



## Breeding bird response to season of burn in an upland hardwood forest

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### ABSTRACT

Upland hardwood forest managers are increasingly burning during the growing-season in hopes of improving wildlife habitat and attaining or accelerating other restoration goals, highlighting the need for research addressing how season of burning affects wildlife, including breeding birds. We used 1-ha strip transects in nine units before (2011) and after (2013–2016) treatments to experimentally assess how breeding birds respond to early growing-season (26 April 2013; GSB) and dormant-season (5 March 2014; DSB) burns, and controls (C). Burn effects on forest structure were minor and transitory, regardless of burn season. Burns did not affect overstory or midstory tree basal area or density; shrub cover did not significantly differ among treatments, but within GSB and DSB it decreased for 1–2 years post-burn. A trend of reduced leaf litter depth immediately following both burn treatments was apparent, and recovery was rapid. Percent canopy cover decreased slightly in GSB within four years post-burn. Total bird species richness and density did not differ among treatments or years, and no treatment  $\times$  year interaction effects were detected. A treatment effect was detected for one of the 10 species tested; Red-bellied woodpecker density was greater in GSB than C in 2013 and 2015. No treatment  $\times$  year interaction effect was detected for any tested species. Density of birds within the tree-, cavity-, and shrub and midstory-nesting guilds were not detectably affected by either burn treatment, but ground-nester density was lower in GSB and DSB than C in 2014. Our results indicate that single, low-intensity burns, regardless of burn season, are not an effective tool in creating suitable forest structure for disturbance-dependent breeding bird species, or changing breeding bird community composition. In the short-term, substantial forest overstory reduction by timber harvests or high-severity burns will likely be required to improve forest conditions for disturbance-dependent species, and to increase species richness and abundance of breeding birds.

### 1. Introduction

Prescribed burning in upland hardwood forests is commonly recommended as a management tool to improve wildlife habitat, among other objectives such as reducing fuels, promoting oak regeneration, or restoring shortleaf pine or open oak woodlands (Waldrop et al., 2016) that were once common, due, in part, to frequent burning by Native Americans and European settlers (Greenberg et al., 2015). Most burns are conducted during the dormant-season, when fuels are generally drier and environmental conditions more predictable (Sparks et al., 2002). Several recent studies indicate that single- or repeated dormant-season burns in closed canopy upland hardwood forest have a minor and transitory effect on most wildlife communities, including reptiles and amphibians (Greenberg et al., 2016, 2018a, 2018b), and breeding birds (Greenberg et al., 2014, 2018c). Forest managers are increasingly burning during the growing-season in efforts to attain or accelerate restoration goals (Waldrop et al., 2016), highlighting the need for

research addressing how season of burning affects wildlife, including breeding bird communities.

Many bird species are closely associated with specific forest structural characteristics that could be affected by prescribed burns, such as forest canopy cover and height, the presence a midstory, snags, shrub cover, or leaf litter conditions (MacArthur and MacArthur, 1961; Thompson et al., 1995). However, species vary in their requirements according to their nesting and foraging habits (Annand and Thompson, 1997). Generally, overstory tree mortality is low (Keyser et al., 2018), and changes to canopy cover are minor after dormant-season burns; shrubs are burned, but rapidly re-sprout and grow, and leaf litter is replenished by autumn, when leaves drop from deciduous trees (Greenberg et al., 2016; Waldrop et al., 2016; Greenberg et al., 2018b). Correspondingly, most breeding bird species are unaffected by low-intensity dormant-season burns (Aquilani et al., 2000; Artman et al., 2001; Greenberg et al., 2018b). In contrast, breeding bird species richness and abundance increases dramatically after high-severity

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burns (Klaus et al., 2010; Rush et al., 2012; Rose and Simons, 2016; Greenberg et al., 2018c) or other silvicultural (Annand and Thompson, 1997; Perry et al., 2018) or natural (Greenberg and Lanham, 2001) disturbances where canopy cover is substantially reduced.

