

Fire and Fuels Management

Delayed Tree Mortality After Prescribed Fires in Mixed Oak Forests in Northwestern Ohio

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Abstract

Delayed tree mortality can contribute to variability in fire effects in forests, but its prevalence is not well understood in eastern North American oak forests where a management goal is using prescribed fire to shape forest density and composition. To assess potential delayed mortality after prescribed fires, we tracked the fates of 690 trees of four species in burned and 542 trees in unburned oak forests in northwestern Ohio, USA, and modeled survival using tree diameter and bole char. Delayed mortality, occurring 3–4 growing seasons after fire and in addition to initial mortality (1–2 growing seasons after fire), varied with species and tree diameter. Compared to initial mortality, delayed mortality resulted in eleven times more small-diameter (1–13 cm) red maple (*Acer rubrum*) dying after fire. White oak (*Quercus alba*), 1–25 cm in diameter, also incurred delayed mortality (five-times increase in dead trees). Neither sassafras (*Sassafras albidum*) nor black oak (*Quercus velutina*) displayed delayed mortality. Background tree mortality in unburned sites was minimal (0.4% per year across species). Logistic regression to model canopy survival selected only stem diameter for burned red maple trees, whereas both diameter and bole char related to survival in other species. Results suggest that (1) monitoring postfire tree mortality in oak forests should extend for at least four growing seasons to detect delayed mortality in some species, and that (2) single surface fires may eventually reduce encroaching red maple in oak forests more than initial postfire years indicate.

Study Implications: Delayed tree mortality, a poorly understood phenomenon in eastern North American oak forests, varied with species and stem diameter after prescribed fires in northwestern Ohio oak forests. Accompanied by minimal mortality of large oaks (>25 cm in diameter), single surface fires appear capable of reducing density of encroaching red maple stems, but these reductions may not appear for at least 3–4 growing seasons after fires.

Keywords: *Acer rubrum*, burns, hardwoods, *Sassafras albidum*, *Quercus alba*, *Quercus velutina*, survival models

After fires, delayed tree mortality can influence trajectories of forest composition and functional change, such as carbon storage, wildlife habitat, and timber values (Hood et al. 2018). However, delayed mortality is a poorly understood process and its significance after fires can be hard to predict, adding uncertainty to fire outcomes (Bär et al. 2019). Postfire delayed mortality has ranged from absent to extensive within tropical, semi-arid, and temperate forests (e.g., Fowler and Sieg 2004, Granzow-de la Cerda et al. 2012, Roula et al. 2020). A current research focus is identifying how delayed mortality may vary with contingency factors such as species, tree diameter, fire severity, climate, site factors, and biotic agents including insects, pathogens, and herbivory (e.g., van Mantgem et al. 2011, Maringer et al. 2016, Dey and Schweitzer 2018).

In eastern North American fire-adapted oak forests, few studies have examined postfire delayed mortality, and several authors have noted this knowledge gap as interest continues in using prescribed fire as a forest management tool (Regelbrugge and Smith 1994, Yaussy and Waldrop 2010, Keyser et al. 2018). Historically, frequent fires were thought to have kept these forests open and conducive to regeneration and maintenance of fire-promoted oak species (Nowacki and

Abrams 2008). Cessation of human-ignited fires and other forest changes since the early 1900s in many areas has hindered oak recruitment, resulting in vast areas of contemporary forest with oak overstories but understories and midstories dominated by other species, such as red maple (*Acer rubrum*; Vander Yacht et al. 2019). As a result, and owing to the habitat and timber values of oak forests, there is keen interest in using prescribed fire to reduce trees competing with oak and maintain future regeneration opportunities for oak (Keyser et al. 2018). Effectiveness of prescribed fires at meeting these objectives can hinge on fire's ability to selectively reduce fire-sensitive trees before they attain sizes where they too may resist all but the most severe fires (Arthur et al. 2015). Delayed mortality could influence the degree to which these management objectives are achieved if greater mortality occurs than initially apparent in either non-oak or oak tree species in long-unburned forests (Taft 2003). However, the limited existing literature in eastern oak forests suggests that delayed mortality after single prescribed fires may be minimal or highly variable. Loomis (1973) and Harmon (1984) concluded that tree mortality was completed quickly, within 1–2 growing seasons after fire. Yaussy and Waldrop (2010) found that some mor-

