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Alterations to the fuel bed after single and repeated prescribed fires in an Appalachian hardwood forest

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ABSTRACT

To manage upland oak forests across the central and southern Appalachian mountain and plateau regions, prescribed fire is applied more frequently and across larger areas than in the recent past. The often stated objective is to reduce fuels, but there is a paucity of information on the impacts of repeated burning on fuels, including woody materials and the soil organic layer. These are complex components of forest ecosystems with significant impacts on nutrient and carbon cycling, forest successional dynamics, and soil protection from loss via erosion. Thus, understanding fuel bed response to prescribed fire is essential for predicting future forest function. Using study sites distributed across a range of landscape positions in the Daniel Boone National Forest on the Cumberland Plateau of eastern Kentucky, we examined changes to the fuel bed over six years in response to a single fire (burned once in six years), repeated fire (burned four times in six years), and fire-excluded treatments to determine prescribed fire impacts on fuel loads and mineral soil exposure. Prior to burning, fuel loads were generally similar among landscape positions, although the duff layer was lowest on sub-mesic and greatest on sub-xeric positions. A single fire reduced duff depth by 50%, whereas repeated burning led to depth reductions of > 60%. Repeated burning also significantly increased mineral soil exposure (25%) compared to single burn and fire-excluded (2–4%) treatments, with the greatest effects on sub-mesic and intermediate landscape positions. Repeated burning significantly reduced fine woody (1-h) fuels, but only after three burns, whereas fine fuel mass on sites burned once was similar to those where fire was excluded. There were no statistically significant effects of burning on large woody fuels (100- and 1000-h fuels). Overall, the primary impact of prescribed fire on the fuel bed was to consume the organic horizon and expose mineral soil, which has the potential to reduce fuel continuity for subsequent burns. Fire behavior in this region is driven primarily by fine fuels (litter and duff) and fuel continuity, both of which recover in relatively short periods of one to several years. Reduction of woody fuels is more intractable under a prescribed fire regime.

1. Introduction

Forest ecosystem function is strongly linked to the accumulation of dead organic matter both in the soil organic horizon and woody materials. Dead organic matter serves as the substrate for the detrital food web, storage and cycling of carbon (C) and nutrients, seedbed for plant germination and establishment, habitat for plant roots and wildlife, soil protection from surface runoff, and fuel for fires (Certini, 2005). Fluctuations in the pools of dead organic matter are controlled by a complex suite of factors, from highly stochastic disturbance events like ice and wind storms that can contribute large volumes of woody materials quickly, to the ongoing processes affecting organic matter accumulation, such as decomposition rates, forest productivity, and vegetation species composition (Kalbitz et al., 2000, Vesterdal et al., 2013).

Throughout much of North America, fire is a frequent disturbance agent that serves as an important regulator of dead organic matter. Frequent fire can reduce soil C pools by combusting litter and reducing forest floor depth (Boerner et al., 2009; Royse et al., 2010), but frequent burning may also increase litter C:N (nitrogen) through changes in species composition and/or in response to decreased soil N availability, potentially mitigating such losses through reduced decomposition rates (Hernandez and Hobbie, 2008). Alternatively, long fire-free intervals resulting from fire suppression can increase dominance of tree species with fast decomposition rates, potentially lowering soil C accumulation rates (Alexander and Arthur, 2014; Knoepp et al., 2009).

In upland oak forests of the central and southern Appalachian

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regions, there is growing evidence that the historic fire regime was significantly altered or even entirely suppressed following Euro-American settlement (Flatley et al., 2013; Guyette et al., 2002; Nowacki and Abrams, 2008). While fire suppression during Euro-American settlement has been posited to have increased woody fuel accumulation with potential for increased fire activity (Spetich et al., 1999), others have demonstrated a lack of evidence for this assertion (Polo et al., 2013). Significant impacts of fire suppression to forest structure and species composition include increased dominance of mesophytic species, i.e., species that may reduce forest flammability by creating cool, moist fuels with lower abundance (Nowacki and Abrams, 2008). Recent findings suggest that key mesophytic species, such as red maple (*Acer rubrum* L.), increase litter moisture (Kreye et al., 2013), increase leaf litter decomposition rates (Alexander and Arthur, 2014), and alter precipitation distribution (Alexander and Arthur, 2010), which taken together, could reduce forest flammability. This emerging understanding of the role of fire in the central and southern Appalachian regions has led to increased use of prescribed fire as a component of forest management. The broad goals for forest management with prescribed fire in this region contributed to the motivation for this research on woody fuels, which was part of a larger study that examined the use of prescribed fire at 'frequent' and 'less frequent' intervals and the subsequent changes to stand structure and tree vigor (Arthur et al., 2015), survival and growth of upland oak and competitor seedlings (Alexander et al., 2008; Alexander and Arthur, 2009), establishment of new oak seedlings after burning (Royse et al., 2010), and response of understory vegetation (Keyser et al., 2017).

There are several reasons we need more information about fuels and fire in Appalachian hardwood forests. First, a concern about fuel accumulation and the potential for unplanned fire has been one justification for the frequent and repeated use of prescribed fire across the central and Appalachian hardwood forest regions (USDA Forest Service Mark Twain National Forest, 2005; Waldrop et al., 2016). A related knowledge gap is how prescribed fire alters the fuel bed and whether this influences the potential for wildfire or wildfire behavior. Second, surface organic matter serves an important role for various ecosystem functions, altering C storage, water and nutrient availability, and increasing invasibility. With more planned burning on the landscape (Arthur et al., 2012) and potential for greater unplanned burning if global temperatures rise as predicted (Liu et al., 2013), there is a need to understand how prescribed fire alters the organic horizon from an ecosystem perspective. Finally, there are relatively few studies quantifying how repeated prescribed fire alters the fuel bed, including whether prescribed fire has differential impacts to the fuel layer in different landscape positions. Most studies that have examined the effects of prescribed fire on fuel-loading in this region have been based on short-term studies with one or two burns (Chiang et al., 2008; Graham and McCarthy, 2006; Hubbard et al., 2004; Loucks et al., 2008); only a few have reported on long-term effects of repeated burning (Neill et al., 2007; Polo et al., 2013; Waldrop et al., 2016).

We used a replicated study of prescribed fire implemented as US Forest Service management burns, across topographically varied terrain within the Cumberland Plateau of eastern Kentucky to examine the effects of single versus repeated prescribed fire on forest floor and woody fuels. We previously reported that single and repeated prescribed fires reduced stem density and basal area, but to a greater extent on sub-xeric and intermediate landscape positions compared to sub-mesic (Arthur et al., 2015). Further, we found greater char height, tree mortality, and lower crown vigor on sub-xeric and intermediate sites compared to sub-mesic landscape positions (Arthur et al., 2015). We have also previously reported on the effects of a single fire on fuels, demonstrating that fire significantly reduced litter mass and depth, both of which recovered to levels statistically indistinguishable from pre-burn measurements 10 months post-burn (Loucks et al., 2008). The burn had no significant effect on other fuel components (Loucks et al., 2008).

Here, we extend this previous research for three additional prescribed fires to determine the effects of repeated fire on dead woody fuels and the soil organic horizon. We asked two primary questions about prescribed fires and fuels in an Appalachian hardwood forest: (1) Do single and repeated prescribed fires alter forest floor and woody fuel mass, and if so, do the effects differ by landscape position?; and, (2) Do single and/or repeated fires lead to increased mineral soil exposure?

