fire & fuels management

Influence of Repeated Prescribed Fire on Tree Growth and Mortality in *Pinus resinosa* Forests, Northern Minnesota

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Prescribed fire is widely used for ecological restoration and fuel reduction in fire-dependent ecosystems, most of which are also prone to drought. Despite the importance of drought in fire-adapted forests, little is known about the cumulative effects of repeated prescribed burning on tree growth and related response to drought. Using dendrochronological data in red pine (*Pinus resinosa* Ait.)-dominated forests in northern Minnesota, USA, we examined growth responses before and after understory prescribed fires between 1960 and 1970 to assess whether repeated burning influences growth responses of overstory trees and vulnerability of overstory tree growth to drought. We found no difference in tree-level growth vulnerability to drought, expressed as growth resistance, resilience, and recovery, between areas receiving prescribed fire treatments and untreated forests. Annual mortality rates during the period of active burning were also low (less than 2%) in all treatments. These findings indicate that prescribed fire can be effectively integrated into management plans and climate change adaptation strategies for red pine forest ecosystems without significant short- or long-term negative consequences for growth or mortality rates of overstory trees.

Keywords: controlled burns, dendrochronology, drought vulnerability, underburning

Prescribed fire is a management practice applied to forest ecosystems around the globe, with prescriptions varying in seasonality and frequency (Knapp et al. 2009), depending on ecosystem and site-specific objectives. These objectives may include fuel reduction, forest regeneration, and ecological restoration (Fernandes and Botelho 2003, Noss et al. 2006). Repeated prescribed fires may affect tree health directly via root, crown, or stem injury (Busse et al. 2000, Stephens and Finney 2002) or indirectly via changes to forest soil and nutrients (Alban 1977, Boerner et al. 2008), seed banks (Keyser et al. 2012), understory vegetation structure and composition (Buckman 1964, Neumann and Dickmann 2001), or forest structure (Agee and Skinner 2005). Under impending changes in climate, repeated prescribed fires may interact with drought to affect tree health. This is a potentially important consequence, considering that droughts are projected to increase in fre-

quency, duration, and intensity with possible negative outcomes for many forested regions (Allen et al. 2015). Yet, little is known on the long-term effects of repeated prescribed fires on vulnerability of tree growth (as a metric of tree health) to drought. These knowledge gaps assume particular importance in fire-dependent forest ecosystems, where decades of fire exclusion have radically altered historical fire regimes, leading to excessive fuel accumulation, and consequently raising the probability for high severity crown fires (Fulé 2008, Ryan et al. 2013). In these ecosystems, the use of prescribed fire for fuel management and ecosystem restoration may therefore become increasingly important as we move toward a more fire-prone climate (Seager et al. 2007, Giorgi and Lionello 2008, Dai 2013).

Red pine forests of the US Great Lakes region, like many other fire-dependent systems around the globe, have experienced nearly 100 years of wildfire suppression leading to significant alterations to

This article uses metric units; the applicable conversion factors are: centimeters (cm): 1 cm = 0.39 in.; meters (m): 1 m = 3.3 ft; square meters (m^2): 1 m^2 = 10.8 ft^2; hectares (ha): 1 ha = 2.47 ac.

forest structure and composition (Aaseng et al. 2003). Before this period of active fire suppression, surface fires were common (every 5–50 years), whereas crown fires were infrequent (150–250 years) in this forest type (Heinselman 1973, Frelich 2002, Fraver and Palik 2012). Suppression of surface fires has resulted in a dramatic increase in live and dead fuels (Sands and Abrams 2011) and has elevated concerns about severe fires with behavior that may be outside the natural range of variability (Scheller et al. 2005). As with other pine-dominated systems, the use of understory prescribed fire has been suggested as a strategy to reduce fuels, reduce competition from shrubs, and prepare seedbeds for pine regeneration, while maintaining a living, productive overstory (Alban 1977). However, because few long-term prescribed burning studies in red pine forests exist to validate this recommendation, the effects of prescribed fire on near-term survival and long-term patterns of tree health remain poorly understood. In particular, there is no information on how prescribed fire interacts with drought to affect tree health.

