

Breeding Bird Community Responses to Growing Versus Dormant Season Prescribed Fire

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Abstract: Prescribed fires are frequently used to restore and maintain pine savanna in the southeastern United States. Although several declining bird species occur within these pine savannas, few studies have directly compared the effects of growing versus dormant season prescribed fires on breeding birds. Therefore, we compared the effects of growing- versus dormant-season prescribed fires on breeding bird communities in mature pine (*Pinus* spp.) stands within the Fort Benning Military Reservation in west-central Georgia. We used 50 m fixed-radius point counts to sample breeding bird communities in growing and dormant season burned stands for two years post-treatment. We detected 50 bird species between 1–2 years post-burn, with 43 species detected on growing-season burned sites and 48 species detected on dormant season burned sites. We detected few effects of season of burn on vegetation, individual bird species, bird habitat associations, or migratory strategy. Of the 25 bird species with >1.5 individuals 10 ha⁻¹, only prairie warbler (*Setophaga discolor*), red-cockaded woodpecker (*Picoides borealis*), and red-headed woodpecker (*Melanerpes erythrocephalus*) abundance differed by season of burn, and all three species were more commonly encountered in growing- season burned stands 1–2 years post-burn. Although our results indicate that season of burn may have little effect on breeding bird communities after a single growing season burn, multiple growing season burns may result in more profound changes in the vegetation that could affect breeding bird communities within fire-maintained pine forests in the southeastern United States.

Key words: breeding birds, dormant season burn, Georgia, growing season burn, prescribed fire, songbirds, upland pine forests

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Many southeastern United States forest types are fire-dominated or fire-influenced systems. Historically, most natural fires in this region were set by lightning during the growing season due to dry conditions and high temperatures (Komarek 1968). Additionally, Native Americans frequently set fires year-round to improve conditions for game, maintain travel corridors, and improve food production (Carroll et al. 2002, Van Lear et al. 2005). Today, prescribed fire is used to reduce forest fuel loads, prevent or minimize destructive wildfires (USDA 2000, Baeza et al. 2002), reduce hardwood cover to improve pine growth (Van Lear and Waldrop 1991, Shiver and Martin 2002), and create early successional plant communities (Sparks et al. 1998).

Although pine (*Pinus* spp.) savannas once encompassed approximately 30 million ha in the southeastern United States prior to European settlement, changing land uses and fire suppression have reduced pine savanna to only about 1.2 million ha in widely scattered patches (Van Lear et al. 2005). Some bird species that depend on mature pine stands with open understories maintained by fire have declined in response to decreasing pine savanna, such as Bachman's sparrow (*Peucaea aestivalis*), northern bobwhite (*Colinus virginianus*), and the federally endangered red-cockaded woodpecker (*Picoides borealis*) (Ligon et al. 1986, Hunter et al. 2001, Tucker et al. 2004). Additionally, several bird species of Partners in Flight conservation priority are common in fire-maintained vegetation

associations, such as brown-headed nuthatches (*Sitta pusilla*), loggerhead shrikes (*Lanus ludovicianus*), prairie warblers (*Setophaga discolor*), and yellow-breasted chats (*Icteria virens*) (Panjabi et al. 2005).

Prescribed burns can affect bird communities by altering structure and composition of understory vegetation (Brockway and Lewis 1997, Brawn et al. 2001). Dormant-season prescribed burns are frequently preferred by land managers due to lower ambient temperatures, higher fuel moisture, and consistent winds that minimize the risk of fires escaping control (Wade and Lunsford 1989). However, use of growing-season burns to mimic natural fire regimes has received attention, particularly to control understory hardwood vegetation (Waldrop et al. 1992). Because growing-season burns occur during the nesting season of many bird species, growing season fire may affect ground and shrub nesting species (Petersen and Best 1987, Lang et al. 2002).

Several studies have demonstrated the effects of burned versus unburned areas in a variety of fire-maintained habitats on breeding bird communities (Wilson et al. 1995, Allen et al. 2006). However, few studies have directly compared the effects of growing versus dormant season prescribed fires on breeding bird communities in mature upland pine forests. Therefore, to determine if season of burn affected individual bird species abundance and altered composition of breeding bird communities, we compared breeding bird communities on growing and dormant season burned upland pine forests for two years in west-central Georgia.

