Research Article

Managed Wildfire, Drought, and Overstory Survival: A Case Study in the Ouachita Mountains of Arkansas

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ABSTRACT

Fire managers are increasingly using natural ignitions to restore and maintain fire-adapted ecosystems. Managing natural ignitions after decades of suppression, however, holds inherent ecological, economic, and personal risks. During the severe drought of 2011, fire management staff on the Ouachita National Forest used a range of strategies to manage the lightning-ignited High Peak Wildfire because full suppression was infeasible and there were potential resource benefits (e.g., restoration of desired forest structure and composition). Because of concerns that the combined effect of drought and fire could adversely affect overstory tree health and timber value, we established plots in burned and unburned areas in dry and dry-mesic pine-oak forests and loblolly pine (*Pinus taeda*) plantations immediately after the fire, and remeasured them 2 and 5 y post-burn. Between inferred pre-burn condition and 2 y post-burn, overstory trees across community types were resistant to fire (94% survival) while midstory trees were less resistant (35% survival). Survival between 2 and 5 y post-burn was near 100% for both overstory and midstory. Both the density and diversity of the midstory were reduced by 2 y post-burn, which had the desired effect of moving forest structure and composition toward an open woodland condition. This study provides evidence that wildfire, even in a drought, can be used in restoration efforts and that overstory trees in dry and dry-mesic pine-oak forests and pine plantations of the Ouachita Mountains can survive with little effect on resource values, provided that weather and fuel conditions are conducive to low- and moderate-severity burning.

Index terms: fire effects; fire management; natural ignitions; overstory survival; woody species richness

INTRODUCTION

Fire can restore forest community species composition and structure in fire-adapted communities by reducing the frequency and abundance of younger mesophytic, fire-intolerant species that have increased during decades of fire suppression, while maintaining the older, fire-tolerant species (Flatley et al. 2015). While reducing the density of fire-intolerant tree species often reduces woody diversity, greater light levels reaching the forest floor often increase diversity and cover of herbaceous species (Brewer 2016; De Jong and Zollner 2018), which in turn can improve habitat for wildlife (Masters et al. 1996; Thill et al. 2004; Rudolph et al. 2006). The desire to restore ecosystems to their historical structure and composition has led to the development of many prescribed burn programs and created opportunities to manage wildfires for multiple objectives beyond immediate suppression (Ouachita National Forest 2005).

Although wildfire can now be seen as an important ecological process and resource management tool, managing natural ignitions for multiple objectives after decades of suppression, and under new and changing climate regimes, holds inherent ecological and social risks (Parsons et al. 2003). For example, fuel buildup due to fire suppression can increase fire intensity and severity (Keyser et al. 2008), ladder fuels increase the probability of destructive crown fires (Agee and Skinner 2005), smoldering duff around tree trunks can cause root or basal cambial damage, especially in mature trees (Varner et al. 2005, 2007, 2009; Jenkins et al. 2011), and increased temperatures and prolonged droughts can add stress to forest trees and make them more susceptible to fire damage (van Mantgem et al. 2013). Fire and drought, whether acting independently or together, are known to cause physiological stress on woody species (Mueller et al. 2005; Jenkins et al. 2011; van Mantgem et al. 2011; Berdanier and Clark 2016). The ability of individual trees to survive such stress is dependent on a number of factors, including functional traits of the individual tree (e.g., species-specific adaptations such as stem size, bark thickness, or xylem structure), the fire event (e.g., severity, intensity, and length of the event, or the season of the fire), primary and secondary fire effects (e.g., crown scorch, bole char, and cambial damage, or insect and fungal invasions), and weather (e.g., temperature, humidity, and winds during the fire, as well as the pre- and post-fire climate) (Regelbrugge and Smith 1994; Graves et al. 2014; Hood et al. 2015; Dunn and Bailey 2016; Keyser et al. 2018).

Ideally, some overstory tree mortality can be accepted when meeting restoration goals, but preventing excessive mortality is key to the delicate balancing act of quality fire management. In much of the United States, tree survival and ecosystem response to wildfire and drought have not been well studied. Keyser et al. (2018) found that larger stem sizes and lower levels of bole char resulted in higher survival rates following prescribed fire for 10 deciduous species in the southeastern U.S., but species had different survival rates. Dunn and Bailey (2016) also found larger stem sizes and lower burn severity increased the likelihood of survival of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) in Oregon. Mueller et al. (2005), however, found drought had the opposite effect on pinyon pine (*Pinus edulis* Engelm.), where survival increased for smaller trees. Vega et al. (2011) found that fire increased the likelihood of pine bark beetle (Scolytidae) attack on maritime pine (*Pinus pinaster* Ait) in Spain, whereas Hood et al. (2015) found low-severity fire increased tree defense in ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) against western bark beetle (*Dendroctonus* spp. Erichson) attack by inducing the production of more resin ducts. Predicting management outcomes with such conflicting results can be challenging, and further emphasizes the need for region-specific studies.