We examined the effects of repeated prescribed fire treatments on tree growth (as a metric of tree health), mortality, and vulnerability of tree growth to drought in a red pine-dominated forest in northern Minnesota, USA. By linking long-term plot measurements with dendrochronological data, we assessed tree growth response and drought vulnerability after the implementation of three treatments including stand density reduction via thinning followed by either no burning, annual growing season prescribed fire, or periodic dormant season prescribed fire. In this study, we focus on burn frequency and assume that burning season has a negligible effect on postfire overstory tree growth. Although burning season can have significantly different effects on understory vegetation (Alban 1977, Knapp et al. 2009), there were few to no effects of burning season on red pine growth reported for the Red Pine Prescribed Burning Experiment (Alban 1977) and in several other studies (e.g., Robbins and Myers 1992, Sala et al. 2005, Hatten et al. 2012). Specifically, we asked whether and how burning treatments influenced the following: red pine basal area growth before, during, and after an active burn period; tree mortality during the period of active burning; and growth responses of trees during a severe drought event that occurred after cessation of burnings.

Methods
Study Area

This study used treatments from the Red Pine Prescribed Burning Experiment (Cutfoot Experimental Forest [CEF], Chippewa National Forest) established by the US Department of Agriculture (USDA) Forest Service in northern Minnesota, USA (47°32′ N, 94°05′ W) in 1959. The experiment was designed to assess prescribed burning impacts on regeneration, woody shrub encroachment, fuels reduction, and soil characteristics. All stands used in the experiment had regenerated naturally after a wildfire in the late 1860s and were approximately 90 years old when stands were thinned in the winter of 1959–1960. At that time, stand density of experimental units was reduced by thinning from below to a basal area of 28 m^2/ha to create uniform overstory conditions. No other thinnings have occurred since. Subsequently, various combinations of frequency (annual, biennial, and periodic) and season (dormant and summer) of prescribed fire were applied between 1960 and 1970 (see Buckman (1964) for more details on site conditions and the experiment).

Here, we contrast two prescribed fire treatments and a control treatment to gain insights on the long-lasting impacts of different

Table 1. Mean proportion of total basal area (%) and SD for the main species occurring across the three study treatments (reference year 1959).

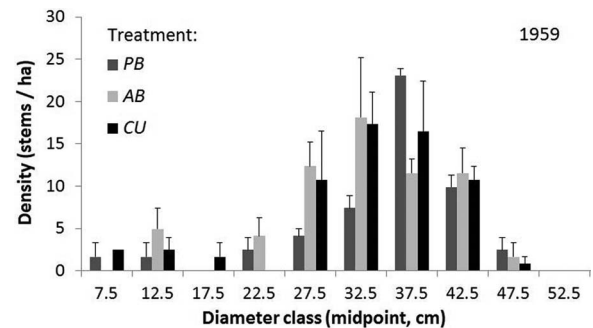|	PB		AB		CU	
	Mean	SD	Mean	SD	Mean	SD
Pinus resinosa	98.6	2.3	97.9	2.6	94.4	1.7
Pinus strobus	1.1	1.9	0.1	0.1	0	0
Betula papyrifera	0	0	0.2	0.4	2.8	2.8
Other	0.3	0.5	1.8	2.1	2.8	1.2

Figure 1. Tree diameter distribution for the three treatments (PB, AB, and CU) at the onset of the burning experiment (1959). Means are based on three replications per treatment, and bars represent 1 SE. Diameter classes: 7.5 (5–9.99 cm), 12.5 (10–14.99 cm), 17.5 (15–19.99 cm), 22.5 (20–24.99 cm), 27.5 (25–29.99 cm), 32.5 (30–34.99 cm), 37.5 (35–39.99 cm), 42.5 (40–44.99 cm), 47.5 (45–49.99 cm), and 52.5 (50–54.99 cm).