Methods

We conducted our study at the 74,000-ha Fort Benning Military Reservation near Columbus, Georgia. Fort Benning is situated on the Fall Line between the Piedmont and Upper Coastal Plain physiographic regions. The Reservation has loamy latosolic soils, and elevations range from 61–213 m above sea level. It contains one of the largest populations of red-cockaded woodpeckers in the state (Schneider et al. 2010), and prescribed fire on a three-year rotation is used to control hardwoods to maintain low basal area pine forests with an open understory as part of red-cockaded woodpecker habitat management. Mixed and pure forest stands of longleaf (*Pinus palustris*), loblolly (*P. taeda*), and shortleaf pine (*P. echinata*) dominated the landscape. Sweetgum (*Liquidambar styraciflua*), persimmon (*Diospyros virginiana*), flowering dogwood (*Cornus florida*), broomsedge (*Andropogon* spp.), blackberry (*Rubus* spp.), gallberry (*Ilex glauca*), and wax myrtle (*Morella cerifera*) were common understory plants.

We selected nine stands burned in the dormant season (January–March) and eight stands burned in the growing season (April–May) in 1994. Stands varied from 50–150 ha in size. Plot selection

was limited to areas that were planned for burns according to the installation management plan, and random sites were not logistically feasible because of safety concerns. All burns contained similar fuel loads and intensities, and all fires were controlled, creeping, and slow-to-moderate with no torching of canopy trees. Between September 1995 and March 1996, one dormant and four growing season stands were burned by wildfires and removed from sampling for the second year. All stands were dominated by loblolly pine. Unburned plots were not included because fire exclusion is considered a treatment rather than a control in fire maintained habitats (Platt et al. 1988).

During July and August 1995 and 1996, we measured vegetation at nine random sampling points (0.04 ha) in each study stand (James and Shugart 1970). We measured stand age, woody basal area, shrub density, percent canopy closure, percent vegetative ground cover, vegetation profile, and snag density. We recorded species and diameter at breast height (DBH) of woody vegetation ≥ 3.0 cm to measure woody basal area. We measured shrub density by recording all woody stems < 3.0 within 1 m of 11.3 m transects in each cardinal direction from each center point. We estimated percent canopy closure with a spherical densiometer (Lemmon 1956). We recorded 10 estimates of vegetative ground cover at each sampling point using an ocular tube at 1 m intervals in each cardinal direction. Vegetative profile was visually estimated using a 0.5×3 m density board situated 11.3 m from the center point (Noon 1981). We measured snag density by counting snags along 122 m transects through each avian and vegetation sampling point within 61 m of each side of each transect. We used analysis of variance (ANOVA) for vegetation comparisons between burn treatments and years, and present the average of all points within each stand as the value for each vegetation variable.

We sampled birds at nine point counts located 122 m apart along a line transect within each stand. We conducted breeding bird counts biweekly from 1 April through mid-June in 1995–1996 using the fixed-radius point count method (Hutto et al. 1986). Point counts were surveyed between sunrise and 1030 and were rotated among observers to reduce bias. All birds seen or heard during a five-minute interval and within 50 m were counted. Each point count was minimally situated 80 m from stand edges, creating a 30-m buffer between the farthest extent of the point count and a stand's edge. We did not conduct counts on days with wind ≥ 4 Beaufort Wind Scale or in moderate rain (Robbins 1981). Birds flushed when approaching or leaving a survey point were noted but not used in our analysis. Likewise, birds observed flying over a point or late overwintering species that do not breed in central Georgia (e.g., yellow-rumped warbler, *Setophaga coronata*) were excluded from analysis. Because adjacent point counts only had a

22-m buffer between their farthest extents, we were careful not to double count by paying attention to bird movements. Birds were classified according to habitat association (forest interior, forest edge, and pine-grassland) (Wilson et al. 1995) and migratory strategy (neotropical migrant, short-distance migrant, or resident) (Poole 2010).

We examined the hypothesis that season of burn did not affect bird abundance, species richness, or diversity. We calculated detection probabilities and absolute abundance estimates using a Royle repeat count model in program PRESENCE (Royle 2004, Hines 2006, Donovan and Hines 2007). We attempted to calculate abundance estimates for each stand by analyzing each stand independently using the nine nested point counts as independent counts over six visits. Unfortunately, sample size was small and generated such large confidence intervals that we had little confidence in the detection probabilities and absolute abundance estimates. Nevertheless, we calculated the maximum number of individual birds encountered by species at each point within a stand over all visits and summed these maximum values to provide a single count index for each stand. We assumed our maximum values represented perfect detection of all individuals within a stand for our count index model, although not accounting for heterogeneous detection probabilities among species likely underestimated the number of birds actually present (Royle et al. 2007). However, because all stands were maintained in low basal area pine with a relatively open understory, we believed that assuming detection was uniform between growing and dormant season burns was reasonable for our analysis. Additionally, although we had uneven replications between treatments and years, our standard errors were relatively

unaffected despite reduced sample sizes. Thus, we assume that any observed differences in count abundance, species richness, and diversity was attributable to season of burn and not to errors associated with small sample sizes.

