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Changes in stand structure and tree vigor with repeated prescribed fire in an Appalachian hardwood forest

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ABSTRACT

Without large scale disturbances to alter forest structure and open the canopy, historically oak-dominated forests of the central and Appalachian hardwood regions of eastern North America are shifting to dominance by shade-tolerant, 'mesophytic' species. In response, prescribed fire is applied with increasing frequency and spatial extent to decrease non-oak species and promote dominance of oak species. However, relatively few studies have examined impacts of repeated fire to forest structure and tree vigor across multiple years and varied terrain. In this study, we examined tree vigor, tree mortality, and stand structure in response to different burn treatments: Frequent (burned 4 times in eight years), Less Frequent (burned 2 times in eight years), and Fire-Excluded. We hypothesized that fire-driven decreases in stem density and basal area would be greatest for small size classes, especially of shade-tolerant species on drier landscape positions, and would increase with burn frequency and fire temperature. We expected trees surviving fire to exhibit increased crown vigor over time since fire. Prescribed fire effects depended on tree size-class and landscape position. About 60% of surviving midstory trees (10–20 cm diameter at breast height (DBH)) and 25% of overstory trees ($\geqslant$20 cm DBH) on sub-xeric and intermediate landscape positions experienced crown dieback. Fire-Excluded sites had fewer trees with crown dieback (11–28% across size classes) compared to burned sites (21–87%). Throughout the duration of the study, midstory and overstory maples had significantly greater likelihood of increased crown dieback compared to oaks. Paradoxically, midstory maples had a higher survival probability than similarly-sized oaks, while overstory maples had lower survival than overstory oaks. The greatest reductions in density and basal area occurred in saplings (trees 2–10 cm DBH) and midstory trees on sub-xeric and intermediate (but not sub-mesic) landscape positions. Both Less Frequent and Frequent burning reduced density and basal area of sapling and mid-story shade-tolerant species, but also of mid-story chestnut oaks. Individual tree mortality was positively correlated with char height after the first burn regardless of burn frequency. A large and significant initial sprouting response to fire dissipated over time and with repeated burning. Future assessments of mortality and vigor of residual trees following fire are essential for evaluating the long-term effectiveness of prescribed fire management in shifting species composition away from 'mesophytic' species and toward oaks, and could help guide management choices regarding repeated prescribed burning.

1. Introduction

Upland oak ecosystems in the central and Appalachian hardwood forests of eastern North America are shifting away from oak (*Quercus* spp.) dominance. In the relative absence of large scale disturbances of the past, shade-tolerant, 'mesophytic' species (such as red maple (*Acer rubrum* L.), sugar maple (*A. saccharum* Marsh.), and blackgum (*Nyssa sylvatica* Marsh.)) are proliferating in increasingly shaded understories (Abrams, 1992; Fei et al., 2011; Nowacki and Abrams, 2008). An understanding of prehistoric (Delcourt and Delcourt, 1997; Delcourt et al., 1998; Hart et al., 2008) and historic (Hutchinson et al., 2008; McEwan et al., 2007) fire regimes in this region has emerged rapidly over the past several decades from

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archaeological, dendrochronological, and fossil pollen and charcoal records. Evidence reveals that anthropogenic ignitions dominated the sources of fire on the landscape, occurred at varying intervals over the past 4000 years, and likely helped to shape forests in myriad ways (Delcourt and Delcourt, 1998). While the national policy of fire suppression since the 1930s has undoubtedly contributed to these changes in forest species composition and forest structure, it is increasingly apparent that the history of fire use and suppression across the landscape is insufficient both for explaining contemporary forest species composition and structure and for understanding past changes. What is emerging is a strong sense of the importance of multiple interacting disturbance agents and ecosystem drivers (*sensu* McEwan et al., 2011) that potentially include climate variability (Buchanan and Hart, 2012; McEwan et al., 2011), changing human uses of the landscape (Guyette et al., 2002), and impacts of other organisms, such as the passenger pigeon (Ellsworth and McComb, 2003), white-tailed deer and other herbivores (McEwan et al., 2011), and chestnut blight (Keever, 1953; Woods and Shanks, 1959).

Nonetheless, in response to documented shifts in long-term forest stand structure and species composition and the apparent primacy of fire as a disturbance agent, prescribed fire has been incorporated into Land and Resource Management Plans (USDA Forest Service Daniel Boone National Forest, 2005; USDA Forest Service Mark Twain National Forest, 2005; USDA Forest Service Ozark-St. Francis National Forests, 2005) throughout the region to address the restoration of oak-dominated ecosystems (Arthur et al., 2012; Yaussy et al., 2008). Yet, it is proving difficult to discern the potential for prescribed fire, applied to contemporary forests, to shift species composition and stand structure in a manner that achieves forest management goals for a more open canopy structure with lower densities of fire-sensitive and shade-tolerant species. Many stated objectives for prescribed burning, such as increased oak regeneration, require alterations to stand structure including reduced stem density and basal area. Management to achieve such objectives could, of course, incorporate forest thinning targeted at shade-tolerant, fire-sensitive species, but one of the challenges of such management is that in eastern forests, the sprouting response to top-kill is prolific, thus requiring herbicide application following cutting. In an effort to treat large areas with limited budgets, and operating on knowledge of the prehistoric and historic roles of fires on the landscape, fire is often the tool of choice, albeit one that requires repeated application and significant refinement (Arthur et al., 2012; Brose et al., 2013).

Over the past two decades, many studies have focused on the effects of prescribed fire on stand structure, species composition, and oak regeneration in central and Appalachian hardwood forests and surrounding region, with somewhat mixed results. Several studies in New York (McGee et al., 1995), Georgia (Loftis, 1990), Connecticut (Ducey et al., 1996), and North Carolina (Elliott et al., 1999) have shown reduced importance value, survival, or density of oak following a single, low-intensity prescribed fire and increased competitive status of mesophytic species like red maple, American beech (*Fagus grandifolia* Ehrh.), and black birch (*Betula lenta* L.). In Ohio (Hutchinson et al., 2005, 2012a), Kentucky (Alexander et al., 2008; Blankenship and Arthur, 2006; Green et al., 2010), and Georgia (Loftis, 1990), studies have found differing effects of prescribed fire on oak seedling success following fire-driven reductions in stem density. With this focus on the success of oak regeneration in response to prescribed fire, several studies have measured longer-term effects of multiple fires on stand structure (Barnes and Van Lear, 1998; Blankenship and Arthur, 2006; Burton et al., 2010; Fan et al., 2012; Hutchinson et al., 2012b; Signell et al., 2005), again with somewhat varied results that likely reflect variability in geography and species composition, as well as nuances of burn behavior. Measures of tree vigor are rarely included in these studies, yet data stemming from such measures may be key to how future stand dynamics will unfold, especially if fires lead to declined vigor, or even mortality, of residual, seed-producing overstory oaks.

This study examined the effects of prescribed fire implemented over eight years at differing frequencies on crown dieback (measured using crown dieback classes, and changes in crown dieback classes), tree mortality, stand structure, species composition, and sprouting response, as well as the relationship between fire temperatures and char height, mortality, and crown dieback class. We hypothesized (H1) that prescribed burning would impact stand structure and species composition, and that these effects would be greater on sub-xeric and intermediate sites compared to sub-mesic sites because of greater flammability and burn severity on drier sites. We also expected greater impacts to maples versus oaks after the first fire (H1a), and that the initial pulse of tree mortality after the first fire would cause the greatest change to stand structure (H1b). We further hypothesized (H2) that char height and fire temperature would be positively correlated with tree mortality and negatively correlated with the vigor (measured using crown dieback classes) of surviving trees. Additionally, we hypothesized (H3) that prescribed burning would initially increase sprouting response, but that subsequent fires would lead to fewer surviving sprouts. Finally, we hypothesized (H4) that in subsequent years, surviving trees would recover from the effects of fire, as measured by increasing crown dieback class, based on the assumption that lower stem density and basal area would allow for increased access to limited availability of soil moisture, light and nutrients.

2. Materials and methods

2.1. Study area

This study was conducted on the Cumberland Plateau in eastern Kentucky, in the Cumberland Ranger District of the Daniel Boone National Forest (DBNF) in the Low Hills Belt (Braun, 1950). Climate is humid, temperate and continental. Mean annual air temperature is 12.8 °C, with cool winters (January mean daily temperature is 0.5 °C) and warm summers (July mean daily temperature is 24 °C; Foster and Conner, 2001). Annual mean precipitation of 122 cm is distributed fairly evenly throughout the year (Foster and Conner, 2001). The study area encompasses varied topographic terrain and aspect, with elevations ranging from 260 to 360 m and slopes ranging from 0% to 75% (median 45%). Steep slopes and unglaciated terrain lead to variation in topography from shallow coves to exposed ridges, which influence soil moisture conditions. Soils are variable in depth and texture, and are classified as Typic Hapludults, Typic Hapludalfs, Ultic Hapludalfs, and Typic Dystrochrepts (Avers, 1974).

Prior to burning, the study sites were second-growth forests dominated by oaks and hickories (*Carya* spp.) in the midstory (stems 10–20 cm diameter at breast height (DBH)) and overstory (stems $\geqslant$ 20 cm DBH). The forests regenerated since extensive logging and range in age from 80 to 110 years, on site indices ranging from SI 15 to 34 m based on white oak (*Q. alba* L.), base year 50. The sapling stratum (stems 2–10 cm DBH) was dominated by red and sugar maple, downy serviceberry (*Amelanchier arborea* (Michx. f.) Fern., blackgum, and sourwood (*Oxydendrum arboreum* (L.) DC.). Stem density and basal area of saplings (2–10 cm DBH), midstory (10–20 cm DBH), and overstory ($\geqslant$20 cm DBH) trees were similar among burn treatments and landscape positions (Tables 1 and 2). Density of saplings averaged 899 stems ha^{-1}. Tree densities of midstory and overstory stems were similar, averaging 249 and 211 stems ha^{-1}, respectively, across all landscape positions. Basal area averaged 2 m^2 ha^{-1} for saplings, 4 m^2 ha^{-1} for midstory, and

Table 1
Pre-burn relative stem density (%) and total stem density (stems ha^{-1}) and total basal area ($m^2\ ha^{-1}$) by species and landscape position for saplings (stems 2–10 cm DBH) in 2002 on sites in the Daniel Boone National Forest, Kentucky.

