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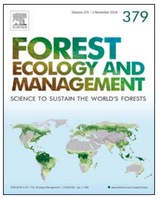

Reptile and amphibian response to season of burn in an upland hardwood forest

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ABSTRACT

Growing-season burns are increasingly used in upland hardwood forest for multiple forest management goals. Many species of reptiles and amphibians are ground-dwelling, potentially increasing their vulnerability to prescribed fire, especially during the growing-season when they are most active. We used drift fences with pitfall traps to experimentally assess how herpetofaunal species and communities responded to early, growing-season burns, dormant-season burns, and unburned controls. We documented no adverse effects of either growing-season burns or dormant-season burns on any common herpetofaunal taxa, but capture rates of total, adult, and juvenile five-lined skinks (*Plestiodon fasciatus*) were greater following growing-season burns. Most measurements reflected little or transient change in forest structure. However, canopy cover decreased by an average of 16% in growing-season burns within four growing-seasons of burning, with some tree mortality in patches where fire temperature likely was hotter. Our study suggests that even modest reductions in canopy cover may positively affect relative abundance and reproductive success of *P. fasciatus*. We cautiously suggest that a higher mean ground-level fire temperature and the physiologically active condition of vegetation in growing-season burns interacted to damage a greater proportion of trees, resulting in more canopy thinning than in dormant-season burns. However, weather, fuel types and condition, vegetation structure, and topography interact to affect fire intensity and the level of mortality or damage to canopy trees within and among stands, regardless of season conducted. We suggest that herpetofaunal response, for the species we studied, is more closely linked to change in canopy cover than to season of burn *per se*.

1. Introduction

Prescribed burning, a common management tool in eastern deciduous forests, is most commonly conducted during dormant-season due to drier fuels and more predictable environmental conditions (Sparks et al., 2002). As burning has become a preferred tool in restoration of disturbance-adapted ecosystems, forest managers are increasingly widening the ‘burn window’ to include growing-season burns. However, little is known about how growing-season burns impact the wide diversity of wildlife taxa. Many species of reptiles and amphibians are ground-dwelling, potentially increasing their vulnerability to prescribed fire, especially during the growing-season when they are most active. Land managers need to know how season of burn affects herpetofauna to inform and direct wildlife conservation in conjunction with ecosystem restoration, or other forest management objectives.

In the southern Appalachians, many terrestrial reptile and amphibian species retreat belowground and become dormant during winter, with aboveground activity occurring primarily during the warmer spring and summer months (Camp, 1988; Fitch and von Achen, 1977; Greenberg, *unpubl. data*). Increased surface activity during spring and summer months could potentially increase amphibian and reptile vulnerability to growing-season prescribed burns. In addition, most herpetofaunal species have small home ranges and poor dispersal abilities, potentially limiting their ability to evade fire. For example, the home range of Plethodontid salamanders in the southern Appalachians is estimated to be $\leq 14.5\,m^2$ (Merchant, 1972). Whereas box turtles (Howey and Roosenberg, 2013; Melvin, 2017) and some snakes (Beaupre and Douglas, 2012) may be vulnerable to fire-caused injury or mortality, many herpetofaunal species may exhibit adaptive behaviors in response to fire that could minimize mortality (O’Donnell et al.,

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2016; Pitt et al., 2013). Although difficult to gauge and poorly studied, evidence suggests that direct mortality from fire is rare, and unlikely to affect most reptile and amphibian species at the population level (Harper et al., 2016).

In contrast, fire-caused change to forest structure may indirectly influence relative abundance of some reptile and amphibian species by altering microhabitats such as availability of leaf litter or coarse woody debris, or microclimatic conditions such as light, moisture, and temperature at the forest floor. Research shows that low-intensity dormant-season prescribed burns in upland hardwood forest generally have a negligible impact on herpetofauna (e.g., Greenberg et al., 2016, 2017; O'Donnell et al., 2015; Perry et al., 2009; Raybuck et al., 2015), whereas high-severity dormant-season burns that kill overstory trees result in greater relative abundance of lizards (Greenberg et al., 2017; Fouts et al., 2017). Because of seasonal differences in fuel condition, air and fire temperature, and physiological activity of vegetation, dormant-season and growing-season burns are likely to differ in their effects on forest structure, and thereby indirectly affect relative abundance of some herpetofaunal species.

Reptiles and amphibians are important components of biological diversity, and play important ecological roles as predator and prey (Greenberg et al., 2017). In general, terrestrial reptiles and amphibians differ considerably in their microhabitat requirements due to phylogenetic and physiological differences (Moorman et al., 2011). Reptiles have dry scaly skin, require warmer temperatures associated with higher light levels for thermoregulation and egg incubation, and lay their eggs on land (Moorman et al., 2011). In contrast, most amphibians have moist, permeable skin that increases their susceptibility to desiccation; they require moist microenvironments, and water bodies for egg deposition (Moorman et al., 2011). These physiological distinctions between reptiles and amphibians likely influence their response to altered forest structure and microclimate after fire or other disturbances (Moorman et al., 2011). Despite their importance, relatively little is known about how season of burn, and especially growing-season burns, affect herpetofauna in upland hardwood forests. To the best of our knowledge, our study is the first to experimentally compare season of burn effects on relative abundance of reptile and amphibian species, as measured by surface activity.

We used a replicated, completely randomized design with repeated measures to experimentally assess how herpetofaunal species and communities responded to early growing-season burns (GSB), dormant-season burns (DSB), and controls (C). Our objective was to determine if, and how, species richness or capture rate of common reptile and amphibian taxa differed among the season of burn treatments.

2. Methods

2.1. Study area

Our study was conducted at the Bent Creek Experimental Forest, a part of the Pisgah National Forest in Buncombe County, North Carolina (Fig. 1). Located within the Southern Appalachian region of western North Carolina, the area encompasses 2500 ha with annual precipitation averaging 140 cm (Owenby and Ezell, 1992) and elevations ranging from 700 m to 1070 m (McNab et al., 2004). Monthly average temperatures range −4.2° to 8.6 °C in January, to 16.0° to 28.9 °C in July (Owenby and Ezell, 1992). Common tree species in this upland hardwood forest site include black oak (*Quercus velutina*), chestnut oak (*Q. montana*), scarlet oak (*Q. coccinea*), white oak (*Q. alba*), sourwood (*Oxydendrum arboreum*), red maple (*A. rubrum*), dogwood (*Cornus florida*), and interspersed shortleaf pine (*Pinus echinata*) (McNab et al., 2004).