We used multivariate ANOVA to compare count abundance (hereafter referred to as abundance) for individual bird species with >1.5 individuals 10 ha^{-1} and Shannon H' diversity between growing and dormant season burns and year, and we tested for a burn by year interaction. Shannon diversity index values range from 0 where there is no diversity to ~ 5 for the most diverse communities (Shannon and Weaver 1949). We calculated evenness as part of our diversity calculations, which varied from 0–1 with 0 where most species were rare and just a few were abundant to 1 where all species were almost equally abundant. We used multivariate ANOVA to compare abundance and species richness for habitat association and migratory status between growing and dormant season burns and year. We checked for possible violations of normality and outliers by examining graphs of residuals, Cook's distance, and quantile-quantile plots in Program R (R Development Core Team 2008). When necessary, we used $\log(x+1)$ transformations to normalize our data; however, we present untransformed values for ease of interpretation. We classified differences as significant at $\alpha = 0.05$.

Results

Vegetation characteristics and profile were similar between growing and dormant season burned areas (Table 1). Only ground cover showed a treatment response and was greater in growing-season burned areas ($P=0.04$). We detected 50 bird species be-

Table 1. Mean vegetation characteristics and standard errors of growing- and dormant-season burns at one-year (1995) and two-years (1996) post-burn on Fort Benning Military Reservation, Georgia.

Habitat characteristic	1995					1996				
	Growing		Dormant		P-value	Growing		Dormant		P-value
	Mean	SE	Mean	SE		Mean	SE	Mean	SE	
Stand age ^a (years)	51.6	4.3	57.7	4.0	0.31					
Snag density ^a (n/0.04 ha)	29.1	19.0	37.2	28.6	0.49					
Basal area ^a (m ² /acre)	51.3	5.9	60.5	7.7	0.35					
Canopy closure ^a (%)	31.7	0.9	31.0	0.8	0.56					
Shrub density (n/0.04 ha)	10.2	0.8	8.0	0.8	0.45	15.1	1.6	13.4	1.2	0.73
Vegetative ground cover (%)	6.3	0.1	5.0	0.2	0.04	6.5	0.4	5.7	0.2	0.30
Vegetation profile (%)										
0.0–0.5 m	58.8	1.9	48.1	2.0	0.15	64.4	2.9	56.4	2.4	0.26
0.5–1.0 m	36.1	2.1	30.6	2.0	0.41	44.7	3.1	38.9	2.5	0.48
1.0–1.5 m	18.9	1.7	17.7	1.6	0.79	25.7	2.7	23.5	2.1	0.72
1.5–2.0 m	13.3	1.5	15.5	1.6	0.57	23.3	2.7	20.4	2.0	0.59
2.0–2.5 m	11.2	1.4	11.7	1.4	0.90	20.1	2.5	14.3	1.7	0.30
2.5–3.0 m	11.7	1.5	10.6	1.3	0.74	16.9	2.3	14.6	1.8	0.66

a. Measurements only taken in 1995.

Table 2. Breeding bird count abundance (individuals/10 ha) comparisons for birds with >1.5 individuals 10 ha⁻¹ between growing and dormant season burns at one-year (1995) and two-years (1996) post-burn on Fort Benning Military Reservation, Georgia.

