: *Pinus edulis*; RONE: *Robinia Neomexicana*.

Acknowledgements

The authors would like to thank the BLM staff of the Arizona Strip District and the Arizona State Office, particularly Brandon Boshell, Maegan Cooper, Jennifer Fox, Greta Goff, and Aaron Wilkerson. We would also like to thank the staff and students at Northern Arizona University, Ecological Restoration Institute, and the School of Forestry, particularly Wally Covington, Margaret M. Moore, and Doc Smith. NAU is an equal opportunity provider.

Authors' contributions

JPR, DWH, TAH, PZF conceived and designed the research. All authors analyzed the data. JPR, KCR, JEC prepared tables and figures. JPR, DWH, KCR wrote the main manuscript text. All authors reviewed the manuscript.

Funding

This project was funded by a grant provided through the US Forest Service, agreement #23-DG-11030000-013, from congressional appropriation to the Southwest Ecological Restoration Institutes, authorized under the Southwest Forest Health and Wildfire Prevention Act (Public Law No. 108-317).

Data availability

The datasets used and/or analyzed during the current study are available at <https://doi.org/10.5281/zenodo.17253694> and/or from the corresponding author on reasonable request.

Declarations

Ethics approval and consent to participate

Not applicable.

Consent for publication

Not applicable.

Competing interests

The authors declare no competing interests.

Author details

¹Ecological Restoration Institute, Northern Arizona University, Flagstaff, AZ 86011, USA. ²Bureau of Land Management National Headquarters, Western Office, Grand Junction, CO 81505, USA. ³School of Forestry, Northern Arizona University, Flagstaff, AZ 86011, USA.

