



Short-term changes in summer and winter resident bird communities following a high severity wildfire in a southern USA mixed pine/ hardwood forest



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ABSTRACT

High severity forest fires are increasing in large areas of the southern and western United States as the climate becomes warmer and drier. Given their strong roles in ecosystem dynamics, documenting the response of bird communities to wildfires is important for improving our understanding and management of post-wildfire ecosystems. However, because wildfires are unplanned events, relatively few studies have been conducted to assess local-scale impacts on forest bird communities. In this study, we had the opportunity to use a before–after, control–impact (BACI) approach to assess the response of resident birds to high severity wildfires that occurred in the Lost Pines ecoregion of Texas in September and October 2011. We replicated a previous study using point count surveys to assess summer and winter bird community changes ca. 1 year after the wildfires. We found that total number of detected individuals did not change following the wildfires, but winter bird species richness increased in burned habitat. Changes were apparent at the foraging guild-level for the winter bird community, with an increase in aerial insectivores and decrease in bark insectivores. Summer and winter bird community composition changes were apparent at the species-level and generally agreed with species-specific habitat preferences. For example, species such as eastern bluebirds and chipping sparrows that prefer open woodlands were positively associated with burned habitat. Our results provide quantitative evidence that high severity forest fires increased habitat suitability for many resident bird species. At the landscape-scale, fire-induced increases in habitat heterogeneity could result in higher bird diversity in the Lost Pines ecoregion. We expect bird community composition will be dynamic in the Lost Pines ecoregion over the next few decades as the burned habitat continues to change through successional processes and post-fire management actions.

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1. Introduction

The climatic pattern towards warmer and drier conditions in much of the southern and western United States, coupled with longstanding broad-scale fire suppression, have resulted in an increase in high severity forest fires (Litschert et al., 2012; Crotteau et al., 2013; Hurteau et al., 2014), particularly in

pine-dominated and mixed-pine forests (Miller et al., 2009; Miller and Safford, 2012). Previous research has shown that fire severity is a primary driver of fire influences on many ecosystem components (Knapp et al., 2009; Brown et al., 2014a), including influences on bird responses (Smucker et al., 2005; Fontaine and Kennedy, 2012; Lindenmayer et al., 2014). Thus, improving our understanding of community responses to high severity forest fires (i.e., wildfires that kill or top-kill the majority of live vegetation and consume the majority of dead organic matter) is currently of high interest to assist land managers with post-fire management decisions (Bisson et al., 2003; Beschta et al., 2004). These decisions range from allowing natural recovery and regeneration regimes

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with minimal human intervention up through to high levels of manipulation, such as salvage logging and active revegetation (Driscoll et al., 2010; Powers et al., 2013).

Understanding the response of bird communities to wildfires is important given their strong roles in ecosystem dynamics (Sekercioglu et al., 2004; Anderson et al., 2011; Cavallero et al., 2013), potential use as ecological indicators (Niemi and McDonald, 2004; Howe et al., 2007), and aesthetic, cultural, and intrinsic value (Burger et al., 1995; Bowen-Jones and Entwistle, 2002). Given the vagility of birds, fire-induced changes in abundance and occupancy are likely driven by spatial responses to changes in habitat suitability (i.e., food resource and nesting site availability and structural habitat preferences; Hutto, 1995; Saab and Powell, 2005; Fontaine and Kennedy, 2012). At the extreme end, species such as Kirtland's warbler (*Setophaga kirtlandii*), red-cockaded woodpecker (*Picoides borealis*), and black-backed woodpecker (*Picoides arcticus*) are fire specialists, relying on wildfires to generate high suitability habitat under natural conditions (Probst, 1986; Saenz et al., 2001; Hutto et al., 2008). Wildfires can also result in reduced habitat suitability, particularly for species that are primarily foliage insectivores (Saab and Powell, 2005) and species that prefer mature forests (Ager et al., 2007). Given the diversity and complexity of ecological requirements and preferences of bird species, studies that investigate local-scale responses to habitat changes are useful both for informing post-wildfire management decisions and for understanding landscape-scale species and community patterns and trends (Dickson et al., 2009; Pons and Clavero, 2010; Fontaine and Kennedy, 2012).