Landscape position	Sub-xeric	Intermediate	Sub-mesic
Species	Relative density (%)		
Acer rubrum L.	35	23	6.2
A. saccharum Marsh.	0	16	50
Amelanchier arborea (Michx. f.) Fern.	21	14	1.5
Carya spp.[a]	4.2	3.1	0.89
Cercis canadensis L.	0	1.8	8.6
Cornus florida L.	2.4	9.8	4.2
Fagus grandifolia Ehrh.	0.60	6.4	7.4
Fraxinus spp.[b]	0	1.8	3.9
Nyssa sylvatica Marsh.	25	13	8.3
Oxydendrum arboreum (L.) DC.	8.4	3.4	0.30
Other spp.[c]	1.2	4.3	7.7
Total *Acer* spp.	35	39	56
Total *Quercus* spp.[d]	2.4	3.7	1.2
Total density (stems ha^{-1})	879	798	1021
Total basal area ($m^2\ ha^{-1}$)	2.1	1.7	2.3

[a] Includes *Carya glabra* Sweet, *C. ovata* K. Koch., and *C. tomentosa* (Poiret) Nutt.
[b] Includes *Fraxinus americana* L., *F. pennsylvanica* Marsh., and *F. quadrangulata* Michx.
[c] Includes *Asimina triloba* (L.) Dunal, *Betula lenta* L., *Crataegus spp.*, *Juglans nigra* L., *Liriodendron tulipifera* L., *Magnolia acuminata* (L.) L., *Morus rubra* L., *Ostrya virginiana* (Mill.) K. Koch, *Pinus rigida* Mill., *P. strobus* L., *P. virginiana* Mill., *Prunus serotina* Ehrh., *Robinia pseudoacacia* L., *Sassafras albidum* (Nutt.) Nees., *Tilia americana* L., and *Ulmus rubra* Muhl.
[d] Includes *Quercus alba* L., *Q. coccinea* Muenchh., *Q. falcata* Michx., *Q. muehlenbergii* Engelm., *Q. montana* Willd., *Q. rubra* L., *Q. stellata* Wangehn., and *Q. velutina* Lam.

22 $m^2\ ha^{-1}$ for overstory trees (Tables 1 and 2). Oak species dominated the overstory stratum prior to burning, across all landscape positions and treatment areas, whereas species composition of midstory trees and saplings varied with landscape position (Tables 1 and 2). On sub-xeric landscape positions, red maple dominated sapling and midstory size classes, while sugar maple dominated these size classes on sub-mesic landscape positions. On intermediate landscape positions, red and sugar maple combined dominated sapling and midstory size classes. Species composition of the sapling stratum differed from that of the oak-dominated midstory and overstory strata by having low stem density of sapling oaks (<4%) compared to 35–56% of sapling stem density dominated by sugar and red maple combined.

Prior to the implementation of this study, sites had not been burned by wildfire or prescribed fire in the last 30+ years (Loucks et al., 2008).

2.2. Experimental design

Three study sites (Buck Creek, Chestnut Cliffs and Wolf Pen), each ~200 to 300 ha in area, were established in 2002 within an 18-km^2 area. Each of the three sites was subdivided into three treatment areas, 58–116 ha in size, with each of three treatments randomly assigned within each site: 'Frequent burn', to which prescribed fire was applied in 2003, 2004, 2006, and 2008; 'Less Frequent burn', to which prescribed fire was applied in 2003 and 2009; plus a Fire-Excluded treatment. Within each treatment-site combination, 8–12 study plots were located using a stratified-random sampling scheme designed to select plots across the varied terrain of each area for a total of 92 plots (Table 3). In December 2006, 5 of 10 Fire-Excluded plots in one study site (Wolf Pen) burned accidentally during an unplanned fire, leaving 5 Fire-Excluded plots on that site, or 26 Fire-Excluded plots for the entire experimental design starting with measurements made in 2007.

Plots were 10 m × 40 m (0.04 ha), arrayed parallel to the topographic contour (Fig. 1). Following determination of species composition on each plot (see below), plots were assigned one of three landscape positions, sub-xeric, intermediate, or sub-mesic, using a classification system of individual tree species' site preferences to create an objective and quantitative accounting of site quality (McNab et al., 2007; McNab and Loftis, 2013). Each species was subjectively categorized as xerophytic, mesophytic, or ubiquitous based on expert knowledge of site affinity for that species. By

Table 2
Pre-burn relative stem density (%) and basal area (%) and total stem density (stems ha^{-1}) and basal area ($m^2\ ha^{-1}$) by species and landscape position for midstory (10–20 cm DBH) and overstory (⩾20 cm DBH) stems in 2002 on sites in the Daniel Boone National Forest, Kentucky.

Landscape position	Sub-xeric		Intermediate		Sub-mesic		Sub-xeric		Intermediate		Sub-mesic	
Size-class (cm DBH)	10–20	⩾20	10–20	⩾20	10–20	⩾20	10–20	⩾20	10–20	⩾20	10–20	⩾20
Species	Relative density (%)						Relative basal area (%)					
Acer rubrum L.	47	11	23	3.8	8.3	2.8	45	5.7	23	1.7	8.3	1.6
A. saccharum Marsh.	0	0	9.5	1.8	36	5.6	0	0	8.9	0.66	34	2.6
Amelanchier arborea (Michx. f.) Fern.	6.3	0	5.9	0	1	0	4.9	0	4.5	0	0.70	0
Carya spp.[a]	3.1	3.8	11	14	8.3	19	2.5	2.2	13	10	9.0	18
Fraxinus spp.[b]	0	0	0.81	0.29	8	2.1	0	0	0.99	0.12	7.9	2.3
Liriodendron tulipifera L.	0.45	0.63	1.1	8.3	5	15	0.85	0.92	1.6	12	4.6	16
Nyssa sylvatica Marsh.	2.2	0.63	3.8	2.1	2.7	1.1	2.2	1.0	3.3	2.4	2.8	0.67
Oxydendrum arboreum DC.	9.8	0	7.0	0.59	0	0	8.8	0	5.7	0.18	0	0
Quercus alba L.	5.4	8.9	11	14	4.3	27	5.3	12	13	13	5.3	26
Q. coccinea Muenchh.	4.1	22	0.54	6.2	0.33	1.1	6.3	23	0.75	6.7	0.27	0.53
Q. montana Willd.	16	37	15	27	6.0	5.3	19	38	14	27	6.0	8.5
Q. rubra L.	0.45	0	1.6	5.0	2.3	6.7	0.70	0	2.1	3.6	2.7	8.6
Q. velutina Lam.	0.89	12	0.81	15	1.0	5.3	0.95	13	0.80	21	1.1	7.0
Other *Quercus* spp.[c]	0	1.3	0	0	1.0	1.8	0	1.5	0	0	0.77	1.0
Sassafras albidum (Nutt.) Nees	1.3	0	4.6	1.5	6.0	0.70	0.86	0	4.7	0.81	7.0	0.25
Other spp.[d]	3.1	3.1	4.1	0.59	10	6.7	3.0	2.4	3.0	0.55	10	7.3
Total *Acer* spp.	47	11	33	5.6	44	8.4	45	5.7	31.9	2.4	42.3	4.2
Total *Quercus* spp.	27	81	29	67	15	47	32	88	31	71	16	52
Total density (stems ha^{-1})	295	211	226	207	227	215						
Total basal area ($m^2\ ha^{-1}$)							4.7	19	3.7	23	3.7	23

[a] Includes *Carya glabra* Sweet, *C. ovata* K. Koch., and *C. tomentosa* (Poiret) Nutt.
[b] Includes *Fraxinus americana* L., *F. pennsylvanica* Marsh., and *F. quadrangulata* Michx.
[c] Includes *Q. falcata* Michx., *Q. muehlenbergii* Engelm., and *Q. stellata* Wangenh.
[d] Includes *Asimina triloba* (L.) Dunal, *Betula lenta* (L.), *Cercis canadensis* L., *Cornus florida* L., *Crataegus* sp., *Juglans nigra* L., *Magnolia acuminata* L., *Morus rubra* L., *Ostrya virginiana* (Mill.) K. Koch, *Pinus rigida* Mill., *P. strobus* L., *P. virginiana* Mill., *Prunus serotina*, *Robinia pseudoacacia* Ehrh., *Tilia americana* L., and *Ulmus rubra* Muhl.

Table 3
Number of plots in each landscape position (sub-xeric, intermediate, or sub-mesic) for each prescribed fire treatment on each study site within the Daniel Boone National Forest, Kentucky. In December 2006, several fire-excluded plots (one on intermediate landscape position and four on sub-mesic) within the Wolf Pen study area accidentally burned and were removed from analyses thereafter (remaining number of plots shown in parentheses).

	Fire-excluded			Less Frequent			Frequent			
	Sub-xeric	Intermediate	Sub-mesic	Sub-xeric	Intermediate	Sub-mesic	Sub-xeric	Intermediate	Sub-mesic	Total
Buck Creek	4	3	3	3	3	4	4	8	0	32
Chestnut Cliffs	0	4	7	1	6	2	1	5	4	30
Wolf Pen	0	2 (1)	8 (4)	4	7	1	1	3	4	30 (25)
Total	4	9 (8)	18 (14)	8	16	7	6	16	8	92 (87)
Total per treatment	31 (26)			31			30			

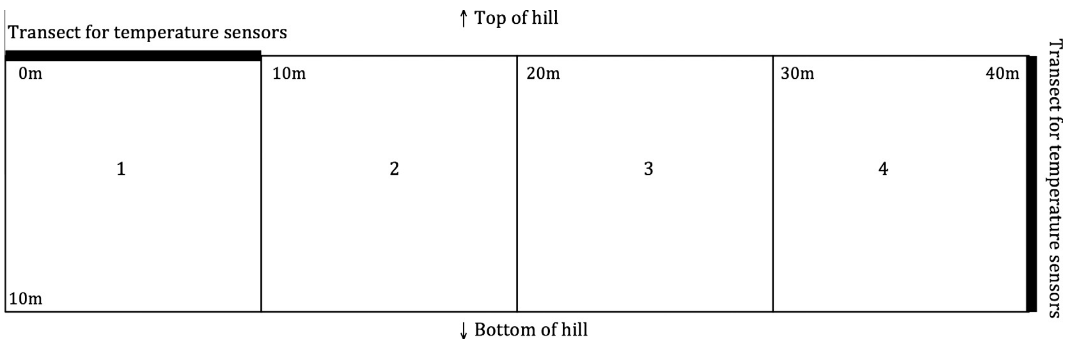

Fig. 1. Diagram of 10 × 40 m plot used for forest measurements in all treatments in the Daniel Boone National Forest, Kentucky. Three fire temperature sensors were installed along each of two 10 m transects on either end of each plot for a total of 6 per plot. All trees ⩾10 cm DBH were tagged and measured across the entire plot. Trees 2–10 cm DBH were tagged and measured in the first quadrant.

tallying and averaging the weight values for the species occurrences on each plot, we placed each plot into one of the three landscape position classes noted above.