Received: 21 May 2025 Accepted: 10 September 2025

Published online: 23 October 2025

References

- Alfred, B. W., B. T. Bestelmeyer, C. S. Boyd, C. Brown, K. W. Davies, M. C. Duniway, L. M. Ellsworth, T. A. Erickson, S. D. Fuhlendorf, T. V. Griffiths, V. Jansen, M. O. Jones, J. Karl, A. Knight, J. D. Maestas, J. J. Maynard, S. E. McCord, D. E. Naugle, H. D. Starns, D. Twidwell, and D. R. Uden. 2021. Improving Landsat predictions of rangeland fractional cover with multitask learning and uncertainty. *Methods in Ecology and Evolution* 12 (5): 841–849. <https://doi.org/10.1111/2041-210X.13564>.
- Altschul, J.H. and H.C. Fairley. 1989. *Man, models, and management: an overview of the archaeology of the Arizona Strip and the management of its cultural resources*. Report prepared for USDA Forest Service and USDI Bureau of Land Management by Statistical Research, Plateau Archaeology, Dames and Moore, Inc. Contract # 53-8371-6-0054.
- Aycrigg, J. L., T. R. Mccarley, R. T. Belote, and S. Martinuzzi. 2022. Wilderness areas in a changing landscape: Changes in land use, land cover, and climate. *Ecological Applications* 32 (1): e02471. <https://doi.org/10.1002/eap.2471>.
- Bakker, J.D., A.J. Sánchez Meador, P.Z. Fulé, D.W. Huffman, and M.M. Moore. 2008. "Growing trees backwards": description of a stand reconstruction model. Pp. 136–147 in Olberding, S.D. and M.M. Moore. (tech cords). *Fort Valley Experimental Forest – a century of research 1908–2008*. US Forest Service Proceedings RMRS-P-53CD.
- Barger, R.L. and P.F. Ffolliott. 1972. *Physical characteristics and utilization of major woodland tree species in Arizona*. USDA Forest Service Research Paper INT-RP-481. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.
- Belote, R. T., M. S. Dietz, C. N. Jenkins, P. S. McKinley, G. H. Irwin, T. J. Fullman, J. C. Leppi, and G. H. Aplet. 2017. Wild, connected, and diverse: Building a more resilient system of protected areas. *Ecological Applications* 27 (4): 1050–1056. <https://doi.org/10.1002/eap.1527>.
- Boerigter, C. E., S. A. Parks, J. W. Long, J. D. Coop, M. Armstrong, and D. L. Hankins. 2024. Untrammeling the wilderness: Restoring natural conditions through the return of human-ignited fire. *Fire Ecology* 20: 76. <https://doi.org/10.1186/s42408-024-00297-5>.
- Breiman, L. 2001. Random forests. *Machine Learning* 45 (1): 5–32.
- Buma, B., S. Weiss, K. Hayes, and M. Lucash. 2020. Wildland fire reburning trends across the US West suggest only short-term negative feedback and differing climatic effects. *Environmental Research Letters* 15 (3): 034026. <https://doi.org/10.1088/1748-9326/ab6c70>.
- Bureau of Land Management (BLM). 1990. *Wilderness Management Plan: Mt. Trumbull Wilderness, Mt. Logan Wilderness, Arizona*. USDI Bureau of Land Management, Arizona Strip District, St. George, UT.
- Cansler, C. A., M. C. Wright, P. J. van Mantgem, T. M. Shearman, J. M. Varner, and S. M. Hood. 2024. Drought before fire increases tree mortality after fire. *Ecosphere* 15 (12): e70083. <https://doi.org/10.1002/ecs2.70083>.