tality of hardwoods occurred through four postfire growing seasons but that it was only a few percent greater than background mortality of unburned trees. Contrasting with these studies, Brose et al. (2007) reported substantial delayed mortality in striped maple (*Acer pensylvanicum*). In that study, 34% of the total stems dying after spring prescribed fires died between the first and third growing season after fire.

The objective of our study was to track tree survival of four hardwood species in burned and unburned sites up to four growing seasons after prescribed fires in eastern oak forests. We tested a hypothesis of no change in status of trees between two postfire periods representing initial (first 1–2 growing seasons postfire) and delayed mortality (3–4 growing seasons postfire). To identify factors potentially related to fire effects, we developed models of tree canopy survival based on stem diameter and bole char variables.

Methods

Study Sites and Prescribed Fires

We performed the study in the 1692-ha Oak Openings Preserve (41°33'12" N, 83°50'8" W) within the 47,000 ha Oak Openings region, northwestern Ohio, USA. The temperate climate averages 86 cm/year of precipitation (Midwestern Regional Climate Center, Champaign, IL). Upland Udipsamment soils support forest with black oak-white oak (*Quercus velutina*-*Quercus alba*) overstories and mostly red maple-sassafras (*Sassafras albidum*) understories (Abella et al. 2020). Surface fires in the region prior to European-American settlement in the 1800s are thought to have been historically frequent, in part maintaining open oak ecosystems (Brewer and Vankat 2004). A fire history reconstruction in a similar habitat in Wisconsin reported a mean presettlement fire interval of 3.7 years (Wolf 2004).

We studied three burned sites, 7–18 ha in size and at least 0.5 km apart. Sites had sandy-textured soil, elevations ~215 m, and slope gradients <2% typical of the flat to rolling sandplain landscape. Two sites were in mature oak forest containing black oak and white oak trees 80–200+ years old, and the third site was in a young oak forest (~10–20 years old). Both mature forest sites had basal areas of 26 m²/ha (87% oak, 7% red maple, 4% sassafras, 2% other species) for live trees ≥1 cm in diameter at 1.4 m (DBH). Tree canopy cover, above a height of 3 m from the ground, averaged 67%. Fire is not thought to have occurred at these sites in at least the past 70 years. The young forest site had a basal area of 4 m²/ha (all oak). Understory fuels among sites were primarily forest floor leaves, Pennsylvania sedge (*Carex pensylvanica*), tree seedlings, and the shrubs lowbush and Blue Ridge blueberry (*Vaccinium angustifolium* and *V. pallidum*) and huckleberry (*Gaylussacia baccata*).

One mature forest site received prescribed fire on April 24, 2017, and the other site on April 23, 2018. The young forest site was burned on April 26, 2018. During ignition for each burn, field observations indicated that relative humidity was 32–46%, wind speeds ranged from 1 to 18 km/hour, and high temperatures were 17°C–21°C. The prescribed fires included backing, flanking, and head fires and had flame lengths mostly ≤2 m, minimal torching of understory tree canopies as flames charred boles of trees up to a maximum height of ~3 m, and rates of spread were typically <2–3 m/minute. This fire behavior is similar to that previously described for dormant-season, relatively low-severity surface fires in eastern hardwood for-

ests (Brose et al. 2014). Precipitation in the 14 days after each burn was as follows: mature forest first site 2017 burn, 10.3 cm of precipitation and nine of 14 days received measurable precipitation; mature forest second site 2018 burn, 7.0 cm and seven days; and young forest site 2018 burn, 7.2 cm and seven days (Toledo Express Airport climate station, National Centers for Environmental Information, Asheville, NC). Postfire growing seasons were moist or near average. Summer (May through August) precipitation was above average (103–151%, 2017–2019) and the summer Palmer Drought Severity Index (PDSI) “unusually moist” (2.1–4.5) for all years except 2020. That year was nearly normal (91% of average summer precipitation and a 1.7 summer PDSI).