2. Methods

2.1. Study area

Study sites were located in the Cumberland Ranger District of the Daniel Boone National Forest (DBNF) on the Cumberland Plateau in eastern Kentucky. The region's climate is humid, temperate and continental. Winters are cool with mean daily temperature in January of 0.5 °C, and summers are warm with mean daily temperature in July of 24 °C. Annual mean air temperature is 12.8 °C (Foster and Conner, 2001). Precipitation is distributed fairly evenly throughout the year with an annual mean of 122 cm (Foster and Conner, 2001). The terrain of the study area varied in topography and aspect with elevations ranging from 260 to 360 m and slopes ranging from 0 to 75% slope (median 45%). Topographical variation ranged from shallow coves to exposed ridges, including steep slopes and unglaciated terrain, which influences soil moisture conditions (Jones, 2005). Soils are variable in depth and texture and classified as Typic Hapludults, Typic Hapludalfs, Ultic Hapludalfs, and Typic Dystrochrepts, (Avers, 1974).

The study sites were second-growth forests (80–110 years of age since extensive logging) dominated by upland oaks (*Quercus* spp.) and hickories (*Carya* spp.) in the overstory (stems ≥10 cm DBH), with site index (SI) of 50–110. Red and sugar maple (*A. saccharum* Marsh.) dominated the midstory (2–10 cm DBH) along with downy serviceberry (*Amelanchier arborea* Michx. F.), black gum (*Nyssa sylvatica* Marsh.), and sourwood (*Oxydendrum arboreum* L.). The sites had not been burned by wildfire or prescribed fire in the last 30+ years prior to this study (Loucks et al., 2008). Due to a lack of live fuels in this closed-canopy forest ecosystem, fuel measurements included only dead fuels; live fuels such as grasses and vines, though occasionally present on the site, occur only in isolated circumstances and with low stature.

2.2. Experimental design

In 2002, three study sites (Buck Creek, Chestnut Cliffs, and Wolf Pen), each ~200 to 300 ha in area, were established within an 18 km^2 area. Each study site was subdivided into three treatments: repeated burn, to which prescribed fire was applied in 2003, 2004, 2006, and 2008; a single burn, to which prescribed fire was applied in 2003; and fire-excluded. Using a stratified-random sampling scheme designed to select plots across the varied terrain of each area, 8–12 study plots were located within each treatment-site combination for a total of 92 plots. Plots were 10 m × 40 m, laid out parallel to the topographic contour. Plots were assigned a landscape position (sub-xeric, intermediate, or sub-mesic) using a classification system based on tree species composition (McNab et al., 2007; McNab and Loftis, 2013). Five fire-excluded plots in one study site (Wolf Pen) burned accidentally in an unplanned fire in December 2006, leaving 26 fire-excluded plots beginning with measurements made in 2007 (Table 1).

2.3. Prescribed fires and temperature measurements

USDA Forest Service personnel conducted the prescribed fires according to established prescription parameters (USDA Forest Service, 2011). Fires were typically ignited by hand using drip torches, although in 2003, two of the three sites were ignited aerially (Table 2). Burns were conducted between March 24th and April 16th, when air temperatures were between 20–27 °C, and relative humidity was between

Table 1
Schedule of burning and fuel measurements before and after each year's prescribed burning on all treatments in the Daniel Boone National Forest, Kentucky. "X" denotes fuel measurement, and "B" denotes burning for each treatment and time period. Number of plots per treatment is shown for each study year. In 2006, five fire-excluded plots burned in an unplanned fire in 2006, leaving 26 fire-excluded plots for measurements in 2007 and later.

	2003				2004				2006				2008			
	Jan–Feb	Burned Mar–Apr	Apr–May	# plots	Jan–Feb	Burned Mar–Apr	Mar–Apr	# plots	Jan–Feb	Burned Apr	Apr–May	# plots	Jan–Feb	Burned Apr	Apr–May	# plots
Fire-excluded	X		X	31	X			31	X		X	31	X		X[a]	26
Single burn	X	B	X	31	X			31	X		X	31	X			31
Repeated burn	X	B	X	30	X	B	X	30	X	B	X	30	X	B	X	30

[a] Only forest floor blocks measured post-burn on fire-excluded. Woody fuels measured pre-burn only.

23 and 45%. The highly dissected and steep terrain, along with concerns for personnel safety, prevented efforts to record flame height and spread rates as an assessment of fire intensity (Loucks et al., 2008). Fire temperatures measured in plots and tree char heights were used as surrogates for fire intensity. Pyrometers were constructed using temperature-sensitive Tempilac® paints applied to aluminum tags and covered with aluminum foil, which provided a range of minimum temperatures recorded from 79 °C to 644 °C, the latter being the melting point of aluminum. Tags were affixed to pin flags and positioned at 0, 20 and 40 cm above the forest floor surface, with six flags per plot, systematically arrayed along two transects, as described in Loucks et al. (2008). The mean minimum temperature was then calculated across the plot for each position above the forest floor. The mean temperature measured by pyrometers was highest at the forest floor surface (462 °C) and decreased with increasing height above the surface (258 °C at 20 cm, 149 °C at 40 cm height).

2.4. Fuel measurements

Fuels were measured in all treatments in the winter following leaf fall and prior to each burn in order to compare fuel mass among treatments (Table 1). Fuels on the repeated burn treatment were also measured within 2–4 weeks after every prescribed fire to measure fuel consumption. The fire-excluded treatments were also sampled after the 2006 and 2008 burns because statistical analyses for consumption relied on relative changes in fuel on the fire-excluded. Due to budget constraints, the fire-excluded treatment was not measured post-burn 2004, and the single burn treatment was not measured post-burn 2004 and post-burn 2008 (Table 1).

Forest floor mass was measured by collecting 0.073 m^2 (27 × 27 cm) sections of forest floor from four pre-determined locations in each plot. The forest floor block was moved if it crossed woody material greater than the 10-h timelag size class (2.5 cm diameter), and the sampling location was shifted each year to avoid resampling the same spot. The litter (Oi), duff (Oea), and wood, bark, and seeds (WBS) were separated, dried at 60 °C for 48 h, and then weighed. Duff depth was measured at 0.5 and 1.5 m on each of two transects (described below) per plot. Woody fuel loading was measured by tallying four fuel size-classes along planar intercept transects (Brown, 1974). Fuel classes were nested along two 17-m transects with 1-h (0–0.64 cm in diameter) and 10-h (0.64–2.5 cm in diameter) timelag fuels tallied along 2 m, 100-h (2.5–7.6 cm in diameter) timelag fuels tallied along 4 m, and 1000-h (> 7.6 cm in diameter) timelag fuels measured along the full 17 m. Transects were located at opposite ends of each plot, perpendicular to each other, in locations that would be minimally disturbed during plot installation. The litter was palpated for 1-h fuels that might be hidden by litter and exposed during fire to prevent underestimating consumption of this fuel class.

Woody fuel load weight (w) was calculated by converting the number of intersections tallied to mass/area for size classes using Brown's (1974) formulas. Specific gravity (s) was estimated for the 1-, 10-, and 100-h timelag fuel classes based on Anderson (1978) for southern and southeast forest types as follows: 1- and 10-h, 0.7; 100-h, 0.58. Specific gravity for 1000-h fuels was based on the ratios of rotten and solid 1000-h fuels found on these study sites and on specific gravity values used in central hardwood forests. Initially in 2003, 1000-h fuels were separated into rotten and solid fuels. Later, rotten and solid 1000-h fuels were combined because of difficulty in determining condition class during winter when wet logs were frozen. Based on specific gravity values found in Franklin et al. (1995) of s = 0.58 for solid and 0.30 for rotten 1000-h fuels, 0.58 and 0.30 were applied to the proportion of solid and rotten wood measured in 2003. This resulted in a weighted mean value of 0.40 that we applied to all 1000-h fuels in subsequent sampling.

Table 2
Dates, conditions, and mean fire temperature for prescribed fires conducted on all treatments in three study sites in the Daniel Boone National Forest, Kentucky. Temperature is listed by height above ground and by landscape position.