- Chambers, M. E., P. J. Fornwalt, S. L. Malone, and M. A. Battaglia. 2016. Patterns of conifer regeneration following high severity wildfire in ponderosa pine – dominated forests of the Colorado Front Range. *Forest Ecology and Management* 378:57–67. <https://doi.org/10.1016/j.foreco.2016.07.001>.
- Clary, W. P., and A. R. Tiedemann. 1986. Distribution of biomass within small tree and shrub form *Quercus gambelii* stands. *Forest Science* 32 (1): 234–242. <https://doi.org/10.1093/forestscience/32.1.234>.
- Collins, B. M., M. Kelly, J. W. van Wagtenonk, and S. L. Stephens. 2007. Spatial patterns of large natural fires in Sierra Nevada wilderness areas. *Land-scape Ecology* 22:545–557. <https://doi.org/10.1007/s10980-006-9047-5>.
- Collins, B. M., R. G. Everett, and S. L. Stephens. 2011. Impacts of fire exclusion and recent managed fire on forest structure in old growth Sierra Nevada mixed-conifer forests. *Ecosphere* 2 (4): 51. <https://doi.org/10.1890/ES11-00026.1>.
- Coop, J. D., S. A. Parks, C. S. Stevens-Rumann, S. D. Crausbay, P. E. Higuera, M. D. Hurteau, A. Tepley, E. Whitman, T. Assal, B. M. Collins, K. T. Davis, S. Dobrowski, D. A. Falk, P. J. Fornwalt, P. Z. Fulé, B. J. Harvey, V. R. Kane, C. E. Littlefield, E. Q. Margolis, M. North, M. A. Parisien, S. Prichard, and K. C. Rodman. 2020. Wildfire-driven forest conversion in western North American landscapes. *BioScience* 70 (8): 659–673. <https://doi.org/10.1093/biosci/biaa061>.
- Coop, J. D., S. A. Parks, C. S. Stevens-Rumann, S. M. Ritter, and C. M. Hoffman. 2022. Extreme fire spread events and area burned under recent and future climate in the western USA. *Global Ecology and Biogeography* 31 (10): 1949–1959. <https://doi.org/10.1111/geb.13496>.
- Coppoletta, M., K. E. Merriam, and B. M. Collins. 2016. Post-fire vegetation and fuel development influences fire severity patterns in reburns. *Ecological Applications* 26 (3): 686–699. <https://doi.org/10.1890/15-0225.1>.
- Covington, W. W., and M. M. Moore. 1994. Southwestern ponderosa pine forest structure: Changes since Euro-American settlement. *Journal of Forestry* 92 (1): 39–47. <https://doi.org/10.1093/jof/92.1.39>.
- Covington, W. W., P. Z. Fulé, M. M. Moore, S. C. Hart, T. E. Kolb, J. N. Mast, S. S. Sackett, and M. R. Wagner. 1997. Restoring ecosystem health in ponderosa pine forests of the southwest. *Journal of Forestry* 95 (4): 23–29. <https://doi.org/10.1093/jof/95.4.23>.
- Covington, W. W., R. L. Everett, R. Steele, L. L. Irwin, T. A. Dear, and A. N. D. Auclair. 1994. Historical and anticipated changes in forest ecosystems of the Inland West of the United States. *Journal of Sustainable Forestry* 2 (1–2): 13–63. https://doi.org/10.1300/J091v02n01_02.
- Crawford, L.J., R. Heinse, M.J. Kimsey, D.S. Page-Dumroese. 2021. *Soil sustainability and harvest operations: a review*. Gen. Tech. Rep. RMRS-GTR-421. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 39 p. <https://doi.org/10.2737/RMRS-GTR-421>.
- Davis, K. T., J. Peeler, J. Fargione, R. D. Haugo, K. L. Metlen, M. D. Robles, and T. Woolley. 2024. Tamm review: A meta-analysis of thinning, prescribed fire, and wildfire effects on subsequent wildfire severity in conifer dominated forests of the Western US. *Forest Ecology and Management* 561: 121885. <https://doi.org/10.1016/j.foreco.2024.121885>.