Data Collection

We inventoried trees at each burned site in July–August 2018 and 2020, corresponding with periods of initial (2018, 1–2 growing seasons after fires) and delayed postfire mortality (2020, 3–4 growing seasons after fires). We used the National Park Service (2003) fire effects monitoring protocol with 20 m × 50 m (0.1 ha) plots. We randomly located three plots in each of the two burned mature forest sites and one plot in the smaller young forest site. To track fates of individual trees, on each plot in 2018, we mapped the *x* and *y* coordinates to the nearest 0.1 m for each tree (≥1 cm DBH). We recorded species, DBH, and status as live canopy, top-killed (canopy dead) but with resprouting around the base of the trunk, or completely dead. We also recorded maximum char height (nearest 0.1 m) up the bole and the percent surface area of the bole charred from the ground up to 1 m. In 2020, we reinventoried the status of each tree.

Although it is possible that some trees were already dead or dying from nonfire causes when the dormant-season fires occurred, this would be expected to be minimal owing to the infrequency of standing dead, small trees in unburned forests and to the procedures we used to minimize the possibility. In 2018, for the initial postfire measurement of plots, the few small trees likely already dead before fires were usually readily identifiable by having loose bark, few or no branches, and lack of old leaves. Those trees were not included in measurements, enhancing confidence that measured mortality was in fact associated with fire. We further assessed expected mortality in the absence of fire in unburned forests.

For comparison with mortality on burned sites by identifying background mortality of unburned trees, we studied three unburned, mature oak forest sites and an unburned young oak site. On two randomly located plots within each unburned site, we inventoried trees in July–August 2018 and 2020 using the same methods as for burned plots.

Statistical Analyses

For analyses, we grouped trees into burned and unburned groups by species (red maple, sassafras, black oak, and white oak) combined across plots and sites. We simplified analyses by combining across plots and sites because postfire tree responses were similar across burned plots and sites. For example, among plots in 2018, a similar percentage (91–97%) of small red maple stems 1–13 cm DBH were top-killed with basal trunk resprouting. When plots were averaged on a site basis, the percentage was nearly identical (94% and 95% red maple top-killed and resprouting) between the two burned, mature forest sites.

We performed three statistical analyses to (1) assess change in tree status using categorical data analysis, (2) model postfire canopy survival as a function of continuous variation in predictor variables (logistic regression), and (3) identify potential changepoints in predictor variables at which postfire canopy survival predominately shifted between live and dead (classification tree modeling). By DBH classes (saplings, 1–13 cm; poles, >13–25 cm; and overstory, >25 cm; Brose et al. 2014), we assessed whether the proportions of trees among the three status categories (live canopy, top-killed but resprouting around the trunk base, or completely dead) changed between 2018 and 2020 for burned and unburned groups using McNemar-Bowker tests of symmetry for repeated measures (PROC FREQ, SAS 9.4). In canopy survival models, we focused on a functional outcome of whether burned trees in 2020 retained a live canopy as compared collectively with trees entirely dead or that had dead canopies and basal resprouting but that were no longer part of canopy or subcanopy layers. To model 2020 canopy survival status (canopy alive or dead based on a 0.5 probability cutoff) on burned sites by species using predictors (DBH and bole char height and percentage) screened for inclusion at $P < 0.15$, we used logistic regression (PROC LOGISTIC, SAS 9.4; Woolley et al. 2012). To identify changepoints in predictors for 2020 canopy survival, we computed classification tree models using the j48 algorithm, five-observation minimum for terminal nodes, and 10-fold cross-validation in Weka 3.8 (University of Waikato, Hamilton, New Zealand).