prescribed fire regimes on growth response to burning and drought. From the Red Pine Prescribed Burning Experiment, we sampled three replicate experimental units from the periodic late dormant season burning treatment (hereafter PB for periodic burning: prescribed burned twice, May 1960 and 1969), three from the annual growing season burning (hereafter AB for annual burning: prescribed burned 11 times in June–July, from 1960 to 1970), and three from the control treatment (hereafter CU for control unburned). In total, our study consisted of nine 0.4-ha experimental units, each with one circular 0.08-ha permanent sample plot.

At the onset of the burning experiment, red pine was the dominant canopy species among trees >10 cm dbh (1.3-m height) in all treatments, ranging from 94 to 100% of total stand basal area (Table 1, reference year 1959). Less abundant species were mainly in the subcanopy layer and include eastern white pine (*Pinus strobus* L.), paper birch (*Betula papyrifera* Marsh.), northern red oak (*Quercus rubra* L.), balsam fir (*Abies balsamea* [L.] Mill.), red maple (*Acer rubrum* L.), white spruce (*Picea glauca* [Moench] Voss), and bigtooth aspen (*Populus grandidentata* Michx.). All treatments were characterized by comparable and unimodal tree diameter distributions (Figure 1).

Field and Laboratory Methods

Within each experimental unit, a single 0.08-ha circular plot was placed near the center and permanently marked before treatment in 1959. Within this plot all living trees (dbh >10 cm) were measured (species, dbh, and vigor), and all dead trees were identified to species and recorded. Inventory measurements were collected in 1959, 1964, 1969, 1997, 2005, 2010, and 2014 by the USDA Forest Service Northern Research Station.

In 2010 and 2014, we resampled each of the nine 0.08-ha plots selected for this study, collecting overstory data (species, dbh, height, and vigor) from all trees (dbh >10 cm). Each tree was visually examined for the presence of fire scars via observation of damage to the outer stem. Fire scars and stem damage were recorded to characterize aspects of tree vigor, to provide a possible indication of lasting alterations in tree functionality that may be related to fire-induced modifications to stem function (Ducrey et al. 1996).

In addition, each tree (dbh >10 cm) was cored (one core per tree) at breast height, resulting in 21–56 increment cores per treatment plot. Increment cores were prepared, cross-dated, and measured using standard dendrochronological procedures (Speer 2010). The dating and ring-width measurements of each series were checked for errors with time series correlation analyses using the program COFECHA (Grissino-Mayer 2001). To refine age estimates for cores that missed the pith, we applied Duncan's (1989) geometric pith location method and checked these corrections with Applequist's (1958) visual method. Establishment dates were obtained from cross-dated cores, refining ring counts when necessary. Ring-width chronologies were converted to annual tree basal area increment (BAI) based on back-reconstructed dbh values derived from dbh inside bark at the time of coring and radial increments over time (Bunn 2008). Bark thickness was estimated by the bark factor equation proposed by Fowler and Damschroder (1988) and subtracted from dbh to obtain the corresponding dbh inside bark. BAI was used instead of ring width, because BAI is less dependent on tree diameter and thus reduces the need for detrending (Biondi 1999), which could also remove low-frequency variability and produce larger errors toward the end of the tree-ring chronology (Kohler et al. 2010). The period examined for drought impacts in this study was beyond the juvenile growth trend commonly observed for BAI series and therefore was unaffected by ontogenetic changes in growth rate.

Data Analysis

Short-term effects of repeated understory prescribed fire were evaluated by overstory tree density and mortality. Mean differences in stem density between just before the beginning (1959) and the end (1970) of the active burning treatment period were used to compare the effects of the three treatments (PB, AB, and CU) on stem density by diameter class. Annual rates of mortality (percentage) were calculated from inventory measurements for the active burn period 1959–1970, according to the method proposed by Sheil and May (1996):

$$\text{Mortality} = 1 - [1 - (M_1/N_0)]^{1/t} \tag{1}$$

where M_1 is the total number of stems that died during the measurement interval (t, years) and N_0 is the total number of live stems at the beginning of the measurement interval.