- Donager, J. J., A. J. Sánchez Meador, and D. W. Huffman. 2022. Southwestern ponderosa pine forest patterns following wildland fires managed for resource benefit differ from reference landscapes. *Landscape Ecology* 37:285–304. <https://doi.org/10.1007/s10980-021-01352-1>.
- Egan, D., and E. A. Howell. 2001. The historical ecology handbook: A restorationist's guide to reference ecosystems. Island Press, Washington. <https://doi.org/10.5751/es-00468-070111>.
- Fulé, P.Z., C. McHugh, T.A. Heinlein, and W.W. Covington. 2001. *Potential fire behavior is reduced following forest restoration treatments*. In: Vance, R.K., W.W. Covington, C.B. Edminster, and J.A. Blake. (Eds.), *Ponderosa Pine Ecosystems Restoration and Conservation: Steps Toward Stewardship*. Proceedings of the Symposium, 25–27 April 2000, Flagstaff, Arizona. USDA Forest Service Proceedings RMRS-P-22, Ogden, UT, pp. 28–35.
- Fulé, P. Z., W. W. Covington, and M. M. Moore. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. *Ecological Applications* 7 (3): 895–908. [https://doi.org/10.1890/1051-0761\(1997\)007\[0895:DRCFEM\]2.0.CO;2](https://doi.org/10.1890/1051-0761(1997)007[0895:DRCFEM]2.0.CO;2).
- Fulé, P. Z., W. W. Covington, H. B. Smith, J. D. Springer, T. A. Heinlein, K. D. Huisinga, and M. M. Moore. 2002. Comparing ecological restoration alternatives: Grand Canyon, Arizona. *Forest Ecology and Management* 170 (1–3): 19–41. [https://doi.org/10.1016/S0378-1127\(01\)00759-9](https://doi.org/10.1016/S0378-1127(01)00759-9).
- Ganey, J. L., and W. M. Block. 1994. A comparison of two techniques for measuring canopy closure. *Western Journal of Applied Forestry* 9 (1): 21–23. <https://doi.org/10.1093/wjaf/9.1.21>.
- Ganey, J. L., B. J. Bird, L. S. Baggett, and J. S. Jenness. 2015. Density of large snags and logs in northern Arizona mixed-conifer and ponderosa pine forests. *Forest Science* 61 (2): 353–362. <https://doi.org/10.5849/forsci.13-125>.
- Gorelick, N., M. Hancher, M. Dixon, S. Ilyushchenko, D. Thau, and R. Moore. 2017. Google Earth Engine: Planetary-scale geospatial analysis for everyone. *Remote Sensing of Environment* 202:18–27. <https://doi.org/10.1016/j.rse.2017.06.031>.
- Grier, C. C., K. J. Elliott, and D. G. McCullough. 1992. Biomass distribution and productivity of *Pinus edulis-Juniperus monosperma* woodlands of north-central Arizona. *Forest Ecology and Management* 50 (3–4): 331–350. [https://doi.org/10.1016/0378-1127\(92\)90346-B](https://doi.org/10.1016/0378-1127(92)90346-B).
- Guiterman, C. H., E. Q. Margolis, C. D. Allen, D. A. Falk, and T. W. Swetnam. 2017. Long-term persistence and fire resilience of oak shrubfields in dry conifer forests of northern New Mexico. *Ecosystems* 21:943–959. <https://doi.org/10.1007/s10021-017-0192-2>.
- Hagmann, R. K., P. F. Hessburg, S. J. Prichard, N. A. Povak, P. M. Brown, P. Z. Fulé, R. E. Keane, E. E. Knapp, J. M. Lydersen, K. L. Metlen, M. J. Reilly, A. J. Sánchez Meador, S. L. Stephens, J. T. Stevens, A. H. Taylor, L. L. Yocom, M. A. Battaglia, D. J. Churchill, L. D. Daniels, D. A. Falk, P. Henson, J. D. Johnston, M. A. Krawchuk, C. R. Levine, G. W. Meigs, A. G. Merschel, M. P. North, H. D. Safford, T. W. Swetnam, and A. E. M. Waltz. 2021. Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests. *Ecological Applications* 31 (8): e02431. <https://doi.org/10.1002/eap.2431>.
- Harrod, R. J., B. H. McRae, and W. E. Hartl. 1999. Historical stand reconstruction in ponderosa pine forests to guide silvicultural prescriptions. *Forest Ecology and Management* 114 (2–3): 433–466. [https://doi.org/10.1016/S0378-1127\(98\)00373-9](https://doi.org/10.1016/S0378-1127(98)00373-9).
- Heinlein, T.A., P.Z. Fulé, A.E.M. Waltz, and J.D. Springer. 1999. *Changes in ponderosa pine forests of the Mt. Trumbull wilderness*. Unpublished report. Ecological Restoration Institute, Northern Arizona University, Flagstaff, AZ. Available at <https://cdm17192.contentdm.oclc.org/digital/collection/p17192coll1/id/373/rec/13>.
- Hessburg, P. F., C. L. Miller, S. A. Parks, N. A. Povak, A. H. Taylor, P. E. Higuera, S. J. Prichard, M. P. North, B. M. Collins, M. D. Hurteau, A. J. Larson, C. D. Allen, S. L. Stephens, H. Rivera-Huerta, C. S. Stevens-Rumann, L. D. Daniels, Z. Gedalof, R. W. Gray, V. R. Kane, D. J. Churchill, R. K. Hagmann, T. A. Spies, C. A. Cansler, R. T. Belote, T. T. Veblen, M. A. Battaglia, C. Hoffman, C. N. Skinner, H. D. Safford, and R. B. Salter. 2019. Climate, environment, and disturbance history govern resilience of western North American forests. *Frontiers in Ecology and Evolution* 7:1–27. <https://doi.org/10.3389/fevo.2019.00239>.