Common name	Scientific name	Status ^a	1995				1996				Burn		Year	
			Growing		Dormant		Growing		Dormant		F	P ^b	F	P
			Mean	SE	Mean	SE	Mean	SE	Mean	SE				
Bachman's sparrow	<i>Peucaea aestivalis</i>	B, F	6.3	1.2	7.6	2.2	8.2	3.8	4.5	1.3	0.08	0.78	0.35	0.56
Blue-gray gnatcatcher	<i>Poliaptila caerulea</i>	A, E	2.1	0.9	4.1	1.2	5.0	3.1	4.8	1.2	0.70	0.41	1.11	0.30
Brown-headed cowbird	<i>Molothrus ater</i>	B, E	3.0	0.9	2.3	0.9	3.9	1.2	3.0	0.9	0.67	0.42	0.68	0.42
Brown-headed nuthatch	<i>Sitta pusilla</i>	C, F	4.1	1.3	4.0	1.4	2.1	1.4	3.6	1.0	0.12	0.74	0.54	0.46
Blue jay	<i>Cyanocitta cristata</i>	B, E	1.6	0.8	1.6	0.5	0.7	0.4	0.9	0.4	0.00	0.94	1.51	0.23
Carolina chickadee	<i>Poecile carolinensis</i>	C, D	3.0	1.1	4.4	1.0	2.9	1.2	5.9	1.7	2.28	0.14	0.41	0.53
Carolina wren	<i>Thryothorus ludovicianus</i>	C, E	7.3	1.5	5.6	1.3	6.4	2.1	4.3	0.9	1.87	0.18	0.67	0.42
Common yellowthroat	<i>Geothlypis trichas</i>	A, E	3.9	1.4	3.6	1.6	4.6	1.4	2.3	1.1	1.76	0.20	0.03	0.87
Downy woodpecker	<i>Picoides pubescens</i>	C, D	0.4	0.2	0.9	0.2	2.9	1.2	1.6	0.5	0.12	0.73	9.50	<0.01
Eastern towhee	<i>Pipilo erythrophthalmus</i>	B, E	9.1	1.6	9.7	2.1	12.5	1.5	10.9	2.6	0.00	0.92	0.78	0.38
Eastern wood-pewee	<i>Contopus virens</i>	A, F	4.1	1.4	3.6	0.7	4.3	1.3	3.8	0.7	0.04	0.84	0.36	0.55
Great crested flycatcher	<i>Myiarchus crinitus</i>	A, D	5.5	1.4	5.7	1.4	2.9	1.5	5.7	1.2	0.66	0.42	0.48	0.49
Indigo bunting	<i>Passerina cyanea</i>	A, F	8.4	1.1	7.9	0.7	8.9	1.2	9.3	1.8	0.02	0.88	0.71	0.41
Northern cardinal	<i>Cardinalis cardinalis</i>	C, E	5.2	1.0	6.6	1.4	6.8	1.2	5.7	1.8	0.09	0.76	0.00	0.97
Pine warbler	<i>Setophaga pinus</i>	B, F	17.5	4.8	15.0	1.9	6.1	2.0	12.1	1.9	0.05	0.82	3.73	0.06
Prairie warbler	<i>Setophaga discolor</i>	A, F	13.9	2.4	6.7	1.7	17.9	2.6	10.4	2.7	9.41	<0.01	0.13	0.13
Red-bellied woodpecker	<i>Melanerpes carolinus</i>	C, D	2.0	0.5	2.9	0.6	2.5	0.4	2.5	0.9	0.44	0.51	0.00	0.96
Red-cockaded woodpecker	<i>Picoides borealis</i>	C, F	6.4	1.9	2.7	1.1	4.6	3.4	1.4	1.2	4.55	0.04	2.37	0.14
Red-eyed vireo	<i>Vireo olivaceus</i>	A, D	3.8	0.9	2.7	0.7	3.9	1.6	3.8	1.1	0.53	0.47	0.53	0.47
Red-headed woodpecker	<i>Melanerpes erythrocephalus</i>	B, F	2.7	1.1	1.1	0.5	3.6	2.1	0.5	0.4	5.29	0.03	0.00	0.96
Summer tanager	<i>Piranga rubra</i>	A, D	3.8	1.1	6.0	1.1	0.0	0.0	0.0	0.0	2.09	0.09	75.50	<0.01
Tufted titmouse	<i>Baeolophus bicolor</i>	C, D	2.9	0.9	4.0	1.3	3.9	0.4	2.9	0.7	0.08	0.78	0.08	0.78
White-breasted nuthatch	<i>Sitta carolinensis</i>	B, E	3.2	1.5	2.3	0.8	0.0	0.0	0.0	0.0	0.34	0.57	6.99	0.02
White-eyed vireo	<i>Vireo griseus</i>	A, E	0.9	0.5	2.7	1.0	1.8	1.4	1.1	0.5	1.02	0.32	0.65	0.43
Yellow-breasted chat	<i>Icteria virens</i>	A, E	2.3	1.0	4.7	1.2	7.1	1.7	3.8	1.2	0.03	0.86	0.89	0.35

a. A = Neotropical migrant, B = Short-distance migrant, C = Resident, D = Forest interior, E = Forest edge, F = Pine-grassland
 b. Degrees of freedom: season of burn = 1, year = 1, year × season of burn = 1, error = 24.