To examine the long-term effects of repeated understory fires, we evaluated overstory tree growth vulnerability to drought. Two droughts were chosen, one before the beginning of the active burn period and one after burning ceased. To identify and characterize drought years, we used the standardized precipitation evapotranspiration index (SPEI, unitless) (Vicente-Serrano et al. 2010), a multiscalar index based on precipitation and temperature data and therefore suitable for detecting, monitoring, comparing, and analyzing different drought types and impacts in the context of global warming. The SPEI reflects both water surplus (positive values) and water deficit (negative values) as standardized deviations from the average monthly climatic water balance (Vicente-Serrano et al. 2010). The SPEI was obtained using the "spei" function (Beguería and Vicente-Serrano 2013). The function returns a time series of the SPEI, given a time series of the climatic water balance (precipitation minus potential evapotranspiration). The computation of potential evapotranspiration was obtained according to the Hargreaves equation ("Hargreaves" function) (Hargreaves 1994). All functions were performed using the statistical computing software R (version 3.2.1) (R Core Team 2014), within the package SPEI (version 1.6) (Beguería and Vicente-Serrano 2013). The SPEI based on a 6-month integration period was chosen because of a detected stronger response of red pine to cumulative droughts over 6 months. Such responses were analyzed in a related work on the CEF (A. Bottero, A.W. D'Amato, B.J. Palik, S. Fraver, J.B. Bradford, 2014, unpubl. data). Here, a cross-dated increment series of 42 trees from the CEF were standardized using the ARSTAN software (Laboratory of Tree-Ring Research, University of Arizona). Each growth-ring series was standardized with a cubic smoothing spline with 50% frequency responses at 100 years to remove the influences of age-related growth and stand dynamics (Cook and Holmes 1986, Kipfmueller et al. 2010, Speer 2010). The mean standard chronology was calculated via robust estimation of the mean value function to increase the common signal and minimize the effect of outliers (Cook 1985). Correlations between the index curve and temperature, precipitation, SPEI based on different integration periods (1, 3, 6, 12, and 18 months), self-calibrating Palmer Drought Severity Index (sc-PDSI), and ratio between precipitation and potential evapotranspiration (P/PET) were calculated using a sliding window of months from June (previous year) to October (current year) for the period 1901–2009. We used the "dcc" function of the package "bootRes" (Zang and Biondi 2013) in R. The function calculates response and correlation functions from tree ring chronologies and monthly climatic data, using a bootstrapped procedure to calculate their significance and confidence intervals. Growth in the chronologies examined was positively related to May–July precipitation of the current year ($P < 0.05$). On the contrary, inverse relationships between growth and June temperature of the current year were noted. Correlations with SPEI based on a 6-month integration period were significantly greater compared with only temperature, precipitation, sc-PDSI, P/PET, or SPEI based on other integration periods. Input meteorological data (monthly temperature and precipitation) were obtained from the PRISM Climate Group database.[1]