In this study, we had the opportunity to assess the short-term outcomes on the bird community following a high severity wildfire in the Lost Pines ecoregion of central Texas. To our knowledge, this represents the first published study to assess wildfire impacts on birds in this mixed pine/hardwood ecoregion. Further, although several studies have investigated fire impacts on grassland and shrubland bird communities in Texas (e.g., Reynolds and Krausman, 1998; Marx et al., 2008; Roberts et al., 2012), to our knowledge this is the first published study to assess wildfire impacts on bird community composition in forested regions of Texas. We hypothesized that the substantial changes in forest structure would result in local-scale species composition changes through indirect impacts on food resource and nesting site availability, and species-specific habitat preferences. Specifically, we predicted that insectivorous species that primarily forage on tree bark and foliage would decrease in the short-term, whereas other insectivorous guilds and omnivores would increase due to greater understory vegetation diversity (Brown et al., 2014a) potentially producing greater insect diversity and abundance (Swengel, 2001; Buddle et al., 2006). We also predicted that species associated with open forest habitats would increase in burned areas due to substantial tree mortality, whereas species associated with mature closed-canopy forests would decrease in burned areas.

2. Methods

2.1. Study area

This study was conducted in the 34,400-ha Lost Pines ecoregion in Bastrop County, Texas, USA (Fig. 1). The Lost Pines is a remnant patch of loblolly pine (*Pinus taeda*)-dominated forest that was likely isolated from the East Texas Piney Woods ecoregion between 10,000 and 14,000 years ago, when east-central Texas transitioned from primarily forest to primarily open savanna and grassland (Bryant, 1977). Genetic data indicate the loblolly pines of the area began to differentiate genetically up to 30,000 years ago