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 (Knapp et al., 2009; Waldrop et al., 2016), and potentially breeding bird communities. Higher-intensity burns have the potential to increase forest structural heterogeneity and create “habitat variability,” thereby providing suitable conditions for more species of breeding bird species with different requirements (Hiers et al., 2016). Several studies address season of burn effects on southern pine-dominated forests (e.g., Robertson and Hmielowski, 2014; Glitzenstein et al., 1995), but very little is known about effects on upland hardwood forests (Keyser et al., 2019) or associated wildlife. Forest managers need information regarding how season of burning in upland hardwood forests affects breeding bird communities to better manage wildlife in conjunction with ecosystem restoration or other forest management objectives.

We used a replicated Before-After/Control-Impact (BACI; Smith, 2002) approach to experimentally assess how breeding birds responded to early growing-season burns (GSB), dormant-season burns (DSB), and controls (C). Our objective was to determine if, and how, breeding bird species richness and relative abundance of total birds, common species, and nesting guilds differed between dormant-season and growing-season burns, or unburned controls.

## 2. Methods

### 2.1. Study area

Our study was conducted at the Bent Creek Experimental Forest (BCEF), a 2500 watershed within the Southern Appalachians in Pisgah National Forest, Buncombe County, North Carolina. Elevations within BCEF range from 700 m to 1070 m. Average annual precipitation is 140 cm and is evenly distributed year-round. Monthly average temperatures range  $-4.2^{\circ}$  to  $8.6^{\circ}$  C in January, to  $16.0^{\circ}$  to  $28.9^{\circ}$  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 (*Acer rubrum*), dogwood (*Cornus florida*), and interspersed shortleaf pine (*Pinus echinata*). Common shrub species include mountain laurel (*Kalmia latifolia*) and deerberry (*Vaccinium stamineum*) (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. Units were separated by fire lines as needed. All units were comprised of mature ( $> 100$  years old), oak-dominated closed canopy stands, and portions of each were within 500 m of perennial streams. 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 including 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. Each GSB and DSB burn unit was ignited off of the established fire lines using backing and flanking fires; interior ignition was completed using strip-head firing techniques (Keyser et al., 2019).

### 2.3. Forest structure and fire temperature

We measured overstory and midstory tree density and basal area in three, randomly located circular plots spaced  $\geq 30$  m apart within each treatment unit pretreatment (2011), and again three years after growing-season and dormant-season burns (see Keyser et al., 2019 for details). We measured maximum fire temperatures at ground level and 30 cm above ground level using temperature-sensitive paints on tags placed at two locations, 8 m apart, in each of three, randomly located vegetation plots per treatment unit (Keyser et al., 2019). Maximum fire temperatures varied, with higher temperatures at both ground-level (range  $93.3\text{--}426.7^{\circ}$  in GSB;  $93.3\text{--}204.5^{\circ}$  in DSB) and 30 cm above ground-level (range  $< 79.5\text{--}315.6^{\circ}$  in GSB;  $< 79.5\text{--}176.7^{\circ}$  in DSB), at a greater proportion of subplots in GSB than in DSB (Keyser et al., 2019). We measured additional post-burn forest structure variables in GSB and C units during summer 2013, and in DSB, GSB, and C units in summers 2014, 2015, and 2016. Percent cover of shrubs (woody understory) was measured along each of four 15-m line transects originating at randomly chosen distances and perpendicular directions out from the bird transect centerlines bisecting each unit (see Section 2.4). We recorded ‘start’ and ‘stop’ distance for shrubs (both above and below transect lines) along each transect, summed the total distance, and divided the sum by the transect length to obtain percent cover. Litter depth was measured at 7.5 m and 15 m along each transect line. We measured canopy closure with a spherical densiometer at two (in 2013) or three (2014–2016) random locations  $\geq 75$  m apart (Greenberg et al., 2018a) within each unit. We used average percent shrub cover, litter depth, and canopy closure within treatment units for data analyses.