In this article, severe drought was defined as extraordinary departure from the mean SPEI, lower than the mean by 1 SD for the period 1901–2009. The two selected droughts were chosen among previously identified severe events and further confirmed by comparison with historical records (e.g., national water summary—floods and droughts: Minnesota). In addition, SPEI values for the 5 years before and after the drought periods were taken as background years for comparison. Thus, for each selected drought, the SPEI values during the 5 years predrought and the 5 years postdrought were compared to evaluate and exclude the potential confounding influence of differences in climatic conditions in the periods before and after the selected droughts. The first drought (1948, with SPEI during the growing season ranging from −1.4 to −0.7) (Figure 2) was selected to evaluate the drought response before the establishment of the burning experiment in 1960. The second drought (1976–1977, with SPEI during the growing season ranging from
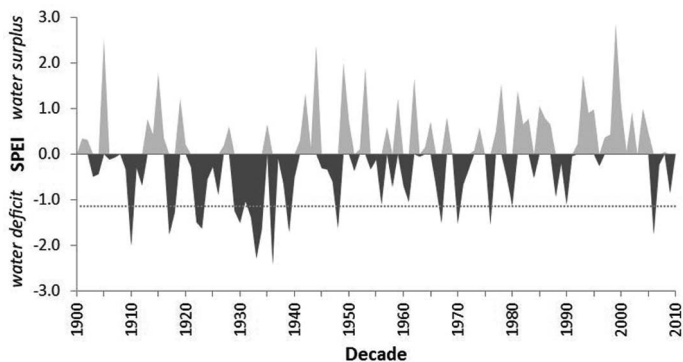
Figure 2. SPEI for the period 1901–2009. The horizontal dotted line represents the threshold (mean SPEI − 1 SD for the period 1901–2009) to identify severe droughts.

Figure 3. Age structures of the forest stands in the three treatments (PB, AB, and CU), reconstructed from cores collected in 2010 and 2014. Means are based on three replications per treatment, and bars represent 1 SE. The triangle denotes the known fire event. Note: the main recruitment period occurred between 1886 and 1892. Therefore, the protracted recruitment period observed (decades 1880 and 1890) is to some degree a visual artifact.

−1.9 to −0.6 in 1976, and from −0.4 to −0.3 in 1977) allowed us to determine the drought response after cessation of burning.

Growth responses to drought were quantified at the tree level and are expressed as growth resistance, resilience, and recovery using 5-year windows, adapting the approaches developed by Kohler et al. (2010), Lloret et al. (2011), and D'Amato et al. (2013). The tree-level BAI for each year was used as the unit of analysis for examining tree-level growth resistance, resilience, and recovery from past droughts. Tree-level resistance was defined as the ability to avoid growth reduction during drought, expressed as BAI_D/BAI_{pre}, where BAI_D is average tree-level BAI during a drought event and BAI_{pre} is the average tree-level BAI during the 5 years before the event. Resilience was defined as the ability to regain predrought growth after drought, calculated as BAI_{post}/BAI_{pre}, where BAI_{post} is the average tree-level BAI during the 5 years after a drought. Finally, recovery was defined as the increase in growth after drought-induced growth depression, expressed as BAI_{post}/BAI_D.

For the periods before, during, and after (1950–2009) the burn period, the effects of the application of understory fire (burned versus control) on tree-level growth (BAI) were analyzed using a mixed-model analysis of variance (ANOVA). Tree-level BAI was included as the dependent variable, application of fire as the fixed effect, and experimental units as a random effect. A similar analysis was conducted using treatment (PB, AB, and CU) as the fixed effect. The effects of prescribed burning on resistance, resilience, and recovery were examined using separate ANOVAs for each drought event. ANOVAs examining the influence of prescribed fire on tree density change and mortality rates over the active burning period (1959–1970) were also conducted. Verification of the homoscedasticity of variance and the normal distribution of residuals before all ANOVAs were conducted and Tukey's honest significant difference method was used to compare treatment means. The analyses performed were conducted using the statistical computing software R (version 3.2.1) (R Core Team 2014).

Results

All treatments showed primarily single-cohort age structures (Figure 3). In all experimental units, red pines were the oldest trees and preceded the establishment of other species. Additional recruitment, primarily of species other than *P. resinosa*, occurred in both burning treatments (PB and AB) mainly after the cessation of burnings.