		Buck Creek					Chestnut Cliffs					Wolf Pen				
		Single	Repeated				Single	Repeated				Single	Repeated			
		2003	2003	2004	2006	2008	2003	2003	2004	2006	2008	2003	2003	2004	2006	2008
Date	Rx parameters	4/14	4/14	3/26	4/11	4/8	3/25	3/24	4/7	4/13	4/8	4/16	4/16	4/7	4/11	4/8
Ignition method		Aerial	Aerial	Hand	Hand	Hand	Hand	Hand	Aerial	Hand	Hand	Aerial	Aerial	Aerial	Hand	Hand
Air temp (°C)[a]	≤ 29.4[b]	26	26	25	24	26	25	24	23	25	26	27	27	23	20	25
Rel. Hum. (%)[a]	≥ 25%[b]	23	23	45	33	38	33	35	40	43	40	39	39	39	33	42
Wind speed (km/hr)	≤ 18[b]	0–9.7	0–9.7	3.2–9.7	1.6–3.2	0–3.0	0–9.7	0–9.0	3.2–6.4	4.8–7.6	0–8.0	0–15	0–15	8–13	3.2–6.4	0–7.0
Fuel moisture (%)	≥ 7%[b]	15[c]	15[c]	7.5[d]	12[c]	8.3[c]	14[c]	18[c]	7.6[d]	10.9[d]	18.1[c]	7[e]	7[e]	7.8[d]	10[c]	14[c]
Mean fire temp @0 cm (°C)		473	564	149	529	300	533	475	411	485	288	584	561	152	476	345
Sub-xeric		546	548	40	531	146	550	554	102	439	369	574	617	95	510	532
Intermediate		537	572	203	528	389	522	494	474	477	301	595	560	20	503	274
Sub-mesic		371	ND[f]	ND[f]	ND[f]	ND[f]	542	371	409	514	251	564	536	266	454	352
Mean fire temp @20 cm (°C)		182	270	54	293	176	316	233	155	197	152	351	257	48	187	124
Sub-xeric		215	282	23	276	89	537	282	30	225	105	295	332	54	128	83
Intermediate		203	264	70	302	226	346	221	198	217	221	411	253	20	230	124
Sub-mesic		141	ND[f]	ND[f]	ND[f]	ND[f]	202	199	133	154	76	267	230	68	181	133
Mean fire temp @40 cm (°C)		139	187	29	181	106	235	158	116	124	90	242	200	32	105	56
Sub-xeric		167	215	20	177	62	416	194	20	181	54	208	256	20	37	20
Intermediate		160	173	33	182	131	254	167	146	149	142	279	202	20	135	69
Sub-mesic		102	ND[f]	ND[f]	ND[f]	ND[f]	149	111	104	61	34	188	169	44	106	55

[a] Average through duration of fire.
[b] Burn prescription parameters are detailed in the Interagency Prescribed Fire Planning and Implementation Procedures Reference Guide for Region 8.
[c] Fuel moisture data collected on site prior to ignition.
[d] Fuel moisture data accessed from the Remote Automated Weather Station (RAWS) located at Triangle Mountain, Kentucky, (https://wrcc.dri.edu/cgi-bin/rawMAIN.pl?ncKTRI) prior to ignition.
[e] Fuel moisture data collected from the National Fire Danger Rating System, accessed by EJ Bunzendahl, Assistant Fire Management Officer, Daniel Boone National Forest.
[f] ND = no data – Buck Creek repeated burn contains no sub-mesic plots.

2.5. Mineral soil exposure

In early winter 2004 and 2006, at the same time as the fuel measurements, the proportion of exposed mineral soil was measured in each plot on all treatments in response to land managers' observations of a perceived increase in mineral soil exposed in burned treatments that was outside of the intended effects of the prescribed burn plan. Concerns were also articulated regarding potential negative impacts to oak seedling regeneration, a key management goal of prescribed burning (J. Lewis, District Silviculturist, Cumberland District, USFS Daniel Boone National Forest, July 17, 2017). Along two 20-m transects in each plot, substrate (organic matter, mineral soil, rock, tree, stump, woody debris, or moss/lichen) was recorded every 10 cm by placing a vertical probe perpendicular to the horizontal transect and touching the forest floor or soil surface. This allowed us to capture the effects of the 2003 fire on repeated burn and single burn treatments (measurements made in early 2004), and two years later (in early 2006) after the second (2004) fire on repeated burn treatments. The 2006 measurements allowed us to assess potential recovery of organic horizon 3 years after one burn (single burn treatment) and 2 years after two burns (repeated burn treatment).

2.6. Statistical analysis

The experimental design was a split-plot design with three replications, with fire treatment as the whole-plot factor and landscape position as the split-plot factors. Within each site, data were collected on multiple plots (a total of 92, except where unplanned events led to a reduction, as noted in Section 2.2), arrayed across each treatment-landscape position combination. To compare changes in measured variables through time (from 2002 to 2008), the repeated-measures factor (year) was modeled on the plot level. Repeated-measures analyses were performed using PROC MIXED. Due to the unequal time intervals between data collection, a spatial power covariance function (TYPE = SP(POW)), a generalization of the commonly used auto-regressive 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. Relationships between pre-burn fuel mass and mean plot burn temperatures and between burn temperatures and fuel consumed (by fuel class) were examined using Pearson correlation analysis (PROC CORR). Statistical analyses were conducted using SAS™ software, version 9.2 (SAS, 2011).

3. Results

3.1. Characterization of pre-burn fuels across the landscape

Prior to burning, fuel load was similar across all landscape positions (41 Mg ha^{-1}) and consisted mostly of duff (46%), followed by 1000-h fuels (22%), 100-h fuels (12%), litter wood, bark, seeds (11%), leaf litter (7.5%), 10-h fuels (6.5%), and 1-h fuels (1.4%). Before burning, duff mass and depth varied significantly among landscape positions ($p < 0.003$ for both; Fig. 1). Duff mass was significantly lower ($p < 0.006$) on sub-mesic landscape positions (16 Mg ha^{-1}) than on both intermediate (20 Mg ha^{-1}) and sub-xeric (23 Mg ha^{-1}). Duff depth was significantly lower on sub-mesic (2.1 cm) and intermediate (2.2 cm) landscape positions compared to sub-xeric (3.5 cm; $p < 0.002$). No other fuel classes differed by landscape position prior to burning.

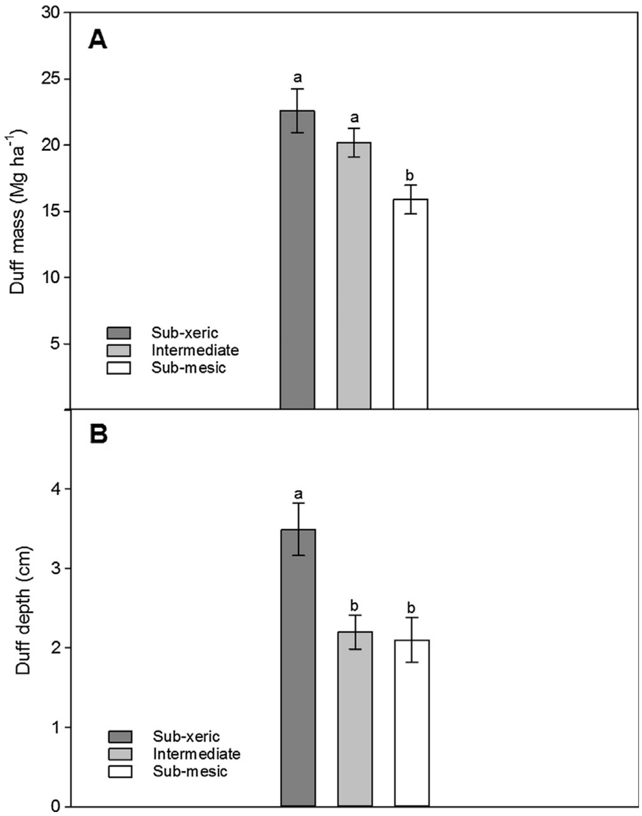

Fig. 1. Duff mass (A, Mg ha^{-1}) and duff depth (B, cm) by landscape position (all treatments combined) as measured before (January and February 2003) prescribed burning in the Daniel Boone National Forest, Kentucky. Duff mass and depth differed significantly among landscape positions before 2003 prescribed fires ($p = 0.002$ and $p = 0.003$, respectively). Significant differences among landscape positions are noted by different lower-case letters above each.