### 2.4. Breeding bird sampling

Breeding birds were surveyed twice per season by the same observer between 15 May and 30 June in 2011 (pretreatment), 2013 (after the GSB; DSB units not yet burned and were not surveyed), 2014 (after the DSB), 2015, and 2016 along a  $200\text{ m} \times 50\text{ m}$  (1 ha) strip transect bisecting each treatment unit beginning and ending 25-m from the edge. Transect centerlines were walked slowly (about 15 min/transect) between sunrise and approximately four hours after sunrise. All individual birds (excluding flyovers) seen or heard within 25 m on either side of the centerline were recorded by species (one record per bird if the same bird was detected more than once). Survey start times were rotated among treatment units and between both visits to avoid day- or time-of-day detection bias. Each unit was surveyed earlier and later within the six-week survey period to avoid bias associated with differences in singing rates as the breeding season progressed; all nine units were surveyed on both survey days. We did not estimate detectability of different bird species (Allredge et al., 2008), and assumed that bird detection error was minimal and consistent among units due to a short (25-m) detection distance from transect centerline, surveys conducted by the same observer, repeated surveys within each unit, and rotating repeat surveys over time of day and breeding season. Relative density of birds for each treatment unit was calculated by averaging over both visits per unit for each year. Species richness represented the total number of species detected during both surveys in each unit each year.

### 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  $\times$  year interactions for all analyses of breeding birds and forest structure. Response variables analyzed were species richness and total density of breeding birds, densities of common (minimum 10 observations and mean  $\geq 1.0$  individual per ha in any treatment and year) species, and density of birds within nesting guilds (tree-, cavity-, shrub and midstory-, and ground-nesting) (Hamel, 1992). 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. Our primary interest was in treatment × year interaction effects as indicators that at least one treatment was responding differently from the others between pre- (2011) and post-treatment years. A nonsignificant treatment × year interaction indicated that treatment differences were consistent between pretreatment and post-treatment years, and that there was a consistent difference between years across treatments. Where significant ( $p \leq 0.05$ ) treatment × 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 bird data were non-estimable because of missing 2013 data for DSB (the DSB treatment was not yet implemented or sampled in 2013). 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 ( $p \leq 0.05$ ).

Several individual species were detected infrequently, resulting in many zero observations, and raising concern that normality assumptions required for repeated measures analyses may be violated. We limited our analyses to common species ( $\geq 10$  observations and mean  $\geq 1.0$ /ha in a given treatment and year). We square-root transformed individual species density data, and checked it for conforming approximately to the normal distribution by inspecting the residuals. We were quite liberal and included a species in the analysis if the skewness of the residuals was between  $-2$  and  $2$  and kurtosis of residuals was less than  $5$ . Thus, although some species did not pass a typical normality test, we feel they were sufficiently normal to perform a valid ANOVA, which is generally considered to be robust to non-normality (e.g., Blanca et al., 2017). Percentage data were arcsine square root transformed for analyses.

### 3. Results

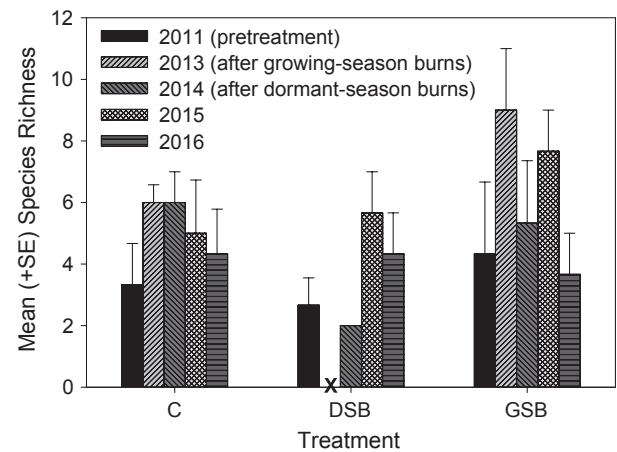
#### 3.1. Forest structure

Overstory and midstory tree basal area and stem density did not differ among treatments, or significantly change after growing-season or dormant-season burns (Keyser et al., 2019). Post-burn leaf litter depth differed among years; no treatment or treatment × year effects were detected, but a trend of reduced depth immediately after both the growing-season (2013) and dormant-season (2014) burns, followed by recovery, was apparent (Table 1). Post-burn percent shrub cover did not

**Table 1**

Mean ( $\pm$  SE) leaf litter depth (cm), shrub cover, and canopy closure (%), and results of general mixed model ANOVA comparing treatment, year, and treatment × year interaction effects. Treatments were: growing-season burn (April 26, 2013; GSB); dormant-season burn (March 5, 2014; DSB), and; unburned control (C). 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.