2.3. Prescribed fires

All prescribed fires were conducted by USDA Forest Service personnel according to pre-established prescription parameters (USDA Forest Service, 2011). Fires were typically ignited using drip torches, although in 2003, two of the three sites were ignited aerially (Table 4). Burns were conducted between March 24th and April 18th, when air temperatures were between 20 and 27 °C and relative humidity was between 23% and 45%. Efforts to record flame height and spread rates for an assessment of fire intensity were thwarted by the highly dissected and steep topography of the study sites coupled with concerns for personnel safety (Loucks et al., 2008). Hence, we rely on plot-based measures of fire temperature and tree char heights as surrogates for fire intensity. Fire temperatures (Table 4) were measured using temperature-sensitive Tempilac® paints, applied to aluminum tags and covered with aluminum foil, providing a range of minimum temperatures reached (as recorded by the highest temperature surpassed on each pyrometer) including 79 °C, 163 °C, 246 °C, 399 °C, 482 °C and 644 °C in 2003, and 79 °C, 163 °C, 246 °C, 316 °C, 399 °C, 510 °C, and 644 °C, with 644 °C being the melting point of aluminum, in 2004, 2006, 2008 and 2009. Tags were affixed to pin flags for positioning at 0, 20 and 40 cm above the forest floor surface, with six flags per plot, systematically arrayed along two fuel transects, as described in Loucks et al. (2008) (Fig. 1). Mean minimum temperature reached was averaged across the six pyrometers per plot at each height above the forest floor. Fire temperatures varied among years in Frequent sites ($p < 0.01$), but not in Less Frequent sites ($p = 0.33$). Fire temperatures were highest in 2003 and 2006 and significantly lower in 2004 and 2008. There was no significant variation in mean fire temperatures among landscape positions ($p = 0.28$ for Frequent and $p = 0.32$ for Less Frequent).

2.4. Forest measurements

A suite of forest characteristics were measured in each study plot starting in 2002 (prior to burning), and subsequently in 2003, 2004, 2006, 2008, 2009, and 2010 (Fig. 1). Throughout each 0.04 ha plot, all stems ⩾10 cm DBH were identified to species and tagged with a metal tag at the base of the tree, each with a unique number. Similarly, stems 2–10 cm DBH were tagged and measured in one 10 m × 10 m subplot per plot. As trees grew into measurable size classes for their location in the plot, they were tagged, measured, and added to the dataset. During the summer (June through August) of each measurement year and for each tagged tree, measurements and information recorded were DBH, species, crown dieback class, number of basal sprouts, and char height (starting in 2003). Crown dieback class categories were based on percent dieback: 0 = dead, 1 = >50% dieback with some live canopy, 2 = 25–50% dieback, and 3 = <25% dieback. Char height was measured as the highest point of char on the tree bole.

2.5. Statistical analysis

The experiment was analyzed as a split-plot design with three replications (three sites), fire treatment as the whole-plot factor, and landscape position as the split-plot factor. For testing the effect of fire treatment, the error term was the treatment-by-site interaction, while the error term for the effect of landscape position was the treatment-landscape position-site interaction. To investigate changes in measured variables through time (from 2002 to 2010), the repeated-measures factor (year) was modeled on the plot (within landscape position) level, except for measures of sprouts, which were modeled on the split-plot level due to the strong prevalence of zeros among the plots. Stems were divided

Table 4
Burn dates and prescribed fire conditions for each burn within each study site and treatment within the Daniel Boone National Forest, Kentucky.

	Buck Creek					
	Less Frequent		Frequent			
	2003	2009	2003	2004	2006	2008
Date	4/14	4/18	4/14	3/26	4/11	4/8
Ignition method	Aerial	Hand	Aerial	Hand	Hand	Hand
Air temp °C[a]	26	25	26	25	24	26
Rel. Hum. (%)[1]	23	38	23	45	NA[b]	38
Wind speed (km hr^{-1})	0–9.7	NA[b]	0–9.7	3.2–9.7	1.6–3.2	0–3
Mean fire temp (°C)						
Sub-xeric	309	319	348	28	328	99
Intermediate	300	368	336	102	337	249
Sub-mesic	205	125	ND[c]	ND[c]	ND[c]	ND[c]

	Chestnut Cliffs					
	Less Frequent		Frequent			
	2003	2009	2003	2004	2006	2008
Date	3/25	4/18	3/24	4/7	4/13	4/8
Ignition method	Hand	Hand	Hand	Aerial	Hand	Hand
Air temp °C[a]	25	25	24	23	25	26
Rel. Hum. (%)[1]	33	38	35	40	NA[b]	40
Wind speed (km hr^{-1})	0–9.7	NA[b]	0–9	3.2–6.4	4.8–7.6	0–8
Mean fire temp (°C)						
Sub-xeric	501	383	343	51	282	176
Intermediate	374	198	294	273	281	221
Sub-mesic	297	204	227	215	243	121

	Wolf Pen					
	Less Frequent		Frequent			
	2003	2009	2003	2004	2006	2008
Date	4/16	4/17	4/16	4/7	4/11	4/8
Ignition method	Aerial	Hand	Aerial	Aerial	Hand	Hand
Air temp °C[a]	27	24	27	23	20	25
Rel. Hum. (%)[1]	39	38	39	39	NA[b]	42
Wind speed (km hr^{-1})	0–15	NA[b]	0–15	8–13	3.2–6.4	0–7
Mean fire temp (°C)						
Sub-xeric	359	386	402	56	225	212
Intermediate	428	334	338	20	290	156
Sub-mesic	340	20	312	126	247	180

[a] Calculated as the average through duration of fire.
[b] NA means 'not available'.
[c] ND means 'no data'; Buck Creek Frequent contains no sub-mesic plots.

into three size classes for analysis of treatment effects: 2–10 cm DBH (saplings), 10–20 cm DBH (midstory), and ⩾20 cm DBH (overstory). All analyses were performed for each size class separately, or for mid- and overstory combined where explicitly indicated. Statistical analyses for this study were performed using SAS™ software, version 9.2 (SAS, 2011).

Stem density, basal area, and sprout measures were log transformed to satisfy assumptions of homoscedasticity and normality. Repeated-measures analyses were performed using PROC MIXED. Due to the unequal time intervals between data collections, a spatial power covariance function (TYPE = SP(POW)), a generalization of the commonly used autoregressive covariance function, was used to model the repeated-measures covariance structure. Where treatment interactions with year and/or landscape position were significant, the SLICE option in PROC MIXED was used to perform F-tests for treatment effects within year, landscape position, or both. To limit the number of overall tests performed, pairwise post hoc mean comparisons were only conducted where these slice effects were significant. All hypothesis tests were considered statistically significant at p-values <0.05.

Fire temperatures were summarized on the plot level for each burn by calculating the mean temperature measured at three positions (0, 20 and 40 cm above the forest floor) in six locations per plot. The effect of landscape position on fire temperatures was tested using a linear mixed model in PROC MIXED. Relationships between mean plot fire temperatures and response measures (change in sprouts from previous year, change in basal area and stem density, mean change in crown dieback class, and mean plot char height in 2003) were examined using Pearson correlation analysis (PROC CORR).

To examine the relationship between char height and tree mortality, we fit a generalized linear mixed model (in PROC GLIMMIX) with the binary response of 'dead' or 'alive' in 2010. The explanatory variables included in the analysis were maximum char height in 2003, landscape position, and treatment. For testing the effects of treatment and landscape position in the model, the experimental design of a split-plot was retained, but char height was modeled as a subject (tree)-specific covariate such that mortality was modeled for each individual tree as a function of char height. All interactions were included in the model, and non-significant terms removed in a stepwise fashion until all remaining terms were found to have a significant relationship with tree mortality. Where effects were significant, post hoc comparisons of survival odds between treatments or landscape positions were performed. For the continuous variable char height, linear contrasts were used to compare char height effect among treatments and landscape positions. Similarly, differences in mortality between oaks and maples were analyzed using a generalized linear mixed model with explanatory variables treatment, landscape position and species, for midstory and overstory stems only (there were insufficient numbers of oaks in the sapling size class to support the analysis).

Analyses of treatment differences and changes in crown dieback class over time were applied only to trees that remained standing during the entire study period (2002–2010); mortality of trees no longer standing was captured in changes to stem density and mortality analysis. Changes in crown dieback class were analyzed as difference in class (0–3) between pre-burn measurements in 2002 and post-burn observations in 2003 and 2010. Pearson's Chi-squared test was used to test for differences in the relative frequency of each change category (−3 to +2) between treatments. Finally, to test for differences among maples and oaks in the probability of a decline in crown dieback class on fire treated plots, while controlling for potential between-species differences even on control plots, we fit a generalized linear mixed model (PROC GLIMMIX). The binary response variable for this model was defined as 'decline' (for change category −3 to −1) or 'no decline' (0 to +2).

In addition to analyzing stem density and basal area across species, we also examined the effect of treatment and time (2002 versus 2010) on changes in the density and basal area of standing live trees by species group. However, not all species were represented in sufficient numbers in each size class and treatment combination, so only species that occurred in at least three plots in each treatment combination in 2002 were included in each analysis. For saplings, the species (or species groups) that met these criteria were red and sugar maple combined, downy serviceberry, flowering dogwood (*Cornus florida* L.), blackgum, and sourwood. Species analyzed in the midstory size class were red and sugar maple combined, downy serviceberry, all hickory species combined, sourwood, white oak, chestnut oak (*Q. montana* Willd.), and sassafras (*Sassafras albidum* (Nutt.) Nees). In the overstory size class, species analyzed were red and sugar maples combined, hickories combined, white oak, chestnut oak, scarlet oak (*Q. coccinea* Muench.) and northern red oak (*Q. rubra* L.) combined, and black oak (*Q. velutina* Lam.). We combined red and sugar maple because together they spanned all landscape positions and are similarly shade-tolerant. Scarlet and northern red oaks were combined because they are both red oak species (Section *Lobatae*), and together they grow on all the landscape positions considered in this study. To expand the number of species in the analysis, we also analyzed the combined midstory and overstory size classes. This included the species noted above as well as yellow-poplar (*Liriodendron tulipifera* L.) and blackgum.

3. Results

3.1. Char height, crown dieback class and tree mortality

Char height, measured in 2003 after the first burn, was significantly and negatively correlated with crown dieback class and with change in crown dieback class, in every year and for all tree size classes. Maples and oaks differed somewhat in the strength of the relationship between char height and crown dieback class, with stronger correlations for maples (‘*r*’ ranging from −0.33 to −0.39 among years for maples compared to −0.20 to −0.32 for oaks). In addition, a greater proportion of trees on burned sites exhibited a decline in crown dieback class over the study period compared to those in Fire-Excluded areas, regardless of fire frequency (Table 5). Pearson Chi-squared tests revealed significant differences in the relative occurrence of crown dieback class change between each fire treatment and Fire-Excluded ($p < 0.001$ for saplings and midstory, $p < 0.009$ for overstory, in both 2003 and 2010). For saplings, the Chi-squared test also indicated a difference between Frequent and Less Frequent burns in 2003 ($p = 0.04$) and 2010 ($p < 0.001$). Surprisingly, Less Frequent burns had somewhat greater decline in crown dieback class than Frequent burns. In addition, it appears that throughout the duration of the study, trees declined in crown dieback class; there was a decline in crown dieback class of live trees from 2002–2010 that exceeded the change between 2002 and 2003, even in the Fire-Excluded sites. Fire frequency had little or no effect on change in crown dieback class; both burn treatments had a higher proportion of trees that declined in crown dieback class over the duration of the study compared to Fire-Excluded. Furthermore, the large increase in the proportion of trees declining in crown dieback class from 2002 to 2010 (Table 5) compared to the change measured in the first growing season after burning (from 2002 to 2003; data not shown), suggests that live trees continue to decline in vigor as measured by the proportion of crown dieback.