2.2. Study design

We established nine, approximately 5 ha (range 3.5–7.0) units (two treatments and control, three replicates each) within the Bent Creek Experimental Forest (Fig. 1). Units were separated by fires lines as needed. All units were comprised of mature (> 70 years old), oak-dominated closed canopy stands, and portions of each were within 500 m of perennial streams. Randomly assigned treatments were: (1) growing-season prescribed burn (GSB), (2) dormant-season prescribed burn (DSB), and (3) control (C). We defined growing-season based on vegetation phenology, including the presence of new, small leaves on several deciduous tree species such as white oak, dogwood, yellow poplar (*Liriodendron tulipifera*), and red maple, and full flowering by dogwood and several oak species. We defined dormant-season as the absence of live leaves on any deciduous tree species. We conducted the three growing-season burns on 26 April 2013, and the three dormant-season prescribed burns on 5 March 2014.

2.3. Forest structure and fire temperature measurements

We tagged all live overstory trees ≥25 cm diameter at breast height (dbh) within three, 0.05 ha, randomly located, permanent circular vegetation plots separated by > 30 m, within each treatment unit; we also tagged live midstory trees ≥5 cm and < 25 cm dbh within a 0.01 ha subplot concentrically nested within the 0.05 ha plot. We recorded status (live or dead) of all tagged trees each year (2013–2016) to assess treatment effects on live tree density. We measured additional forest structure variables in GSB and C during summer 2013, and in DSB, GSB, and C in summers 2014, 2015, and 2016. We measured percent cover of shrubs (woody understory) and leaf litter depth along each of four 15 m transect lines. Transects originated from a center line bisecting each unit, at a randomly chosen distance along the center line, oriented in a randomly selected perpendicular direction out. We recorded 'start' and 'stop' distance for shrubs along each transect, summed the total distance, and divided the sum by the transect length to obtain percent cover. We used average percent cover across all transects within treatment units for data analyses. We measured litter depth at 7.5 m and 15 m along each transect line. We used a spherical densiometer at the center bucket of each trapping array in each unit (see Section 2.4) to measure percent canopy cover. We measured fire temperatures at ground level using temperature-sensitive paints on tags placed at two locations, 8 m apart, in each of the three vegetation plots spaced throughout each GSB and DSB unit.

2.4. Herpetofaunal sampling

We installed two drift fence arrays in each GSB and C unit in 2013. Arrays were located at least 35 m from unit boundaries and ≥75 m apart using a randomly chosen direction and distance (0–50 m) perpendicular to a center transect bisecting each unit. In 2014, a third array was added to each GSB and C unit, and three arrays were established in DSB (Fig. 1). We determined the location of the third array in each unit using a randomly chosen azimuth from the center bucket of each second array, and set 75 m apart. We constructed arrays with three 7.6 m, 50 cm high sections of aluminum flashing positioned at approximately 120° angles (in a "Y" configuration), with one, 19-l bucket buried at the center, and at the end of each "arm," for a total of four pitfall traps per array. We placed a double-ended funnel trap, constructed from aluminum screening, along both sides of each arm for six funnel traps total per array. We drilled holes in the bottoms of pitfalls to prevent flooding. We shaded all traps with a small board, and placed a sponge in pitfall traps that was moistened as needed to provide cover and humidity for captured animals; we additionally placed a small piece of styrofoam in buckets that were frequently flooded, for flotation.

We identified, weighed, measured (snout-vent and total length), and sexed (when possible) all reptiles and amphibians. We marked animals by year and treatment by toe-clips (lizards, frogs, and salamanders), scale-clips (snakes), or scute-notching (turtles). We trapped during late

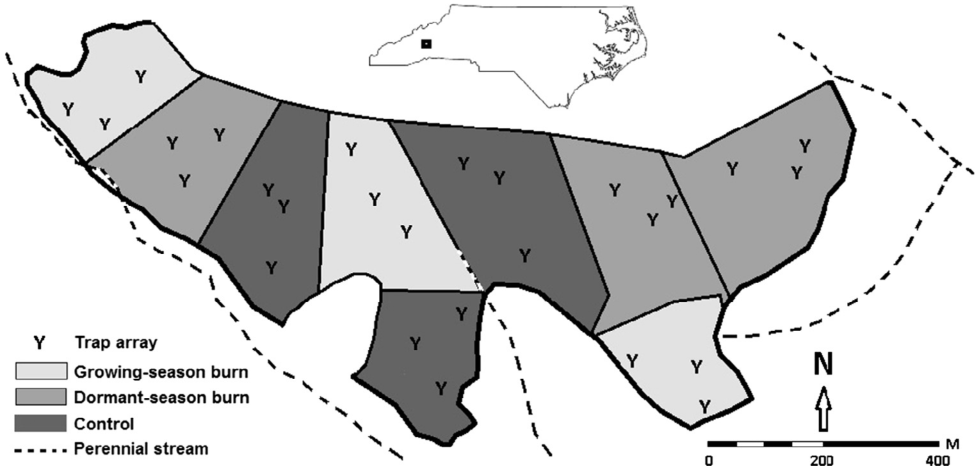

Fig. 1. Map of study sites and locations of herpetofaunal trapping arrays, Bent Creek Experimental Forest, Pisgah National Forest, Buncombe County, NC.

spring and summer each year, when herpetofauna are most active aboveground (Greenberg *unpubl. data*). We opened drift fence arrays continuously and concurrently in C and GSB from June 4 to August 9 in 2013, and in C, DSB, and GSB from May 27 to August 14 in 2014; May 19 to July 31 in 2015, and; May 18 to August 8 in 2016, for a total of 7110 “array nights” (number of arrays x the number of nights operational) across all units and years.

2.5. Statistical analyses

We used repeated measures general mixed model ANOVAs (Proc Mixed; SAS 9.3) in a completely randomized design with compound symmetry covariance structure to examine effects of treatment, year, and treatment x year interactions for all analyses of forest structure variables and herpetofauna. In all repeated measures ANOVAs, we considered treatment, year, and their interaction as fixed effects, and unit within treatment as a random effect and the repeated subject factor. Where significant ($p \leq .05$) treatment x year interactions were present, we used the least square means for partitioned *F*-tests (SLICE option) in PROC MIXED (SAS 9.3) to examine the significance of treatments within years, and years within treatments. Least squares means pairwise comparisons that included 2013 data were non-estimable because of missing 2013 data for DSB (DSB was not sampled until 2014, after treatment implementation). Therefore, when mixed model main effects indicated that treatments differed overall, with or without an interaction effect, we used least squares means comparisons within years to determine which treatments differed ($p \leq .05$). Shrub and canopy cover percentage data were arcsine square root transformed for analyses.