Furthermore, while the restoration potential of prescribed fires in the southeastern U.S. has been documented (Jenkins et al. 2011; DeJong and Zollner 2018; Keyser et al. 2018), it is not well documented for wildfires. Despite this lack of evidence, fire managers are increasingly considering using natural ignitions, in combination with prescribed burns, to restore and maintain fire-adapted ecosystems (Cohen et al. 2007). It has been argued that lightning ignitions hold greater potential to restore and maintain fire-adapted ecosystems because they occur under conditions (e.g., seasonality, intensity) to which plant and animal communities are adapted (e.g., Foti and Glenn 1991; Frost 1998). Prescribed burns are conducted under conditions that minimize many risks, which can result in a fire regime that differs from the natural fire regime (Knapp et al. 2009). Because prescribed fires generally are conducted under more conservative weather conditions than wildfire events, typically resulting in low severity, it may take several prescribed burns to achieve the same outcomes as only one naturally ignited fire.

In late July of 2011, a lightning strike on High Peak Mountain on the Ouachita National Forest (ONF) ignited the High Peak Wildfire. Located in a remote area characterized by rugged terrain where previous fires had proven difficult to suppress and accompanied by extreme heat, ONF staff decided against fully suppressing this wildfire in the interest of firefighter safety and with the hope of potential ecological benefits from letting it burn. Because different plant communities and ecosystems in different regions may respond differently to such practices, managers on the ONF wanted to know the ecological effects of the High Peak Wildfire, specifically the impacts on woody plant species in the dry and dry-mesic pine-oak forests (dry forests have excessively drained or shallow soils whereas dry-mesic forests have well drained soils that drain readily but not rapidly [Nelson 2005]) of the central Ouachita Mountains. Given the lack of published research on the combined effects of drought and fire in Arkansas, it was not known how the vegetation would respond to the High Peak Wildfire. In order to address this concern, we investigated the following questions:

1. What was the survival rate of midstory and overstory trees between pre-burn and post-burn years 2 and 5 on the High Peak Wildfire, and did these survival rates differ over time?
2. Did the wildfire accomplish the restoration goals of reducing midstory stem density, specifically the mesophytic species?

METHODS

Study Site

The ONF is located in the Ouachita Mountains Ecoregion of western Arkansas and eastern Oklahoma (Woods et al. 2004). The ridges of this mostly east–west trending mountain range are covered with pine, pine-oak, and oak woodlands and forests (USDA Forest Service 1999). We conducted our study in the vicinity of High Peak Mountain approximately 5 km northeast of the township of Norman and 10 km south of the city of Mount Ida on the Caddo-Womble Ranger District of the ONF (Figure 1), in Montgomery County, Arkansas. The elevation of High Peak ranges from 256 m at the southern foot to 552 m at its peak. High Peak is underlain with Mazarn Shale and Crystal Mountain Sandstone, which are Ordovician aged rock (Haley et al. 1993). Upper elevations are covered with steep scree slopes of massive, coarse-grained light gray sandstone (McFarland 2004). Average high and low temperatures in Mount Ida are 22 °C and 8 °C, respectively, and annual precipitation is about 1450 mm (https://www.usclimatedata.com/climate/mount-ida/arkansas/united-states/usar0959).

The forests of our study area were dominated by white oak (*Quercus alba* L.), shortleaf pine (*Pinus echinata* Mill.), hickory (*Carya* L. spp.), blackgum (*Nyssa sylvatica* Marshall), northern red oak (*Quercus rubra* L.), and red maple (*Acer rubrum* L.). Loblolly pine (*Pinus taeda* L.) occurred in single-species planted stands, but was not found in the naturally regenerated forest stands because the region is just to the north of the natural range of that species. While pine plantations are not naturally occurring, they are a common component of the Ouachita Mountain landscape across National Forest and private timber company lands.

Fire Conditions and Behavior

In 2011, the Southeast experienced a severe drought (Fernando et al. 2016). In the central Ouachita Mountains, the Keetch-Byrum Drought Index (KBDI; Keetch and Byram 1968) was over 700 (noting that the maximum index level is 800, indicating extremely dry soil moisture conditions). The average high temperature and low relative humidity during July in Oden, Arkansas, were 37.2 °C and 33.8%, respectively (Oden RAWS [Remote Automatic Weather Station]). These weather conditions, coupled with higher than usual lightning activity, led to more than 60 lightning ignitions on the ONF during 2011. Most of these ignitions were fully suppressed, but the High Peak Wildfire was managed for resource benefit and burned nearly 607 ha of the 729 ha designated containment area between 29 July and 11 August 2011, when the fire was extinguished by rain.