3.2. Changes in fuel mass with repeated prescribed fire

Leaf litter was the primary fuel consumed by each fire; however, this fuel recovered to pre-burn (2003) levels by the subsequent sampling due to annual litter fall inputs (Fig. 2). For example, the 2003 prescribed fires consumed ~2.8 Mg ha^{-1} of leaf litter, but by 2004, leaf litter had recovered to near pre-burn mass (Fig. 2). Despite this recovery to pre-burn mass, litter mass differed significantly among treatments in 2004, with greatest mass on the fire-excluded (fire-excluded vs. single: $p < 0.0001$; fire-excluded vs. repeated: $p = 0.02$; Fig. 2). The repeated burn also had greater litter mass than the single burn ($p = 0.03$) in 2004. The 2006 fire on the repeated burn treatment significantly consumed more litter (3.6 Mg ha^{-1}) than losses without burning on the single burn treatment (0.77 Mg ha$^{-1,}$ $p = 0.001$), which was not burned, and on the fire-excluded (1.6 Mg ha^{-1}; $p = 0.007$). Again, by the next measurement in 2008, litter mass recovered. A similar trend was observed following the 2008 burn ($p = 0.04$: fire-excluded vs. repeated). Landscape position had no significant impact on the change in leaf litter mass with burning (no treatment by landscape position interaction).

Although the results for the wood, bark and seed (WBS) mass component of fuels were highly variable across years and among treatments, the most consistent finding was a significant trend of repeated burning leading to a reduction in this fuel compared to fire-excluded and single burn treatments, and then rebounding during the fire-free interval to levels that exceeded that on fire-excluded treatments (Fig. 3). In 2003, WBS mass on sub-mesic landscape positions increased more than on intermediate landscape positions ($p = 0.006$; data not shown). Landscape position had no significant effect on change in WBS mass with burning for any other measurement period. WBS

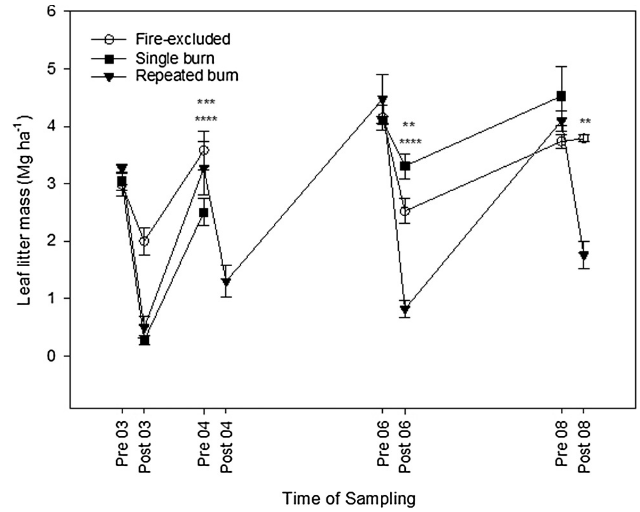

Fig. 2. Leaf litter mass (Mg ha^{-1}) as measured before and after burning on all treatments in Daniel Boone National Forest, Kentucky. The single burn treatment was burned in 2003. The repeated burn treatment was burned in 2003, 2004, 2006, and 2008. The fire-excluded and single burn treatments were not sampled post-burn in 2004, and the single burn treatment was not sampled post-burn in 2008. Error bars are ± 1 standard error of the mean. Significant differences between treatments for a given year pre-burn are indicated as follows: *fire-excluded vs. single burn, **fire-excluded vs. repeated burn, ***fire-excluded vs. single burn and repeated burn, ****repeated burn vs. single burn. Significant differences between treatments for consumption (change from pre-burn to post-burn) are indicated above post-burn dates.

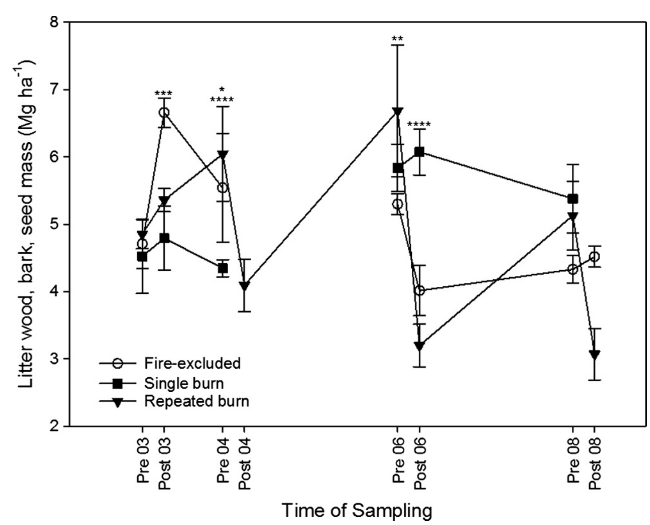

Fig. 3. Mass of wood, bark and seeds in the litter (Oi) (Mg ha^{-1}) as measured before and after burning on all treatments in the Daniel Boone National Forest, Kentucky. The single burn treatment was burned in 2003. The repeated burn treatment was burned in 2003, 2004, 2006, and 2008. The fire-excluded and single burn treatments were not sampled post-burn in 2004, and the single burn treatment was not sampled post-burn in 2008. Error bars are ± 1 standard error of the mean. Significant differences between treatments for a given year pre-burn are indicated as follows: *fire-excluded vs. single burn, **fire-excluded vs. repeated burn, ***fire-excluded vs. single burn and repeated burn, ****repeated burn vs. single burn. Significant differences between treatments for consumption (change from pre-burn to post-burn) are indicated above post-burn dates.

mass increased between pre-burn and post-burn measurements in 2003 on all treatments, but significantly more on the fire-excluded than on the burned (2.0 Mg ha^{-1} vs. 0.37 Mg ha^{-1}; $p = 0.03$). Pre-burn 2006 WBS mass on repeated burn treatments was significantly greater than in the fire-excluded (6.7 vs. 5.3 Mg ha^{-1}, $p = 0.01$), but the 2006 burn significantly consumed WBS mass on the repeated burn (3.4 Mg ha^{-1}) compared to single burn treatments (not burned in 2006), where WBS mass increased during the same time period (an addition of