Variable	Year	GSB	DSB	C	ANOVA results		
					P <sub>trt</sub>	P <sub>yr</sub>	P <sub>trt×yr</sub>
Leaf litter depth (cm)	2013	0.9 ± 0.2	–	2.9 ± 0.8	0.0704	0.0326	0.0780
	2014	2.4 ± 0.5	1.8 ± 0.5	4.5 ± 0.4			
	2015	2.6 ± 2.1	3.9 ± 0.7	4.2 ± 0.6			
	2016	3.3 ± 0.1	3.0 ± 0.3	2.7 ± 0.6			
Canopy Closure (%)	2013	90.3 ± 2.2	–	95.8 ± 1.2	0.0240	< 0.0001	0.0300
	2014	94.5 ± 0.6	98.0 ± 0.2	98.1 ± 0.3			
	2015	92.7 ± 2.8	95.9 ± 1.3	99.0 ± 0.2			
	2016	74.0 ± 8.8	92.5 ± 0.5	94.0 ± 0.7			
Shrub Cover (%)	2013	8.8 ± 3.8	–	15.5 ± 2.5	0.2559	< 0.0001	0.0295
	2014	23.1 ± 5.8	10.2 ± 0.3	17.6 ± 0.7			
	2015	28.3 ± 5.3	15.2 ± 2.4	17.6 ± 2.4			
	2016	36.2 ± 12.4	21.9 ± 6.1	23.2 ± 0.5			

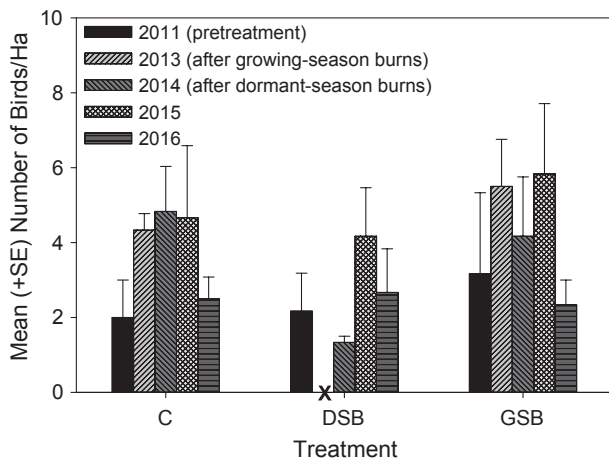


**Fig. 1.** Mean ( $\pm$  SE) total breeding bird species richness in growing-season burn (26 April 2013; GSB), dormant-season burn (5 March 2014; DSB), and unburned control (C) treatments, 2011 (pretreatment), 2013 (after the GSB), 2014 (after the DSB), 2015, and 2016, Bent Creek Experimental Forest, Pisgah National Forest, Buncombe County, NC.

differ among treatments, but differed among years, and a treatment × year interaction effect was detected (Table 1). Shrub cover did not differ among treatments within any year, but was dynamic 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. Post-burn percent canopy closure differed among treatments and years, and a treatment × year interaction effect was detected (Table 1). Changes in canopy closure in C and DSB were small relative to GSB. Within C, canopy closure was lower in 2013 than in 2015; in DSB, it was greater immediately after the burns (2014) than in 2016. Within GSB, canopy closure was greater immediately after the burns (2013) than in 2016, and greater in 2014 and 2015 than in 2016. In 2015, canopy closure was greater in C than in GSB; in 2016, canopy closure was greater in both C and DSB than in GSB.

#### 3.2. Breeding birds

We detected 298 birds of 34 species within transects, during the five years sampled. Total bird species richness (Fig. 1) and density (Fig. 2) did not differ among treatments or years, and no treatment × year interaction effects were detected (Table 2). A treatment effect was



**Fig. 2.** Mean (+SE) total number of breeding birds/ha in growing-season burn (16 April 2013; GSB), dormant-season burn (5 March 2014; DSB), and unburned control (C) treatments, 2011 (pretreatment), 2013 (after the GSB), 2014 (after the DSB), 2015, and 2016, Bent Creek Experimental Forest, Pisgah National Forest, Buncombe County, NC.

detected for one of the 10 species tested; Red-bellied woodpecker (*Melanerpes carolinus*) density was greater in GSB than C in 2013 and 2015. Year effects were detected for Red-eyed vireos (*Vireo olivaceus*) and Carolina chickadees (*Poecile carolinensis*). No treatment  $\times$  year interaction effect was detected for any tested species (Table 2).

