Article

Forest Resistance to Extended Drought Enhanced by Prescribed Fire in Low Elevation Forests of the Sierra Nevada

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Abstract: Prescribed fire reduces fire hazards by removing dead and live fuels (small trees and shrubs). Reductions in forest density following prescribed fire treatments (often in concert with mechanical treatments) may also lessen competition so that residual trees might be more likely to survive when confronted with additional stressors, such as drought. The current evidence for these effects is mixed and additional study is needed. Previous work found increased tree survivorship in low elevation forests with a recent history of fire during the early years of an intense drought (2012 to 2014) in national parks in the southern Sierra Nevada. We extend these observations through additional years of intense drought and continuing elevated tree mortality through 2017 at Sequoia and Kings Canyon National Parks. Relative to unburned sites, we found that burned sites had lower stem density and had lower proportions of recently dead trees (for stems ≤ 47.5 cm dbh) that presumably died during the drought. Differences in recent tree mortality among burned and unburned sites held for both fir (white fir and red fir) and pine (sugar pine and ponderosa pine) species. Unlike earlier results, models of individual tree mortality probability supported an interaction between plot burn status and tree size, suggesting the effect of prescribed fire was limited to small trees. We consider differences with other recent results and discuss potential management implications including trade-offs between large tree mortality following prescribed fire and increased drought resistance.

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1. Introduction

Drought is a common condition in seasonally dry coniferous forests in the western U.S., but as temperatures rise the length and intensity of droughts in this region are likely to increase in coming decades [1]. The occurrence of “hotter droughts” is associated with vegetation stress and forest diebacks [2]. Drought can compromise tree water relations (loss of xylem function and carbon starvation), and if sufficiently severe can lead directly to mortality [3]. Drought may also weaken defenses, so that trees are susceptible to secondary pest attack, particularly bark beetles (e.g., *Dendroctonus* spp., *Ips* spp., and *Scolytus* spp.). Severe drought may also be a trigger for bark beetle outbreaks where both weakened and unaffected trees may be killed [4].

A history of fire exclusion in western U.S. forests may exacerbate drought effects. Many coniferous forests in this region historically experienced high frequency, low severity fires, but fire exclusion over the past century has led to surface fuel accumulations and the ingrowth of high numbers of smaller, shade tolerant trees (reviewed in [5]). These conditions not only represent increased fire hazards, but high stem density has also been linked to increased competition and drought-related mortality [6,7]. Stands with high

stem density may also be more susceptible to drought-related bark beetle attack and pest-mediated mortality [8,9].

Forest managers are keen to identify actions that could promote resistance and resilience to drought. Mechanical thinning and prescribed fire (often used in combination) are two primary approaches managers use to manipulate stand conditions at large scales. These actions are typically undertaken as fuels reduction tactics, but reductions in stem density may result in improved growth of residual trees (a key index of individual tree vigor) [10]. Reductions in stem density may sufficiently reduce competition to improve residual tree growth [11] and potentially may reduce drought-related mortality [12] and mortality associated with bark beetles [11,13]. However, under outbreak conditions it is possible that biotic agents could override effects of stand density reductions.

It is uncertain whether a recent history of fire without mechanical treatments influences drought response. Fire often injures surviving trees, directly impacting photosynthetic capacity and vascular system function (reviewed in [14]). However, fire may improve growing conditions by removing competing vegetation [15], and stimulate the development of structures that may improve resistance to bark beetle attack [16]. Prescribed fire is designed to disproportionately kill small trees, so that large reductions in stand basal area are uncommon [17]. Yet, prescribed fire may remove a substantial number of small trees [18], potentially reducing the competitive environment for surviving trees.

Recently, van Mantgem et al. [19], found that individual tree mortality probabilities were lower in stands with a recent history of fire in low elevation forests of the southern Sierra Nevada during the early phase of an intense multi-year drought. The drought in this area lasted from 2012 to 2016 [4], exceeding by several years the final year of data collection reported in van Mantgem et al. [19]. Here, we revisit forests in this area to determine how continued drought may have changed forest-tree mortality patterns in stands with and without a recent history of prescribed fire.