We standardized all herpetofaunal capture data for small differences in trapping effort, by using first-captures per 100 array nights. Response variables analyzed were species richness of all herpetofauna, amphibians, and reptiles, and relative capture rate of all reptiles, amphibians, frogs, salamanders, lizards, snakes, and individual species that were sufficiently common ($n \geq 45$ captures). We also examined relative capture rate of adults and juveniles separately for species with ≥ 45 captures of both adults and juveniles, including the southern Appalachian salamander (*P. teyahalee*) ($J \leq 58$ mm SV; Homyack and Haas, 2009), American toad ($J \leq 41$ mm; *Anaxyrus americanus*), and five-lined skink, (*Plestiodon fasciatus*) ($J \leq 52$ mm SV; Vitt and Cooper, 1986).

We were unable to accurately calculate detection probability due to low (< 2.5%) recapture rates. Further, herpetofaunal species likely differ in their detectability. For example, *Plethodon* salamanders spend most of their time underground or under cover objects (O’Donnell et al., 2014; O’Donnell and Semlitsch, 2015; O’Donnell et al., 2016), whereas lizards, such as *P. fasciatus,* spend more time aboveground except during hibernation (Fitch and von Achen, 1977). Therefore, we considered our capture data to be a measure of relative surface activity, rather than relative abundance *per se*.

3. Results

3.1. Prescribed burns and forest structure

In DSB, an average of 66% of subplots burned (≥ 79.4 °C) per unit; in GSB, an average of 78% of subplots burned per unit. Based on burned subplots, mean ground-level fire temperature was significantly lower in DSB (172.1 ± 1.2 °C; range 135.0°–197.5 °C per unit) than in GSB (277.0° ± 6.5 °C; range 246.1°–324.8 °C per unit) (*t*-test; $t(4) = -3.41$, $p = 0.0270$). Overstory live tree density did not differ among treatments or years, and no treatment x year interaction effect was detected; midstory tree density decreased each year, but no treatment or treatment x year interaction effects were detected (Table 1). Average (± SE) overstory density was 161.1 ± 10.3/ha, and midstory density was 721.0 ± 69.8/ha, across all treatments and years. Leaf litter depth differed among years; no treatment or treatment x year effects were detected (Table 1), but a trend of reduced depth immediately following both the growing-season (2013) and dormant-season (2014) burns was apparent (Fig. 2). Percent shrub cover did not differ among treatments, but differed among years, and a treatment x year interaction effect was detected (Table 1). Shrub cover was dynamic over time in DSB and GSB. In DSB, shrub cover was lower immediately after burns (2014) than in 2016. In GSB, shrub cover was lower immediately after burns (2013) than in subsequent years, and lower in 2014 than in 2016 (Fig. 2). Shrub cover did not differ among treatments

Table 1
Results of general mixed model ANOVA comparing midstory (≥ 5 cm and < 25 cm dbh) and overstory (≥ 25 cm dbh) live tree density, and leaf litter depth, shrub cover (%) and canopy cover (%) by treatment, year, and treatment x year interaction effects. Treatments were: growing-season burn (April 26, 2013; GSB); dormant-season burn (March 5, 2014; DSB), and; unburned control (C). Tree measurements were made in GSB, DSB, and C in 2011 (pretreatment), 2013 (after GSB implemented), 2014 (after DSB implemented), and 2015; other forest structure variables were measured in GSB and C units in 2013, and in all treatments 2014–2016, Bent Creek Experimental Forest, Pisgah National Forest, Buncombe County, NC.

Forest structure variable	ANOVA results		
	P_{trt}	P_{yr}	P_{trtXyr}
Overstory tree density (ha)	0.5164	0.4932	0.5766
Midstory tree density (ha)	0.5145	0.0039	0.6601
Leaf litter depth (cm)	0.0704	0.0326	0.0780
Canopy openness (%)	0.0240	< 0.0001	0.0300
Shrub cover (%)	0.2559	< 0.0001	0.0295

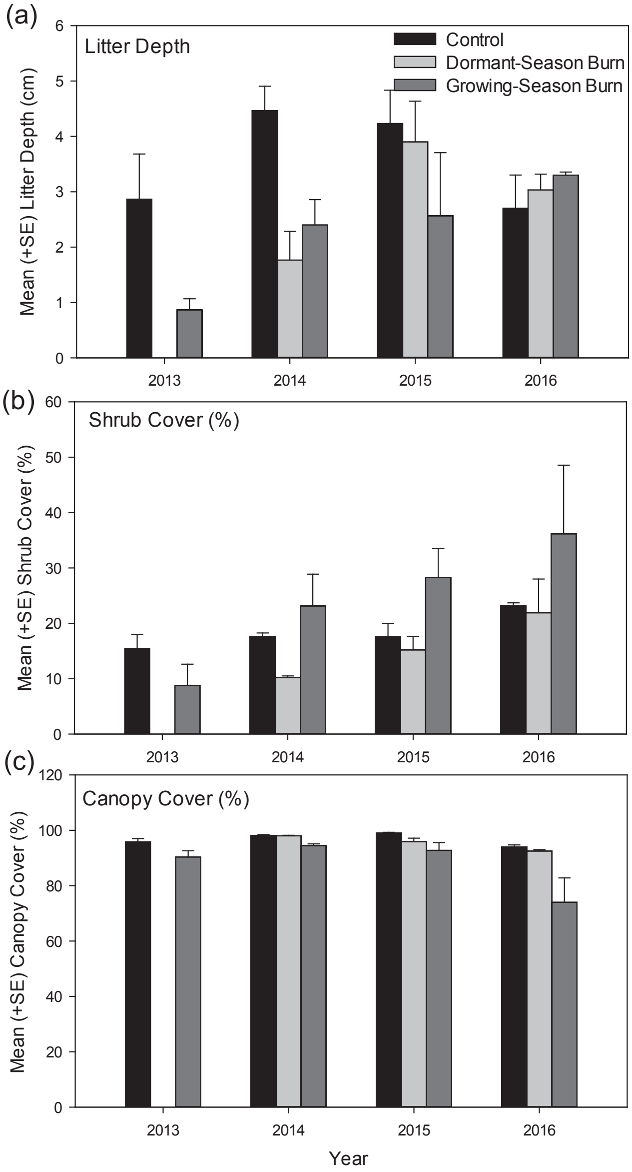

Fig. 2. Mean (± SE) (a) leaf litter depth; (b) percent shrub cover, and; (c) percent canopy cover in early growing-season burn (April 26, 2013; GSB), dormant-season burn (March 5, 2014; DSB), and unburned controls (C), 2013–2016. GSB and C units were sampled 2013–2016; DSB units were sampled 2014–2016, Bent Creek Experimental Forest, Pisgah National Forest, Buncombe County, NC.