Density of birds within the tree-nesting guild differed among years but not treatments, and no treatment  $\times$  year interaction effect was detected (Fig. 3a; Table 2). Density of cavity-nesting birds differed among treatments, but least squares means comparisons within years showed no significant ( $p < 0.05$ ) among-treatment differences; no year or treatment  $\times$  year interaction effect was detected (Fig. 3b; Table 2). Shrub and midstory-nester density did not differ among treatments or years, and no treatment  $\times$  year interaction effect was detected (Fig. 3c; Table 2). Density of ground-nesters differed among treatments but not years, and no interaction effect was detected; least squares means comparisons within years indicated that ground-nester density was greater in C than GSB or DSB in 2014, but no differences were detected for any other year (Fig. 3d; Table 2).

#### 4. Discussion

Our results indicated that single, low-intensity prescribed burns had a negligible and ephemeral effect on upland hardwood forest structure and breeding bird communities, regardless of whether they were conducted in the dormant-season or early growing-season. Overstory and midstory tree density were not significantly affected by prescribed burns, regardless of season burned (Keyser et al., 2019). Shrub cover and leaf litter depth was reduced after burns in both burn treatments, but recovered rapidly. Canopy closure decreased slightly (average 16%) within four growing seasons in GSB. Correspondingly, breeding bird community and species responses to low-intensity burning in the dormant-season or growing-season, were also negligible.

Occurrence and abundance of many breeding bird species are closely linked to forest structure (MacArthur and MacArthur, 1961), and canopy cover in particular (Annand and Thompson, 1997; Greenberg et al., 2014; Perry et al., 2018). In our study, breeding bird species richness and total density remained similar before and after burning, and did not differ among season of burn treatments or unburned controls. Similarly, we found no differences in numbers of birds within of canopy-, or shrub and midstory-nesting guilds among treatments. This was not surprising, given that neither GSB nor DSB killed trees or substantially altered the forest canopy, and reductions in shrub cover following both GSB and DSB were relatively small and short-term. A

**Table 2**

Total number of individual bird detections within 200  $\times$  50 m (1-ha) strip transects (all years, units, and both transect surveys each year), and results of mixed-model ANOVA with repeated measures comparing treatment, year, and treatment  $\times$  year interaction effects on breeding bird species richness, total density (no/ha), density of common (minimum 10 observations; mean  $\geq 1.0$ /ha in any treatment and year) species<sup>1</sup> and density within tree-, cavity-, shrub and midstory-, and ground-nesting guilds, Bent Creek Experimental Forest, Pisgah National Forest, Buncombe County, NC. Treatments were: growing-season burn (April 26, 2013; GSB); dormant-season burn (March 5, 2014; DSB), and; unburned control (C). All units were sampled pretreatment (2011); GSB and C units were sampled in 2013 (after the GSB), and all units were sampled 2014 (after the DSB), 2015, and 2016.