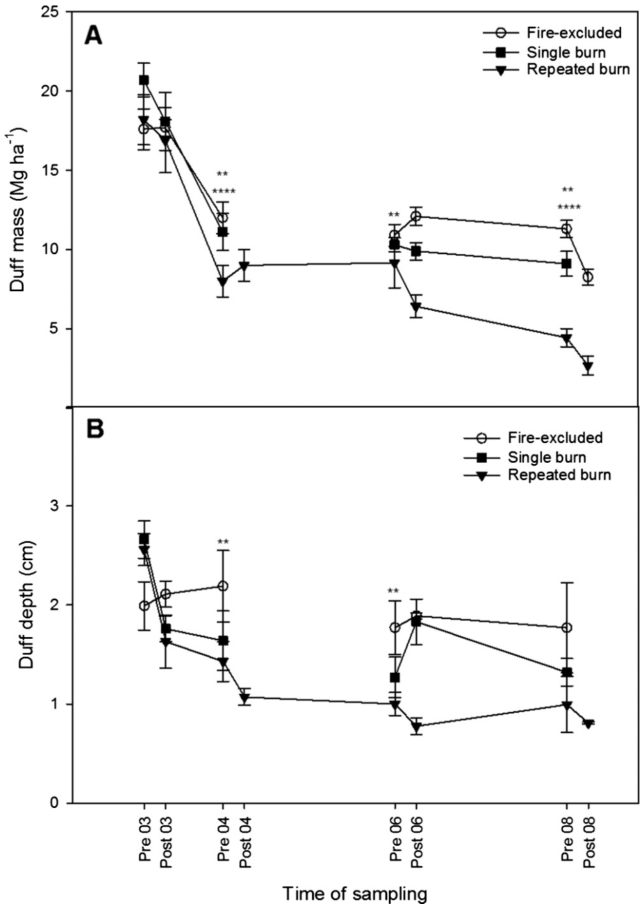

Fig. 4. Duff mass (A, Mg ha^{-1}) and duff depth (B, cm) as measured before and after each prescribed fire in the Daniel Boone National Forest, Kentucky, from 2003 to 2008. The single burn treatment was burned in 2003. The repeated burn treatment was burned in 2003, 2004, 2006, and 2008. The fire-excluded and single burn treatments were not sampled post-burn in 2004, and the single burn treatment was not sampled post-burn in 2008. Error bars are ± 1 standard error of the mean. Significant differences between treatments for a given year pre-burn are indicated as follows: *fire-excluded vs. single burn, **fire-excluded vs. repeated burn, ***fire-excluded vs. single burn and repeated burn, ****repeated burn vs. single burn.

0.26 Mg ha^{-1}; $p = 0.009$).

While duff mass varied significantly by landscape position before burning, burn treatment affected duff mass more than landscape position after the initial burn. Only in 2003, was landscape position a significant factor for the change in duff mass from pre-burn to post-burn measurements, whereas treatment was not a significant factor. After the 2003 burn, intermediate landscape positions lost significantly more duff mass (3.1 Mg ha^{-1}) than sub-mesic landscape positions, which inexplicably gained duff mass (an addition of 2.1 Mg ha^{-1}, $p = 0.01$). Throughout the remainder of the study, duff mass was significantly lower on repeated burn treatments compared to the fire-excluded (2004: 8.0 Mg ha^{-1} vs. 12 Mg ha^{-1}; $p = 0.004$; 2006: 9.2 Mg ha^{-1} vs. 11 Mg ha^{-1}; $p = 0.04$; 2008: 4.4 Mg ha^{-1} vs. 11 Mg ha^{-1}; $p < 0.0001$) and compared to the single burn treatments (2004: 8.0 Mg ha^{-1} vs. 11 Mg ha^{-1}; $p = 0.006$; 2008: 4.4 Mg ha^{-1} vs. 9.1 Mg ha^{-1}; $p < 0.0001$; Fig. 4A). Duff depth on repeated burn treatments was also significantly lower than on the fire-excluded treatments in 2004 and 2006 ($p < 0.05$) and marginally insignificant in 2008 ($p = 0.06$; Fig. 4B). While not significantly different, the single burn treatments typically had shallower duff depth than the fire-excluded (Fig. 4B).

For fine woody (1-h) fuels, mass loss by burning was replaced the following year, presumably by fairly constant fine woody additions to the forest floor, at least initially. Only following three fires in 2008 did repeated burn treatments have lower 1-h fuels compared to fire-excluded (0.45 vs. 0.84 Mg ha^{-1}, $p < 0.0001$) and single burn treatments (0.84 Mg ha^{-1}, $p = 0.001$; Fig. 5A). Sub-xeric landscape positions had greater 1-h fuel mass (0.73 Mg ha^{-1}) than intermediate (0.60) and sub-mesic (0.59; $p = 0.002$; data not shown) positions, but there were no significant interactions among year, landscape position and treatment, suggesting that treatment did not differentially affect the mass of 1-h fuels among landscape positions.

Ten-hour fuels were initially unaffected by burning, after which different trends were found for repeated and single burn treatments. In single burn treatments, 10-h fuels gradually increased relative to fire-excluded and repeated burn treatments. The repeated burns decreased 10-h fuels between pre and post-burn measurements, except after the 2008 fire, in which there was a reverse trend (Fig. 5B). The only significant difference among treatments in 10-h fuels was in 2008, when the single burn treatments had significantly greater mass than the fire-excluded (3.5 vs. 2.0 Mg ha^{-1}, $p = 0.02$) and the repeated burn (2.0 Mg ha^{-1}, $p = 0.02$) (Fig. 5B). Landscape position was not a significant factor in 10-h fuels. Burning had no significant impact on mass of large woody fuels (100- and 1000-h fuels) (Fig. 5C and D).

For mass of all fuels combined (Fig. 5F), there was a marginally insignificant interaction of treatment by year by landscape position ($p = 0.08$). On sub-xeric landscape positions in 2008, fire-excluded treatments had significantly lower total woody fuel mass than repeated burn treatments ($p = 0.009$, data not shown). On intermediate landscape positions, single burn treatments had significantly greater total fuel mass than the repeated burn treatments ($p = 0.002$, data not shown).

3.3. Mineral soil exposure

Mineral soil exposure was greater on burned treatments in 2004 (single burn: 11%, repeated burn: 20%) compared to fire-excluded (5.2%) treatments. While mineral soil exposure on repeated burn treatments did not change significantly from 2004 to 2006 following an additional fire, there was significantly more exposed mineral soil on these treatments than on the single burn treatments in both 2004 ($p = 0.002$) and 2006 ($p < 0.001$). The single burn treatments appeared to recover post fire as mineral soil exposure decreased significantly from 2004 to 2006 (11–2.1, $p < 0.0001$; Fig. 6). Sub-xeric landscape positions (Fig. 6A) had less exposed mineral soil than intermediate (Fig. 6B) and sub-mesic (Fig. 6C) positions (year × treatment × landscape position $p = 0.01$).

3.4. Relationship of burn temperatures to fuel mass and fuel consumption

Across years, total fuel mass before burning was weakly but significantly and positively correlated with burn temperatures at the surface and at 40 cm height (Table 3); there was no significant relationship at 20 cm. This relationship was driven by strongly positive correlations between duff mass and burn temperatures measured at all pyrometer positions (Table 3). Pre-burn measurements of duff depth were also positively correlated with measured fire temperatures. Since mass and depth of duff differed significantly by landscape position, we also analyzed the correlation between duff mass and depth and burn temperature by landscape position. The correlations were significant for all landscape and pyrometer positions, except for the 20 cm position in sub-mesic landscapes ($p = 0.09$). The correlations for burn temperature and duff depth were not borne out across landscape positions, however. Duff depth was not correlated with burn temperature in sub-mesic landscape positions, and was significant only at the surface for intermediate landscape positions. For sub-xeric landscape positions, duff depth was significant at 20 cm and 40 cm pyrometer positions.

With the exception of 1-h woody fuel mass, which was positively correlated with fire temperatures at 40 cm only, there were no other positive correlations between pre-burn fuel mass and burn temperature.