Landscape position affected char height and mortality (Table 6). For all size classes, mean char height was higher on trees in subxeric and intermediate plots than on those in sub-mesic plots

Table 5
Percent change in crown dieback class from 2002 to 2010 for saplings (2–10 cm DBH), midstory (10–20 cm DBH) and overstory (≥20 cm DBH) trees, where 0 indicates no change, negative numbers indicate decline in crown dieback class, and positive numbers indicate improvement in crown dieback class. This is for all prescribed fire treatments within the Daniel Boone National Forest, Kentucky, calculated by including only those stems still alive in 2010. Percent change in crown dieback class is shown for all species, then for *Acer* spp. and for *Quercus* spp. Significant differences in change in crown class dieback are noted with different lower case letters in parentheses for each treatment within each size-class.

Size-class	Treatment	Change in crown dieback class 2002–2010 (%)																	
		−3			−2			−1			0			1			2		
		All spp.	Acer spp.	Quercus spp.	All spp.	Acer spp.	Quercus spp.	All spp.	Acer spp.	Quercus spp.	All spp.	Acer spp.	Quercus spp.	All spp.	Acer spp.	Quercus spp.	All spp.	Acer spp.	Quercus spp.
Saplings	Fire-Excluded (a)	11	9.4	0	5.1	2.8	33	12	10	33	67	73	33	5.1	4.7	0	0	0	0
	Less Frequent (b)	66	72	64	11	4.7	27	6.5	3.3	9.1	15	19	0	0.97	0.66	0	0	0	0
	Frequent (c)	63	61	25	13	10	0	11	4.6	50	13	24	25	0	0	0	0	0	0
Midstory	Fire-Excluded (a)	8.6	2.1	1.8	4.1	2.1	7.1	6.9	1	8.9	73	88	73	6.5	7.2	7.1	0.41	0	1.8
	Less Frequent (b)	41	49	34	7.1	5.5	14	11	5.5	6.9	36	38	32	4.3	1.4	12	0.93	0.69	1.4
	Frequent (b)	36	41	24	9.9	5.8	17	14	7.7	21	37	45	35	2.0	0	3.0	0.68	0	0
Overstory	Fire-Excluded (a)	2.8	6.3	1.7	6.1	0	7.7	1.9	0	3.4	79	88	74	9.4	6.2	13	0.47	0	0.85
	Less Frequent (b)	19	42	13	5.2	0	7.0	6.8	7.7	7.0	62	50	63	6.8	0	9.5	0.40	0	0.63
	Frequent(b)	14	61	11	4.4	0	4.2	3.0	0	3.0	68	39	66	10	0	15	0.37	0	0.60

Table 6
Mean char height (m) after 2003 prescribed fires and% dead stems in 2010 for all species combined, *Acer* spp., and *Quercus* spp., for saplings (2–10 cm DBH), midstory (10–20 cm DBH) and overstory ($\geq$20 cm DBH) trees, in each landscape position in all prescribed fire treatments within the Daniel Boone National Forest, Kentucky. For all size-classes, mean char height on sub-xeric and intermediate landscape positions was significantly higher than on sub-mesic plots; there were no significant differences in char height between Less Frequent and Frequent treatments. Mortality was significantly higher on sub-xeric and intermediate landscape positions, but only for saplings and midstory stems. NA means not available as there were no char heights on the Fire-Excluded plots.

Size-class	Treatment	Landscape position: Sub-xeric						Intermediate						Sub-mesic					
		Mean char height 2003 (m)			% Dead in 2010			Mean char height 2003 (m)			% Dead in 2010			Mean char height 2003 (m)			% Dead in 2010		
		All spp.	*Acer* spp.	Quercus spp.	All spp.	*Acer* spp.	Quercus spp.	All spp.	*Acer* spp.	Quercus spp.	All spp.	*Acer* spp.	Quercus spp.	All spp.	*Acer* spp.	Quercus spp.	All spp.	*Acer* spp.	Quercus spp.
Saplings	Fire-Excluded	NA	NA	NA	6.7	10	0	NA	NA	NA	18	6.9	100	NA	NA	NA	22	18	0
	Less Frequent	0.37	0.19	0.38	93	100	0	0.27	0.17	0.60	87	86	100	0.10	0.068	NA	63	58	100
	Frequent	0.22	0.19	NA	93	100	0	0.36	0.15	0.40	94	95	100	0.065	0.052	0.20	63	57	0
Midstory	Fire-Excluded	NA	NA	NA	13	4.6	27	NA	NA	NA	19	4.2	12	NA	NA	NA	13	2.0	43
	Less Frequent	0.63	0.54	0.59	72	84	59	0.52	0.31	0.58	47	44	47	0.21	0.17	0.16	25	23	38
	Frequent	0.67	0.42	0.75	73	86	53	0.54	0.27	0.62	58	47	65	0.21	0.17	0.13	21	5.9	0
Overstory	Fire-Excluded	NA	NA	NA	12	50	10	NA	NA	NA	8.6	0	2.7	NA	NA	NA	6.6	0	14
	Less Frequent	0.79	0.73	0.78	28	63	21	0.65	0.63	0.62	26	50	8.2	0.33	0.20	0.49	26	25	30
	Frequent	0.76	1.3	0.68	30	88	16	0.64	1.1	0.56	17	67	15	0.14	0.18	0.16	16	0	23

($p < 0.04$; Table 6). Furthermore, char height in 2003 outweighed fire frequency as a significant factor for sapling or midstory tree mortality between 2002 and 2010. There was no difference in stem mortality between sub-xeric and intermediate landscape positions; however, sub-xeric and intermediate landscape positions had significantly greater mortality than sub-mesic positions. For overstory trees, the effect of char height on tree mortality was strongest on sub-xeric landscape positions and significantly weaker on intermediate landscape positions.

The significant effect of landscape position on tree mortality differed somewhat between oaks and maples. For midstory trees there was a significant effect of landscape position for survival probability of maples ($p < 0.001$), but not for oaks ($p = 0.78$). Midstory maples had the greatest mortality on sub-xeric landscape positions, followed by intermediate, with the lowest rates of mortality on sub-mesic landscape positions. Differences in mortality between midstory oaks and maples were found on intermediate ($p < 0.001$) and sub-mesic ($p = 0.007$) landscape positions; on these sites, midstory maples had greater survival probability than oaks. On sub-xeric landscape positions there were no differences in survival probability of oaks and maples. For overstory trees, there was also a significant effect of landscape position for survival probability of maples ($p = 0.014$) but not oaks ($p = 0.093$). As with midstory maples, probability of mortality of overstory maples was highest on sub-xeric landscape positions, followed by intermediate, with the lowest rates of mortality on sub-mesic landscape positions. Differences in survival probability for overstory oaks versus maples were the opposite of those found for midstory stems; on intermediate and sub-xeric landscape positions, maples had greater mortality than oaks; there was no significant effect of landscape position on oak versus maple mortality on sub-mesic sites. There were not enough sapling-sized oaks to test for differences in mortality between oaks and maples for that size class.

Char heights were correlated with fire temperatures, both measured in 2003 (Fig. 2). This relationship between fire temperature and char height was strongest for fire temperatures recorded at 40 cm ($p < 0.001$ for all size classes), and the strength of the relationship increased with increasing tree size class (Pearson correlation 0.48 for saplings; 0.63 for midstory trees; and 0.74 for overstory trees). Although the limitations imposed by using Tempilac paints to measure fire temperature may have influenced the sensitivity of these measurements at the local scale, differences among years and the significant correlation of fire temperature with char height suggest that these measurements are adequate for our purposes.

3.2. Stem density and basal area

Burning significantly reduced stem density and basal area of saplings and midstory trees, but had no significant effect on stand structure of overstory trees. On a plot basis, higher fire temperatures in 2003 were correlated with lower stem density and basal area of saplings ($p = <0.01$) and midstory stems ($p < 0.005$). A similar effect was detected in 2009, but only between fire temperatures recorded at the surface of the forest floor and midstory stem density ($p = 0.04$) and basal area ($p = 0.01$); burns in 2009 occurred on the Less Frequent sites, which had not been burned since 2006.

Burn impacts were most pronounced on drier (sub-xeric and intermediate) landscape positions and often were delayed several years following initial treatment (Figs. 3 and 4). On sub-xeric and intermediate landscape positions, mortality of saplings occurred immediately after burning and continued to occur for 1–2 years after the initial burn, with little effect of burn frequency on this trend. For midstory trees on these landscape positions, both Less Frequent and Frequent burning reduced basal area, but only

Frequent burning reduced stem density on intermediate landscape positions. These changes in midstory stand structure were not statistically significant until 2008, following four burns on the Frequent treatment and only one burn on the Less Frequent treatment. In contrast, on sub-mesic landscape positions, both Less Frequent and Frequent burning had minimal effect on stem density and basal area of saplings and midstory trees. Only in 2009 and 2010 was sapling stem density on the Frequent burn areas significantly less compared to that on Fire-Excluded areas ($p = 0.04$ for both years).

3.3. Species composition

Prescribed fire reduced the number of smaller stems on all landscape positions relative to the Fire-Excluded treatments; the effect was strongest on sub-xeric and intermediate landscape positions (Fig. 5). Although maples had lower maximum DBH than oaks throughout the study area, the lack of larger maples (dominated by red maple) is more striking on sub-xeric and intermediate sites, with diameters typically peaking at <30 cm DBH compared to oaks with diameters as large as 55–80+ cm DBH. On sub-mesic landscape positions, where maples were predominantly comprised of sugar maple, stem diameters peaked at just over 40 cm DBH, compared to oak species with diameters as large as 74 cm DBH. Thus, while both Frequent and Less Frequent prescribed fire significantly reduced stem density and basal area of red maple saplings and midstory stems (Table 7), the reduction in maple stems was attenuated with increasing stem diameter on sub-mesic landscape positions (Fig. 5).

Changes in density and basal area of saplings analyzed by species (maples, downy serviceberry, dogwood, blackgum, and sourwood) over the study period (2002–2010) varied depending on species identity and burn treatment (Table 7). Among saplings, there was a significant reduction in maple density ($p < 0.001$) and basal area ($p = 0.002$) on burned sites regardless of frequency, whereas changes in maple sapling density and basal area on Fire-Excluded sites were not significant. However, as of 2010, maples still dominated saplings on intermediate and sub-mesic landscape positions, regardless of treatment; on sub-xeric landscape positions, genera other than maple and oak dominated the sapling stratum (Fig. 5). Density of flowering dogwood and sourwood saplings decreased significantly and substantially in both burned treatments compared to Fire-Excluded areas. Decreases in dogwood basal area were significant only in the Frequent burn ($p = 0.001$) sites, while sourwood basal area changes were not significant ($p = 0.09$). Blackgum and downy serviceberry decreased in stem density and basal area over time, although lack of statistical power prevents detection of any differences among treatments.