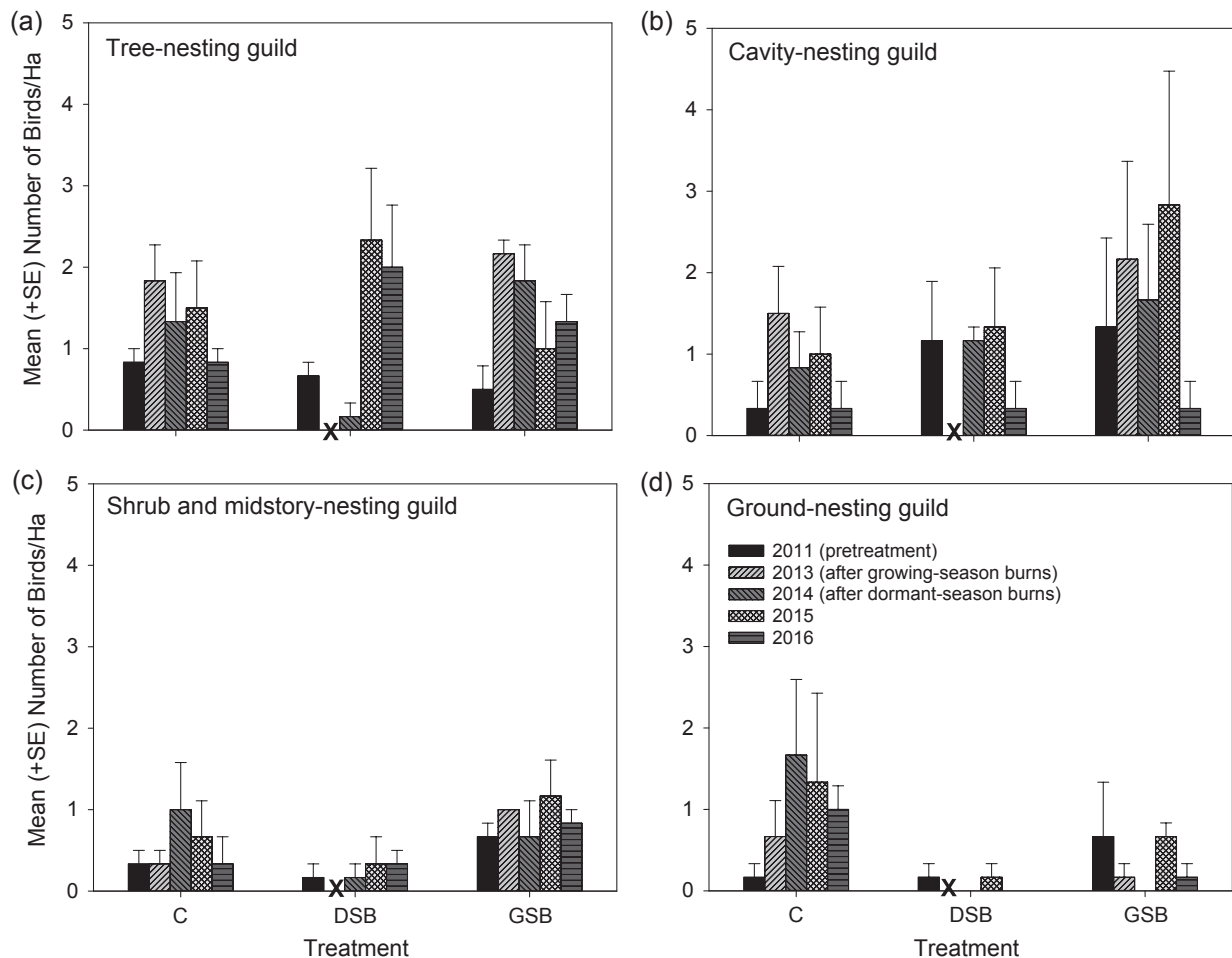
Variable	Obs	ANOVA results		
		P <sub>trt</sub>	P <sub>yr</sub>	P <sub>trtYr</sub>
<b>Tree-nester</b>	<b>107</b>	<b>0.8235</b>	<b>0.0451</b>	<b>0.1130</b>
Black-throated green warbler ( <i>Setophaga virens</i> )	12	0.5825	0.5521	0.1549
Red-eyed vireo ( <i>Vireo olivaceus</i> )	42	0.7137	0.0253	0.5722
Scarlet tanager ( <i>Piranga olivacea</i> )	14	0.5891	0.7577	0.0902
<b>Cavity-nester</b>	<b>98</b>	<b>0.0083</b>	<b>0.2794</b>	<b>0.9741</b>
Carolina chickadee ( <i>Poecile carolinensis</i> )	26	0.7184	0.0354	0.8717
Carolina wren ( <i>Thryothorus ludovicianus</i> )	14	0.2167	0.2912	0.8583
Tufted titmouse ( <i>Baeolophus bicolor</i> )	18	0.5286	0.6363	0.9604
Red-bellied woodpecker ( <i>Melanerpes carolinus</i> )	10	0.0445	0.3171	0.3337
<b>Shrub- and midstory-nester</b>	<b>48</b>	<b>0.1055</b>	<b>0.6990</b>	<b>0.7139</b>
Blue-headed vireo ( <i>Vireo solitarius</i> )	22	0.6333	0.3070	0.5217
<b>Ground-nester</b>	<b>41</b>	<b>0.0302</b>	<b>0.7578</b>	<b>0.5375</b>
Ovenbird ( <i>Seiurus aurocapillus</i> )	18	0.3377	0.5866	0.2459
Worm-eating warbler ( <i>Helmitheros vermivorus</i> )	13	0.3886	0.1228	0.5806
<b>Total</b>	<b>298</b>	<b>0.2872</b>	<b>0.1321</b>	<b>0.8433</b>
<b>Richness</b>	<b>34</b>	<b>0.2123</b>	<b>0.0706</b>	<b>0.6105</b>