Results

Change in Tree Status from Initial to Delayed Mortality Periods

Categorical analyses of change in tree status across the three categories (live canopy, top-killed but basally resprouting, or completely dead) indicated that postfire delayed mortality varied with species and DBH (Figure 1). Red maple saplings (1–13 cm DBH) and white oak saplings and poles (>13–25 cm) exhibited significant changes in proportions of trees among status categories associated with delayed mortality, 3–4 growing seasons (2020) after fire, following initial mortality 1–2 growing seasons (2018) after fire. The percentage of red maple saplings completely dead increased 11-fold, from 3% in 2018 to 34 percent in 2020. Furthermore, canopies subsequently died in 30% of red maple trees <25 cm DBH that had retained live canopies 1–2 growing seasons after fire. By 3–4 growing seasons after fire, only 30 (8%) of 366 red maple had live canopies. Although many sassafras and black oak saplings were top-killed by 1–2 growing seasons after fire, no significant delayed mortality followed. This contrasted with white oak saplings and poles, which experienced a four-times and eight-times increase, respectively, in individuals completely dead between 1–2 and 3–4 growing seasons after fire. Neither oak species had significant mortality in trees >25 cm DBH.

In unburned sites, mortality was low between 2018 and 2020. Only 1 of 216 (0.5%) red maple, 0 of 41 sassafras, 2 of 76 (0.9%) white oak, and 1 of 209 (0.5%) black oak trees died.

Modeling Canopy Survival Using Diameter and Bole Charring

In modeling 2020 canopy survival for burned trees, logistic regression indicated that DBH was related to canopy survival

in all four species (Figure 2, Table S1, S2). Bole char was also important in all species except red maple. Classification tree models identified changepoints in DBH and bole charring at which canopy survival shifted (Figure 3). Canopy survival was negligible for red maple trees <15 cm DBH (only 1.5 percent of burned trees this size retained live canopies). Canopies of all sassafras <12 cm DBH died. The DBH changepoint was lower for canopy survival in black oak and contingent on bole charring. A total of 97% of black oaks >2.5 cm DBH retained live canopies regardless of bole charring, whereas 93 percent of even smaller 1–2.5-cm DBH black oaks retained live canopies if ≤15% of the bole was charred.

Discussion

Our findings of extensive top-killing of understory red maple following single fires supported findings of several previous studies in eastern oak forests (e.g., Loomis 1973, Regellbrugge and Smith 1994, Brose et al. 2014). However, a difference between our findings and those of many previous studies was that, instead of single fires maintaining or even increasing red maple density through resprouting (e.g., Albrecht and McCarthy 2006, Brose et al. 2014, Izbicki et al. 2020), we found that a third of resprouting, sapling-sized individuals (1–13 cm DBH) died completely via delayed mortality. As no new red maple stems grew to 1 cm DBH during the study, the single prescribed fires nearly eliminated red maple saplings with live canopies and reduced the total density of live red maple saplings, including resprouts, by one-third.

There could be several factors or combinations of factors associated with our finding of delayed postfire mortality in red maple. After fire, the oak canopy remained intact, as there was almost no mortality of large oaks >25 cm DBH, so red maple resprouts needed to grow below canopy in shade. Red maple is considered to be shade tolerant and physiologically a generalist able to inhabit forests spanning broad light gradients (Abrams 1998). However, top-killing red maple when in shaded conditions can deplete the tree's carbon reserves or compromise resource acquisition, especially when compared with oaks (Reich et al. 1990). For example, Huddle and Pallardy (1999) reported that 1–2-year-old seedlings of white oak and red oak (*Quercus rubra*) maintained greater root starch concentrations than did red maple, possibly accounting for red maple's poorer performance after seedlings were top-killed by spring burns in their study. In another study, complete defoliations triggered increased root biomass, favoring belowground resource acquisition in red oak but not in red maple (Canham et al. 1999). Among seedlings of five tree species (including black oak), red maple experienced the greatest decline in nonstructural carbohydrate reserves when shade was experimentally combined with other stresses (Maguire and Kobe 2015). It is possible that physiological challenges facing postfire red maple sprouts in our study were exacerbated by competition for resources with expanding postfire populations of understory plants, particularly the clonal shrubs huckleberry and Blue Ridge blueberry that grow vigorously after low-severity fires (Matlack et al. 1993).