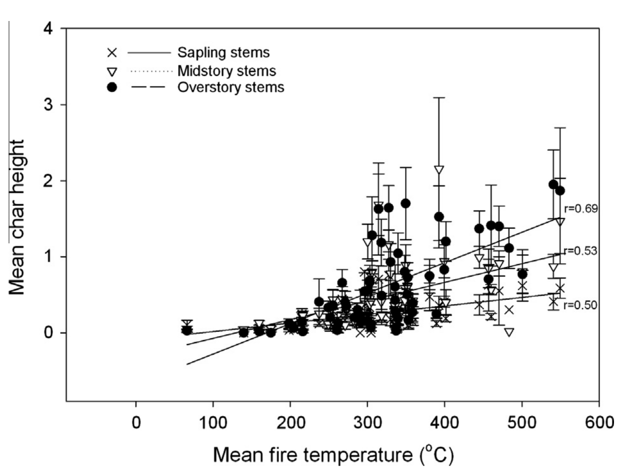

Fig. 2. Mean char height (m) by mean fire temperature (°C) for saplings (2–10 cm DBH), midstory (10–20 cm DBH), and overstory (⩾20 cm DBH) trees in plots with fire temperature sensors in 2003 prescribed fires. Each point represents the mean char height for the specified size-class at the mean fire temperature across all positions of the temperature sensors in the plot. Error bars represent ± one standard error. Plots were located in the Daniel Boone National Forest, Kentucky. Correlation coefficients (r) shown in figure are for Pearson correlation analysis using mean fire temperatures.

Over the study period, changes in density and basal area of midstory species (focusing on those species with sufficient numbers for species-level analysis) depended on species identity and burn treatment. Basal area and density of midstory hickories, chestnut oak, and white oak declined across all treatments from 2002 to 2010. For hickories and chestnut oak, the decrease was most pronounced on burned sites ($p < 0.004$), while for white oak, the decrease was greatest on Fire-Excluded sites (~80% compared to only 45% in burned areas; Table 7). In contrast, density and basal area of midstory maples increased in Fire-Excluded sites but decreased on burned sites (Table 7).

Species composition of combined midstory and overstory stems (all stems ⩾10 cm DBH) did not shift significantly during the study period with the exceptions of hickories and the combined grouping of scarlet oak and northern red oak, which declined in density (by 20%, $p = 0.03$ and 22%, $p = 0.009$, respectively; data not shown) but not in basal area. There were no significant changes in density or basal area of any of the overstory (⩾20 cm DBH) species tested (maples, hickories combined, white oak, chestnut oak, scarlet/northern red oak combined, or black oak).

3.4. Sprouting response

Prior to burning, the number of basal sprouts per tree was similar across burn treatments for all size classes (Fig. 6). There was an initial large increase in sprouts for all size-classes of trees after burning, and the density of sprouts remained greater in 2004 ($p < 0.01$, Fig. 6). After 2004, the density of stump sprouts generally decreased in both Frequent and Less Frequent treatments. However, in 2008 and 2010, the Less Frequent burn treatments had twice as many sprouts per sapling (4.1 and 4.9) compared to the Frequent burn treatments (1.8 and 2.0), where multiple burns reduced the number of sprouts, and 6 times more sprouts per sapling as Fire-Excluded areas (0.59 and 0.86) (Fig. 6). Burning 4 times in 6 years on Frequent sites reduced the number of sprouts per sapling, whereas on Less Frequent sites, the burn in 2009 reduced sprout density, but was followed by significant resprouting in 2010.

In 2003 and 2004, sprouts per midstory stem on burned treatments were about 4× greater (8.6 and 9.0) than on Fire-Excluded treatments (2.1 and 2.2; Fig. 6). Overstory sprouts per stem were about 6× greater on burned treatments than on Fire-Excluded (6.1 versus 0.64) in 2003 and 3× greater (3.4 versus 1.3) in 2004, but differences were not statistically significant. Sprouts per stem on midstory and overstory trees declined in subsequent years, but there were no significant differences compared to 2002 or among treatments (Fig. 6).

Mean fire temperatures were correlated with the change in sprouts per stem from the preceding year in ways that varied with burn, tree size class and year. Correlations between fire temperature and change in sprouts per stem were consistently positive for the initial fire in 2003, regardless of tree size class (r ranged from 0.28 to 0.60; $p < 0.03$; Fig. 7). In 2004, change in sprouts from the previous year on midstory trees was positively correlated with higher fire temperatures (r ranged from 0.51 to 0.62; $p < 0.004$); the relationships were not significant for saplings or overstory trees. Fire temperatures were not significantly correlated with changes in sprouts per stem in 2006 or in 2009 regardless of tree

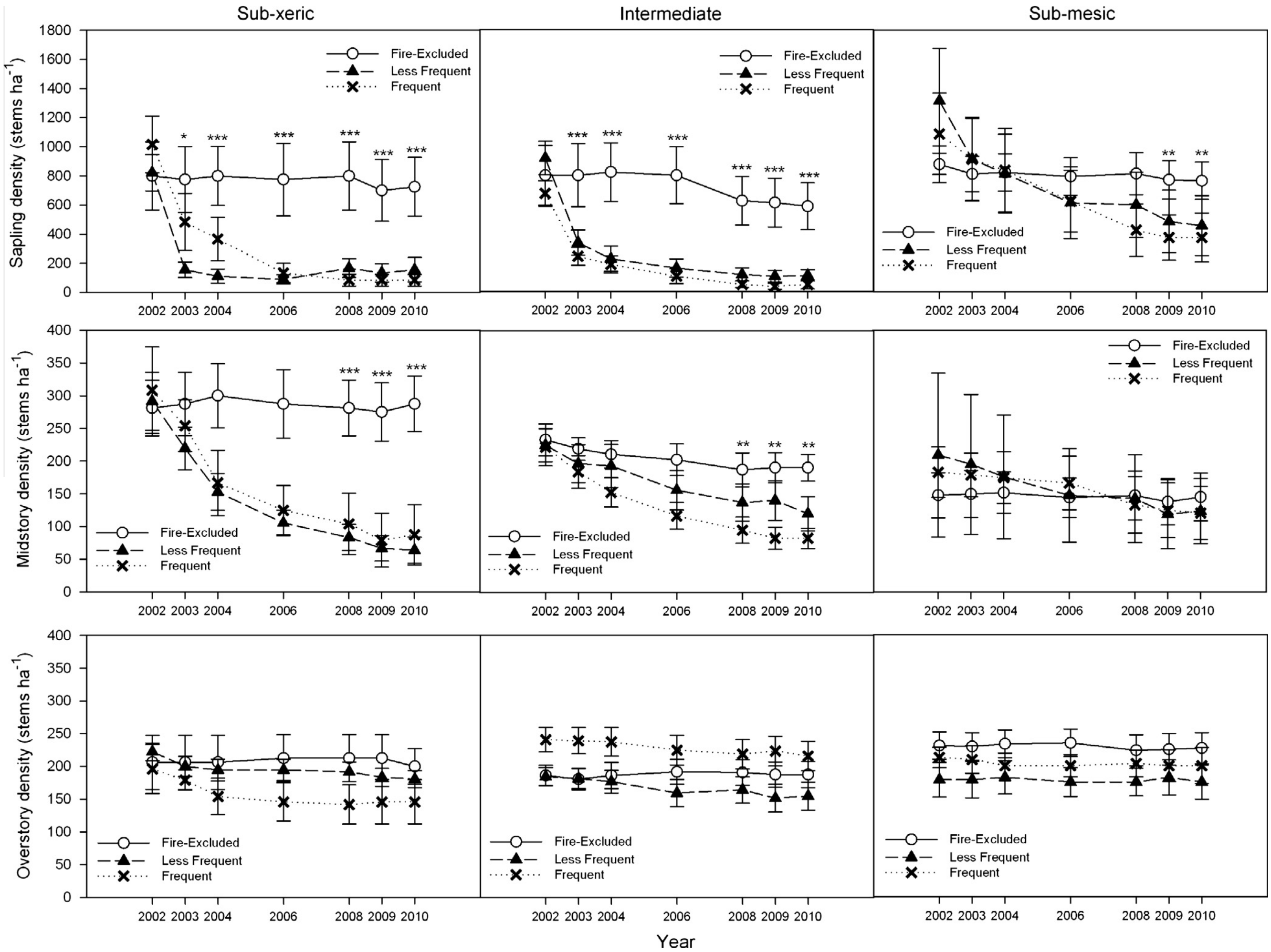

Fig. 3. Density (stems ha^{-1}) of saplings (2–10 cm DBH), midstory (10–20 cm DBH) and overstory ($\geqslant$20 cm DBH) trees on all landscape positions on all prescribed fire treatments for each study year within the Daniel Boone National Forest, Kentucky. The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, and 2008. Error bars represent ± one standard error. Significant differences between treatments within a year are indicated as follows: *Fire-Excluded was significantly different from the Less Frequent treatment, **Fire-Excluded was significantly different from the Frequent treatment, ***Fire-Excluded was significantly different from the Less Frequent and the Frequent treatments.

size class. Although generally not significant, correlation coefficients between fire temperatures and changes in sprouts per stem were more often negative in 2006 and 2008. In 2008, higher fire temperatures resulted in significantly lower numbers of sprouts per stem on saplings (r ranged from −0.22 to −0.42; $p < 0.03$); negative correlations between fire temperature and sprouts per stem were not significant for mid- and overstory trees in 2008. In 2009, when only the Less Frequent sites were burned, there were no significant correlations.

Irrespective of burn treatment and year, sprouting of saplings and midstory trees was significantly greater on drier landscape positions compared to sub-mesic positions ($p < 0.006$, Fig. 8). For saplings, sub-xeric landscape positions had more sprouts per stem than intermediate and sub-mesic landscape positions, and sprouts per stem on midstory trees were greater for sub-xeric and intermediate positions compared to sub-mesic (Fig. 8).