Fire scars on overstory trees were infrequent 40+ years after the last fire in the study area. Only a few of the individuals sampled in
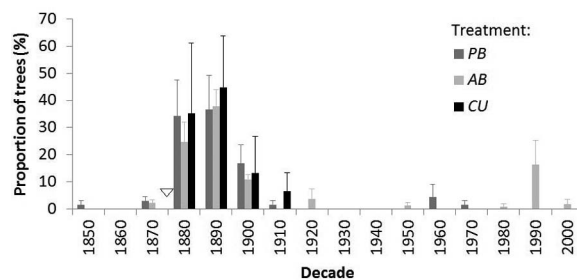
PB and AB (3 and 4%, respectively) had fire scars related to the prescribed fires of 1960–1970. Scar faces were mostly closed in PB and partially rotten in AB (reference year 2014). No fire scars were recorded on any of the individuals sampled at the control treatment.

Tree-level growth (i.e., BAI) for the three treatments is shown in Figure 4. Tree-level growth fluctuated markedly over time in all treatments, but growth patterns were comparable among treatments. Severe drought events similarly reduced growth in all treatments. The analysis of growth changes in relation to burning treatment, conducted for each year of the period 1950–2010, showed no significant effects of the control or the treatments considered separately on tree-level BAI (ANOVA tests, $P > 0.05$). Tree-level BAIs were the same in AB, PB, and control treatments. A significant decrease in BAI was only observed in 1989 at AB (ANOVA tests, $P = 0.0102$) in comparison to PB and control.

The analysis of stand density change from 1959 to 1970, the active burning period, showed that AB experienced the most severe reduction in total number of stems per hectare (Figure 5a). Similarly, the reduced density of trees in lower diameter classes (12.5 cm) did not show an equivalent increase in adjacent, larger diameter classes (17.5 cm or higher) in AB. Annual mortality rates from 1959 to 1970, were generally low (<1%) (Figure 5b). The highest annual mortality rate was recorded on small diameter classes (0.6% for the class 12.5 cm and 0.2% for the class 17.5 cm) at AB, which had significantly greater mortality rates than PB and CU. There was no difference in annual mortality rates between PB and CU, with rates of <0.4% for both these treatments during the active burning period.

For the two selected drought events (1948 and 1976–1977), we found no significant differences in SPEI values between the 5 years predrought (respectively, 1943–1947 and 1971–1975) and the 5 years postdrought (respectively, 1949–1953 and 1978–1982) ($P > 0.05$, Student's *t*-tests). Likewise, we found no significant differences in tree-level growth vulnerability to drought, expressed as growth resistance, resilience, and recovery, among the three treatments before the establishment of the experiment (reference drought year 1948, $P > 0.05$, ANOVA tests) (Figure 6a). Similarly, no significant differences in resistance, resilience, and recovery were found after cessation of burnings (reference drought years 1976–1977, $P > 0.05$, ANOVA tests) (Figure 6b).
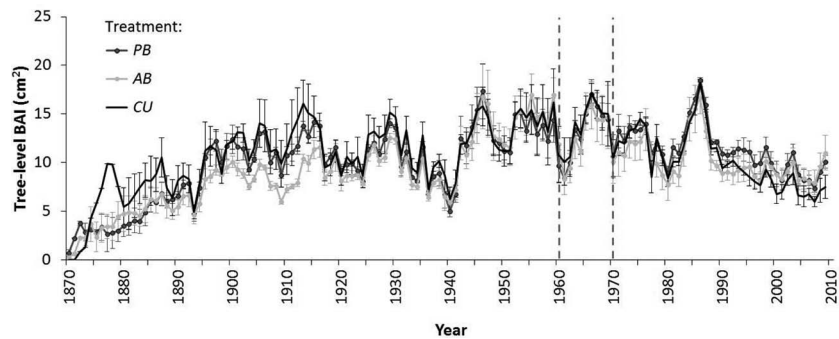
Figure 4. Tree-level BAI for the three treatments (PB, AB, and CU). Means are based on three replications per treatment, and bars represent 1 SE. Vertical dashed lines indicate the beginning and the end of the burning experiment.
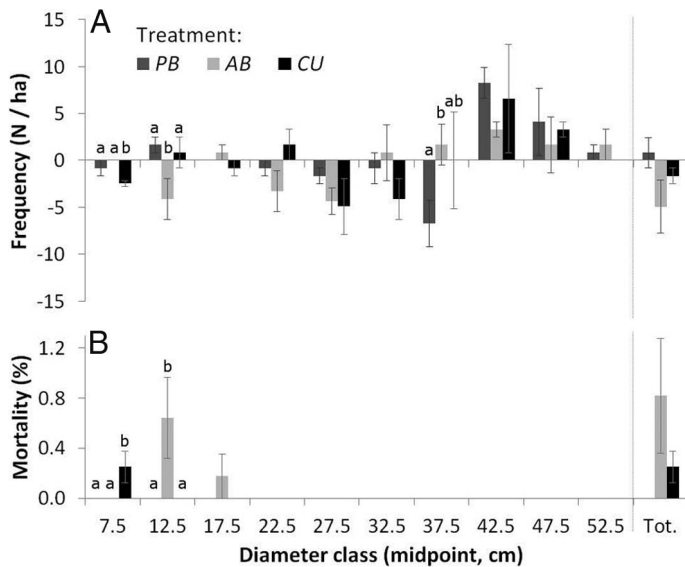