The High Peak Wildfire containment lines to the north and south were unpaved roads, and to the east and west a combination of unpaved roads, dozer lines, and an abandoned crystal mine. Ignition occurred on an eastern ridge of High Peak and gradually spread in all directions, burning on all aspects for the next 6 d. The majority of the fire was backing and flanking with a few short up-slope runs. As the nighttime relative humidity recovery increased, fire was held to the southerly aspects and not able to carry on northerly aspects. Overall, the low-intensity High Peak Wildfire had flame lengths ranging from 0.2 to 0.5 m, with an average char height of 0.30 m (±0.04

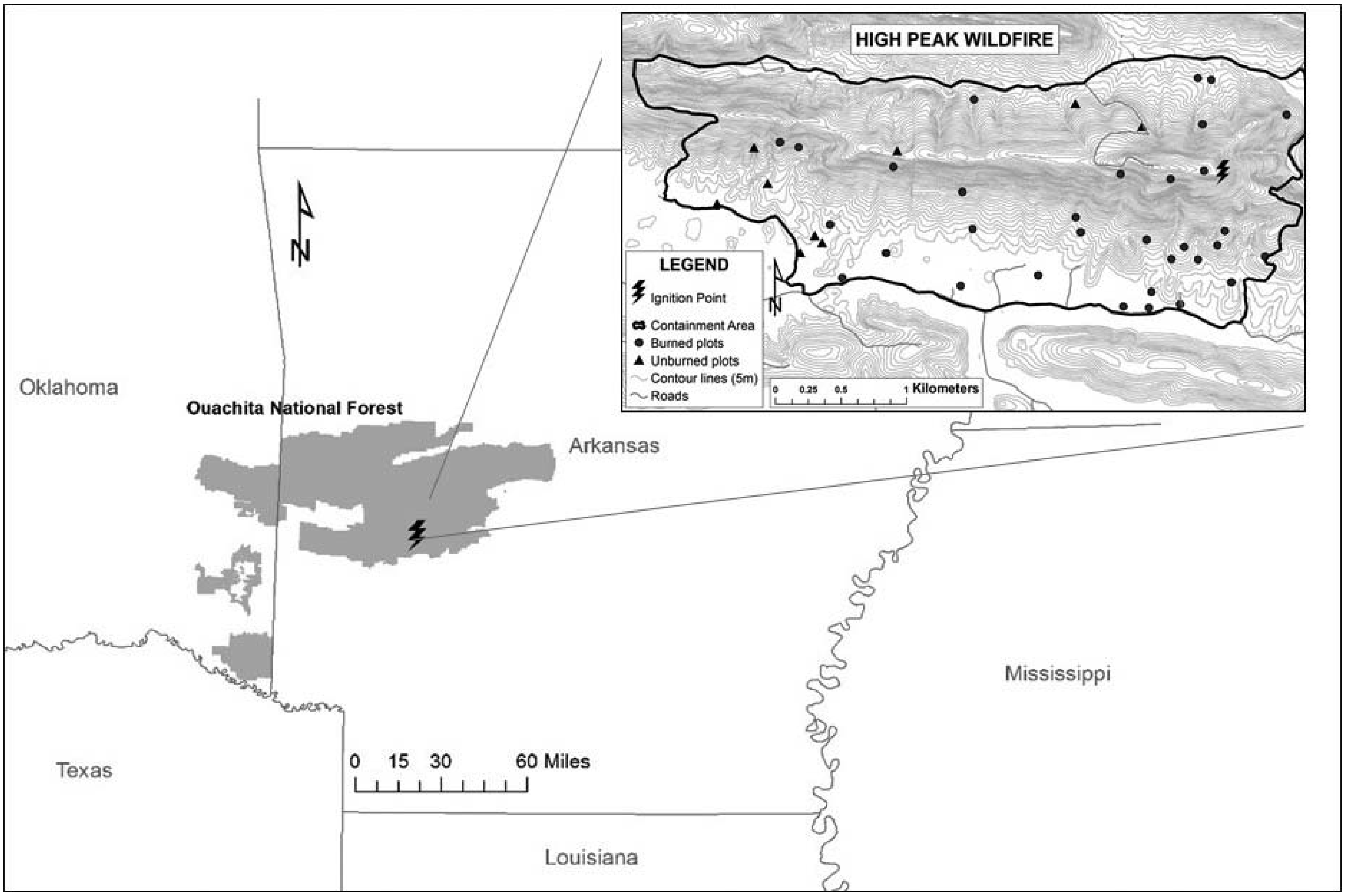

Figure 1.—Location of the 2011 High Peak Wildfire on the Caddo-Womble Ranger District of the Ouachita National Forest in Arkansas.

m SE) and scorch height of 4.62 m ($\pm$0.42 m) (McDaniel et al. 2016a) in a forest with an average canopy height of 20 m (range = 12–33 m).

Data Collection

Directly after the fire in 2011, we randomly generated 41 points within the High Peak Wildfire containment area. Within 2 mo of the fire, we visited each point and installed rebar at the center of a 10 m radius circular plot (0.03 ha) in burned areas (32 plots). Due to time constraints and lack of personnel, only two plots were established in unburned areas at this time. Two years after the burn, seven additional plots were established in the unburned area and served as additional controls (nine plots). We collected slope, aspect, elevation, and community type data for all plots (Table 1).