- Higgins, A. M., K. M. Waring, and A. E. Thode. 2015. The effects of burn entry and burn severity on ponderosa pine and mixed conifer forests in Grand Canyon National Park. *International Journal of Wildland Fire* 24 (4): 495–506. <https://doi.org/10.1071/WF13111>.
- Holmes, R. L. 1983. Computer-assisted quality control in tree-ring dating and measurement. *Tree-Ring Bulletin* 43:69–78.
- Huffman, D. W., A. J. Sánchez Meador, M. T. Stoddard, J. E. Crouse, and J. P. Roccaforte. 2017. Efficacy of resource objective wildfires for restoration of ponderosa pine (*Pinus ponderosa*) forests in northern Arizona. *Forest Ecology and Management* 389:395–403. <https://doi.org/10.1016/j.foreco.2016.12.036>.
- Huffman, D.W., M.M. Moore, W.W. Covington, J.E. Crouse, and P.Z. Fulé. 2001. *Ponderosa pine forest reconstruction: comparisons with historical data*. Pp. 3–8 in Vance, R.K., Edminster, C.B., Covington, W.W., Blake, J.A. (comps.). *Ponderosa pine ecosystems restoration: steps toward stewardship*. US Forest Service Proceedings RMRS-P-22.
- Huffman, D. W., J. E. Crouse, A. J. Sánchez Meador, J. D. Springer, and M. T. Stoddard. 2018. Restoration benefits of re-entry with resource objective wildfire on a ponderosa pine landscape in northern Arizona, USA. *Forest Ecology and Management* 408:16–24. <https://doi.org/10.1016/j.foreco.2017.10.032>.
- Huffman, D. W., J. P. Roccaforte, J. D. Springer, and J. E. Crouse. 2020. Restoration applications of resource objective wildfires in western US forests: A status of knowledge review. *Fire Ecology* 16:18. <https://doi.org/10.1186/s42408-020-00077-x>.
- Iniguez, J. M., A. M. Evans, S. Dadashi, J. D. Young, M. D. Meyer, A. E. Thode, S. J. Hedwall, S. M. McCaffrey, S. D. Fillmore, and R. Bean. 2022. Comparing geography and severity of managed wildfires in California and the Southwest USA before and after the implementation of the 2009 policy guidance. *Forests* 13 (5): 793. <https://doi.org/10.3390/f13050793>.
- Iniguez, J. M., T. W. Swetnam, and C. H. Baisan. 2009. Spatially and temporally variable fire regime on Rincon Peak, Arizona, USA. *Fire Ecology* 5:3–21. <https://doi.org/10.4996/fireecology.0501003>.
- Keane, R. E., S. Arno, and L. J. Dickinson. 2006. The complexity of managing fire-dependent ecosystems in wilderness: Relict ponderosa pine in the Bob Marshall Wilderness. *Ecological Restoration* 24 (2): 71–78. <https://doi.org/10.3368/er.24.2.71>.
- Key, C.H. and N.C. Benson. 2006. Landscape Assessment (LA): Sampling and Analysis Methods. In D. C. Lutes (Ed.), FIREMON: Fire Effects Monitoring and Inventory System. RMRS-GTR-164-CD (pp. LA1–LA51). Rocky Mountain Research Station, USDA Forest Service.
- Kreider, M. R., M. R. Jaffe, J. K. Berkey, S. A. Parks, and A. J. Larson. 2023. The scientific value of fire in wilderness. *Fire Ecology* 19 (1): 36. <https://doi.org/10.1186/s42408-023-00195-2>.
- Lindenmayer, D.B. and J.F. Franklin. 2002. Conserving forest biodiversity: a comprehensive multiscaled approach. Island Press, Washington, D.C. <https://doi.org/10.1071/pc030291>.
- Lydersen, J. M., B. M. Collins, M. Coppoletta, M. R. Jaffe, H. Northrop, and S. L. Stephens. 2019. Fuel dynamics and reburn severity following high-severity fire in a Sierra Nevada, USA, mixed-conifer forest. *Fire Ecology* 15 (1): 1–14. <https://doi.org/10.1186/s42408-019-0060-x>.
- Mast, J.N. P.Z. Fulé, M.M. Moore, W.W. Covington, and A.E.M. Waltz. 1999. Restoration of presettlement age structure of an Arizona ponderosa pine forest. *Ecological Applications* 9 (1): 228–239. <https://doi.org/10.2307/2641181>.
- Miller, C. 2012. The hidden consequences of fire suppression. *Park Science* 28 (3): 75–80.
- Miller, J. D., and A. E. Thode. 2007. Quantifying burn severity in a heterogeneous landscape with a relative version of the delta normalized burn ratio (dNBR). *Remote Sensing of Environment* 109 (1): 66–80. <https://doi.org/10.1016/j.rse.2006.12.006>.
- Miller, J. D., B. M. Collins, J. A. Lutz, S. L. Stephens, J. W. van Wagtenonk, and D. A. Yasuda. 2012. Differences in wildfires among ecoregions and land management agencies in the Sierra Nevada region, California, USA. *Ecosphere* 3 (9): 80. <https://doi.org/10.1890/ES12-00158.1>.
- Moore, M. M., D. W. Huffman, P. Z. Fulé, W. W. Covington, and J. E. Crouse. 2004. Comparison of historical and contemporary forest structure and composition on permanent plots in southwestern ponderosa pine forests. *Forest Science* 50 (2): 162–176. <https://doi.org/10.1093/forestscience/50.2.162>.
- Moore, M. M., W. W. Covington, and P. Z. Fulé. 1999. Reference condition and ecological restoration: A southwestern ponderosa pine perspective. *Ecological Applications* 9 (4): 1266–1277. [https://doi.org/10.1890/1051-0761\(1999\)009\[1266:RCAERA\]2.0.CO;2](https://doi.org/10.1890/1051-0761(1999)009[1266:RCAERA]2.0.CO;2).