Precipitation following fires and initially sub-lethal injuries could be additional factors. Although the above-average summer precipitation during initial postfire years could be theorized to increase tree survival by reducing moisture stress, moist years may not actually be most conducive to tree survival. In offering possible explanations for postfire delayed mortality of striped maple in Pennsylvania, Brose et al. (2007) hypothesized that ex-

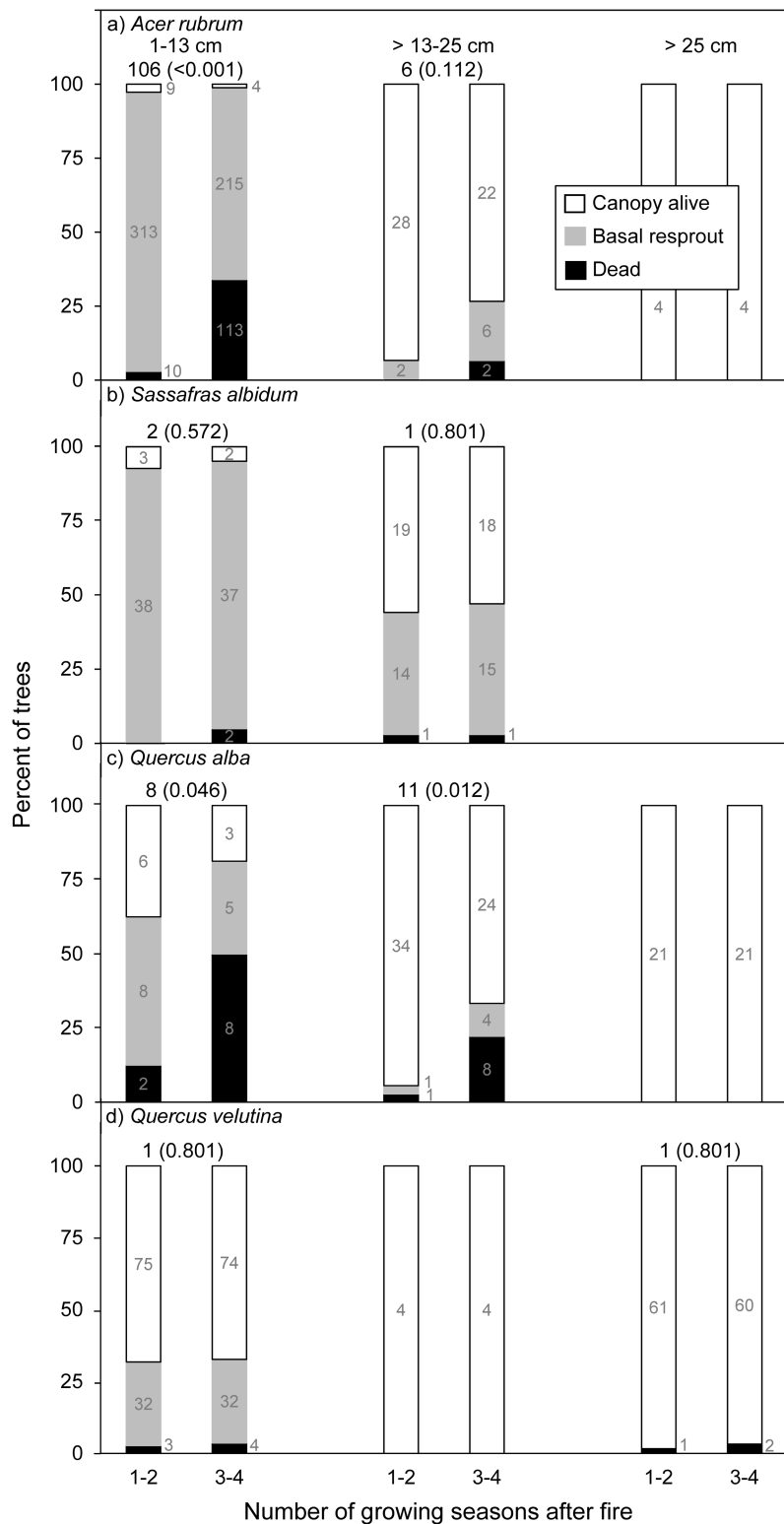


Figure 1. Change in status of trees on burned sites between 1–2 and 3–4 growing seasons after prescribed fires in oak forests, Oak Openings region, Ohio. Trees are divided by species, diameter (saplings, 1–13 cm in diameter at 1.4 m; poles, >13–25 cm; and overstory, >25 cm), and status including live canopy, dead canopy but resprouting around the trunk base, or completely dead. Statistics at the top of bars show McNemar-Bowker tests of symmetry for repeated measures (S-statistic with P-values in parentheses) and compare proportions among status categories between 2018 and 2020 (no statistics are given if <15 trees occurred or if there was no variation between years). No sassafras (*Sassafras albidum*) trees >25 cm in diameter were present. Numbers on or adjacent to bars are numbers of trees by status category.

ceptionally wet growing seasons after fire could have promoted fungal pathogens. Dey and Schweitzer (2018) noted that injur-

ies sub-lethal immediately after fire could become lethal over time, such as by enabling pathogen entry. We observed that dead