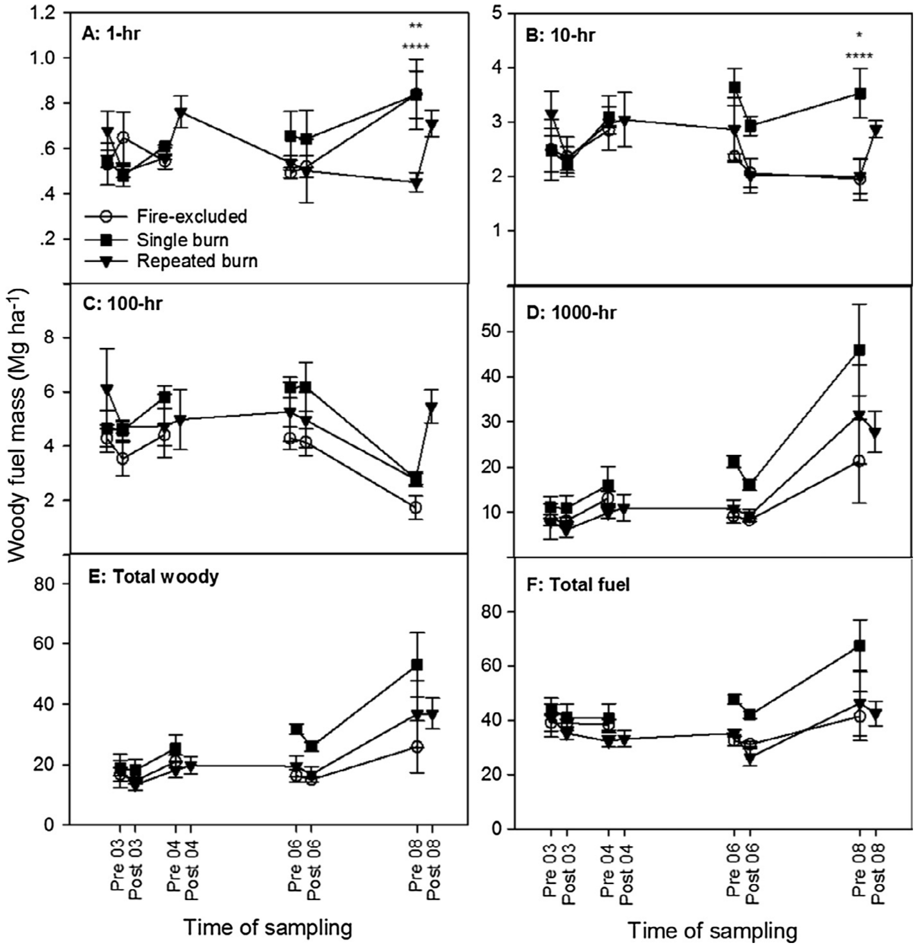

Fig. 5. Woody fuel mass by size class (A–E) and total fuel mass (F) in Mg ha^{-1} as measured before and after burning on all treatments in Daniel Boone National Forest, Kentucky. The single burn treatment was burned in 2003. The repeated burn treatment was burned in 2003, 2004, 2006, and 2008. The fire-excluded and single burn treatments were not sampled post-burn in 2004, and the single burn treatment was not sampled post-burn in 2008. Error bars are ± 1 standard error of the mean. Significant differences between treatments for a given year pre-burn are indicated as follows: *fire-excluded vs. single burn, **fire-excluded vs. repeated burn, ***fire-excluded vs. single burn and repeated burn, ****repeated burn vs. single burn.

Perhaps not surprisingly, there was a significant negative correlation between 1000-h fuel mass and fire temperatures at the surface (Table 3).

Fuel consumption (change in fuel mass between pre-burn and post-burn measurements), evaluated relative to that in fire-excluded treatments over the same period, was more strongly correlated with fire temperatures than fuel mass alone. Relative consumption of litter, 1-h, 10-h, 100-h, 1000-h, total woody mass, and total fuel mass were positively correlated with fire temperatures at all pyrometer positions (Table 4). Relative consumption of duff mass was not correlated with burn temperature despite the strong positive correlation between pre-burn duff mass and temperature; however the change in duff depth was correlated with burn temperatures. There were no significant correlations across all landscape and temperature pyrometer positions for duff mass. For duff depth, we found significant correlations for duff depth in sub-xeric landscape positions at all pyrometer locations, no significant correlations for intermediate landscape positions, and significant correlations for 20 and 40 cm pyrometer positions in sub-mesic landscape positions Relative consumption of litter WBS was negatively correlated to fire temperature at the surface.

4. Discussion

4.1. Effects of single and repeated fire

Our results demonstrate that litter and duff were the primary fuels consumed during each prescribed fire, a finding corroborated in the literature for this region (Boerner et al., 2009; Graham and McCarthy, 2006; Knoepp et al., 2009; Waldrop et al., 2016, 2010). We found no evidence that prescribed fire, conducted as management burns and

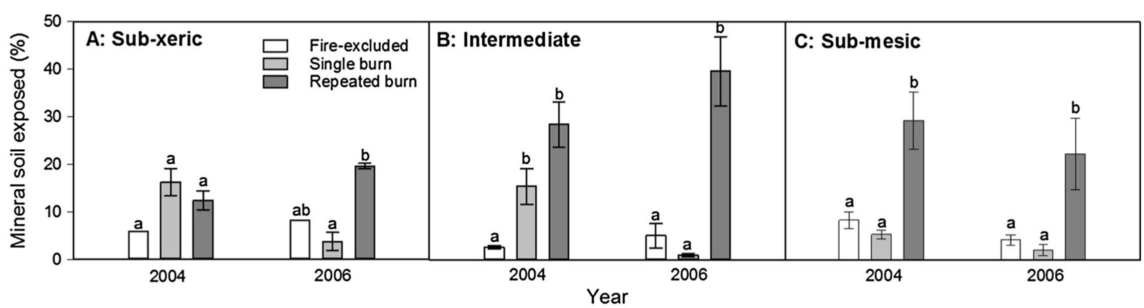

Fig. 6. Proportion of mineral soil exposed (%) as measured in all treatments on each landscape position in 2004 and 2006 in the Daniel Boone National Forest, Kentucky. The single burn was burned in 2003; the repeated burn was burned in 2003, 2004, 2006, and 2008. Different lower-case letters denote significant differences among treatments within a given year on that landscape position. Error bars represent ± 1 standard error of the mean.

Table 3
Correlation coefficients (and p-values in parentheses) for the correlations between pre-burn duff depth, the pre-burn mean mass by fuel component, and pre-burn total fuel mass with temperature measured using temperature-sensitive paints on aluminum tags placed on the surface of the forest floor and at 20 cm and 40 cm above the surface. Study sites were located in the Daniel Boone National Forest, Kentucky. Values in bold denote statistically significant p-values for the correlations shown.

Fuel component	Surface	20 cm	40 cm
Duff depth	**0.31 (0.0003)**	**0.19 (0.03)**	**0.30 (0.0004)**
Duff mass	**0.54 (< 0.0001)**	**0.36 (< 0.0001)**	**0.49 (< 0.0001)**
Litter	−0.07 (0.44)	0.08 (0.36)	−0.05 (0.57)
Litter WBS	−0.04 (0.63)	0.04 (0.66)	−0.05 (0.55)
1-h fuel	0.09 (0.27)	0.12 (0.16)	**0.19 (0.03)**
10-h fuel	−0.06 (0.48)	−0.02 (0.83)	0.02 (0.85)
100-h fuel	0.15 (0.09)	0.11 (0.22)	0.08 (0.36)
1000-h fuel	**−0.24 (0.004)**	−0.11 (0.19)	−0.14 (0.10)
Total woody fuel	−0.15 (0.07)	−0.09 (0.31)	−0.10 (0.25)
Total fuel mass	**0.18 (0.04)**	0.13 (0.13)	**0.19 (0.03)**

Table 4
Correlation coefficients (and p-values in parentheses) for the correlations between the mean temperature measured on the surface of the forest floor, at 20 cm and 40 cm above the surface, and the change in duff depth and the mass of fuel consumed, after accounting for the change in duff depth and fuel mass measured on fire-excluded treatments. Study sites were located in the Daniel Boone National Forest, Kentucky. Values in bold denote statistically significant p-values for the correlations shown.