4. Discussion

Stand structure and tree species composition of the study area were fairly typical of second- and third-growth oak forests in the central and Appalachian hardwood region: oaks dominated the overstory, while sugar and red maples dominated the sapling and midstory strata, suggesting these shade-tolerant species are younger than the overstory oaks, as has been shown for a different study site in the Daniel Boone National Forest (Washburn and Arthur, 2003). In the current study, differences in DBH between maples and oaks were most noticeable on sub-xeric and intermediate landscape positions. As these areas are highly flammable, historically they would have burned more readily than mesic landscape positions (Delcourt and Delcourt, 1997). However, density and basal area of maple saplings were greater than that of oak saplings on all landscape positions, and for midstory stems on all but intermediate landscape positions. Taken together, this is the combination of stand structure and divergence in species composition between overstory and midstory strata that forest managers are attempting to manipulate, either with prescribed fire alone or with thinning and fire, with the goal of reducing stem density of shade-tolerant species to increase oak regeneration (Arthur et al., 2012; Brose et al., 2013; Iverson et al., 2008; Waldrop et al., 2008; Yaussy et al., 2008). Previous studies have documented that without the additional tool of forest thinning and/or herbicide application, repeated prescribed fire, rather than single fires, will be necessary to alter stand structure in substantive ways (Blankenship and Arthur, 2006; Hutchinson et al., 2012a). This study provides further documentation of mid-term responses of forest structure and species composition to repeated prescribed

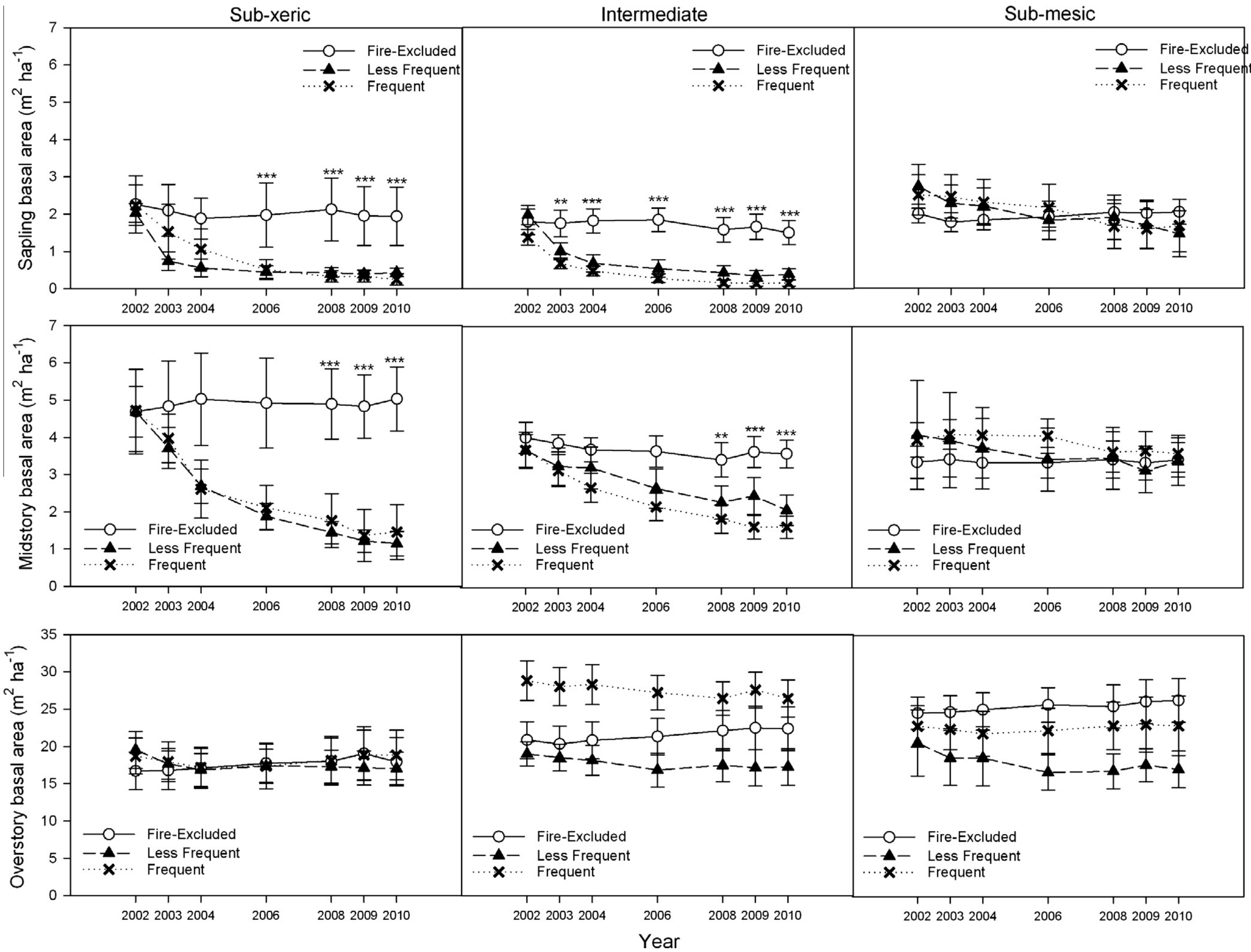

Fig. 4. Basal area ($m^2\,ha^{-1}$) of saplings (2–10 cm DBH), midstory (10–20 cm DBH) and overstory (⩾20 cm DBH) trees on all landscape positions on all prescribed fire treatments for each study year within the Daniel Boone National Forest, Kentucky. The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, and 2008. Error bars represent ± one standard error. Significant differences between treatments within a year are indicated as follows: *Fire-Excluded was significantly different from the Less Frequent treatment, **Fire-Excluded was significantly different from the Frequent treatment, ***Fire-Excluded was significantly different from the Less Frequent and the Frequent treatments.

burning, while also providing evidence for continued mortality and decline in tree vigor (measured by crown dieback class) well beyond the first prescribed fire, and linking fire temperatures with char height and tree mortality.

This study confirmed our hypothesis (H2) that char height was positively correlated with fire temperatures. As hypothesized (H2), char height, which has been used as a surrogate for fire intensity (Sah et al., 2006; Waldrop and Brose, 1999), was positively correlated with fire temperatures and negatively correlated with crown dieback class and changes in crown dieback class over time. The height of charring provides both an effective visual metric for managers assessing the extent of tree damage, but was also reflective of tree mortality, both initially following fire and through time. The relationship between char height and decline in crown dieback class was stronger for maples than for oaks, likely because oaks typically have thicker bark than maples (Bova and Dickinson, 2005; Hengst and Dawson, 1994), and bark thickness is a strong factor in determining the impacts of fire intensity and flame residence time on tissue necrosis (Bova and Dickinson, 2005).

This study supported the hypothesis (H1) that repeated prescribed burning alters stand structure by reducing stem density and basal area of saplings and midstory trees, and to a lesser extent, overstory trees, corroborating findings of previous long-term studies conducted in the region (Blankenship and Arthur, 2006; Fan et al., 2012; Hutchinson et al., 2012b), as well as findings from the Fire and Fire Surrogates network of studies showing that fire is effective at reducing saplings across a wide range of ecosystems (Schwilk et al., 2009). As expected, changes in stand structure were most pronounced on dry sites (sub-xeric and intermediate), despite similar fire temperatures across landscape positions. Red and sugar maple sapling density and basal area decreased significantly following prescribed fire; however, maples continued to dominate the sapling stratum on intermediate and sub-mesic landscape positions even on sites burned four times over eight years (Fig. 5). This is similar to findings in the Ohio Fire and Fire Surrogate Study, which found increased dominance of red maple seedlings following fire on xeric and intermediate sites but not mesic sites (Albrecht and McCarthy, 2006). Consistent with region-wide observations that maples are highly competitive in contemporary upland forests (Blankenship and Arthur, 2006; Fei and Steiner, 2007; Hutchinson et al., 2008), we measured increased stem density and basal area of midstory maples on Fire-Excluded sites during the study period. While burning reduced density and basal area of midstory maples, especially on sub-xeric and intermediate landscape positions, burning also reduced midstory oak and hickory stems.

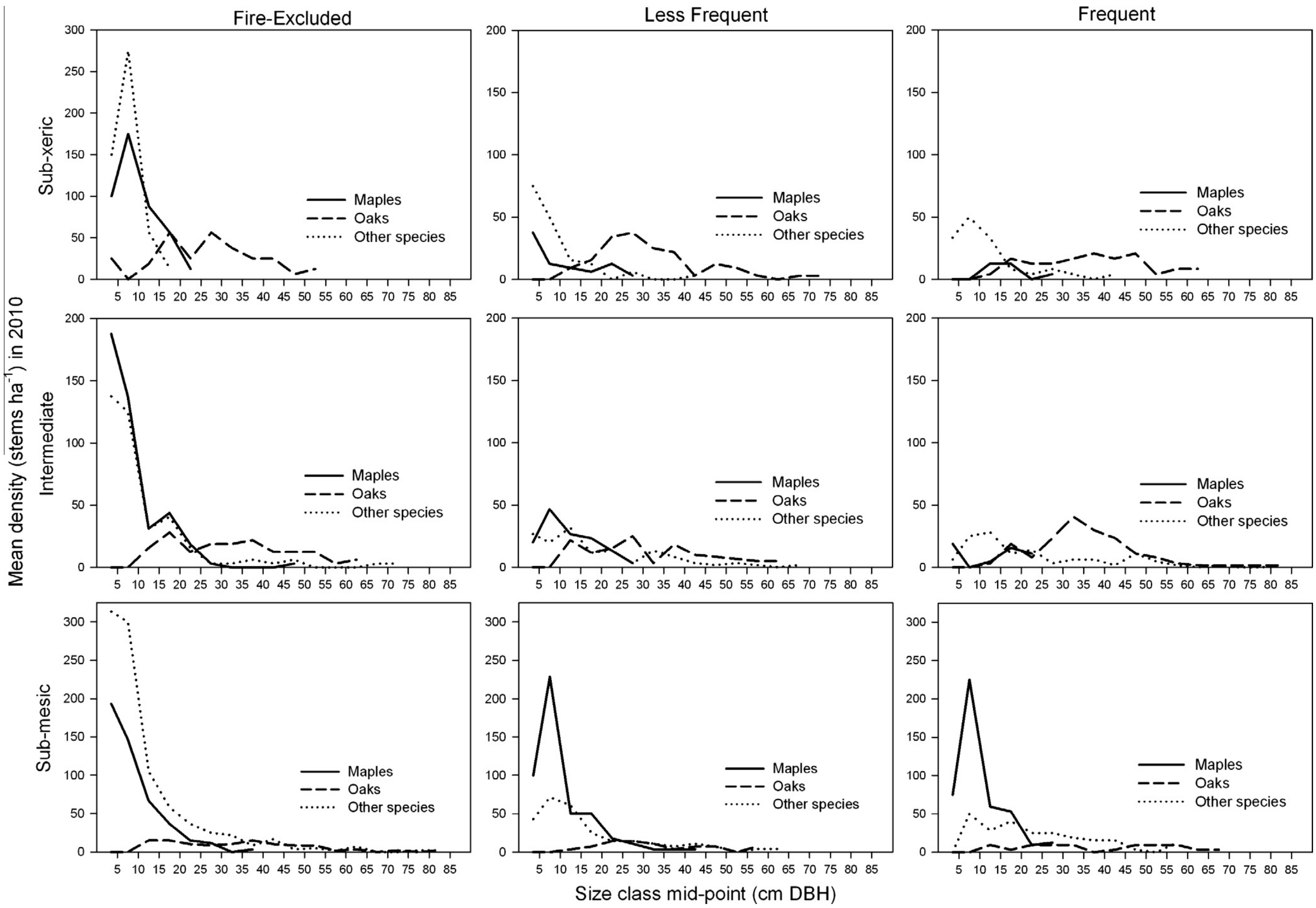

Fig. 5. Mean stem density (stems ha^{-1}) by size-class midpoint (cm DBH) in 2010 (last year of study) on all landscape positions on all prescribed fire treatments within the Daniel Boone National Forest, Kentucky. Maples and oaks are shown separate from all other species. The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, and 2008. The *y*-axis for the sub-xeric Fire-Excluded is different from those of the sub-xeric Less Frequent and sub-xeric Frequent.