<sup>1</sup> Other less common ( $< 10$  observations or mean  $< 1.0$ /ha in any treatment or year) species (and associated nesting guild; T = tree-; C = cavity-; S = shrub and midstory-; G = ground-nesting; O = Other) detected within strip transects were: Acadian flycatcher (*Empidonax virens*; T); American crow (*Corvus brachyrhynchos*; T); American goldfinch (*Carduelis tristis*; S); Black-and-white warbler (*Mniotilta varia*; G); Blue-gray gnatcatcher (*Poliopitla caerulea*; T); Black-throated blue warbler (*Setophaga caerulescens*; S); Blue jay (*Cyanocitta cristata*; T); Broad-winged hawk (*Buteo platypterus*; T); Common grackle (*Quiscalus quiscula*; T); Downy woodpecker (*Picoides pubescens*; C); Eastern bluebird (*Sialia sialis*; C); Eastern phoebe (*Sayornis phoebe*; O); Eastern wood-pewee (*Contopus virens*; T); Eastern towhee (*Pipilo erythrophthalmus*; S); Great-crested flycatcher (*Myiarchus crinitus*; C); Hairy woodpecker (*Picoides villosus*; C); Hooded warbler (*Setophaga citrina*; S); Indigo bunting (*Passerina cyanea*; S); Northern cardinal (*Cardinalis cardinalis*; S); Pileated woodpecker (*Dryocopus pileatus*; C); Pine warbler (*Setophaga pinus*; T); Ruffed grouse (*Bonasa umbellus*; G); White-breasted nuthatch (*Sitta carolinensis*; C); Wood thrush (*Hylocichla mustelina*; S).

treatment effect was detected for the cavity-nesting guild, but least square means comparisons did not detect any differences among treatments within years, and the absence of a treatment  $\times$  year interaction effect indicated that pre- and post-treatment years did not differ. Only one (Red-bellied woodpecker) of 10 tested species showed a positive response to GSB in two of the four post-treatment years; this also may not be biologically meaningful, as no differences were detected between pretreatment (2011) and post-treatment years (e.g., no treatment  $\times$  year interaction effect).

Although tested individual ground-nesting species showed no response to season of burn, total birds within the ground-nesting guild decreased in both burn treatments in 2014 (immediately after DSBs, and the second breeding season after GSBs). Other short-term studies in upland hardwood forests indicate that abundance or occupancy of some ground-nesting species including Black-and-white warblers (*Mniotilta varia*) (Aquilani et al., 2000; Greenberg et al., 2007), Ovenbirds (*Seiurus aurocapillus*) (Artman et al., 2001; Aquilani et al., 2000) and Worm-





**Fig. 3.** Mean (+SE) number of breeding birds/ha within (a) tree-; (b) canopy-; (c) shrub and midstory- and; (d) ground-nesting guilds, in growing-season burn (26 April 2013; GSB), dormant-season burn (5 March 2014; DSB), and unburned control (C) treatments, 2011 (pretreatment), 2013 (after the GSB), 2014 (after the DSB), 2015, and 2016, Bent Creek Experimental Forest, Pisgah National Forest, Buncombe County, NC.

eating warblers (*Helmitheros vermivorus*) (Artman et al., 2001; Greenberg et al., 2007) decrease temporarily after low-intensity, dormant-season burns. This pattern of short-term decreases and rapid recovery of ground-nesting birds generally corresponds with decreased leaf litter (nesting substrate) depth or cover after burns, and recovery as leaves drop from deciduous trees each autumn.