Fuel component	Surface	20 cm	40 cm
Duff depth	**0.26 (0.003)**	**0.30 (0.0005)**	**0.31 (0.0003)**
Duff mass	0.05 (0.55)	0.05 (0.54)	0.13 (0.13)
Litter	**0.67 (< 0.0001)**	**0.56 (< 0.0001)**	**0.61 (< 0.0001)**
Litter WBS	**−0.21 (0.02)**	−0.10 (0.25)	−0.16 (0.064)
1-h fuel	**0.42 (< 0.0001)**	**0.38 (< 0.0001)**	**0.39 (< 0.0001)**
10-h fuel	**0.29 (0.0008)**	**0.18 (0.04)**	**0.26 (0.002)**
100-h fuel	**0.43 (< 0.0001)**	**0.24 (0.006)**	**0.31 (0.0004)**
1000-h fuel	**0.21 (0.02)**	**0.20 (0.02)**	0.12 (0.17)
Total woody fuel	**0.30 (0.0005)**	**0.24 (0.006)**	**0.25 (0.004)**
Total fuel mass	**0.27 (0.002)**	**0.21 (0.02)**	**0.26 (0.003)**

implemented within the constraints of current prescription parameters, significantly reduced the total mass of woody fuels.

Litter mass was highly variable throughout the study, reflecting loss to combustion followed by re-accumulation due to annual leaf fall. Litter mass differed among treatments following burning, recovered by 2006 and again by 2008, despite significant decreases in stem density and basal area with burning (Arthur et al., 2015). We observed significant alterations to fuel continuity in the repeated burn treatments, with greater fuel discontinuity on xeric and intermediate landscape positions where the overstory was dominated by oaks, compared to mesic landscape positions. Greater fuel discontinuity could be due to a combination of increased mineral soil exposure and the greater amount of litter redistribution of oak litter compared to other species, as noted by Boerner and Kooser (1989) in an Allegheny Plateau study site, combined with the steeper slopes found on xeric and intermediate landscape positions in this study. An inverse relationship between fuel continuity and fire frequency, as shown in forests of the Sierra Nevada (Miller and Urban, 2000), is likely operating here as well.

For the WBS litter mass, differences among treatments were more nuanced, but trended toward non-significant differences leading up to the 2008 burn in the repeated burn treatment, again suggesting recovery of this fuel class after each fire. In the absence of fire, WBS fluctuated, likely due to temporal variation in seed inputs and twig fall as well as weather-related effects, such as the region-wide ice storm in February 2003, suggesting that inputs to this fuel class are inherently variable. On the burn treatments, WBS mass also increased between pre-burn and post-burn measurements in 2003, though less so than on fire-excluded treatments, reflecting a near-balance between WBS mass consumed by the fires and new inputs from ice-damaged trees, leading to a significant difference among treatments. Starting with the pre-burn 2006 sampling, WBS in single burn treatments increased, likely due to wood and bark inputs from trees with delayed mortality (Arthur et al., 2015). In contrast, on the repeated burn treatments, except in 2003, each fire caused a reduction in WBS mass, which rebounded during the fire-free interval to levels that exceeded that on fire-excluded treatments.

Fire consumed a large proportion of the duff layer, and unsurprisingly, rebounding of duff mass and depth were much less pronounced following fire compared to that of the litter. Repeated fire had a persistent effect of further reducing the duff layer, so that by the end of the study, there were large (twofold) differences in duff mass on the single and repeated burn treatments. Between 2003 and 2008 there was a 76% decrease in duff mass on repeated burn treatments, compared to a 56% decrease on single burn treatments. Inexplicably, duff mass also decreased on fire-excluded treatments between 2003 and 2008, by 36%, suggesting limitations to sampling accuracy due to high spatial variation in organic matter mass (Yanai et al., 2000). Duff depth declined by 60% over the course of the study (pre-burn 2003 to pre-burn 2008) on repeated burns, 56% on single burn treatments, and only 11% on fire-excluded treatments. Other studies in the region have also found significant reductions in duff mass with fire (Boerner et al., 2009; Knoepp et al., 2009; Waldrop et al., 2010).

It is difficult to evaluate the ecological importance of a reduction in this essential soil layer. On the one hand, the duff serves several important functions, including water absorption, protection of the top layer of mineral soil from erosion, and serves as a pool of nutrients (Certini, 2005). Thus, significant loss of this layer with repeated fire could lead to reduced site productivity. On the other hand, Boerner et al. (2008) postulated that returning fire to sites that were burned consistently by Native Americans in the past could potentially return historically nitrogen (N)-limited forests impacted by chronic N deposition to lower N status. Fire management in this region often targets sustainability of upland oak forests (Arthur et al., 2012), which are known to form belowground relationships with ectomycorrhizal fungi. Unfortunately, there are insufficient data available across the region to evaluate which of these potential scenarios are most relevant for these ecosystems, though a recent study by Oliver et al. (2015) demonstrated that fire-adapted soil fungal communities differ from those on unburned sites, and may therefore support fire-adapted plant communities.

Through time, repeated fire had a more negative impact on duff mass than a single fire, but pre-fire differences in mass and depth of the duff layer led to some important differences in fire effects on the landscape. For example, although the percent loss of duff mass and depth after repeated burning was greater on sub-xeric and intermediate sites between measurements made in 2003 and 2008 (74 and 79%, respectively) compared to sub-mesic sites (67%), there was less mineral soil exposure on sub-xeric sites compared to intermediate and sub-mesic sites, presumably because of the larger mass of duff prior to burning. Thus, despite the greater mass of duff consumed by fire on sub-xeric landscape positions, the greater depth and mass of the duff layer provided a buffer against mineral soil exposure. We also found that, despite differences in fuel mass across landscape positions, there were no parallel differences in burn temperatures (Arthur et al., 2015). We did find significantly greater char height on sub-xeric and intermediate landscape positions compared to sub-mesic (Arthur et al., 2015), suggesting that the greater consumption of duff leads to greater burn intensity through longer residence time of fire smoldering within slow-burning fuels.

Burning significantly increased mineral soil exposure, and the effect was much greater for repeated burn treatments. We observed greater discontinuity in litter (personal observations) after burning which, coupled with the decline in duff mass and depth with burning, likely led to increases in mineral exposure which were far more pronounced on repeated burns (25%) compared to single burn treatments (2%), even after only two burns compared to one burn, respectively. Boerner et al. (2009) also found that fire increased mineral soil exposure in two eastern forests in the Fire and Fire Surrogate Study (the central Appalachian Plateau of Ohio and the southern Appalachian Mountains in North Carolina) two to four years after burning. Unfortunately, a reduction in funding prevented us from measuring mineral soil exposure after 2006, so we have no measure of recovery of this essential soil layer, or of further impact that may have occurred with additional burns. Even so, coupled with our results demonstrating continued reduction of the duff layer with repeated burning, it is likely that there was no recovery of mineral soil exposure over the course of this study.

Regarding woody fuels, our findings largely mirror those of other studies in the region: litter and duff are the primary fuels for prescribed fires in this region, with small changes in 1-h and 10-h fuels, and no measureable reduction in 100-h and 1000-h fuels (Graham and McCarthy, 2006; Loucks et al., 2008; Waldrop et al., 2010). Fire reduced 1-h fuels, but the losses were non-significant and quickly replaced. Repeated burning, however, reduced fine woody fuels after three burns, when repeated burn treatments had lower fine fuels compared to single burn and fire-excluded treatments, which were similar to each other. This finding corroborates that of Graham and McCarthy (2006) who found that any reductions to smaller fuel classes quickly recovered. In this study, we measured greater mass of 10-h fuels in sites burned just once compared to fire-excluded and repeated burn treatments. Ongoing tree mortality (Arthur et al., 2015), perhaps coupled with shedding of branches on fire-damaged trees, likely contributed to this increase in woody fuels in this size class above that measured in the fire-excluded treatments. Repeated burning may have kept fuels of this size class at a lower mass despite continued inputs. The lack of detectable prescribed fire effects on large woody fuels (100–1000 h fuels), which corroborates the findings of Loucks et al. (2008) following the first fire, suggests that burn prescription parameters dictated by contemporary social and management constraints are insufficient for reducing coarse woody fuel consumption. Relaxation of burn prescription parameters to allow for burning during dryer conditions would likely lead to measurable reductions in large woody fuels.