- Naficy, C. E., E. G. Keeling, P. Landres, P. F. Hessburg, T. T. Veblen, and A. Sala. 2016. Wilderness in the 21st century: A framework for testing assumptions about ecological intervention in wilderness using a case study of fire ecology in the Rocky Mountains. *Journal of Forestry* 114 (3): 384–395. <https://doi.org/10.5849/jof.15-010>.
- North, M., R. York, B. Collins, M. Hurteau, G. Jones, E. Knapp, L. Kobziar, H. McCann, M. Meyer, S. Stephens, R. Tompkins, and C. L. Tubbesing. 2021. Pyrosilviculture needed for landscape resilience of dry western United States forests. *Journal of Forestry* 119 (5): 520–544. <https://doi.org/10.1093/jofore/fvab026>.
- Parks, S. A., C. Miller, C. R. Nelson, and Z. A. Holden. 2014. 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>.
- Parks, S. A., J. D. Coop, and K. T. Davis. 2025. Intensifying fire season aridity portends ongoing expansion of severe wildfire in western US forests. *Global Change Biology* 31 (8): e70429. <https://doi.org/10.1111/gcb.70429>.
- Parks, S. A., L. M. Holsinger, M. J. Koontz, L. Collins, E. Whitman, M. A. Parisien, R. A. Loehman, J. L. Barnes, J. F. Bourdon, J. Boucher, Y. Boucher, A. C. Caprio, A. Collingwood, R. J. Hall, J. Park, L. B. Saperstein, C. Smetanka, R. J. Smith, and N. Soverel. 2019. Giving ecological meaning to satellite-derived fire severity metrics across North American forests. *Remote Sensing* 11 (14): 1–19. <https://doi.org/10.3390/rs11141735>.
- Prichard, S. J., P. F. Hessburg, R. K. Hagmann, N. A. Povak, S. Z. Dobrowski, M. D. Hurteau, V. R. Kane, R. E. Keane, L. N. Kobziar, C. A. Kolden, M. North, S. A. Parks, H. D. Safford, J. T. Stevens, L. L. Yocom, D. J. Churchill, R. W. Gray, D. W. Huffman, F. K. Lake, and P. Khatri-Chhetri. 2021. Adapting western North American forests to climate change and wildfires: 10 common questions. *Ecological Applications* 31 (8): e02433. <https://doi.org/10.1002/eap.2433>.
- PRISM Climate Group, Oregon State University, 2023. <http://prism.oregonstate.edu>.
- Reynolds R.T., A.J. Sánchez Meador, J.A. Youtz, T. Nicolet, M.S. Matonis, P.L. Jackson, D.G. DeLorenzo, and A.D. Graves. 2013. *Restoring composition and structure in southwestern frequent-fire forests: a science-based framework for improving ecosystem resiliency*. RMRS-GTR-310. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado. <https://doi.org/10.2737/RMRS-GTR-310>.
- Roccaforte, J. P., D. W. Huffman, K. C. Rodman, J. E. Crouse, R. J. Pedersen, D. P. Normandin, and P. Z. Fulé. 2024. Long-term ecological responses to landscape-scale restoration in a western United States dry forest. *Restoration Ecology* 32 (7): e14181. <https://doi.org/10.1111/rec.14181>.
- Roccaforte, J. P., D. W. Huffman, P. Z. Fulé, W. W. Covington, W. W. Chancellor, M. T. Stoddard, and J. E. Crouse. 2015. Forest structure and fuels dynamics following ponderosa pine restoration treatments, White Mountains, Arizona, U.S.A. *Forest Ecology and Management* 337:174–185. <https://doi.org/10.1016/j.foreco.2014.11.001>.
- Roccaforte, J. P., P. Z. Fulé, and W. W. Covington. 2008. Landscape-scale changes in canopy fuels and potential fire behaviour following ponderosa pine restoration treatments. *International Journal of Wildland Fire* 17 (2): 293–303. <https://doi.org/10.1071/WF06120>.
- Roccaforte, J. P., P. Z. Fulé, and W. W. Covington. 2010. Monitoring landscape-scale ponderosa pine restoration treatment implementation and effectiveness. *Restoration Ecology* 18 (6): 820–833. <https://doi.org/10.1111/j.1526-100X.2008.00508.x>.
- Rodman, K. C., J. B. Bradford, A. M. Formanack, P. Z. Fulé, D. W. Huffman, T. E. Kolb, A. T. Miller-ter Kulie, D. P. Normandin, K. Ogle, R. J. Pederson, D. R. Schlaepfer, M. T. Stoddard, and A. E. M. Waltz. 2025. Restoration treatments enhance tree growth and alter climatic constraints during extreme drought. *Ecological Applications* 35 (1): e3072. <https://doi.org/10.1002/eap.3072>.
- Rodman, K. C., K. T. Davis, S. A. Parks, T. B. Chapman, J. D. Coop, J. M. Iniguez, J. P. Roccaforte, A. J. Sánchez Meador, J. D. Springer, C. S. Stevens-Rumann, M. T. Stoddard, A. E. M. Waltz, and T. N. Wasserman. 2023. Refuge-yeah or refuge-nah? Predicting locations of forest resistance and recruitment in a fiery world. *Global Change Biology* 29 (24): 7029–7050. <https://doi.org/10.1111/gcb.16939>.
- Rogers, J.J., J.M. Prosser, L.D. Garrett, and M.G. Ryan. 1984. *ECOSIM: a system for projecting multiresource outputs under alternative forest management regimes*. USDA Forest Service, Administrative Report, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.