In each plot, we recorded species, status (live/dead), and diameter at breast height (1.37 m above groundline; dbh, cm) of all woody stems $\geq$2.5 cm dbh. Fire effects measured on each tree included scorch height (to nearest 1 m), percentage of crown volume scorched, and bole char height (to nearest 0.1 m). Char was defined as the blackening of the boles of trees, whereas scorch was leaf mortality caused by radiant or convection heat. Because we were unable to collect pre-burn data, we used immediate post-burn conditions to reconstruct the pre-burn composition of live versus dead trees. Trees with scorched leaves were assumed to be alive prior to the wildfire. In unburned plots, measured 2 y after the fire, we differentiated recent snags (died within the last 2 y) from long-dead snags (died >2 y ago). Trees that had most of their bark and were only missing fine branches (<1 inch) were considered recently dead. We classified trees $\geq$15 cm dbh as overstory and trees $\geq$2.5 cm dbh and <15 cm dbh as midstory. All plots were remeasured 2 and 5 y post-burn to quantify survivorship and changes in species composition.

Statistical Analyses

In a previous study of the 2011 High Peak Wildfire, McDaniel et al. (2016a) looked at woody species mortality 1 and 2 y post-burn in three forest communities: hardwood forest (9 plots), pine-oak forest (18 plots), and pine plantation (5 plots) (Table 1). In this paper we chose to combine all the community types because several plots had been logged and the number of plots in each community type was not sufficient to obtain a robust statistical analysis. There were no pine plantations in the control plot analysis.

We used two indices—species richness (S) and Shannon-Weiner's index of diversity (H')—to characterize species composition of the midstory and overstory pre-burn and at 2 and 5 y post-burn. Species richness was calculated as the number

Table 1.—General description of plots including burn/control – B/C, community type – CT (PO – pine-oak forest, HW – hardwood forest, PP – pine plantation), number (#) of plots, aspect (N – north, S – south, E – east, W – west), average percent (%) slope (range), elevation in meters (m) (range), MD – percent midstory density reduction 2 y postburn, and OT – percent overstory density reduction 2 y postburn for plots on the High Peak Wildfire, Arkansas. Burn plot data from McDaniel et al. 2016a.

			Aspect							
B/C	CT	# Plots	N	S	E	W	Slope (%)	Elevation (m)	MD (−%)	OT (−%)
B	PO	18	1	12	5	0	28 (2–65)	322 (270–525)	69	6
B	HW	9	3	4	2	0	35 (5–63)	393 (275–540)	52	13
B	PP	5	0	5	0	0	11 (5–15)	270 (270–270)	73	<1
C	PO	6	0	3	2	1	20 (2–62)	285 (270–335)	0	0
C	HW	3	3	0	0	0	30 (26–37)	430 (395–450)	0	0

of species within a plot. Shannon-Weiner's index of diversity was calculated as $H' = -\sum p_i \, ln(p_i)$ where p_i= proportion of total basal area (BA, m^2/ha) of species *i*. Diversity indices were calculated using PC Ord 7.0 (MjM Software Design 2016). Stem density (stems per hectare, SPH) was calculated for the midstory and overstory canopy layers pre-burn and at 2 and 5 y post-burn.

We used general linear mixed effects models under a repeated measures design to analyze the effects of treatment (burned versus unburned) and year (pre-burn, 2 and 5 y post-burn), as well as their interaction on indices of species richness and diversity (*S* and H') and forest structure (SPH) for the midstory and overstory canopy layers. Analyses were conducted using PROC MIXED in SAS 9.4 (SAS Institute, Cary, North Carolina, USA). Plot within treatment was considered a random effect and the year, modeled at the plot level, was considered the repeated measure. A spatial power covariance structure was used to account for the unequal time interval between sampling. Significant interactions were examined using the SLICE option. Following significant *F*-tests, or partitioned *F*-tests in the case of interactions, differences among least-square means were detected using Fisher's Least Significant Difference. When necessary, data were square-root transformed to achieve normality and homoscedasticity. Analyses were significant at alpha = 0.05. All reported values represent the non-transformed means and standard errors.

RESULTS

Survival

Overstory density was not affected by the burn treatment, years post-burn (YPB), or the interaction of the two (Table 2). Survival of the overstory in burned and unburned plots 5 YPB was 94% (425 of 451 stems/ha) and 100% (316 of 314 stems/ha; increase due to ingrowth from the midstory), respectively, with no significant changes in density detected between 2 and 5 YPB (Figure 2a).

Midstory density change was significant between YPB and the interaction between YPB and burn treatment, but not for treatment alone (Table 2). Prior to the High Peak Wildfire, midstory density averaged 1598 stems/ha in burned plots (Figure 2b). Two YPB, survival of midstory stems in the burned plots averaged only 35% (563 stems/ha) and survival was similar 5 YPB (Figure 2b). In unburned plots, midstory density was similar across all years (Figure 2b). The pre-burn midstory density of burned plots was greater than unburned plots because of the inclusion of five pine plantation plots in the burned plot analysis and no pine plantations in the unburned plot analysis. Pine plantations had more stems per hectare than the pine-oak or hardwood forests (McDaniel et al. 2016a).

Composition

Overstory species richness and diversity were not affected by the number of YPB or interaction of YPB and burn treatment (Table 3). Shannon-Weiner's index of diversity was similar in burned and unburned plots, but overstory species richness was greater in unburned plots than in burned plots (Figure 3).