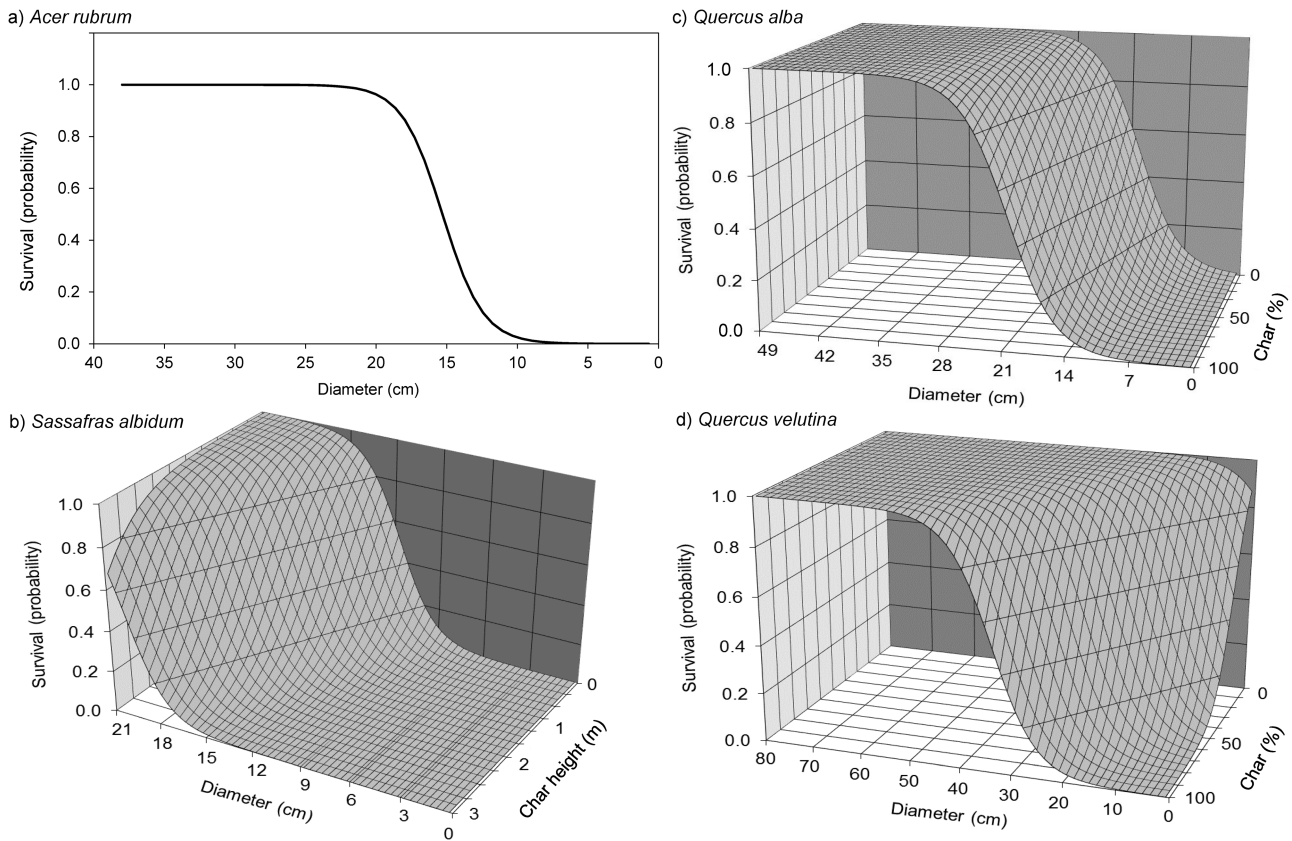


Figure 2. Logistic regression models estimating canopy survival (categorical response as alive or dead) of burned trees of four species in 2020, 3–4 growing seasons after prescribed fires in oak forests, Oak Openings region, Ohio. Trees with dead canopies in 2020 were either resprouting around the trunk base or completely dead. Logistic regressions estimate continuous probabilities of canopy survival using predictor variables. Predictor variables include diameter at 1.4 m (DBH), maximum height of bole char, and percent of the bole charred from the base to a height of 1 m. Tables S1 and S2 provide descriptive statistics, parameter estimates, and classification accuracy for logistic models. Figure S1 provides bivariate relationships between predictor variables.

boles of red maple were cracked and covered with fungus by 2020, though it remains uncertain whether injuries facilitating biotic damaging agents had a role in tree mortality relative to other possible factors (Figure S2; Lindner et al. 2006).

DBH related to postfire canopy survival in all four species, and in three species (all but red maple), bole char variables were secondarily important. The significance of DBH concurred with theoretical expectations of its relationship with bark thickness and thermal insulation (Bova and Dickinson 2005). At a given DBH, red maple has the thinnest bark among the species. For example, 15 cm DBH trees (the changepoint identified in classification tree models for red maple canopy survival) have estimated bark thicknesses of 3.1 mm for red maple, 7.7 mm for sassafras, 6.8 mm for white oak, and 8.1 mm for black oak (Harmon 1984). Top-killed sassafras trees experienced greater bole charring than the other species, possibly accounting for why, despite having thick bark, small sassafras were top-killed similar to thin-barked red