Our results corroborate other studies showing that single (Aquilani et al., 2000; Artman et al., 2001; Greenberg et al., 2007, 2014) or repeated (Greenberg et al., 2018c) low-intensity dormant-season burns have a minor and (or) short-lived effect on forest structure and breeding bird communities. We further show that effects of single, low-intensity, early growing-season burns on forest structure and breeding bird communities are negligible, and do not substantially differ from effects of single, low-intensity dormant-season burns. Results of our study and others clearly indicate that low-intensity prescribed burning in upland hardwood forests, regardless of season conducted, is not an effective tool to “improve habitat” for breeding birds, at least in the short-term.

In contrast, other research shows that mixed- or high-severity burns causing heavy overstory tree mortality can rapidly create suitable conditions for many disturbance-dependent bird species, with little change in the abundance most mature forest species. Klaus et al. (2010) reported higher species richness and diversity in medium- and high-severity burns relative to low-severity burns or unburned southern Appalachian hardwood forest. Rush et al. (2012) found that the abundance of several disturbance-dependent species was highest in high-severity burns, whereas abundances of many other species were unaffected (Rush et al., 2012). Rose and Simons (2016) found that fire

severity had a strong, positive influence on the occurrence of 13 breeding bird species, a negative influence on two ground-nesting species, and no influence on nine breeding bird species in xeric pine oak southern Appalachian forests. Greenberg et al. (2007, 2018c) reported increased total breeding bird abundance and more species after high-severity burns with heavy tree mortality compared to single- or repeated low-intensity burns, midstory reductions, or control treatments, largely due to an influx of disturbance-dependent species, with little change in the abundance or occurrence of species generally associated with mature forest.

Research also indicates that breeding birds respond similarly to changes in upland hardwood forest structure created by high-severity burns (e.g., Rush et al., 2012; Rose and Simons, 2016; Greenberg et al., 2018c) and silvicultural treatments with heavy canopy removal such as shelterwood regeneration harvests (Annand and Thompson, 1997; Greenberg et al., 2014; Perry et al., 2018). This suggests that substantial reduction of forest canopy cover, whether attained by high-severity burning or timber harvesting methods, positively affects many disturbance-dependent bird species with no detectable impact on most species associated with mature forest (e.g., Annand and Thompson, 1997; Rodewald and Smith, 1998; Greenberg et al., 2014; Kendrick et al., 2015), leading to higher species richness and total density of birds within a few years of disturbance. Most studies report that abundance of some ground-nesting species, especially Ovenbirds, decreases after silvicultural regeneration harvests with heavy canopy removal (e.g., Annand and Thompson, 1997; Greenberg et al., 2014; Kendrick et al., 2015; Perry et al., 2018), high-severity burns

(Greenberg et al., 2007; Klaus et al., 2010; Rose and Simons, 2016), or heavy basal area reduction by harvest followed by fall or spring fire (Vander Yacht et al., 2016) but recovers as leaf litter is replenished and disturbed forests mature.

## 5. Conclusions

Our results indicate that single, low-intensity burns, regardless of burn season, are not an effective tool in creating suitable forest structure for disturbance-dependent breeding bird species, or changing breeding bird community composition. Some studies indicate that multiple repetitions of low-intensity dormant-season burns can cause some delayed tree mortality, resulting in a gradual and slight increase structural heterogeneity of forests and a nonsignificant trend of bird species richness (Greenberg et al., 2018c). Repeated growing-season burns have the potential to accelerate delayed tree mortality and greater structural diversity in forests, given that they are generally hotter than dormant-season burns. Further research is needed to examine effects of repeated growing-season and dormant-season burns on forest structure and breeding bird communities. Additionally, research on later growing-season burns could provide insight into how the timing of growing-season burns affects forest structure and breeding birds later in the nesting season; this could be especially important in relation to nest success and productivity, at the burn-unit scale. In the short-term, substantial forest overstory reduction by timber harvests or high-severity burns will likely be required to improve forest conditions for disturbance-dependent species, and to increase the abundance and species richness of breeding bird communities.

## Declaration of Competing Interest

The authors declare that they have no conflict of interest.

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