within any year. Percent canopy cover differed among treatments and years, and a treatment x year interaction effect was detected (Table 1). Changes in canopy cover in C and DSB were small relative to GSB (Fig. 2). Within C, canopy cover was lower in 2013 than in 2015; in DSB, it was greater immediately after the burns (2014) than in 2016. Within GSB, canopy cover was greater in immediately after the burns (2013) than in 2016, and greater in 2014 and 2015 than in 2016. In 2015, canopy cover was greater in C than in GSB; in 2016, canopy cover was greater in both C and DSB than in GSB (Fig. 2).

3.2. Herpetofauna

We captured 489 individuals (six recaptures) of 12 amphibian species, and 192 individuals (11 recaptures) of ten reptile species in 7110 array nights during the four years sampled (2013–2016) (Table 2). *Anaxyrus americanus* (33%), *P. teyahalee* (37%), and *Pseudotriton ruber* (12%) dominated total amphibian captures; *P. fasciatus*

Table 2
Total number of first-captures (and recaptures; not used in analyses) of reptiles and amphibians (all years and units combined; 834 array nights) by species, class, order, and total, and results of mixed-model ANOVA comparing treatment, year, and treatment x year interaction effects on captures per 100 array-nights (AN), and species richness. Treatments were: growing-season burn (April 26, 2013; GSB); dormant-season burn (March 5, 2014; DSB), and; unburned control (C). GSB and C units were sampled 2013–2016; DSB units were sampled 2014–2016, Bent Creek Experimental Forest, Pisgah National Forest, Buncombe County, NC.

	ANOVA results			
	Total	P_{trt}	P_{yr}	P_{trtXyr}
Amphibians	489 (6)	0.2715	0.0090	0.1751
Frogs and toads, Anura	195 (0)	0.0249	0.0003	0.7078
American toad, *Anaxyrus americanus*	162 (0)	0.2607	0.0002	0.7867
Fowlers toad (*A. woodhousii*)	3 (0)	–	–	–
American bullfrog, *Lithobates catesbeianus*	1 (0)	–	–	–
Gray treefrog, *Hyla versicolor chrysoscelis*	1 (0)	–	–	–
Green frog, *Lithobates clamitans*	2 (0)	–	–	–
Pickerel frog, *Lithobates palustris*	1 (0)	–	–	–
Wood frog, *Lithobates sylvaticus*	26 (0)	–	–	–
Salamanders, Caudata	294 (6)	0.6581	0.0843	0.0500
Blue Ridge two-lined salamander, *Eurycea wilderae*	18 (0)	–	–	–
Eastern newt, *Nothophthalmus viridescens*	32 (1)	–	–	–
Southern Appalachian salamander, *Plethodon teyahalee*[a]	182 (3)	0.4991	0.1892	0.1314
Red salamander, *Pseudotriton ruber*	60 (2)	0.7772	0.7505	0.1709
Allegheny Mountain dusky salamander, *Desmognathus ochrophaeus*	1 (0)	–	–	–
Reptiles	192 (11)	0.0815	0.0016	0.0074
Lizards, Lacertilia	139 (8)	0.0885	0.0386	0.1711
Broad-headed skink, *Plestiodon laticeps*	1 (0)	–	–	–
Five-lined skink, *Plestiodon fasciatus*	104 (6)	0.0317	0.0040	0.0051
Eastern fence lizard, *Sceloporous undulatus*	34 (2)	–	–	–
Snakes, Serpentes	46 (1)	0.2183	0.0741	0.1291
Copperhead, *Agkistrodon contortix*	4 (0)	–	–	–
Eastern worm snake, *Carphophis amoenus*	22 (0)	–	–	–
Ring-necked snake, *Diadophis punctatus*	13 (1)	–	–	–
Eastern rat snake, *Pantherophis alleghaniensis*	2 (0)	–	–	–
Common garter snake, *Thamnophis sirtalis*	5 (0)	–	–	–
Turtles, Testudinides	7 (2)	–	–	–
Snapping turtle, *Chelydra serpentine*	1 (1)	–	–	–
Eastern box turtle, *Terrapene carolina*	6 (1)	–	–	–
Total Amphibian Richness	12	0.3289	0.3078	0.3279
Total Reptile Richness	10	0.1216	0.1410	0.1354
Total Richness	22	0.0483	0.1048	0.4070

[a] Northern slimy salamander (*P. glutinosis*) complex (Highton and Peabody, 2000).

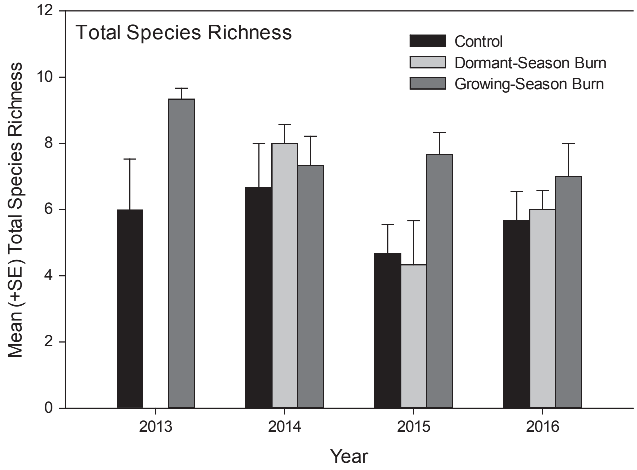

Fig. 3. Mean (± SE) total herpetofaunal species richness in early growing-season burn (April 26, 2013; GSB), dormant-season burn (March 5, 2014; DSB), and unburned controls (C), 2013–2016. GSB and C units were sampled 2013–2016; DSB units were sampled 2014–2016, Bent Creek Experimental Forest, Pisgah National Forest, Buncombe County, NC.