- Roos, C. I., C. H. Guiterman, E. Q. Margolis, T. W. Swetnam, N. C. Laluk, K. F. Thompson, C. Toya, C. A. Farris, P. Z. Fulé, J. M. Iniguez, J. M. Kaib, C. D. O'Connor, and L. Whitehair. 2022. Indigenous fire management and cross-scale fire-climate relationships in the Southwest United States from 1500 to 1900 CE. *Science Advances* 8 (49): eabq3221.
- Sánchez Meador, A. J., P. F. Parysow, and M. M. Moore. 2010. Historical stem-mapped permanent plots increase precision of reconstructed reference data in ponderosa pine forests of northern Arizona. *Restoration Ecology* 18 (2): 224–234. <https://doi.org/10.1111/j.1526-100X.2008.00442.x>.
- Schultz, C. A., T. Jedd, and R. D. Beam. 2012. The collaborative forest landscape restoration program: A history and overview of the first projects. *Journal of Forestry* 110 (7): 381–391. <https://doi.org/10.5849/jof.11-082>.
- Schoennagel, T., and C. R. Nelson. 2011. Restoration relevance of recent National Fire Plan treatments in forests of the western United States. *Frontiers in Ecology and the Environment* 9 (5): 271–277. <https://doi.org/10.1890/090199>.
- Seig, C., B. G. Phillips, and L. P. Moser. 2003. Exotic invasive plants. In *Ecological restoration of southwestern ponderosa pine forests*, ed. P. Freiderici, 251–267. Washington, D.C: Island Press.
- Society for Ecological Restoration. 2004. The SER International primer on ecological restoration. <https://www.ser.org>.
- Speer, J. H. 2010. Fundamentals of tree-ring research. *University of Arizona Press, Tucson, AZ*. <https://doi.org/10.3959/1536-1098-70.2.161>.
- Steel, Z. L., M. J. Koontz, and H. D. Safford. 2018. The changing landscape of wildfire: Burn pattern trends and implications for California's yellow pine and mixed conifer forests. *Landscape Ecology* 33 (7): 1159–1176. <https://doi.org/10.1007/s10980-018-0665-5>.
- Stephens, S. L., and P. Z. Fulé. 2005. Western pine forests with continuing frequent fire regimes: Possible reference sites for management. *Journal of Forestry* 103 (7): 357–362. <https://doi.org/10.1093/jof/103.7.357>.
- Stephens, S.L., Collins, B.M., Biber, E., and Fulé, P.Z. 2005. U.S. federal fire and forest policy: emphasizing resilience in dry forests. *Ecosphere* 7 (11): e01584 <https://doi.org/10.1002/ecs2.1584>.
- Stephens, S. L., M. A. Battaglia, D. J. Churchill, B. M. Collins, M. Coppoletta, C. M. Hoffman, J. M. Lydersen, M. P. North, R. A. Parsons, S. M. Ritter, and J. T. Stevens. 2021. Forest restoration and fuels reduction: Convergent or divergent? *BioScience* 71 (1): 85–101. <https://doi.org/10.1093/biosci/biaa134>.
- Stevens-Rumann, C. S., and P. Morgan. 2019. Tree regeneration following wildfires in the western US: A review. *Fire Ecology* 15:15. <https://doi.org/10.1186/s42408-019-0032-1>.
- Stoddard, M. T., P. Z. Fulé, D. W. Huffman, A. J. Sánchez Meador, and J. P. Roccaforte. 2020. Ecosystem management applications of resource objective wildfires in forests of the Grand Canyon National Park, USA. *International Journal of Wildland Fire* 29 (2): 190–200. <https://doi.org/10.1071/wf19067>.
- Thomas, J.W., R.G. Anderson, C. Maser, and E.L. Bull. 1979. *Snags*. Pages 60–77 In: Thomas, J.W. (ed) *Wildlife habitats in managed forests—the Blue Mountains of Oregon and Washington*. USDA Agricultural Handbook 553 U.S. Department of Agriculture, Forest Service, Washington D.C. <https://doi.org/10.2307/3807910>.
- Thompson, M. P., C. D. O'Connor, B. M. Gannon, M. D. Caggiano, C. J. Dunn, C. A. Schultz, D. E. Calkin, B. Pietruszka, S. M. Greiner, R. Stratton, and J. T. Morissette. 2022. Potential operational delineations: New horizons for proactive, risk-informed strategic land and fire management. *Fire Ecology* 18:17. <https://doi.org/10.1186/s42408-022-00139-2>.
- Turner, M.G. and R.H. Gardner. 2015. *Landscape Ecology in Theory and Practice*. In Springer (2nd ed.). Springer. <https://doi.org/10.1007/978-1-4939-2794-4>.
- USDI National Park Service. 2003. *Fire monitoring handbook*. Fire Management Program Center, National Interagency Fire Center, Boise, ID. 274 p. <https://www.nps.gov/orgs/1965/upload/fire-effects-monitoring-handbook.pdf>.
- US Forest Service. 2022. *Confronting the wildfire crisis: a strategy for protecting communities and improving resilience in America's forest*. FS-1187a. US Forest Service 47 p.
- van Mantgem, P., M. Schwartz, and M. Keifer. 2001. Monitoring fire effects for managed burns and wildfires: coming to terms with pseudoreplication.