maple. Flammable, volatile chemicals in sassafras tissues or the ridged structure of the bark may have contributed to bole charring and top-killing despite the species’ thick bark (Kaler and Setzer 2008). Although susceptible to top-kill, sassafras saplings (1–13 cm DBH) had low complete mortality rates because almost all resprouting sassafras remained alive by 3–4 years after fire. This high degree of fire resilience supported Dey and Hartman’s (2005) conclusion that sassafras had the greatest ability to persist

<p>a) <i>Acer rubrum</i></p> <p>DBH ≥ 15 cm: alive (25/30) DBH < 15 cm: dead (331/336)</p> <p>0.77, 0.05, 0.17</p>	<p>c) <i>Quercus alba</i></p> <p>DBH > 19 cm: alive (37/38) DBH ≤ 19 cm</p> <p>Char ≤ 20%: alive (7/11) Char > 20%: dead (20/24)</p> <p>0.59, 0.23, 0.38</p>
<p>b) <i>Sassafras albidum</i></p> <p>DBH > 12 cm</p> <p>Char ≤ 60%: alive (17/23) Char > 60%</p> <p>DBH ≤ 17 cm: dead (11/11) DBH > 17 cm: alive (3/5) DBH ≤ 12 cm: dead (36/36)</p> <p>0.62, 0.19, 0.35</p>	<p>d) <i>Quercus velutina</i></p> <p>DBH > 2.5 cm: alive (124/128) DBH ≤ 2.5 cm</p> <p>Char ≤ 15%: alive (13/14) Char > 15%: dead (33/34)</p> <p>0.86, 0.08, 0.21</p>

Figure 3. Classification tree models estimating canopy survival (categorical response as alive or dead) of burned trees of four species in 2020, 3–4 growing seasons after prescribed fires in oak forests, Oak Openings region, Ohio. The models identify changepoints in predictor variables at which tree canopy status shifts between alive and dead (numbers in parentheses are numbers of trees categorized with live or dead canopies out of the total trees of the species). Predictor variables include diameter at 1.4 m (DBH) and percent of the bole charred from the base to a height of 1 m. The sequence of three numbers below each model are model statistics: Kappa statistic, mean absolute error, and root mean squared error.