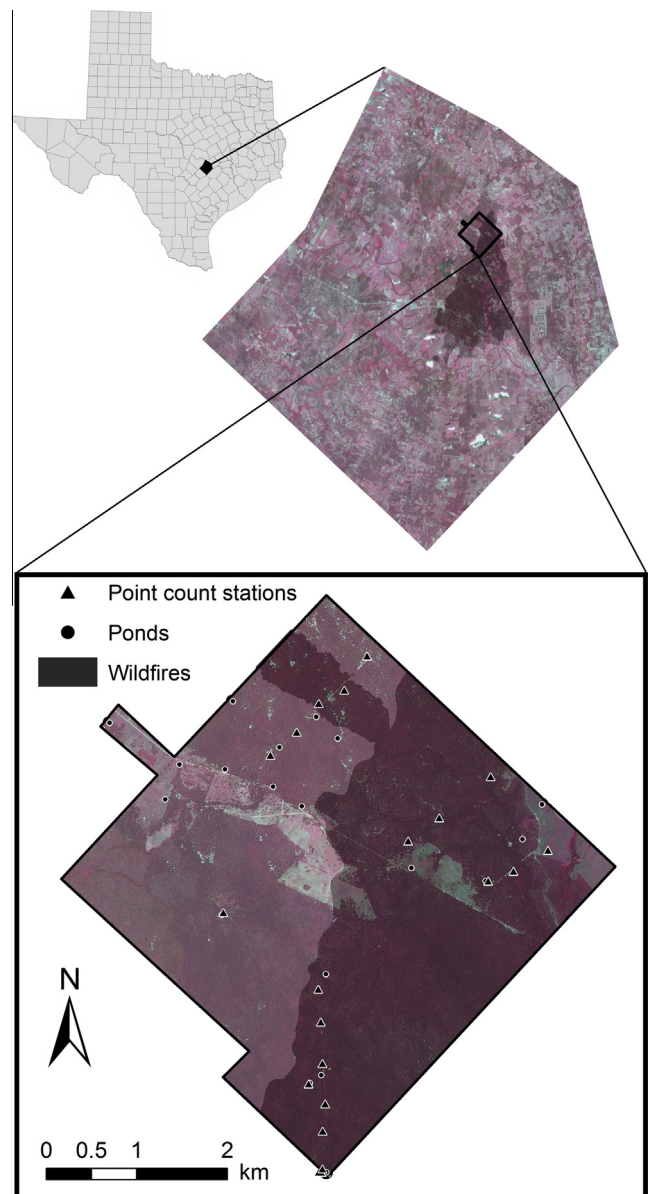


Fig. 1. Aerial image of the Griffith League Ranch (GLR), Bastrop County, Texas, USA, and its location with respect to high severity wildfires that burned 13,531 ha in the Lost Pines ecoregion in September and October 2011. Overlain on the image are the locations of the point count stations used to investigate changes in the summer and winter resident bird communities following the wildfires.

(Al-Rabah'ah and Williams, 2004). The Lost Pines was extensively logged in the 1800s and early 1900s (Moore, 1977). Since the early to mid-1900s, landscape-scale fire suppression has been implemented throughout the ecoregion, resulting in the accumulation of heavy fuel loads.

The study area for this project was the 1948-ha Griffith League Ranch (GLR). The GLR was primarily a forested ranch, with a pre-fire overstory dominated by loblolly pine, post oak (*Quercus stellata*), and eastern red cedar (*Juniperus virginiana*), and a pre-fire understory dominated by yaupon holly (*Ilex vomitoria*), American beautyberry (*Callicarpa americana*), and farkleberry (*Vaccinium arboreum*). The GLR contains 3 permanent ponds (i.e., ponds have not dried in at least 14 years), 10 semi-permanent ponds (i.e., ponds typically dry several times per decade), and 10 or more ephemeral pools that hold water for days to months annually depending on rainfall.

On 4 September 2011, a high severity wildfire began from multiple initial fire outbreaks across the Lost Pines. The fire was unstoppable due to wind gusts in excess of 58 kph resulting from the passage of tropical storm Lee, coupled with extreme drought conditions in central Texas (Lost Pines Recovery Team, 2011). After 18 days the fire was 95% contained, with the total burn area encompassing 13,406 ha. A fire break was installed on the GLR during the burn, restricting the fire on the property to 987 ha. On 4 October 2011 the wildfire breached the fire break on the GLR, burning an additional 125 ha.

Brown et al. (2014a) quantified short-term terrestrial and aquatic habitat changes on the GLR following the wildfire. For this study, we provide summary data of forest structure changes based on 15 randomly placed 50-m \times 20-m vegetation plots that were located within the wildfire area. We surveyed these plots pre-wildfire in summer and fall of 2010, and post-wildfire in summer and fall of 2012, following National Park Service (2003) fire monitoring guidelines. We estimated overstory (DBH \geq 15 cm) tree mortality by calculating the percentage of trees in each plot that were alive in 2010 and dead in 2012. We estimated pole-sized (DBH \geq 2.5 cm and $<$ 15 cm) tree mortality by calculating the percentage of trees in one quarter of each plot that were alive in 2010 and dead in 2012. We estimated percent canopy cover in each plot using a spherical densiometer, with the mean of 5 estimates (i.e., the 4 plot corners and the plot center) used for this study. We estimated understory vegetation species richness and percent vegetation cover using 1 50-m transect per plot, with presence/absence of vegetation and species identification (when applicable) recorded every 0.3 m (i.e., 166 points per transect). We estimated shrub abundance by counting all live shrubs observed in 1 50-m \times 1-m belt within each plot.

2.2. Bird data collection

We designed our study to replicate a previous bird community study on the GLR (White, 2003), which allowed us to assess short-term responses to the wildfires using a before-after, control-impact (BACI) framework. The original study used 25 point count stations, with all birds seen or heard within a 100-m radius of the stations counted over a 10 min survey period. The point count stations were non-randomly placed near forest trails, but were spaced $>$ 250 m apart to minimize the potential for individual birds to be counted at multiple stations. In the original study each station was surveyed 3 times in spring (mid-March to mid-June), summer (mid-June to late-September), fall (late-September to mid-December), and winter (mid-December to mid-March) in 2002 and 2003 (White, 2003).