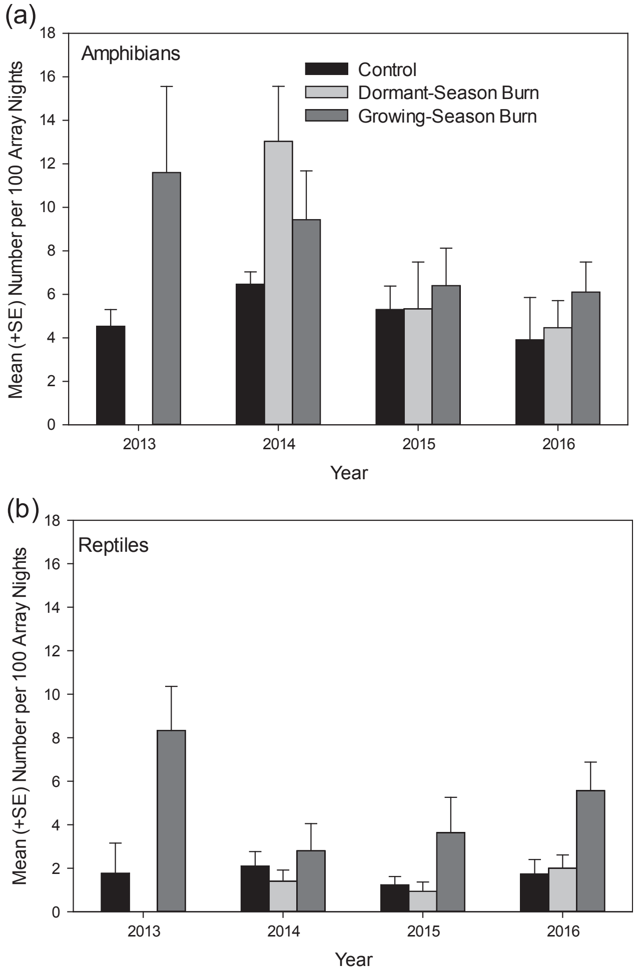

Fig. 4. Mean (± SE) number of first-captured (a) amphibians; and (b) reptiles per 100 array nights in early growing-season burn (April 26, 2013; GSB), dormant-season burn (March 5, 2014; DSB), and unburned controls (C), 2013–2016. GSB and C units were sampled 2013–2016; DSB units were sampled 2014–2016, Bent Creek Experimental Forest, Pisgah National Forest, Buncombe County, NC.

(54%) and *Sceloporus undulatus* (18%) dominated reptile captures. Total herpetofaunal species richness differed among treatments but not among years, and no interaction effects were detected (Table 2; Fig. 3). Least squares means tests indicated that total species richness was higher in GSB than in C in 2013, and higher in GSB than in C and DSB in 2015. Species richness of amphibians or reptiles did not differ among treatments or years, and no treatment x year interaction effects were detected (Table 2).

Capture rate of amphibians differed among years but not among treatments, and no treatment x year interaction was detected (Table 2; Fig. 4). Anuran capture rate differed among treatments and years, but no interaction effect was detected (Table 2; Fig. 5). Anuran captures differed among treatments only in 2014, when fewer were captured in C than in DSB or GSB. Salamander capture rate differed among years but not treatments, and a treatment x year interaction was detected (Table 2; Fig. 5). Salamander capture rate was dynamic over time in DSB and GSB. Within both DSB and GSB, salamander capture rate was greater immediately post-treatment (2013 for GSB; 2014 for DSB) than in subsequent years. Capture rates of *A. americanus* differed among years but not treatments, and no interaction effect was detected (Table 2; Fig. 6). Capture rate of adult *A. americanus* differed among years but not among treatments, and a treatment x year interaction effect was detected (Table 3; Fig. 7). Adult capture rates were dynamic over time in C and DSB; fewer were captured in 2014 than in 2015 in C, and more were captured in 2014 than in 2015 or 2016 in DSB. In 2014, fewer adults were captured in C than in DSB. Capture rate of juvenile *A. americanus* differed among years but not among treatments, and no treatment x year interaction effect was detected (Table 3; Fig. 7). Capture rate of total (Table 2; Fig. 6), adult, or juvenile (Table 3; Fig. 7) *P. teyahalee* did not differ among treatments or years, and no interaction effects were detected. Capture rate of red salamanders (*Pseudotriton ruber*) did not differ among treatments or years, and no interaction effect was detected (Table 2).

Capture rate of reptiles differed among years, but not among treatments; a treatment x year interaction effect was detected (Table 2; Fig. 4). Reptile capture rate was dynamic over time in GSB, with a higher capture rate immediately post-burn (2013) than in subsequent years, and higher capture rate in 2016 than in 2014 or 2015. In 2013, reptile capture rate was greater in GSB than in C. Lizard capture rate differed among years, but not among treatments, and no treatment x year interaction effect was detected (Table 2; Fig. 5). Capture rate of snakes did not differ among treatments or years, and no interaction effect was detected (Table 2; Fig. 5). Capture rate of total *P. fasciatus* differed among treatments and years, and an interaction effect was detected (Table 2; Fig. 6). In GSB, *P. fasciatus* capture rate was greater immediately post-treatment (2013) and in 2016 than in 2014 or 2015. In 2013, *P. fasciatus* capture rate was greater in GSB than C, and in 2016 capture rate was greater in GSB than in DSB or C. Capture rate of adult *P. fasciatus* differed among treatments, but no year or treatment x year interaction effect was detected (Table 3; Fig. 7). In GSB, adult *P. fasciatus* capture rate was greater in 2013 than in 2014 or 2015. In 2013, adult *P. fasciatus* captures were greater in GSB than in C; in 2016, captures were greater in GSB than in C or DSB. Capture rate of juvenile *P. fasciatus* differed among years but not among treatments, and a treatment x year interaction effect was detected (Table 3; Fig. 7). In GSB, juvenile *P. fasciatus* capture rate was greater in 2013 than in 2014 or 2015, greater in 2015 and 2016 than in 2014, and greater in 2016 than in 2015. In 2013, juvenile *P. fasciatus* capture rate was greater in GSB than in C; in 2016, capture rate was greater in GSB than in C or DSB.

4. Discussion

Our results showed no adverse effects of either growing-season burns or dormant-season burns on any herpetofaunal species that was sufficiently common for statistical testing, and species richness of amphibians or reptiles did not differ among treatments. Capture rate of reptiles, and total, adult, and juvenile *P. fasciatus* in particular, was greater in growing-season burns than dormant-season burns or unburned controls. We could not definitively link this response to changes in forest structure, as season of burn had a minor and transitory effect on shrub cover and leaf litter depth, and live overstory and midstory tree density did not differ among treatments or controls. However, canopy cover decreased by an average of 16% in GSB within four growing-seasons of burning, possibly leading to a warmer, drier microenvironment on the forest floor, which was favorable for *P. fasciatus*.