Midstory species richness and diversity were both significantly affected by the YPB and interaction of YPB and burn treatment, but not by the burn treatment alone (Table 3). Midstory species richness in burned plots decreased from 10.3 to 5.4 between pre-burn and 2 YPB, but did not significantly change between post-burn years 2 and 5 (Figure 4a). Midstory species diversity in burned plots showed a similar trend, decreasing from 1.68 pre-burn to 1.21 two YPB, and did not show significant change between post-burn years 2 and 5 (Figure 4b).

In burned plots, the number of woody species decreased from 46 species pre-burn to 37 species 5 YPB (Table 4). Mesic species that were present in small numbers, such as umbrella magnolia (*Magnolia tripetala* (L.) L.), sycamore (*Platanus occidentalis* L.), and devil's walking stick (*Aralia spinosa* L.), were top killed. Early successional species, such as black locust (*Robinia pseudoacacia* L.), were added to the midstory in 5 YPB plots. No changes in woody species were noted in unburned plots (data not shown).

Table 2.—Results from the general linear mixed effects model describing the effects of treatment (TRT; burned vs. unburned), years post-burn (YPB), and the interaction between TRT and YPB on density of the midstory (stems ≥2.5 cm dbh and <15 cm dbh) and overstory (stems ≥15 cm dbh). Midstory data were square-root transformed prior to analysis.

Canopy layer Effect	df	*F*	*P*
Midstory			
TRT	1	1.78	0.2028
YPB	2	23.47	<0.0001
YPB * TRT	2	18.86	<0.0001
Overstory			
TRT	1	1.12	0.2967
YPB	2	2.91	0.0615
YPB * TRT	2	0.17	0.8459

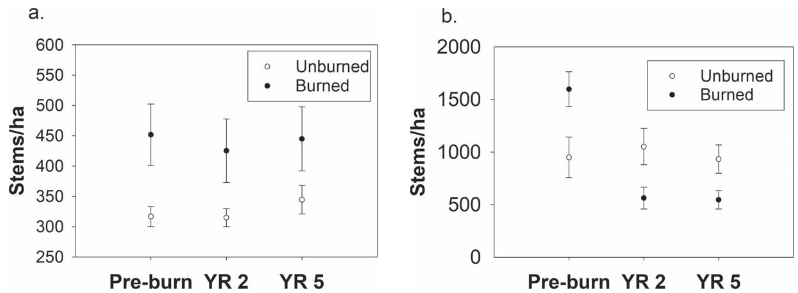

Figure 2.—Stems per hectare of the (a) overstory (stems ≥15 cm dbh) and (b) midstory (stems ≥2.5 cm dbh and <15 cm dbh) canopy layers by treatment (burned vs. unburned) and year (pre-burn and 2 and 5 y post-burn). Lines represent standard error.

DISCUSSION

In the High Peak Wildfire, overstory trees were resistant to mortality despite the combined impact of wildfire and drought (Nagel et al. 2017). The majority of tree mortality occurred by 2 YPB; McDaniel et al. (2016a) reported on woody stem mortality on this same wildfire (2011 High Peak Wildfire) after 1 y and 2 y post-burn and found most mortality occurred 1 YPB and was restricted to trees in the midstory stratum. Fire mortality was insignificant between 2 and 5 YPB, regardless of size class. The resistance of the overstory to mortality from fire and drought has several possible explanations. Varner et al. (2007) noted overstory mortality in old-growth longleaf pine (*Pinus palustris* Mill.) can be caused by smoldering of dry duff around the bases of trees, resulting in root and basal cambial damage, and therefore greater probability of mortality. They also noted mortality of longleaf pine was most prominent during spring (February–April) burns, coinciding with high rates of transpiration during leaf-out. High transpiration rates cause duff moisture content to drop quickly, leaving many fine roots in the duff susceptible to heating during smoldering fires. During summer, when duff moisture is particularly low, fine roots move deeper in the soil where they are better protected from such heating (Knapp et al. 2009).

Table 3.—Results from the general linear mixed effects model describing the effects of treatment (TRT; burned vs. unburned), years post-burn (YPB), and the interaction between TRT and YPB on species richness (*S*) and Shannon-Weiner's index of diversity (*H′*) for the midstory (stems ≥2.5 cm dbh and ≤15 cm dbh) and overstory (stems >15 cm dbh) canopy layers.