We conducted post-wildfire sampling between 27 July 2012 and 24 February 2013, ca. 1 year after the high severity wildfires. We used 19 of the point count stations for this study, with the remaining 6 stations removed prior to analyses because they were located in non-forested habitat. Fifteen of the stations were located in burned forest, and 4 were located in non-burned forest (Fig. 1). We limited our surveys to the summer and winter periods for this study because we were interested in changes to the seasonally-resident bird communities. All surveys were completed within 5 h of sunrise in both studies, and for each season we attempted to survey each station once during early-, mid-, and late-morning to mitigate possible biases associated with survey time. Prior to analyses we removed 4 species from the data set that are known only to occur as migrants in the region (Poole, 2005), as well as flyover observations. In addition, we detected 15 meadowlarks (*Sturnella* sp.) during a post-burn sampling event that we were unable to identify to the species-level; these data were only included in an analysis that assessed changes in total number of detected individuals.

2.3. Bird data analyses

Preliminary analyses indicated species turnover between the summer and winter sampling periods was high for the post-fire data set (i.e., 64% of total detected species were present in only one sampling period). Thus, we analyzed the summer and winter bird communities separately. For each season, we assessed post-fire bird community changes at 3 levels of data resolution. First, we investigated changes in total number of detected individuals and species richness. Second, we investigated changes in nesting and foraging guilds, based on published results showing fire often results in guild-level changes (Saab and Powell, 2005). Finally, we assessed species-level changes in relative abundance.

To assess changes in total number of detected individuals and species richness, we calculated the total number of individuals and species for each sample at each point count station. To analyze these data sets we used linear mixed-effects models, specifying burn status (i.e., pre-fire or post-fire) and treatment (i.e., control or fire) as fixed effects, and point count station replicates as random effects (Pinheiro and Bates, 2000). For each analysis, we assessed assumptions of normality and homoscedasticity using graphical diagnostic plots (Zuur et al., 2009), which indicated no data transformations were necessary. We performed these analyses using the nlme package (version 3.1–113) in the program R (version 3.0.2).

To assess guild-level and species-level changes, we used a two-stage analysis approach. In the first stage, we used multivariate analyses to test for an overall treatment \times status interaction effect (typically referred to as a community-level effect), and inferred responses of individual guilds and species using ordination diagrams (i.e., species-environment biplots). This is a useful data exploration and reduction tool when the number of response or predictor variables is large (McCune and Grace, 2002). In the second stage, we used univariate analyses to investigate differences for individual response variables that appeared to be strongly positively or negatively associated with the treatment \times status interaction effect based on the ordination diagrams. The univariate analyses were identical to those used for assessing changes in total number of detected individuals and species richness, with the exception that we square-root transformed several response variables to improve data normality (Fowler et al., 1998).

To assess relative abundance changes at the foraging and nesting guild-levels, we specified guilds for each species based on Saab and Powell (2005) and Poole (2005). We included 7 foraging guilds (i.e., aerial insectivore, bark insectivore, carnivore, foliage insectivore, ground insectivore, herbivore, and omnivore) and 3 nesting guilds (i.e., cavity, branch, and ground). For the nesting guild analysis, we analyzed only the summer bird community data set.