tween 1–2 years post-burn, with 43 species detected in sites with growing season burned areas and 48 species detected in dormant season burned areas. In the first year post-burn, 40 species were observed in growing season burned areas and 43 species in dormant season burned areas, and in the second year 33 and 37 species were detected in growing and dormant season burned areas, respectively.

We detected few differences in individual bird species abundance between burn treatments. Only prairie warbler, red-cockaded woodpecker, and red-headed woodpecker (*Melanerpes erythrocephalus*) abundance differed by season of burn, and all three species were more commonly encountered in growing-season burned stands 1–2 years post-burn (Table 2). Only downy woodpecker (*Picoides pubescens*), summer tanager (*Piranga rubra*), and white-breasted nuthatch (*Sitta carolinensis*) abundance varied by year irrespective of treatment. Downy woodpecker abundance increased in both burn treatments in the second year, whereas summer tanagers and white-

breasted nuthatches were not detected in either burn treatment in the second year. The yellow-breasted chat was the only species with a year by season of burn interaction ($F_{1,25} = 4.81, P = 0.04$); chats were more abundant on dormant-season burned areas in the first year following burning, and more abundant on growing-season burned areas in the second year.

Avian Shannon H' diversity was similar between burn treatments and years ($F_{1,24} \leq 1.27, P \geq 0.27$) and averaged 2.82 (SE = 0.03) with average evenness of 0.90 (SE = 0.01). Season of burn had no effect on forest interior, forest edge, pine-grassland, neotropical migrant, or resident bird species (Tables 3 and 4). Forest interior birds declined in abundance and species richness in the second year irrespective of treatment (Table 3). We detected more species of short-distance migrants in growing season than dormant season burns in both years, but abundance of short-distance migrants was similar between burn treatments (Table 4).

Table 3. Abundance and species richness means and standard errors for bird habitat associations between growing and dormant season burns at one-year (1995) and two-years (1996) post-burn on Fort Benning Military Reservation, Georgia.

	1995				1996				Burn		Year	
	Growing		Dormant		Growing		Dormant		F	P ^a	F	P
	Mean	SE	Mean	SE	Mean	SE	Mean	SE				
Abundance												
Forest interior	26.6	3.4	32.2	2.5	20.7	3.8	24.6	2.8	2.6	0.12	4.95	0.04
Forest edge	39.8	5.0	44.7	7.4	53.2	5.3	38.9	6.2	0.11	0.74	0.04	0.84
Pine-grassland	65.2	10.8	50.6	5.8	56.4	11.0	47.0	7.4	2.11	0.16	0.40	0.53
Species richness												
Forest interior	8.0	0.8	8.5	0.7	6.5	1.0	6.6	0.7	0.21	0.65	4.82	0.04
Forest edge	8.0	0.4	7.7	0.9	9.5	0.3	8.1	0.8	0.80	0.38	1.11	0.30
Pine-grassland	8.3	0.5	7.1	0.6	7.0	0.6	6.8	0.6	1.65	0.21	1.18	0.29

a. Degrees of freedom: season of burn = 1, year = 1, year × season of burn = 1, error = 24.

Table 4. Abundance and species richness means and standard errors for bird migratory strategies between growing and dormant season burns at one-year (1995) and two-years (1996) post-burn on Fort Benning Military Reservation, Georgia.

	1995				1996				Burn		Year	
	Growing		Dormant		Growing		Dormant		F	P ^a	F	P
	Mean	SE	Mean	SE	Mean	SE	Mean	SE				
Abundance												
Neotropical migrant	53.8	3.4	53.6	4.5	57.9	4.4	48.6	4.9	0.59	0.45	0.12	0.74
Short-distance migrant	38.8	5.3	33.6	3.2	31.4	5.2	29.3	3.1	0.89	0.35	1.61	0.22
Resident	39.1	2.2	40.4	3.1	41.1	8.1	32.7	4.0	0.35	0.56	1.11	0.30
Species richness												
Neotropical migrant	10.4	0.5	10.3	0.8	8.8	0.6	9.3	0.4	0.08	0.78	3.16	0.09
Short-distance migrant	5.4	0.4	4.9	0.4	6.3	0.8	4.5	0.4	4.25	0.05	0.03	0.87
Resident	8.5	0.4	8.1	0.5	8.0	0.7	7.6	0.4	0.60	0.44	0.93	0.34

a. Degrees of freedom: season of burn = 1, year = 1, year × season of burn = 1, error = 24.