- Natural Areas Journal* 21 (3):266–273. https://www.naturalareas.org/docs/v21_3_01_pp266_273.pdf.
- van Mantgem, P. J., J. C. Niesmith, M. Keifer, E. E. Knapp, A. Flint, and L. Flint. 2013. Climatic stress increases forest fire severity across the western United States. *Ecology Letters* 16 (9): 1151–1156. <https://doi.org/10.1111/ele.12151>.
- Van Wagten, J. W. 2007. The history and evolution of wildland fire use. *Fire Ecology* 3 (2): 3–17. <https://doi.org/10.4996/fireecology.0302003>.
- Waltz, A. E. M., P. Z. Fulé, W. W. Covington, and M. M. Moore. 2003. Diversity in ponderosa pine forest structure following ecological restoration treatments. *Forest Science* 49 (6): 885–900. <https://doi.org/10.1093/forestscience/49.6.885>.
- Wasserman, T. N., A. E. M. Waltz, J. P. Roccaforte, J. D. Springer, and J. E. Crouse. 2021. Natural regeneration responses to thinning and burning treatments in ponderosa pine forests and implications for restoration. *Journal of Forestry Research* 33:741–753. <https://doi.org/10.1007/s11676-021-01404-x>.
- White, A. S. 1985. Presettlement regeneration patterns in a southwestern ponderosa pine stand. *Ecology* 66 (2): 589–594. <https://doi.org/10.2307/1940407>.
- White, P. S., and J. L. Walker. 1997. Approximating nature's variation: Selecting and using reference information in restoration ecology. *Restoration Ecology* 5 (4): 338–349. <https://doi.org/10.1046/j.1526-100X.1997.00547.x>.
- Williams, A. P., B. J. Cook, and J. E. Smerdon. 2022. Rapid intensification of the emerging southwestern North American megadrought in 2020–2021. *Nature Climate Change* 12:232–234. <https://doi.org/10.1038/s41558-022-01290-z>.
- Yocom-Kent, L. L., P. Z. Fulé, W. A. Bunn, and E. G. Gdula. 2015. Historical high-severity fire patches in mixed-conifer forests. *Canadian Journal of Forest Research* 45 (11): 1587–1596. <https://doi.org/10.1139/cjfr-2015-0128>.
- Young, J. D., A. M. Evans, J. M. Iniguez, A. Thode, M. D. Meyer, S. J. Hedwall, S. McCaffrey, P. Shin, and C. H. Huang. 2020. Effects of policy change on wildland fire management strategies: Evidence for a paradigm shift in the western US? *International Journal of Wildland Fire* 29 (10): 857–877. <https://doi.org/10.1071/WF19189>.
- Youngblood, A., T. Max, and K. Coe. 2004. Stand structure in eastside old-growth ponderosa pine forests of Oregon and northern California. *Forest Ecology and Management* 199 (2–3): 191–217. <https://doi.org/10.1016/j.foreco.2004.05.056>.

Publisher's Note

Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.