For the multivariate analyses, we used redundancy analysis (RDA), which is an extension of principal components analysis (PCA) to include explanatory variables. Specifically, for RDA each response variable is regressed on each explanatory variable, and then a PCA is performed on the matrix of fitted values (McCune and Grace, 2002). We chose RDA, which assumes response variables are related linearly to predictors, over canonical correspondence analysis (CCA), which assumes response variables are related unimodally to predictors, because our gradient lengths were short (\leq 4) and our predictors were categorical (Lepš and Šmilauer, 2003). We included point count stations as covariates in the analyses. By including this covariate, we subtracted the average values and assessed only value changes within each point count station (Lepš and Šmilauer, 2003). We $\log_{10} + 1$ transformed the response data so that percentage rather than absolute changes were assessed (note this transformation applies only to the RDA analyses). We tested for a burn status \times treatment interaction effect using a Monte Carlo permutation test ($n = 5000$ iterations),

with data randomized within, but not among, point count stations. We note that zero-heavy matrices, such as our species-level data sets, can be problematic for multivariate analyses because joint zeros are perceived as a positive relationship (McCune and Grace, 2002). However, since we were interested only in a burn status \times treatment interaction, this was not an issue for this study. We performed these analyses using the program CANOCO (version 4.5). Due to the small sample size and thus low statistical power, we considered differences to be significant at $\alpha = 0.1$.

3. Results

The wildfires resulted in a mean overstory tree mortality of 87.0% (SD = 28.1%), with a corresponding 39.2% (SD = 12.9%) decrease in overstory canopy cover ca. 1 year after the wildfires. The mean pole-sized tree mortality was 97.0% (SD = 10.6%). The wildfires initially resulted in a ‘moonscaped’ ground layer (Fig. 2b). Prior to the post-wildfire bird surveys, there was substantial understory vegetation recolonization, with pokeweed (*Phytolacca americana*), panic grasses (*Dicanthelium* spp.), and sedges (*Cyperus* spp.) dominating the burned portions of the study area at the time of this study (Fig. 2c). Mean understory vegetation species richness was 3.7 (SD = 2.3) pre-burn and 10.3 (SD = 5.0) post-burn. Mean vegetation cover was 22.8% (SD = 11.4%) pre-burn and 49.9% (SD = 23.5%) post-burn. Mean shrub abundance was 68.1 (SD = 45.9) pre-burn and 33.6 (SD = 27.8) post-burn.

We detected a total of 1232 individuals (617 in summer and 615 in winter) during the pre-burn surveys, and 1274 individuals (518 in summer and 756 in winter) during the post-burn surveys (Appendix A). We detected 29 and 35 resident species in the summer and winter pre-burn data sets, respectively. We detected 34 and 39 resident species in the summer and winter post-burn data sets, respectively. We found no significant burn status \times treatment effect for total number of detected individuals in summer ($F_{1,17} = 0.94$, $P = 0.347$) or winter ($F_{1,17} = 2.44$, $P = 0.136$). We found no significant burn status \times treatment effect for species richness in summer ($F_{1,17} = 0.15$, $P = 0.707$), but there was a significant effect in winter ($F_{1,17} = 6.77$, $P = 0.019$). Mean species richness among point count stations in winter was similar between control (5.67 species) and burned (5.64 species) sites prior to the wildfire. Mean species richness among point count stations in winter decreased at control sites (3.83 species) and increased at burned sites (6.29 species) after the wildfire.

The RDA analyses indicated a significant burn status \times treatment effect for winter foraging guilds ($P = 0.001$), summer species composition ($P = 0.033$), and winter species composition ($P = 0.011$). There was no significant burn status \times treatment effect for summer foraging guilds ($P = 0.141$) or summer nesting guilds ($P = 0.193$). For the winter foraging guild analysis, the species-environment biplot and additional univariate analyses indicated omnivores ($F_{1,17} = 8.34$, $P = 0.010$) and aerial insectivores ($F_{1,17} = 3.01$, $P = 0.101$) were positively associated with burned habitat, while bark insectivores ($F_{1,17} = 6.74$, $P = 0.019$) were negatively associated with burned habitat (Fig. 3). Ground insectivores also appeared to be positively associated with burned habitat, but the burn status \times treatment effect was not significant ($F_{1,17} = 0.65$, $P = 0.432$).