Canopy layer Effect	df	*F*	*P*
Midstory			
S			
TRT	1	0.03	0.8583
YPB	2	16.52	<0.0001
YPB * TRT	2	30.8	<0.0001
H′			
TRT	1	0.00	0.9816
YPB	2	7.00	0.0018
YPB * TRT	2	12.96	<0.0001
Overstory			
S			
TRT	1	6.34	0.0155
YPB	2	1.74	0.1848
YPB * TRT	2	0.20	0.8192
H′			
TRT	1	0.95	0.3372
YPB	2	1.44	0.2432
YPB * TRT	2	0.01	0.9900

Given that the High Peak area had not burned in over 20 y, in a forest with a 7 y fire return interval (Foti and Glenn 1991), and duff was dry (KBDI > 700), mortality from this wildfire was a valid concern several years after the burn. Even though McDaniel et al. (2016a) found little overstory mortality 2 YPB after the High Peak Wildfire, delayed mortality due to insects and fungus was still possible. Several factors may explain the lack of delayed mortality observed in this study. First, it could be related to the dry conditions in Arkansas prior to the High Peak Wildfire, which may have resulted in fine roots moving deeper in the soil, thereby effectively protecting them from the fire. Furthermore, duff depth in the Ouachita Mountains is relatively shallow (0.5–2 cm; McDaniel et al. 2016b), so the likelihood of smoldering fire in duff is low relative to other geographic areas. Additionally, the High Peak Wildfire was a low-intensity fire that consisted of mostly backing and flanking fire, causing minimal char and scorch on overstory trees (McDaniel et al. 2016a). Low levels of char and crown scorch are good predictors of low tree mortality (Hood et al. 2018; Keyser et al. 2018).

The High Peak Wildfire significantly reduced stem densities of the midstory, bringing them closer in line to historical conditions (Figure 2, Table 1). Stem density of woody plants in the Ozark-Ouachita Highlands was historically much lower than it is today (Spetich 2004). Chapman et al. (2006) compared stem density of woody plants on the Sylamore Experimental Forest in northern Arkansas and found overstory stem density (≥14.1 cm dbh) increased from 124 stems/ha to 344 stems/ha between 1934 and 2002, while midstory stem density (4.1–14.0 cm) increased from 240 stems/ha to 688 stems/ha. Using the General Land Office data from the 19th century, Foti (2004) found historical stem densities of 121 stems/ha and 112 stems/ha in the Boston Mountains and Ozark Highlands, respectively. Pre-burn stem densities of overstory and midstory canopy layers on High Peak were higher than historical conditions (McDaniel 2016a), especially in the planted loblolly pine stands. High stem density, often a result of fire suppression, can make stands more susceptible to insect outbreaks (Schowalter et al. 1981) and high-intensity fire (Agee and Skinner 2005) or, in some cases, less susceptible to fire (Nowacki and Abrams 2008).

Fire behavior that is too intense can have many undesired effects. Intense fire behavior can result in more overstory tree mortality (both immediate and delayed due to injury, insects,

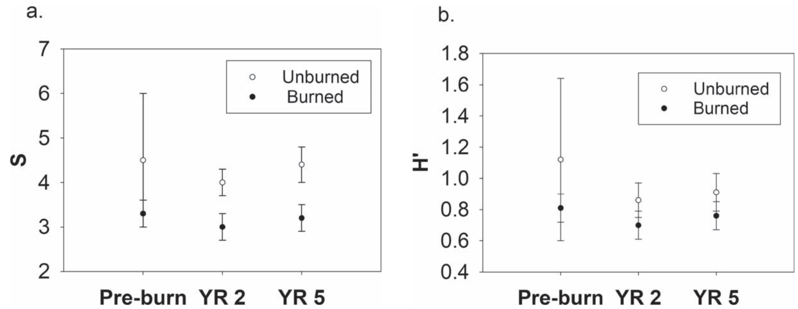

Figure 3.—Species richness (*S*) (a) and Shannon-Weiner index of diversity (H') (b) of overstory (stems >15 cm dbh) by treatment (burned vs. unburned) and year (pre-burn and 2 and 5 y post-burn). Values represent the mean (standard error).

and fungi) than desired, which would negatively impact the production of provisioning ecosystem services, including timber volume and timber value. There is evidence that following long periods of suppression, fire during dry conditions can result in widespread overstory tree mortality (Varner et al. 2007). Van Mantgem et al. (2011) found mortality rates of woody stems in burned areas were higher than controls up to 6 y following a prescribed burn in mixed conifer stands of the Sierra Nevada. Vega et al. (2011) found fire-weakened trees in Spain were more susceptible to insect attack, which increased mortality 2–5 y after the fire.

Over time, the presence or absence of fire can have a profound effect on woody species diversity and composition in a community (Keyser et al. 2017; Knapp et al. 2017). The High Peak Wildfire caused greater mortality among mesophytic, fire-intolerant species like black cherry (*Prunus serotina* Ehrh.), flowering dogwood (*Cornus florida* L.), red maple, serviceberry (*Amelanchier arborea* (Michx. f.) Fernald), northern red oak, sassafras (*Sassafras albidum* (Nutt.) Nees), and white ash (*Fraxinus americana* L.). Lower mortality was observed among the more xeric-adapted, fire-tolerant species like post oak, shortleaf pine, and white oak (Table 4). Flatley et al. (2015) found that fire promoted community differentiation such that dry, fire-adapted communities, like yellow pines (shortleaf, pitch (*P. rigida* Mill.), and Virginia (*P. virginiana* Mill.)) and chestnut oak (*Quercus montana* Willd.), were easily delineated from mesic, fire-intolerant cove forests in Great Smoky Mountains National Park. After fire suppression, the community lines began to blur, as mesic species survived in those more xeric, fire-adapted communities. After years of fire suppression, community types in the High Peak area were becoming less differentiated, with fire-intolerant species invading more xeric forests and woodlands. This was evident in the higher species richness found in the midstory of burned plots (Figure 4a). Reducing species richness and diversity in the midstory appears to be effectively halting or reversing the mesophication process that was underway during fire suppression and will help maintain fire-adapted communities.