(via resprouting and high resprout survival) after even three or more fires among eleven hardwood species they examined in the Missouri Ozarks.

We found that quantitative variation in bole char height and percentage was unrelated to postfire survival in red maple, possibly because only the presence or absence of bole charring was important. Burned red maple with dead canopies showed the least range in bole char height and percentage among species (Table S1). This could result from only small amounts of bole damage being required to kill red maple, stems not having physical features susceptible to much charring although still experiencing heat damage, or other factors relating more closely with canopy mortality than did degree of bole charring.

The DBH changepoint for canopy survival of black oak was at least five times smaller than for the other species, including white oak. To what extent this difference between black and white oak could hinge on the current availability of sample trees and ecological settings in which the species occur is unclear. As is common in eastern North American oak forests (Abrams 2003), white oak is proportionately less abundant than black oak at our study sites compared with before European-American settlement (Brewer and Vankat 2004). Most of the white oaks less than 25 cm DBH, which incurred all of the species' top-kill or complete mortality, including delayed mortality, were in suppressed, subcanopy positions. Tree growth and vigor preceding fires can affect postfire survival independently or interactively with severity of fire-caused injuries (van Mantgem et al. 2020). It seems plausible that white oak's presence as primarily low-vigor, suppressed trees at the study sites contributed to the species' delayed mortality and general lack of fire resistance compared with black oak. This further suggests that, similar to red maple, additional research differentiating primary (e.g., immediate effects of fire to plant tissues) and secondary fire effects (e.g., fire-caused damage eventually compromising tree performance) may help unravel species differences in postfire survival (Bär et al. 2019).

Management Implications

Our results suggest that single prescribed fires in long-unburned oak forests can partially meet management goals of reducing encroaching nonoak trees and potentially making progress toward enabling opportunities for oak regeneration. Delayed fire effects, particularly the complete mortality of a third of red maple saplings not evident until 3–4 growing seasons after fire, were a key part of curtailing encroachment by nonoak trees. Although multiple or potentially higher-severity fires may be needed to eventually reduce larger red maple trees and deter development of dense layers of resprouting red maple or sassafras, a single low-severity fire in mature oak forests may keep understories relatively open for over a decade, given the attrition of red maple saplings and the time needed for surviving resprouts to regrow to sapling sizes (Hart et al. 2012). Our results were consistent with the idea that windows are limited during subcanopy tree maturation in which low-severity burning can effectively reduce nonoak encroachment (Abrams 2005). Most red maple stems were still smaller than the fire-susceptible threshold of 15 cm DBH at our study sites and effectiveness of prescribed fire for reducing encroachment could differ if most red maple had instead been larger. Identifying fire-susceptible size thresholds among tree species, such as in our study, may assist in prioritizing oak forest sites amenable to manage-

ment using low-severity burning, before more intensive fires or mechanical cutting are necessary to reduce encroachment by nonoak trees. Although recruitment of oak saplings ≥ 1 cm DBH did not occur during the study, the postfire reduction in understory nonoak trees could enable opportunities for future oak regeneration if suitable canopy openings become available. Competition with red maple and other species is known to hinder oak regeneration, and the observed postfire reduction in competing trees should enhance opportunities for oak recruitment in canopy gaps (Brose et al. 2014, Izbicki et al. 2020). Our study suggests that delayed mortality can be a significant part of postfire changes in eastern hardwood forests and can continue for at least four growing seasons after fires.

Supplementary Material

Supplementary material is available at *Forest Science* online.

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