Although fire altered stand structure, changes in species composition were minimal because there was mortality across all species (Table 7), and because for some species there was significant mortality on Fire-Excluded sites. In particular, white oak mortality was high across all landscape positions, likely the result of an infestation by three defoliating moths (buck moth (*Hemileuca maia* Drury), eastern tent caterpillar (*Malacosoma americanum* Fabricius), and two-lined chestnut borer (*Agrilus bilineatus* Weber)) that began in 2002 and throughout a period of droughts (noted in more detail below; personal communication, Jeffrey Lewis, Acting District Ranger, Redbird District, USFS Daniel Boone National Forest, November 12, 2014). An ice storm in 2003 that led to branch breakage throughout the study area may also have contributed to mortality in both Fire-Excluded and burned treatments. Changes in stand structure varied depending on landscape position (more so than fire frequency), with much greater impacts on sub-xeric and intermediate landscape positions than on sub-mesic positions, similar to findings in the southern Appalachians (Elliott et al., 1999) and Ohio Fire and Fire Surrogate Study (Albrecht and McCarthy, 2006). These shifts in stand structure did not greatly alter the light environment for regenerating oaks, as shown previously for this site (Alexander et al., 2008) and another study site within the Daniel Boone National Forest (Chiang et al., 2005). Thus, the idea that multiple prescribed fires can affect a fire-mediated shift away from mesophytic species in favor of oaks was not demonstrated convincingly across landscape positions in this study. This study did, however, provide some evidence that prescribed fire is an effective tool for preventing or slowing the expansion of maple dominance, as the Fire-Excluded sites revealed an increase in maple density that was not present in the prescribed fire-treated sites.

A key knowledge gap in elucidating the oak-fire hypothesis is the strength and longevity of the relationship between fire severity and sprouting response (Brose et al., 2013). It has been shown repeatedly that a single fire leads to prolific stump sprouting of hardwood species (Blankenship and Arthur, 2006; Phillips and Waldrop, 2008). This study did not seek to link fire severity to sprouting to address the knowledge gap identified by Brose et al. (2013). However, we did find evidence that repeated burning reduced the number of sprouts per sapling and the capacity for resprouting, supporting hypothesis H3. This is in contrast to the sprouting response on Less Frequent sites, however, where two prescribed fires were separated by five fire-free years. On these sites, the second burn in 2009 initially reduced sprout density, but significant resprouting occurred the following year. These differing trends in sprout density between Frequent and Less Frequent burning suggest that frequency of burning, rather than the actual number of burns, and perhaps the intensity of burns, may determine the resilience of the sprouting response among sapling and midstory stems damaged by fire. Our findings in this study thus partially supports and partially refutes our hypothesis (H3) that prescribed burning would initially increase sprouting response, but that subsequent burns would lead to lower surviving sprouts.

Several studies have examined the relationship between burning and sprouting, with varied responses. Studies using repeated fire, such as the long-term studies of longleaf pine forests

Table 7
Percent change (%) from 2002 to 2010 in stem density (stems ha^{-1}) and basal area ($m^2\ ha^{-1}$) for sapling (2–10 cm DBH) and midstory (10–20 cm DBH) species tested for each treatment within the Daniel Boone National Forest, Kentucky. Species had to be present in a minimum of 3 plots in each treatment to be tested. *P*-values are provided for the main effect of year, i.e., the null hypothesis that there are no differences between 2002 and 2010, across all treatments. Where the treatment by year interaction was significant, *p*-value for interaction is reported in parentheses and individual *p*-values for treatment effect within each year are listed. No significant changes were found for the overstory ($\geq$20 cm DBH) tree species tested.

Size-class	Species	% Change in density (stems ha^{-1})				% Change in basal area ($m^2\ ha^{-1}$)			
		Fire-Excluded	Less Frequent	Frequent	*p*-value	Fire-Excluded	Less Frequent	Frequent	*p*-value
Sapling	*Acer rubrum* L. & *A. saccharum* Marsh.	−10 $p = 0.88$	−69 $p = 0.006$	−77 $p < 0.001$	0.001 (0.009)[a]	+16 $p = 0.21$	−48 $p = 0.008$	−61 $p = 0.008$	0.002 (0.004)[a]
	Amelanchier arborea (Michx. f.) Fern.	−36	−88	−100	0.006	−25	−86	−100	0.018
	Cornus florida L.	−18	−83	−79	0.016	−0.97 $p = 0.89$	−37 $p = 0.52$	−64 $p = 0.001$	0.008 (0.012)[a]
	Nyssa sylvatica Marsh.	−37	−88	−81	0.009	−32	−77	−71	0.013
	Oxydendrum arboreum DC.	−10	−69	−77	0.039	−57	−88	−99	0.091
	All species	−20	−79	−83	<0.001	−10	−69	−68	<0.001
Midstory	*A. rubrum* L. & *A. saccharum* Marsh.	+12	−54	−43	0.024	+18	−44	−32	0.045
	Amelanchier arborea (Michx. f.) Fern.	+14	−49	−72	0.074	+23	−43	−73	0.14
	Carya spp.	−11	−48	−42	0.018	−11	−32	−36	0.020
	Oxydendrum arboreum DC.	+13	−60	−27	0.090	+42	−33	−51	0.18
	Quercus alba L.	−85	−56	−40	0.01	−82	−60	−35	0.047
	Q. montana Willd.	−12 $p = 0.68$	−67 $p = 0.004$	−67 $p = 0.004$	0.001 (0.04)[a]	−3	−61	−50	0.021
	Sassafras albidum (Nutt.) Nees	−38	−62	−34	0.093	−30	−49	−61	0.15
	All species	−7.1	−51	−52	<0.001	−2.3	−47	−45	<0.001

[a] Significant treatment x year interaction for these species. P-values provided for year effect within each treatment.

(Outcalt and Brockway, 2010), have demonstrated that frequent fire can effectively reduce hardwood stem density. Blankenship and Arthur (2006) found more sprouts on sites that were burned three times compared to those that were burned two times; that study had variable antecedent periods between fires which, based on the results of this study, could obfuscate the effects of numbers of burns, versus frequency of burns, on sprout dieback and resprouting. Hutchinson et al. (2012b), working on sites within the unglaciated Allegheny Plateau of southern Ohio, found that more fires led to fewer sprouts. They invoke a common expectation that repeated topkill should deplete belowground carbon reserves, thereby leading to less subsequent resprouting. Differences in the findings of Blankenship and Arthur (2006), Hutchinson et al. (2012b) may reflect different numbers of fires (three to five for the latter study), as well as differences in the antecedent periods between fires (difficult to distinguish in the studies) and differences in the density of red maple and other mesophytic species in the sapling and midstory strata of these sites. Additionally, fire temperature and fire intensity are important factors determining the extent of damage and carbon loss of topkilled trees. In a prescribed burn study in mixed oak forests of southern Ohio, Schwemlein and Williams (2007) found that fall burns were significantly hotter than spring burns and suggest that fall burns may inhibit sprouting due to nutrient leaching during tree dormancy, while spring fires assist sprouting because of nutrient additions. Working on the Bankhead National Forest within the Southern Cumberland Plateau Physiographic Region, Clark and Schweitzer (2013) found that the temperature of a single fire did not predict initial red maple sprouting intensity, which differs from our finding that fire temperature and sprouts per stem were positively correlated for the first burn. The variable results of these studies serve primarily to demonstrate insufficient understanding of the conditions leading to reductions in sprouting due to burning. The timing of repeated burns, and particularly the antecedent period between burns, may be more important to the ultimate outcome of competition among resprouting stems of oaks and maples than the number of burns. Further elucidating key factors influencing this important structural response to burning should be a focus of future studies aimed at improving our understanding of how and when fire alone can be effective in reducing understory stem density.

We hypothesized (H1b) that the first prescribed fire would lead to immediate effects on tree mortality, and therefore that the most significant changes to stand structure would occur in the first growing season after fire treatments were initiated, with relatively small effects in subsequent years, either due to repeated fire or to latent mortality. Further, we hypothesized (H4) that surviving trees would recover from the effects of fire, with improving crown dieback class ratings as time since the initial fire progressed. Our results revealed a more nuanced response to fire. While there was an initial pulse of mortality after the first fire, additional mortality continued in subsequent years. This was particularly true for the saplings and midstory strata in sub-xeric and intermediate landscape positions, which continued to decline in stem density and basal area for several years. This trend toward decreasing stem density and basal area occurred on both Frequent and Less Frequent sites, despite a lack of additional fire treatment on the Less Frequent sites for five years (2004–2008), and is corroborated by the declines in crown dieback class of residual trees found on both Frequent and Less Frequent treatments for the duration of the study. Yaussy and Waldrop (2010) also observed latent mortality up to 4 years after prescribed burning in two sites within the National Fire and Fire Surrogate study, in the Southern Appalachians of North Carolina and the Ohio Hills of Ohio. While tree mortality was not directly attributable to fire temperatures in this study, probably in part because mortality unfolded over multiple

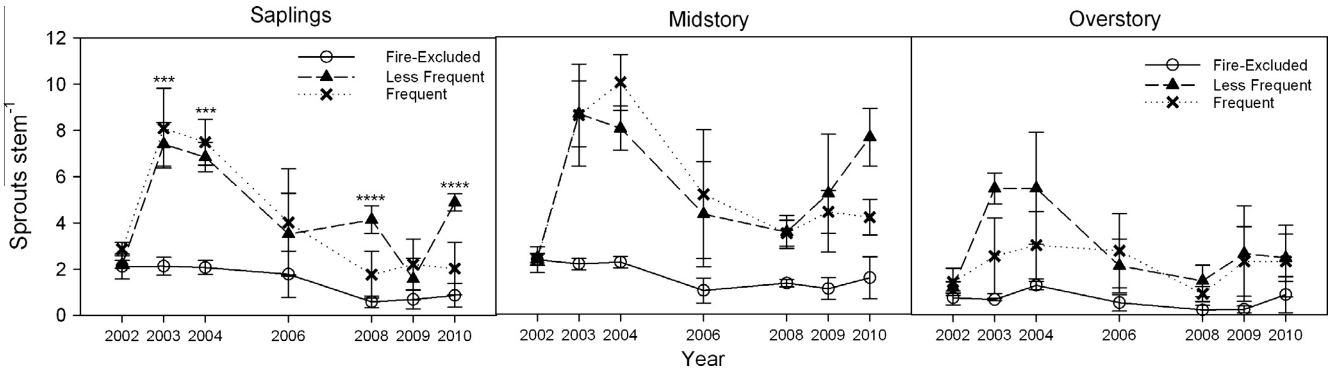

Fig. 6. Mean sprouts stem^{-1} for saplings (2–10 cm DBH), midstory (10–20 cm DBH), and overstory ($\geqslant$20 cm DBH) trees for each prescribed fire treatment for each study year within the Daniel Boone National Forest, Kentucky. The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, and 2008. Error bars represent ± one standard error. Significant differences among treatments is indicated as following: ***Fire-Excluded was significantly different from the Less Frequent and the Frequent treatments, ****the Less Frequent was significantly different from the Fire-Excluded and the Frequent treatments.