2. Materials and Methods

2.1. Study Site

We used observations from low- to mid-elevation yellow pine and mixed conifer forests at Sequoia and Kings Canyon National Parks, California (Figure 1). The region is characterized by a Mediterranean climate, with wet, snowy winters and long, dry summers. Soils are primarily derived from granitic parent material, and portions of the southern Sierra Nevada feature a deep regolith capable of storing plant-accessible moisture [20]. This region is characterized by steep slopes and high topographic variability, resulting in spatially heterogeneous water availability. Fire history reconstructions suggest that low- to mid-elevation forests in this area historically experienced high frequency (approximate fire return interval of 5 to 15 years), low severity fires but generally have not burned since the late 1800s [21]. Starting in the late 1960's park managers initiated a prescribed burning program, including the management of lightning ignited fires. Target conditions for restoration burns in mixed conifer forests include fuel reduction of 60 to 95% and stem densities of 50 to 250 trees ha⁻¹ for trees < 80 cm dbh (stem diameter at breast height, 1.37 m), and 10 to 75 trees ha⁻¹ for trees ≥ 80 cm dbh. Target conditions for subsequent maintenance burns specify a range of fuel loads, varying from 11 to 135 MT ha⁻¹, and a range of forest gap/patch sizes across the landscape (75–95% 0.1 to 1 ha, 50–25% 1 to 10 ha, and <1% 10 to 100 ha).

Figure 1. Location of study sites in Sequoia and Kings Canyon National Parks. Red symbols represent locations of burned plots, blue symbols represent locations of unburned plots.

2.2. Forest Plot Data

We used three data sources that measured individual trees in burned and unburned stands: (1) the National Park Service Fire Effects plots (FMH) at Sequoia and Kings Canyon National Parks and supplemental mortality plots, (2) the Sequoia National Park Fire and Fire Surrogate (FFS) plots, and (3) the U.S. Geological Survey Sierra Nevada Forest Dynamics Plot Network (SNFDPN) (Table 1). This provided a data set that was similar, but not identical, to the data used in van Mantgem et al. [19]. An important difference among these data sources used in this study is how individual tree mortality was assessed. The exact year of individual tree mortality was unknown for the FMH and FFS plots, and assessments of recent, drought-related, tree mortality was based on the retention of dead foliage. The SNFDPN plots are surveyed annually for mortality, so the exact year of individual tree death is known. There is a degree of classification error using foliage characteristics as an indicator of recent mortality, but this method has been shown to follow patterns of mortality obtained from annual observations at our study location [4]. Below we provide details on each data source.

2.2.1. National Park Service Fire Effects Plots (FMH) and Supplemental Mortality Plots

Fire effects monitoring plots at Sequoia and Kings Canyon National Parks follow the protocols provided in the Fire Monitoring Handbook [22]. FMH protocols establish 50×20 m (0.1 ha) plots at random locations prior to burning. At Sequoia and Kings Canyon National Parks all live trees ≥ 1.37 m tall were measured for dbh, and identified to species. Typically trees were assessed for mortality (no green needles) immediately post-fire and at 1-, 2-, 5-, and 10-year intervals, and every 10 years thereafter. For this study we resurveyed plots in 2016 and 2017, noting trees that had recently died based

on retention of dead foliage since the most recent plot measurement (i.e., trees that were recorded as dead during previous plot surveys were excluded).

Table 1. Summary statistics of burned and unburned plots for stems >12.5 cm stem diameter at breast height (dbh), showing the median and range for year of fire (for burned plots), stem density, quadratic mean diameter (QMD), basal area (BA), and plot elevation.

Data Source	Plot Status	Plot Count	Year of Fire	Stem Density (ha)	QMD (cm)	BA (m ² ha ^{−1})	Elevation (m)
FMH	Burned	20	2003 (1990 to 2007)	140 (10 to 270)	73 (47 to 252)	55 (9 to 478)	2030 (1878 to 2421)
	Unburned	8	-	225 (120 to 420)	53 (44 to 256)	70 (26 to 619)	2116 (1869 to 2256)
FMH supplemental	Burned	26	2002 (1969 to 2007)	200 (50 to 780)	57 (32 to 232)	64 (21 to 362)	2039 (1707 to 2330)
	Unburned	32	-	335 (60 to 530)	58 (36 to 108)	74 (16 to 262)	1963 (1701 to 2404)
FFS	Burned	34	2001 (2001 to 2002)	135 (43 to 352)	66 (32 to 118)	45 (7 to 100)	2025 (1928 to 2156)
	Unburned	30	-	256 (91 to 418)	55 (38 to 72)	54 (22 to 89)	1956 (1909 to 2150)
SNFDPN	Burned	5	2001 (2001 to 2004)	104 (47 to 143)	77 (58 to 93)	48 (29 to 52)	2128 (2018 to 2202)
	Unburned	10	-	210 (56 to 304)	58 (49 to 115)	58 (15 to 155)	2117 (1941 to 2405)