Our study corroborates several others using drift fences with pitfall traps showing that single or multiple low-intensity dormant-season burns have little effect on reptile or amphibian capture rates in upland hardwood forest (Ford et al., 1999, 2010; Greenberg et al., 2016, 2017; Matthews et al., 2010). In our study, total amphibians were not affected by either growing-season or dormant-season burns. Capture rate of total salamanders, *P. ruber*, or *P. teyahalee*, which comprised 62% of all salamander captures, was unaffected by either burn treatment. During 2014, fewer anurans (all species) in C than the burn treatments was apparently driven by *A. americanus*, which comprised 83% of total anuran captures. More adult *A. americanus* were captured in DSB than in C in 2014, and an overall trend of greater *A. americanus* captures in DSB was also apparent. Juvenile *A. americanus* captures were greater in 2014 than other years, but did not differ among treatments. Increased captures of *A. americanus* after burns has been reported in some studies

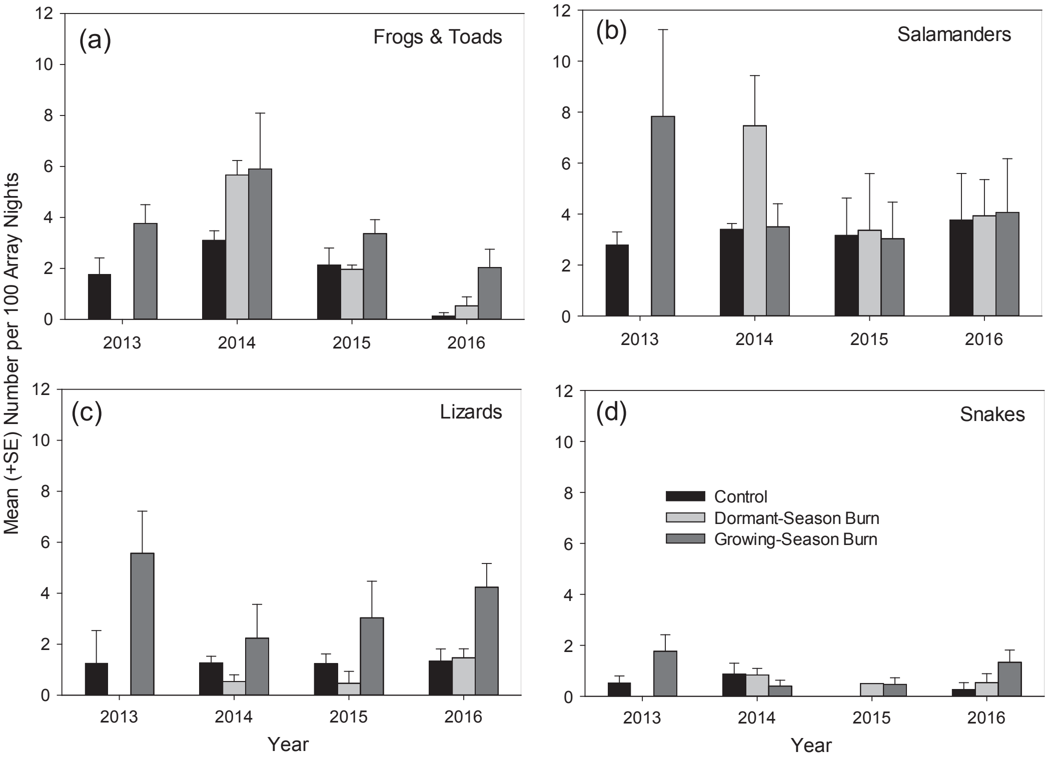

Fig. 5. Mean (± SE) number of first-captured (a) anurans; (b) salamanders; (c) lizards; and (d) snakes per 100 array nights in early growing-season burn (April 26, 2013; GSB), dormant-season burn (March 5, 2014; DSB), and unburned controls (C), 2013–2016. GSB and C units were sampled 2013–2016; DSB units were sampled 2014–2016, Bent Creek Experimental Forest, Pisgah National Forest, Buncombe County, NC.

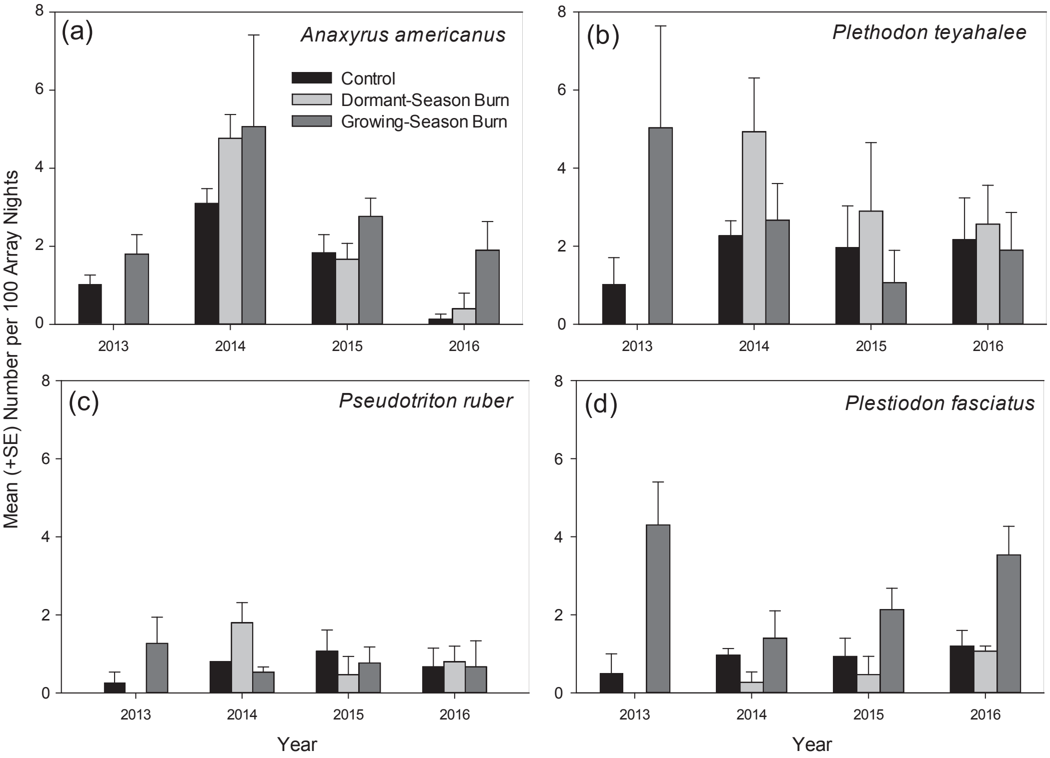

Fig. 6. Mean (± SE) number of first-captured (a) *Anaxyrus americanus*; (b) *Plethodon teyahalee*; (c) *Pseudotriton ruber*; and (d) *Plestiodon fasciatus* per 100 array nights in early growing-season burn (April 26, 2013; GSB), dormant-season burn (March 5, 2014; DSB), and unburned controls (C), 2013–2016. GSB and C units were sampled 2013–2016; DSB units were sampled 2014–2016, Bent Creek Experimental Forest, Pisgah National Forest, Buncombe County, NC.