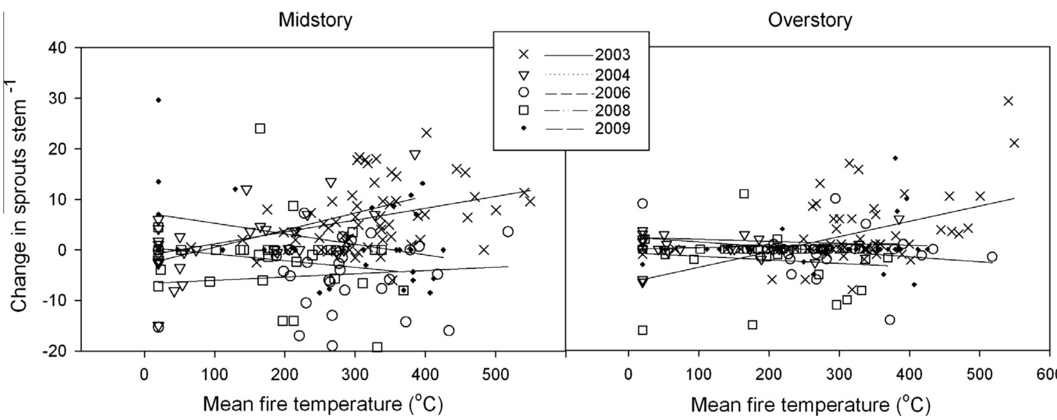

Fig. 7. Change from prior year in mean number of sprouts stem^{-1} on midstory (10–20 cm DBH) and overstory ($\geqslant$20 cm DBH) trees in relation to mean fire temperatures (°C) measured during prescribed fires conducted within the Daniel Bone National Forest, Kentucky. Each point is a single plot, and each plot is represented for each year of measurement, 2003, 2004, 2006, 2008 and 2009. Plots on Less Frequent and Frequent treatments are shown only for those years in which they were burned and for which there are fire temperature data. The fire temperatures shown are means across the three heights of the temperature sensors (0, 20 and 40 cm). Lines show the relationships between mean fire temperature and change in sprouts stem^{-1}. Pearson correlation coefficients for mean temperatures for midstory stems are as follows: 2003: 0.37; 2004: 0.51; 2006: 0.09; 2008: −0.15; 2009: −0.31. Correlation coefficients for overstory stems are: 2003: 0.47; 2004: 0.22; 2006: −0.33; 2008: −0.12; 2009: −0.08.

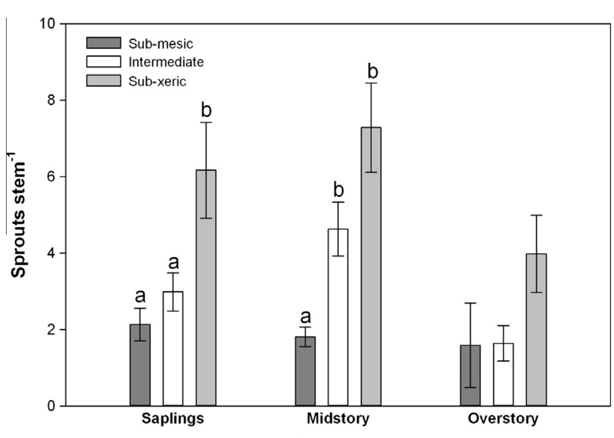

Fig. 8. Mean sprouts stem^{-1} by landscape position, all treatments and years combined, for saplings (2–10 cm DBH), midstory (10–20 cm DBH), and overstory ($\geqslant$20 cm DBH) trees within the Daniel Boone National Forest, Kentucky. Distinct lowercase letters indicate statistically significant differences in sprouts stem^{-1} among landscape positions for saplings and midstory trees. There were no significant differences in sprouts stem^{-1} for overstory trees. Error bars represent ± one standard error.

years and involved multiple interacting factors including drought (see below), it was strongly related to char height. In contrast, Yaussy and Waldrop (2010) found that peak fire temperatures did significantly impact tree mortality.

The latter finding refutes our hypothesis (H4) that surviving trees would recover from the effects of fire with time. We based our hypothesis on the idea that reduced basal area following fire mortality would lead to greater resource availability on sites that are limited in moisture and nutrients, especially on the sub-xeric and intermediate landscape positions, and that residual trees would benefit from this increase in resources. Instead, it appears that fire led to significant tree damage, effectively measured as char height (which was strongly related to fire temperature), and that this fire damage continued to impact the vitality of residual trees for the duration of the study. Large percentages of both oaks and maples, and across all size classes, decreased in crown dieback class between 2002 and 2010, with no indication of improvements for the duration of the study.

Stand-level effects of fire disturbance, whether unplanned or prescribed to achieve management goals, are strongly influenced by other environmental factors as well. In this study, drought likely contributed to the continued stem mortality and decline in crown dieback class we observed throughout the duration of the study. The Palmer Drought Severity Index (PDSI), used as an indicator of moisture excess or deficiency, revealed that three years between 2002 and 2010 experienced moderate to severe drought: 2006, moderate drought; 2007, severe to extreme drought; and 2008, moderate to severe drought (NOAA, 2014, www.ncdc.noaa.gov/temp-and-precip/drought/historical-palmers.php). Indeed, between 2002 and 2010 we found marked rates of declining crown dieback class on Fire-Excluded sites among oaks, with 66% of oak saplings, 16% of oak midstory stems, and 11% of oak overstory stems declining 1–2 crown dieback classes. The proportion of trees declining in crown dieback class on Fire-Excluded sites were less pronounced than on burned sites and did not lead to a significant increase in tree mortality. Nonetheless, this finding supports our contention that prolonged drought, alone and by exacerbating the effects of defoliating insect infestations, may be a contributing factor in both the decline in crown dieback class and mortality of trees on sites treated with prescribed fire.

5. Conclusions and management implications

An increase in the use of prescribed fire in oak forests across the central hardwood and central and southern Appalachian hardwood regions has stemmed from an expanded understanding of the likely role of fire by prehistoric peoples and the shaping of the forest systems that resulted (Delcourt et al., 1998; Hart et al., 2008). Noting apparent changes in forest structure over the past 80 years that include greater stem densities, basal area, and increasing presence of shade-tolerant species, perhaps most notably the maples ('mesophication', *sensu* Nowacki and Abrams, 2008), managers and scientists alike reasoned that adding fire back into these systems could reverse those changes. Early studies of the effects of single prescribed fires demonstrated a lack of ecologically significant impacts to stand structure and species composition (Brose et al., 2006). Reasoning that a single fire would be insufficient to alter changes that took decades to create, and well aware of the potential for single fires to create increased understory stem density and shade through prolific sprouting, researchers began experimenting with multiple fires. Results from several such studies have suggested that even with multiple fires, changes in stand structure are often too minor to significantly alter the course of succession in a direction that favors oaks (Blankenship and Arthur, 2006; Hutchinson et al., 2012b). A notable exception to this result was found when prior burning was followed by gap formation (due to white oak decline), resulting in significantly more abundant and larger oak and hickory regeneration than in unburned gaps where shade-tolerant species had not been reduced by fire (Hutchinson et al., 2012a). This study adds to the accumulating evidence that, in the absence of other disturbances, fire alone is unlikely to alter forest structure sufficiently to create improved regeneration opportunities for oaks and hickories, at least over the timeframe examined here.

Nevertheless, there is still considerable opportunity to refine our use of prescribed fire on these landscapes. For example, we found equivocal evidence for the use of repeated fire to reduce sprouting of (mostly) shade-tolerant species. Repeated burning of top-killed stems and their many root sprouts should, with time, lead to a reduction in root carbon reserves sufficient to slow or eliminate the sprouting response. As demonstrated by Hutchinson et al. (2012a), if shade-tolerant stems and their sprouts can be reduced sufficiently with repeated fire used at the right intervals and with sufficient intensity, opportunities for oak and hickory regeneration will likely occur through gap-phase disturbances. What is still unknown is what type and frequency of prescribed fire would be required to significantly reduce basal sprouts of maples, which significantly reduce light availability in the forest understory (Alexander et al., 2008; Chiang et al., 2005). As long as sapling and midstory size-classes of maples and other shade-tolerant species, including sprouts initiated by burning, exist in high numbers, oak seedlings and saplings will be unable to obtain a competitive edge.

Tree mortality continued over the 8 years of this study, raising questions about how long additional mortality may continue to occur, as well as how mortality may impact future regeneration. Continued mortality could influence future stand composition in different ways, depending on which species are most affected (through mortality of seed-producing trees) and how continued mortality impacts the availability of limiting resources. As noted by Hutchinson et al. (2012a), canopy gaps created by substantial mortality of white oak in an oak forest in Ohio led to improved oak seedling recruitment in areas that had been previously burned. In this study, drought conditions throughout several years may have contributed to mortality; it may be another decade before the impacts of slowly unfolding mortality on regeneration can be evaluated. This long period of response to burning presents significant challenges for experimentally establishing the fire interval required to open the canopy sufficiently for promoting growth and survival of established oaks while maintaining the appropriate seedbed conditions for new oak recruits, but not for shade-tolerant competitors. For example, while burning can reduce the depth of the forest floor horizon and enhance oak seedling germination and establishment (Abrams, 2005; Royse et al., 2010), burning too frequently, or with fires that are too severe, may expose mineral seedbeds, which may favor smaller-seeded species like maples while hindering via desiccation large-seeded species like oak (Royse et al., 2010; Schuler et al., 2010).

Finally, it bears noting that efforts to understand the role of prescribed fire in managing oak ecosystems are unfolding against a backdrop of changing climate and increased pressure from exotic invasive species, both of which have the potential to greatly alter the ecological impacts of fire. Climate warming should promote fire activity, but may also trigger stressors (e.g., drought, insect infestations) that hinder oak ability to respond to fire-driven changes in environmental conditions like light availability. Similarly, there is growing evidence that, under certain circumstances, fire may foster the invasion of exotic species (Rebbeck, 2012; Symstad et al., 2014) such that any potential positive effects of burning on oak regeneration may be outweighed by threat of invasions. Even if fire hinders the growth and survival of shade-tolerant native tree species, invasive species may capitalize on newly available space and resources, creating a new competitive arena for oaks.

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