To supplement the FMH plot data, we surveyed additional areas in 2017 that had burned prior to 2012 or did not have a history of recent fire. The areas were sampled using randomly located 0.1 ha circular plots, measuring all trees ≥ 1.37 m in height for dbh (in 5 cm increments), species identity, and tree status (live or recently dead). Crews in the field made visual judgements on whether an individual tree had been killed by drought, relying on retention of dead foliage as an indicator of recent mortality. Trees judged to be dead prior to drought or killed by fire were not included in analyses.

2.2.2. Sequoia National Park Fire and Fire Surrogate Plots (FFS)

The Fire/Fire Surrogate Research Study at Sequoia National Park is located in mixed conifer forest, and was designed to contrast the outcome of spring versus fall prescribed fire [23]. Units were randomly designated for fall prescribed fire (burned in 2001), spring prescribed fire (burned in 2002), or no treatment. Randomly located 0.1 ha plots were installed prior to prescribed burning and were measured immediately post-fire and again in 2004. The original surveys recorded dbh and species identity for all live stems >10 cm dbh. In 2015 we surveyed plots in the burned units that were sampled by Nesmith et al. [24] and resurveyed all plots in the unburned units, recording tree live/dead status. Crews judged trees to be recently dead based on retention of dead foliage.

2.2.3. Sierra Nevada Forest Dynamics Plot Network (SNFDPN)

The Sierra Nevada Forest Dynamics Plot Network is a plot-based forest monitoring project that, since starting in the early 1980s, has measured birth, death, and growth of individual trees across a steep elevation gradient in the Sierra Nevada [25]. This network includes several plots with and without a recent history of fire. Plots in this network are generally 1.0 ha in size and individual birth and death of trees ≥ 1.37 m in height is tracked annually, with individual stem dbh measured approximately every five years. We used annual observations, with observations of recent mortality from 2014 to 2017, following Stephenson et al. [4]. The drought in California occurred from 2012 to 2016, but Stephenson et al. [4] showed that tree mortality in our area was correlated with drought over a three-year running average of drought conditions, and elevated mortality continued into 2017. We used stem diameters from the most recent dbh measurement before 2014.

2.3. Analyses

We matched plots with local records of recent fire history, including the Sequoia and Kings Canyon National Parks 2020 fire history spatial data (<https://irma.nps.gov/DataStore/Reference/Profile/2284464>, accessed on 24 April 2021). We removed three burned plots where wildfire was the most recent fire event. We also excluded plots that had burned more recently than 2007 to avoid measuring direct effects of fire (including delayed tree mortality) prior to the onset of drought ($n = 29$). We included all other plots with a

recent history of fire (median prescribed fire year = 2002, range = 1969 to 2007). To check if our results were sensitive to the timing of prescribed fire, we conducted analyses excluding plots that had burned more than 20, 30, or 40 years prior to field observations, finding broadly similar results. Drought mortality was concentrated in low- to mid-elevation stands [4], so we excluded plots that were >2500 m in elevation (burned plots $n = 4$, unburned plots $n = 2$). Burned plots did not extend into the lowest elevation range of the unburned plots (burned plot minimum elevation = 1707 m, unburned plot minimum elevation = 1467 m). To improve comparability of burned and unburned plots, we imposed a minimum elevation limit of 1700 m, removing 18 unburned plots. We view this as conservative, as these low elevation unburned plots had high proportions of recently dead trees. Our final data set consisted of 85 burned plots and 80 unburned plots. The Sequoia and Kings Canyon National Parks 2020 fire history spatial data indicated that our plots were located in 24 separately identified fires with a median prescribed fire area of 55 ha (range = 20 to 2504 ha). The Sequoia and Kings Canyon National Parks 2020 fire history spatial data and internal park records showed that most burned plots had a single recorded burn, but 16 plots had burned two times, and one plot had burned three times. The most recent prescribed fires for our plots occurred in the summer and fall (median most recent burn month = September, range = June to December).