Table 3
Results of general mixed model ANOVAs comparing treatment, year, and treatment x year interaction effects on adult and juvenile captures per 100 array-nights for species with ≥45 total captures per age class. Treatments were: growing-season burn (April 26, 2013; GSB); dormant-season burn (March 5, 2014; DSB), and; unburned control (C). GSB and C units were sampled 2013–2016; DSB units were sampled 2014–2016, Bent Creek Experimental Forest, Pisgah National Forest, Buncombe County, NC.

Species	ANOVA results				
	A/J	Total	P_{trt}	P_{yr}	P_{trtXyr}
American toad, *Anaxyrus americanus*	A	49	0.3118	0.0218	0.0087
	J	113	0.3133	0.0004	0.9599
Southern Appalachian salamander, *Plethodon teyahalee*	A	128	0.5044	0.1136	0.1520
	J	54	0.6187	0.1484	0.7997
Five-lined skink, *Plestiodon fasciatus*	A	47	0.0337	0.5811	0.1063
	J	57	0.1076	0.0006	0.0340

(Greenberg and Waldrop, 2008; Kirkland et al., 1996), but others found no change (Keyser et al., 2004). Variable capture rates of many anuran species, including *A. americanus*, appears to be more closely associated with breeding periods, weather, proximity to breeding sites, and juvenile recruitment than to prescribed burns or other disturbance treatments (Greenberg et al., 2016, 2017).

In our study, growing-season burns did not affect any amphibian taxa, but benefited lizards overall, and *P. fasciatus* in particular. Our results for total amphibians and reptiles were clearly driven by dominant species within our study area, and cannot be generalized to all species within these larger groups. For example, other lizard species such as litter-dwelling ground skinks (*Scincella lateralis*), which did not occur in our study area, are likely to respond differently (Sutton et al., 2014). Total reptile captures increased in GSB immediately after burns (2013) and again in 2016, four growing-seasons post-burn. Capture rate of *P. fasciatus*, including both adults and juveniles, was greater in GSB than C immediately after the burns, and higher in GSB than C or DSB in 2016. Other studies also documented increased lizard abundance, especially *S. undulatus* or *P. fasciatus*, following substantial canopy reduction by wind (Greenberg, 2001), high severity fire (Fouts et al., 2017; Greenberg et al., 2017; Matthews et al., 2010; McLeod and Gates, 1998), timber harvest (Greenberg et al., 2016; Renken, 2006), thin-with-burn (Sutton et al., 2013, Sutton et al., 2014), or harvest and burn (Keyser et al., 2004). These studies suggest that canopy reduction, leaf litter reduction, and higher light and ground temperatures following such disturbances likely create thermoregulatory conditions favorable for these common lizard species (Fouts et al., 2017; Moorman et al., 2011). Additionally, some of these studies also reported higher capture rates of juvenile *S. undulatus* or *P. fasciatus* following heavy canopy reduction by harvesting or high-severity fire (Greenberg et al., 2016, 2017; Renken et al., 2006), suggesting that their reproductive rates are also increased in disturbed sites. In contrast, lizard capture rate generally is unaffected by single (Greenberg et al., 2016) or repeated (Greenberg et al., 2017; Matthews et al. 2010) low-intensity dormant-season burns that do not substantially affect canopy cover. In our study, no fire-induced tree mortality occurred in either burn treatment, but canopy cover decreased in GSB within four growing-seasons of burning. We suggest that hotter burn temperatures interacting with physiologically active vegetation in GSB promoted a modest decline in canopy cover, increasing light and associated suitable microclimatic conditions for *P. fasciatus* adults and successful juvenile recruitment.

Although we could not gauge survival of individuals within our study design, our results clearly indicate that season of burn did not adversely affect relative capture rates of any common reptile or amphibian species. Similarly, based on observation and measures of relative abundance, most studies suggest that direct mortality of herpetofauna from prescribed burns is rare (see Harper et al., 2016). Only a few studies have tracked the fate of individuals before and after