Figure 5. Mean density change (A) and mortality rates (B) from 1959 to 1970 by diameter class for the three treatments (PB, AB, and CU). Means are based on three replications per treatment, and bars represent 1 SE. Treatments with different letters are significantly different at $\alpha < 0.1$. Note: letters are reported only for those classes where significant differences were found. Diameter classes: 7.5 (5–9.99 cm), 12.5 (10–14.99 cm), 17.5 (15–19.99 cm), 22.5 (20–24.99 cm), 27.5 (25–29.99 cm), 32.5 (30–34.99 cm), 37.5 (35–39.99 cm), 42.5 (40–44.99 cm), 47.5 (45–49.99 cm), and 52.5 (50–54.99 cm).

Figure 6. Tree-level growth resistance, resilience, and recovery in relation to drought across treatments (PB, AB, and CU) and two drought events: 1948 (A); 1976–1977 (B). Means are based on three replications per treatment, and bars represent 1 SE. There were no statistical differences in resistance, resilience, or recovery among treatments during either drought event.

Discussion

Prescribed fire is a widely used forest management tool in part because it can simultaneously promote fuel reduction and forest regeneration and address restoration goals (Fernandes and Botelho 2003), presumably without causing prolonged negative effects on growth of surviving trees (e.g., Methven and Murray 1974, Sala et al. 2005). Given the projected increases in the frequency and severity of drought and wildfires for many fire-prone forest systems, we extended the evaluation of prescribed fire impacts to include the effects of repeated treatments on tree growth and vulnerability of tree growth to drought. Our results represent the first such examination of these impacts in red pine forest ecosystems and are broadly consistent with past work that has documented a transient impact of burning on tree survival and growth.