Trees were grouped into 5 cm dbh categories to normalize stem size measurements among data sources. We used the midpoint of the size categories as the dbh measurement for all analyses. To compare stem density (trees ha^{-1}), quadratic mean stem diameter (cm), and stand basal area ($\text{m}^2 \text{ha}^{-1}$) in burned and unburned plots we considered only trees that were >12.5 cm dbh (minimum dbh midpoint across all of the data sources). Stand-level stem density, quadratic mean diameter, and basal area in burned and unburned plots were compared using bootstrapped Welch two sample t -tests with 1000 iterations.

Drought-related mortality patterns can vary widely by species. We therefore considered mortality patterns for dominant species, which included white fir (*Abies concolor* (Gordon & Glend.) Lindley), red fir (*A. magnifica* A. Murr.), sugar pine (*Pinus lambertiana* Douglas), and ponderosa pine (*P. ponderosa* Laws). Other species were too sparse or had too few observations of mortality for analysis. Initial data exploration suggested species within genera were responding similarly, so we combined fir and pine into species groups for analysis.

To determine differences in the proportion of recently dead trees, we set a minimum group size of 20 individuals with at least five observations of mortality (with a group defined as a combination of species group, burn status, and tree size). Following these criteria we had sufficient data for three size classes (<22.5, 22.5 to 47.5, and >47.5 cm dbh). We compared the proportion of recently dead trees using Fisher's exact test.

The proportion of recently dead trees may obscure differences due to individual tree characteristics, so we modeled individual tree mortality probabilities using a Bayesian generalized linear mixed model (GLMM) (binomial with a logit link). This approach allowed us to estimate individual tree mortality probabilities (tree status, 0 = live; 1 = dead), based on dbh, elevation, and history of prescribed fire (binary 0 or 1), with plot-level variation estimated from random intercepts (Table 2). To reasonably estimate plot-level intercepts we only considered plots that had at least five fir or pine individuals (removing nine burned plots). Initial models suggested differences between fir and pines for the effect of stem diameter, so we fit an interaction term between dbh and species group. We standardized all continuous predictor variables by subtracting the mean and dividing by the standard deviation, allowing better comparisons among model estimates. Model parameters were estimated using weak priors with four chains of 5000 iterations each. GLMM fit was checked using posterior predictive plots. Model accuracy was assessed using the area under the receiver operating characteristic curve (AUC), with $\text{AUC} > 0.70$ indicating acceptable model accuracy. We compared the performance of the full model against a model that did not include a term for plot burn status using leave-one-out cross-validation (LOO). LOO calculates model predictive accuracy when sequentially holding out

each observation in the data. Model comparisons were done by calculating the differences in expected log predictive density ($\Delta ELPD$). Values of $\Delta ELPD$ less than four indicate similar predictive accuracy between the models, with the standard error of $\Delta ELPD$ supplying an estimate of uncertainty of the comparison (<https://avehtari.github.io/modelselection/CV-FAQ.html>, accessed on 1 July 2021). Analyses were conducted using the R statistical language [26] using the “MKinfer”, “rstanarm”, and “loo” packages [27–29]. The data for our analyses are available on ScienceBase [30].

Table 2. Summary statistics of data used in models of individual tree mortality probabilities. Columns for dbh and elevation show median and range of stem diameters. Plots had to have at least five fir or pine individuals to estimate plot-level intercepts for generalized linear mixed models (GLMMs).

Species Group	Plot Status	Live	Dead	dbh (cm)	Elevation (m)
Fir	Burned	1871	206	22 (2.5 to 232.5)	2124 (1707 to 2400)
	Unburned	5448	991	12 (2.5 to 177.5)	2086 (1701 to 2405)
Pine	Burned	325	38	18 (2.5 to 152.5)	2023 (1707 to 2256)
	Unburned	469	136	18 (2.5 to 237.5)	1941 (1701 to 2405)

3. Results

The burned plots had lower average stem density and greater quadratic mean stem diameter compared to the unburned plots (Figure 2). Burned plots had slightly lower average basal area compared to unburned plots, but this difference was non-significant (bootstrapped Welch two sample *t*-test, $p = 0.43$).

Figure 2. Differences in plot-level stem density (a), QMD (b), and basal area (c) for burned and unburned plots. Bar height represents average values +1 standard deviation (bootstrapped Welch two sample *t*-tests: ** $p < 0.01$, *** $p < 0.001$).