MANAGEMENT IMPLICATIONS

Initial results from the High Peak Wildfire demonstrated that, even during a drought, the ecological effects of summer burning could be positive by improving wildlife habitat (e.g., reduction in midstory) and reducing fuel loading, without causing excessive overstory tree mortality (McDaniel et al. 2016a). The results of this study show the ONF, when burned under conditions similar to the High Peak Wildfire, has overstory trees that are resistant to mortality from wildfire. While many factors are considered in the decision-making process for each fire, studies such as this can provide support for continuing to consider all management options for lightning-ignited fires in dry and dry-mesic pine-oak forests and pine plantations of the Ouachita Mountains and potentially similar ecosystems.

Prescribed fire management objectives often include a reduction in midstory stems, an increase in herbaceous plant diversity, improved wildlife habitat, improved health and vigor of overstory trees, and restoration of historical forest composition and structure (Guldin et al. 2004; Andre et al. 2009). Attaining these objectives is possible with prescribed fire, but will probably require several burns or mechanical treatments (e.g., thinning) because prescribed burns are conducted under

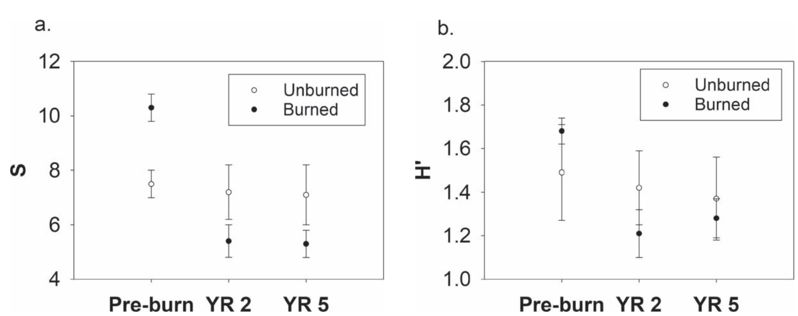

Figure 4.—Species richness (*S*) (a) and Shannon-Weiner index of diversity (H') (b) of midstory (stems ≥2.5 cm dbh and ≤15 cm dbh) by treatment (burned vs. unburned) and year (pre-burn and 2 and 5 y post-burn). Values represent the mean (standard error).

Table 4.—Frequency (Fq) of xeric and mesophytic species found in burn plots before the burn and 5 y after the burn and percent change in frequency (list includes midstory and overstory).

Xeric species	Pre-burn Fq (%)	Post-burn Fq (%)	% Change
Quercus alba	100.0	86.2	−14
Carya tomentosa	79.3	51.7	−35
Carya texana	69.0	41.4	−40
Pinus echinata	55.2	44.8	−19
Quercus velutina	44.8	31.0	−31
Pinus taeda	24.1	24.1	0
Quercus stellata	24.1	24.1	0
Quercus marilandica	20.7	10.3	−50
Vaccinium arboreum	20.7	3.4	−83
Juniperus virginiana	20.7	13.8	−33
Quercus falcata	3.4	3.4	0
Mesophytic Species	**Pre-burn Fq (%)**	**Post-burn Fq (%)**	**% Change**
Acer rubrum	93.1	44.8	−52
Nyssa sylvatica	86.2	55.2	−36
Quercus rubra	82.8	37.9	−54
Prunus serotina	75.9	31.0	−59
Ulmus alata	55.2	24.1	−56
Cornus florida	48.3	10.3	−79
Amelanchier arborea	37.9	13.8	−64
Ostrya virginiana	37.9	20.7	−45
Liquidambar styraciflua	37.9	24.1	−36
Fraxinus americana	34.5	6.9	−80
Sassafras albidum	17.2	3.4	−80
Celtis occidentalis	13.8	3.4	−75
Ilex opaca	13.8	3.4	−75
Rhus copallinum	10.3	17.2	67
Castanea pumila var. *ozarkensis*	6.9	0	−100
Hamamalis virginiana	6.9	0	−100
Platanus occidentalis	6.9	0	−100
Cercis canadensis	6.9	3.4	−50
Viburnam rufidulum	6.9	3.4	−50
Fagus grandifolia	6.9	6.9	0
Prunus mexicana	6.9	6.9	0
Aralia spinosa	3.4	0	−100
Carya glabra	3.4	0	−100
Crataegus marshallii	3.4	0	−100
Crataegus sp.	3.4	0	−100
Diospyros virginiana	3.4	0	−100
Juglans nigra	3.4	0	−100
Asimina triloba	3.4	3.4	0
Carpinus caroliniana	3.4	3.4	0
Fraxinus pennsylvanica	3.4	3.4	0
Magnolia tripetala	3.4	3.4	0
Ulmus rubra	3.4	3.4	0
Morus rubra	3.4	6.9	100
Tilia americana	3.4	6.9	100
Robinia pseudoacacia	0.0	3.4	N/A