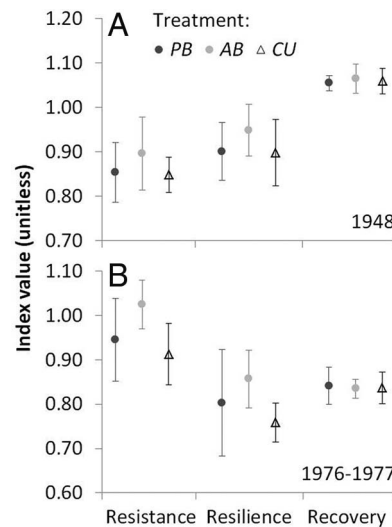Repeated burnings may affect soil organic matter and nutrients (Alban 1977, Carter and Darwin Foster 2004), outweighing the short-term benefits of increased water and nutrient-use efficiency in surviving pines (Schwilk et al. 2009). We found that repeated prescribed fires did not significantly reduce or influence growth in the years immediately after burning, regardless of burning frequency. The significant reduction in growth observed across treatments in 1989 may be reflective of a severe drought that occurred in 1988 and significantly impacted growth in other red pine stands in the region (D'Amato et al. 2013). The minimal presence of fire scars related to the prescribed fires suggests that heat-related damage to the cambium and other live tissues did not significantly reduce tree vigor or affect ecophysiological processes influencing tree growth in contrast to the findings of other studies (e.g., Ryan and Reinhardt 1988, Battipaglia et al. 2014). These findings are in line with the observations of Alban (1977), who reported no significant differences in annual volume increment during and immediately after burning treatments at the Red Pine Prescribed Burning Experiment. Roughly 40 years after the end of the prescribed burning experiment, the growth rates of surviving red pines in the burned treatments were comparable to those of trees growing nearby in periodically thinned stands (Bradford and Palik 2009), suggesting

minimal long-term effects on the productivity of the stands with short-term high or low frequency underburning.

Several studies reported low mortality rates after prescribed fire in pine stands in the Southwestern United States (van Mantgem et al. 2013), in ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) in Oregon (Busse et al. 2000), and in red and white pine in Ontario (Methven 1971). Similarly, at the Red Pine Prescribed Burning Experiment, fire-related mortality rates during the period of active burning were generally low (less than 2%) in all treatments. Mortality was relatively higher in the plots annually burned (AB) and mainly affected trees in small diameter classes. The annual mortality rate for the postexperimental period (1971–2010) was low in all treatments, ranging from 0 to 0.73% (Scherer et al. 2016), showing that no delayed fire-related mortality occurred after cessation of burnings.

The enhancement of forest resistance, resilience, and recovery is a general management goal suggested for adaptation strategies in the context of climate change and increasing drought events (Millar et al. 2007). We found that growth resistance, resilience, and recovery to drought of surviving trees were not affected by prescribed fire treatment after cessation of burnings. Growth vulnerability to drought was not altered by the repeated application of prescribed fire. These results further highlight the minimal short-lived effects of the prescribed fire treatments analyzed on tree growth and related vulnerability to drought, suggesting that understory burning can serve as management tool in red pine forest ecosystems, with no negative trade-off in terms of productivity. However, individual tree drought response is a complex process potentially dependent on individual growth history, local environment, and cumulative effects of responses to previous droughts (Macalady and Bugmann 2014). Future work should aim to better identifying drought response mechanisms by considering other potentially important factors, such as different climatic stressors (e.g., spring glaze and heat waves), pathogen pressure, neighborhood-level competition, and local environmental and edaphic conditions.

Management Implications

The application of fire for fuel reduction treatments represents an opportunity to reduce vulnerability to stand-replacing wildfires in the context of long-term forest resilience (Collins et al. 2014). However, concerns still persist about the potential loss of large overstory trees after the application of fire (Hood 2010), as well as about potential implications for vulnerability to climate change, particularly drought. The results from the Red Pine Burning Experiment in northern Minnesota suggest that a decade of repeated prescribed understory burning in red pine forest systems has the potential to be effectively integrated into management plans and adaptation strategies without causing high mortality rates or severely impacting the growth of overstory trees either in the short or the long term after repeated application of fire (Scherer et al. 2016). In the context of climate change, the application of repeated prescribed fire treatments showed the ability of restoring composition and structure of fire-dependent ecosystems that have undergone decades of fire exclusion (Fraver and Palik 2012, Scherer et al. 2016), building on important attributes of resistance and resilience (DeRose and Long 2014). Although this study did not find evidence of significant prescribed fire effects on tree growth vulnerability to drought, the use of prescribed fire as a forest management tool may warrant conscious implementation, especially considering predictions of increasing drought frequency, duration, and intensity for many fire-prone forest systems.

Endnote

1. For more information, see www.prism.oregonstate.edu/.

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