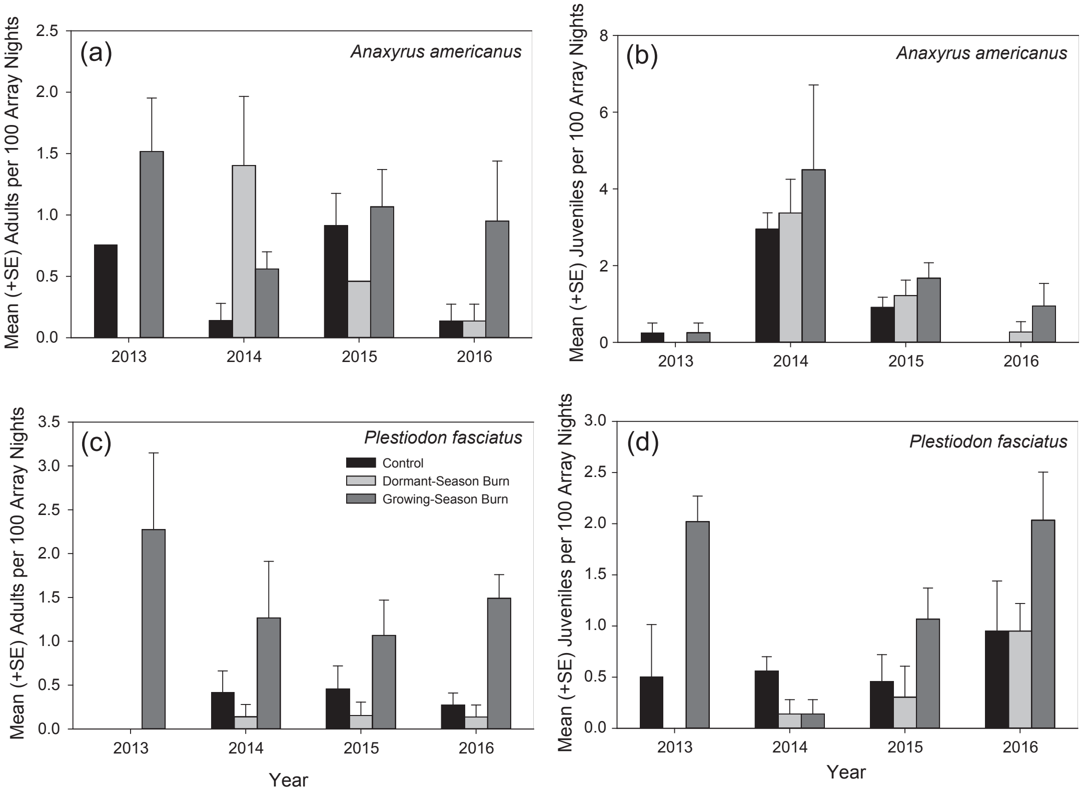

Fig. 7. Mean (± SE) number of first-captured (a) adult *Anaxyrus americanus*; (b) juvenile *A. americanus*; (c) adult *Plestiodon fasciatus*; and (d) juvenile *P. fasciatus* per 100 array nights in early growing-season burn (April 26, 2013; GSB), dormant-season burn (March 5, 2014; DSB), and unburned controls (C), 2013–2016. GSB and C units were sampled 2013–2016; DSB units were sampled 2014–2016, Bent Creek Experimental Forest, Pisgah National Forest, Buncombe County, NC.

burning. Pitt et al. (2013) reported no direct mortality of radio-transmittered *A. americanus* from a dormant-season prescribed burn in the southern Appalachians. O'Donnell et al. (2016) found no evidence of direct mortality of individually PIT-tagged *P. albagula* in upland hardwood forest of the Ozark Highlands.

In studies examining prescribed fire effects on herpetofauna, burn treatments were conducted during winter, when many terrestrial herpetofaunal species are inactive (Camp, 1988; Fitch and von Achen, 1977; Greenberg, *unpubl. data*), and likely belowground or otherwise insulated from high temperatures and potential desiccating effects of fire. In contrast, burns conducted during the growing-season when terrestrial herpetofaunal species are active (Greenberg, *unpubl. data*) could potentially affect survival (Harper et al., 2016). Behavioral avoidance of fire by retreat underground or below or within cover objects such as rocks or coarse woody debris, would likely minimize mortality for most species, with the possible exception of slow-moving species such as the timber rattler (*Crotalis horridus*) (Beaupre and Douglas, 2012) or eastern box turtle (*Terrapene carolina*) (Howey and Roosenburg, 2013; Melvin, 2017). We observed one dead eastern box turtle in one unit immediately following the late April GSB burns, further suggesting that they may be vulnerable to growing season burns.

Our study was not designed to address relative use of cover objects, underground retreats, or leaf litter, but our results indicated that surface activity by terrestrial salamanders was not reduced by either dormant-season or growing-season burns. In fact, we observed a dramatic increase in surface activity by salamanders immediately after both the growing-season (2013) and dormant-season (2014) burns. Notably, other studies using drift fences with pitfall traps in upland hardwood forest also showed a similar increase in terrestrial salamander capture rate immediately following low-intensity dormant-season burns (Greenberg et al., 2016, 2017; Matthews et al., 2010). Post-burn decreases in leaf litter depth or cover, or expanded foraging areas could induce greater activity levels by salamanders (Homyack et al., 2011). In contrast, O'Donnell et al. (2015) reported decreased, short-term abundance and surface activity of *P. serratus*, and increased use of cover objects after winter prescribed burns in the Ozark Highlands, based on repeated, diurnal plot searches. Similarly, Ford et al. (2010) reported increased use of coverboards by *D. ochrophaeus* and *P. cinereus* in twice-burned sites for at least two years compared to pre-burn or unburned controls. Pitt et al. (2013) found radio-transmittered *A. americanus* located closer to CWD in recently burned sites than in unburned sites, where leaf litter was more available and more often used for cover.

Survey methods such as coverboards (e.g.,Hocking et al., 2013; Pough et al., 1987) or searches (e.g.,O'Donnell et al., 2015) can provide a 'snapshot' perspective on changes in microdistribution of some herpetofaunal species following disturbances. In contrast, drift fences with pitfall traps that are continuously and concurrently open in all treatment units, for an entire season, effectively sample reptile and amphibian surface activity across changing diel activity patterns and weather conditions (Heatwole, 1962; O'Donnell et al., 2014). Drift fences with pitfall traps additionally provide insight at the community level, by sampling relative capture rates of more herpetofaunal species than survey methods.

5. Conclusions

Our study indicates that season of burn in upland hardwood forest does not adversely affect common herpetofaunal species, including terrestrial salamanders, but growing-season burns may indirectly increase abundance of lizards, and *P. fasciatus* in particular, likely due to modest decreases in canopy cover. However, further study is warranted for species of that were infrequently captured, or species of conservation concern. Slow-moving species, such as eastern box turtles (Howey and Roosenburg, 2013; Melvin, 2017) or timber rattlers (Beaupre and Douglas, 2012) may be more vulnerable to burning, especially during the growing season when they are most active. Most forest structure measurements following the single prescribed burns in this study reflected little or transient change in overstory, midstory, or ground-level conditions. However, we observed a gradual thinning of the canopy in GSB, and some tree mortality in patches where fire temperature was likely hotter. Mortality of overstory trees may be even greater following repeated application of growing-season burns, possibly leading to more substantial shifts in relative abundance of reptiles and amphibians. Weather, fuel types and condition, vegetation structure, and topography interact to affect fire intensity and the level of mortality or damage to canopy trees, regardless of season of burn. Effects of growing-season burns on tree mortality or damage could additionally vary according to whether they are conducted earlier (as was ours) or later in the growing-season. Additionally, fire effects on forest structure within and across stands are variable, often creating a gradient of structural conditions regardless of burn season. Based on our results and other studies, we suggest that herpetofaunal response is more closely linked to change in canopy cover than to season of burn *per se*, but longer-term monitoring of herpetofauna following repeated application of growing-season burns is warranted. Where prescribed fire is not feasible, reductions in canopy cover for specific wildlife management objectives can be achieved using other silvicultural methods.

Conflict of interest

The authors declare that they have no conflict of interest.

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