Overall proportion of recently dead trees for all members of the fir and pine species group was lower in burned plots (0.10) relative to unburned plots (0.16) (Fisher’s exact test: $p < 0.001$). Both fir and pine mortality patterns varied by stem size, with differences between burned and unburned plots being most dramatic for smaller size classes (Figure 3). For large stems the differences in proportion of recently dead trees between burned and unburned plots was not statistically significant (fir $p = 0.815$, pine $p = 0.095$). Proportion of recently dead trees was similar across size classes for fir but increased sharply with size for pines.

Models for individual tree mortality probabilities indicated that trees in the pine group had higher overall mortality probabilities, although the 95% CI for this term overlapped

0 (Figure 4). Mortality probability increased substantially with increasing stem diameter, with this effect especially pronounced for pines (i.e., positive estimate for the interaction between dbh and pine species group). Dead trees were associated with low elevations, although the estimated effect for elevation had 95% CIs that slightly overlapped 0. The term for burn status showed that trees in unburned plots had higher mortality probabilities, although there was a negative interaction between stem diameter and plot burn status. This comports with the findings for proportional mortality, where the effects of burn status were most pronounced for small trees. The AUC for this model was 0.74, indicating acceptable discrimination. Comparing the full model against a model that did not include terms for burn status resulted in a ΔELPD of -4.7 ($\text{SE} = 3.9$), suggesting a somewhat better model predictive accuracy for the model that contained terms for burn status.

Figure 3. Proportions of trees that died in burned and unburned plots by species group and size class (Fisher’s exact test: ** $p < 0.01$, *** $p < 0.001$).

Figure 4. Model posterior median estimates and 95% credible intervals for probabilities of individual tree mortality in burned and unburned plots at Sequoia and Kings Canyon National Parks. The fir species group and burned plots are the reference conditions used to compare against the pine species group and unburned plots. Terms for dbh*pine and dbh*unburned represent interactions. Observations for dbh and elevation were scaled to better allow comparisons among estimates.

4. Discussion

Our results suggest that general patterns observed during the early years of drought of reduced tree mortality [19] continued to hold across additional years of intense drought. Stem density was lower in burned versus unburned sites, supporting the idea that reduced competition may be at least partly responsible for the contrasting mortality response. Higher quadratic mean diameter in burned versus unburned sites indicate that prescribed fires were associated with stands with larger trees. Like earlier results, we found that the effects of a history of fire on mortality during the drought were most easily seen in smaller trees. Unlike earlier results, models of individual tree mortality probability supported an interaction between burn status and tree size, suggesting that large trees were responding differently during continued drought. Effects of prescribed fire on drought mortality were not readily detectable for trees > 47.5 cm dbh. We interpret these patterns in light of the recent findings of Stephenson et al. [4], who found that bark beetle preference was a strong determinant of mortality during the drought at our study site, with beetles selecting slow growing white fir of all sizes, and both slow and fast growing large pines.

Smaller trees may have benefited from reduced competitive environment in burned stands, reducing mortality during the drought. Competition from surrounding trees is rarely associated with mortality for large trees in this area under non-drought conditions [25], and we assume that reducing small tree stem density would likely have little effect on large tree mortality during drought. Reduction in competition was likely most important for tree mortalities that were directly caused by drought.

Prescribed fire may also influence mortality associated with bark beetles. Thinned stands may contain trees with few competitors and high vigor that are better able to defend against beetle attack. For example, Westlind and Kerns [13] found reduced mortality associated with bark beetles for ponderosa pine stands that were thinned and repeatedly prescribed burned Oregon. Stephenson et al. [4] found beetles preferentially selected slow growing (i.e., low vigor trees) fir that presumably had compromised beetle defenses during the drought in our area. Tree defenses against bark beetles may also be stimulated by prescribed fire [16]. Beyond altering individual tree characteristics, prescribed fire may lead to stand conditions that affect bark beetle activity. This idea is supported by Buonanduci et al. [31], who found low neighborhood stem density decreased probabilities of bark beetle mortality during outbreak conditions for medium to large lodgepole pine (*Pinus contorta* Doug.) in Colorado. Reductions in stem density following prescribed fire may increase air turbulence that can dissipate beetle aggregating pheromones [32]. It is possible that trees in burned stands were better defended or faced reduced beetle pressures, although we did not find strong evidence for differences in large pine mortality which was a hallmark of the recent drought in the Sierra Nevada.