conservative burn prescriptions and often during an altered seasonality (Knapp et al. 2009; Waldrop et al. 2010). Lightning fires, on the other hand, usually burn during the season fires naturally occurred and during drought conditions when prescribed burns would typically be avoided. Thus, these fires may provide an avenue for attaining management objectives more efficiently, so long as fire behavior is of the appropriate intensity. One important caveat is that because prescribed fires are carried out under more or less controlled conditions, and wildfires are larger and more complex events, managers must be open to more variable fire effects with managed wildfires (e.g., a wider range of tree mortality). Also managers need to be realistic about the potential for postfire conditions that may fall outside of desired conditions. Fire intensity in the Ouachita Mountains can be extreme (Rainbow Springs Wildfire, 1984: https://www.wildfirelessons.net/orphans/viewincident?DocumentKey=adb35a37-e78a-4c10-b37d-e1296d5819bf), but in the four managed fires that have occurred on the Ouachita National Forest since the High Peak Wildfire, all have burned with a restoration intensity rather than a destructive intensity. Combining the use of managed wildfire with prescribed burns could add to the number of acres treated and changing the seasonality of burning could benefit species more attuned to the historical fire regime.

A frequent concern for fire managers when considering wildfire response strategies on both public and private land is the potential risk the fires pose to timber value (Dey and Schweitzer 2015). The butt log (the first 5 m length of the tree above a 0.5 m stump) is the most valuable part of a tree and where the danger of damage by fire is highest (Stambaugh et al. 2017). Knapp et al. (2017) found that direct effects (e.g., cambial damage, scorch, or char) of annual and periodical burning did not affect timber value of individual trees in the short term, but over time, the change in species composition did affect the overall value of timber sales. Burned areas tended to be composed of lower value species, such as post oak (*Q. stellata* Wangenh.), and fewer high-value red oak species, such as northern red oak and black oak (*Q. velutina* Lam.). A shortleaf pine timber sale was planned in the High Peak Wildfire prior to the burn. When the area was logged 5 y after the burn, it was valued similarly to other comparable stands that had not burned (Clay VanHorne, Timber Management Assistant, ONF, pers. comm.). Although fire scars were observed in the High Peak Wildfire, scarred trees were mostly hardwoods and occurred on slopes too steep for logging where burn severity was higher.

Fire's importance to the restoration and maintenance of fire-prone ecosystems resulted in the implementation of prescribed burns and policies that allowed lightning fires to burn naturally in wilderness areas (van Wagtendon 2007). The number of fires managed with objectives other than full suppression has increased over the years (from 24,662 ha in 1998 to 196,933 ha in 2005), but suppression remains the standard management option for most wildfires (van Wagtendonk 2007). In theory, the idea of letting a fire burn itself out is simple, but in practice it is complicated (Wells 2009), and managers have to balance multiple priorities and high risk, which often force them toward suppression. The location of wildfire relative to the wildland–urban interface means that consequences for making the wrong decision could be high, including personal liability and the loss of their job. Managers have to deal with the public's reaction to this fire activity, including, among other things, the negative impacts of smoke on local communities. They must also weigh the potential negative ecological and economic consequences (e.g., tree mortality or timber damage) against the ecological benefits. In some cases, managers simply do not have the organizational capacity to handle the size and duration of fires that might result from allowing natural ignitions to run their course. Other times, policy directives wholly prevent these activities from being carried out at all (Wells 2009).

CONCLUSIONS

Over the last century, U.S. fire management policy has progressed from a full suppression "10 a.m." policy, whereby all ignitions were to be extinguished by 10 a.m. the day following ignition, to the current policy, where fire is recognized as an essential ecological process that should be incorporated into planning processes (National Wildfire Coordinating Group 2009). In the case of large wildfires, multiple tactics can be employed simultaneously, including allowing a fire to burn under close monitoring in remote areas where there is potential ecological benefit (e.g., creation of wildlife habitat, hazardous fuels reduction), while actively suppressing the fire in areas where people, property, or other values are at risk. This gives managers the versatility necessary to achieve resource management objectives while also providing for agency and public safety, thereby achieving greater positive outcomes.

The High Peak Wildfire suggests that wildfires, even those that burn during drought conditions, can still have beneficial effects on fire-adapted landscapes. First, the wildfire helped move the forest structure and composition toward a restored condition by reducing the density of mesophytic midstory species. The opening of the forest midstory should allow more light to reach the forest floor, encouraging the growth and spread of warm-season grasses and forbs, thereby benefiting associated fauna. Second, overstory trees were resistant to mortality, with low levels of mortality detected immediately following the fire and through 5 y post-burn. Few trees were damaged by the fire and a timber sale in the burned area soon after the fire demonstrated no loss of value. The decision to limit suppression activities on a fire can have real negative consequences, but in this case we found that no such impacts resulted. Meanwhile, from an ecological perspective, the benefits were numerous.

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