Discussion

Bird communities respond to changes in vegetation structure and composition created by a variety of abiotic and anthropogenic causes (MacArthur and MacArthur 1961). In most cases, differences in bird community abundance and species richness before and after prescribed fire are most obvious during efforts to restore fire-dependent pine-grassland communities after extended periods of fire suppression. Reintroduction of fire to these areas often causes abrupt changes in stand conditions (e.g., reduced understory woody cover, increase in herbaceous vegetation) that encourage bird species that utilize early successional vegetation associations while simultaneously causing declines in forest interior birds (Wilson et al. 1995, Davis et al. 2000, Allen et al. 2006). In areas that have been restored to pyric pine-grassland communities and maintained under similar fire frequencies, differences in plant communities among sites burned in the growing versus dormant season are less distinct (Sparks et al. 1998).

We observed similar vegetation and breeding bird communities between growing and dormant season burns. Similarly, Howell et al. (2008) reported no differences in apparent nest success for all bird species, except for prairie warbler, in dormant versus growing season burned areas in the first year following a burn. However, because the use of growing season fire was limited within Fort Benning, vegetation differences between growing and dormant season fires may not be apparent unless sites are subjected to growing season fires over several fire cycles.

Vegetative responses between sites that have undergone long-term growing and dormant season burns are obscure. Sparks et al. (1998) observed only minor effects after a single growing season burn on a site that was historically burned in the dormant season. Season of fire influenced the distribution and abundance of fewer than 10% of the plant species they encountered. However, season of burn differentially affected seed production of several grass species in Florida (Shepherd et al. 2011); therefore, burning

in the growing season over several fire cycles may shift understory vegetation characteristics. Also, growing season burns had less shrub resprouting than dormant-season burns in Florida and Louisiana after two fire cycles (Drewa et al. 2002). In a 20-year study examining the effects of dormant and growing season burns, Lewis and Harshbarger (1976) observed that grasses were dominant on annually and biennially burned growing season plots, while low-growing shrubs were dominant on dormant season plots. Therefore, multiple and regularly timed growing season burns may alter understory vegetation characteristics and affect bird communities in the long-term.

Although most bird species were unaffected by season of burn, we observed greater abundance of prairie warblers, red-cockaded woodpeckers, and red-headed woodpeckers in growing-season burned areas than dormant-season burned areas. Many species of insects are attracted to recently burned areas (Evans 1971, Hanula and Wade 2003), and growing-season burns may attract wood boring and bark inhabiting insects while these insects are most active, thus increasing the amount of food available to woodpeckers.

Prairie warblers and yellow-breasted chats use patchy low shrub cover to nest and feed (Schneider et al. 2010) and more effective understory hardwood control potentially achieved in growing-season burns may cause temporary reductions in prairie warbler and chat abundance when compared with less effective dormant-season burns (Waldrop et al. 1992). Wilson et al. (1995) observed an increase in prairie warbler abundance following dormant-season fire over unburned stands. Prairie warbler densities peak a few years after fire, then decline as woody shrubs grow into the midstory or the stand is burned again (Nolan 1978). Although season of burn did not affect shrub density in our study, shrub cover may have differed between burn treatments and possibly influenced prairie warbler and chat abundance. Drewa et al. (2002) observed no differences in shrub density between growing and dormant season burns, but they observed fewer shrub resprouts following a growing season burn when compared with a dormant season burn. Therefore, a dormant season burn may already have an excess of woody cover that may discourage yellow-breasted chats and prairie warblers by the second year.

Similar to Howell et al. (2008), our results indicate that season of burn may have little effect on breeding bird communities within fire-maintained pine forests in the southeastern United States. However, we only examined breeding bird communities after a single growing season burn, and multiple growing season burns may result in more profound changes in the vegetation that could affect breeding bird communities. Although our knowledge of the effects of long-term growing versus dormant season burns on vegetation and wildlife is still relatively incomplete, the use of growing

season burns as a management tool appears to pose little risk to breeding and nesting birds.

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