While we hypothesize that lower drought mortality in burned sites could be a result of reduced stand density, our data are insufficient to test this idea. Plot stem density may not reflect tree competition for the small plots used in this study. Many trees may be near plot edges with unknown competitive environments past the plot boundary [33]. A recent study in Yosemite and Sequoia national parks (using some of the same SNFDPN plots as our study) showed that plots that burned > 7 years prior to drought had lower plot-level mortality, although there was not a clear relationship with stand density [34]. A better understanding of the mechanisms driving observed mortality patterns requires more detailed information for individual trees (e.g., recent growth and crown condition), local competitive environments, (e.g., distance based competitive indices), and factors associated with mortality (e.g., assessments of mortality causes) [25,34,35]. Other potentially important mortality predictors are individual tree vigor, insect and pathogen pressure, and plot-level environmental conditions (e.g., soil depth and water holding capacity). Additional shortcomings of our data include field assessments of recently dead trees for the FMH and FFS data which may be subject to error. Combining data sets using somewhat different field methods may also be problematic, although the information we combined were standard forest metrics (species identity, stem diameter). This study provides evidence that

stand conditions created by prescribed fire may influence forest response to drought, but experimental approaches will be needed to identify the biotic and abiotic conditions that drive this response.

Fuel reduction treatments (mechanical thinning, prescribed fire, or both) can create long-lasting reductions in stem density [18]. Treated stands with lower stem density and basal area were found to have lower tree mortality probabilities during the recent drought in the Sierra Nevada [12]. There are, however, conflicting reports of whether prescribed fire without mechanical treatments can create forest structures that promote resistance to drought. Knapp et al. [35] found high mortality in areas that were prescribed burned without mechanical thinning. However, the fires in that study occurred in 2013, coinciding with the drought. In contrast, we did not consider sites that had burned following 2007, well before the onset of drought. Another recent report from the Teakettle Experimental Forest, near our study sites, found that prescribed fires in 2001 led to higher beetle-caused mortality during the drought, particularly for large sugar pine [36]. The authors speculated that fire-caused injuries promoted successful beetle attacks and greater subsequent mortality during the drought. Site-level differences may help explain disparities with our results. Prescribed fires at Teakettle were small (centered around 4 ha study plots) and the prescribed fire without mechanical thinning treatment did not lead to large differences in local competitive environments for individual trees compared to unburned sites [36]. In contrast, the prescribed fires used in our analyses were larger and removed substantial numbers of trees. As a result, fires at our sites may have promoted drought survivorship for small trees due to decreased competition and lower pest activity. Other potential differences include the temporal and spatial range of the study sites, with the Teakettle burn-only treatments measured from three large plots burned in 2001, while our data are from >80 mostly small plots measured over a range of years and distributed over a relatively large area. As a result, likely there are site differences (e.g., topography, hydrology, soils, and local pest dynamics) that may be at least partially responsible for differing outcomes to prescribed fire. Further studies that consider these sources of variation may help to explain the contrasting results.

Burned stands used in this study featured lower stem density relative to unburned stands, suggesting that forest structure following fire may be approaching historical conditions. Empirical data and reconstructions of historical Sierra Nevada mixed conifer forests estimate stem density to range from 59 to 315 trees ha⁻¹, with basal areas from 21 to 56 m² ha⁻¹ (with some variation due to minimum tree diameter used) (reviewed in [37]). Relative to these estimates burned stands in our study had somewhat higher average stand density (197 trees ha⁻¹) and basal area (69 m² ha⁻¹) (Figure 2), although distributions were skewed with lower median values (Table 1). The majority of our burned plots only experienced one fire, and additional prescribed fire treatments may be needed to achieve closer alignment with estimated historical conditions.

It is possible that drought resistance could be improved with further reductions in stem density, but mortality was still high in both burned and unburned stands. It is unrealistic to expect that stand density management will fully protect forests against continued, repeated extreme drought. Such events will likely continue to trigger pest outbreaks, causing mortality that in some cases could be largely insensitive to stand conditions, potentially amplifying the effects of drought [4,38]. A consideration for managers is balancing the potential mortality caused by prescribed fire, relative to prescribed fire's potential to reduce mortality to future disturbance.

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