4.2. Fuel loading compared to region

Woody fuel loading is fairly similar within regions, as noted by Stambaugh et al. (2011) for deciduous forests in the central U.S. We found this to be mostly the case for our study region as well, relative to fuel loading reported across the eastern deciduous forest region. For example, mean total woody fuel mass of 17.9 Mg ha^{-1} measured in this study was very similar to the amount (~20.5 Mg ha^{-1}) measured by Waldrop et al. (2010) in the southern Appalachian Mountains site of the Fire and Fire Surrogate Study (FFS) in North Carolina. However, the amount of woody fuel mass measured in this study was much lower than that measured in the Ohio Hills Region sites of the same FFS study (~37 Mg ha^{-1}), though in that study 1-h fuels measured prior to fire or thinning treatment were exceptionally high, accounting for ~40% of woody fuels (14–17 Mg ha^{-1}; Graham and McCarthy, 2006), compared to ~3% in this study (0.5 Mg ha^{-1}) and ~2% in the Waldrop et al. (2010; 0.3–0.4 Mg ha^{-1}) study. Comparing woody fuels between the Ohio site and our study excluding 1-h fuels, fuel mass was very similar between our sites (17.3 Mg ha^{-1}) and the Ohio sites (20.6 Mg ha^{-1}). Woody fuel mass measured in this study was higher than that reported by Stambaugh et al. (2011) for central US deciduous forest sites in Missouri, Illinois, and Indiana (9.8 Mg ha^{-1}), and considerably lower than that measured by Stottlemyer et al. (2009) in the Blue Ridge Mountain Province of South Carolina (38–77 Mg ha^{-1}, varying from xeric to mesic landscape positions, respectively).

We might expect fuel accumulation to generally mirror basal area, which is an imperfect measure of time since disturbance and productivity. In this study, pre-treatment basal area varied from 23.7 $m^2\,ha^{-1}$ in sub-xeric landscape positions to 26.7 $m^2\,ha^{-1}$ in intermediate and sub-mesic landscape positions (Arthur et al., 2015). By comparison, basal area was similar in both the southern Appalachian FFS study (24.6–31.8 $m^2\,ha^{-1}$; Phillips and Waldrop, 2008) and the Ohio Hills FFS study (25–28 $m^2\,ha^{-1}$) prior to treatment, and with the exception of the unusually large mass of 1-h fuels in the Ohio Hills sites (Graham and McCarthy, 2006), we found similar amounts of fuel accumulation as in those studies. In contrast, the sites in the central US deciduous forest region examined by Stambaugh et al. (2011) ranged in basal area from approximately 11–27 $m^2\,ha^{-1}$ (Michael Stambaugh, email communication, May 26, 2016), on average much lower basal area than was measured in this study, and likely a partial explanation for the lower mass of woody fuels across that region. Larger fuel masses in the Stottlemyer et al. (2009) study might be because the study was designed to remove disturbance-related fuel variation. One of the challenges to modeling fuel accumulation across the landscape is the potentially confounding relationships between variability in stand age and disturbance history and the topographic and geographic variability in site productivity and decomposition environment (Harmon et al., 1986; Stottlemyer et al., 2009; Waldrop et al., 2004).

Prior to burning, we found that the depth of the duff layer was greatest on sub-xeric compared to intermediate and sub-mesic landscape positions (Fig. 1). Mass of duff on sub-xeric sites was also higher than sub-mesic landscape positions. These findings are similar to measurements made by Waldrop et al. (2010). Given that sub-xeric landscape positions in this study site had lower basal area compared to more mesic landscape positions (Arthur et al., 2015), this difference in the depth and mass of the organic horizon is likely the result of biotic and abiotic conditions rather than differences in litterfall, which, all other things being equal, should lead to lower organic mass and depth on drier landscape positions due to lower leaf area associated with lower basal area. The drier conditions found on sub-xeric and intermediate landscape positions would be expected to slow decomposition rates. Differences in species composition among landscape positions could further influence decomposition rates and depth of the organic horizon. On these sites, relative basal area of oaks ≥10 cm DBH varied from 76% to 66% to 47% on sub-xeric, intermediate, and sub-mesic landscape positions, respectively. Conversely, relative basal area of mesophytic species, such as *Acer* spp. and *Liriodendron tulipifera* L., increased relative to oaks from sub-xeric to sub-mesic landscape positions (Arthur et al., 2015). The shifting species composition is a reflection of microclimatic and soil moisture differences, but also signals a shift in litter quality. In general, oaks have greater lignin concentration compared to maples (Alexander and Arthur, 2014; Lovett et al., 2015), contributing to slower decomposition rates and greater organic horizon accumulation on sub-xeric sites compared to sub-mesic landscape positions when coupled with variation in soil moisture availability. While impossible to separate or quantify in this study, the combined effects of shifting species composition with varying litter quality, coupled with greater rates of decomposition on more-productive, moister sites (Mudrick et al., 1994), likely led to the greater mass and thickness of the recalcitrant soil organic matter layer on sub-xeric compared to sub-mesic landscape positions.

5. Conclusions and management implications

The use of prescribed fire is increasing as a management tool in the central and southern Appalachian region in response to a greater understanding of the prehistoric role of fire in the region, shifting species composition with increased dominance by mesophytic species on sites long dominated by oaks, and, sometimes, as an effort to reduce fuel loads across the landscape. These goals for fire management contributed to the motivation for this research.

In response to the specific questions that motivated this fuels study, we found that:

(1) Prescribed fires in the region primarily consume leaf litter and the duff horizon, with surprisingly few differences in fire effects among landscape positions, a somewhat surprising result given the landscape scale differences in fire effects on char height, crown vigor and tree mortality (Arthur et al., 2015). Repeated fire led to larger differences in fuel mass than a single fire: sites burned only once accumulated woody fuels whereas repeated fire reduced 1-h and 10-h fuels. While there were no significant differences among treatments for larger fuels, fuel mass tended to be higher in burn treatments compared to fire-excluded treatments, perhaps reflecting the continued tree mortality in these sites (Arthur et al., 2015). Landscape position was relevant primarily in terms of the accumulation of the duff layer, with thicker horizons of recalcitrant organic matter accumulating on sub-xeric landscape positions compared to intermediate and sub-mesic positions.
(2) Fire, and especially repeated fire, increased mineral soil exposure. We postulate that recovery of the duff layer may be fairly slow on sites that have lost significant organic matter cover, as pre-fire duff depth and mass accumulated during a long fire-free period. This is an important information need for this region.

This study provides evidence that repeated fires of low to moderate intensity conducted on a highly-dissected topographic landscape can significantly reduce soil organic matter and increase exposed mineral soil. These fire effects may alter forest ecosystem components that are rarely measured following prescribed fire in this region, including the soil community and soil physical and chemical structure; this finding suggests the need for inclusion of measurements of the effects of fire on the soil biotic community and feedbacks to tree regeneration. Indeed, these alterations to forest community structure and function may signal the more ecologically significant roles of fire in this landscape, and suggest the need for greater attention to fire impacts on belowground biodiversity.

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