For the summer species community, the species-environment biplot and additional univariate analyses indicated the species most positively associated with burned habitat was the northern cardinal (*Cardinalis cardinalis*; $F_{1,17} = 4.83$, $P = 0.042$), and the species most negatively associated with burned habitat was the white-eyed vireo (*Vireo griseus*; $F_{1,17} = 13.90$, $P = 0.002$). The pine warbler (*Setophaga pinus*) also appeared to be negatively

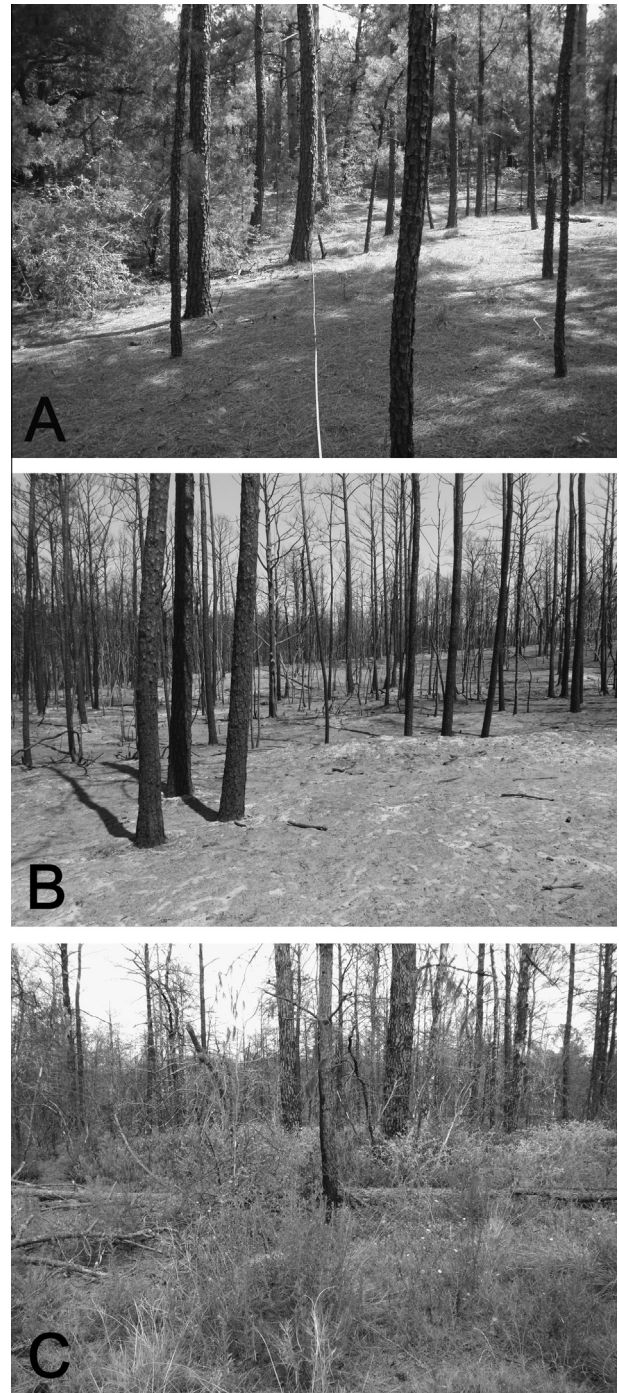


Fig. 2. Representative forest habitat on the Griffith League Ranch (GLR), Bastrop County, Texas, USA, prior to (A), immediately after (B), and ca. 1 year after (C) high severity wildfires that occurred in September and October 2011.

associated with burned habitat in the summer data set, but the burn status \times treatment effect was not significant ($F_{1,17} = 1.21$, $P = 0.286$). For the winter species community, the species-environment biplot and additional univariate analyses indicated the species most positively associated with burned habitat were the eastern bluebird (*Sialia sialis*; $F_{1,17} = 6.76$, $P = 0.019$), chipping sparrow (*Spizella passerina*; $F_{1,17} = 4.94$, $P = 0.040$), and Lincoln's sparrow (*Melospiza lincolnii*; $F_{1,17} = 3.38$, $P = 0.083$), and the species most negatively associated with burned habitat was the pine warbler ($F_{1,17} = 9.45$, $P = 0.007$).

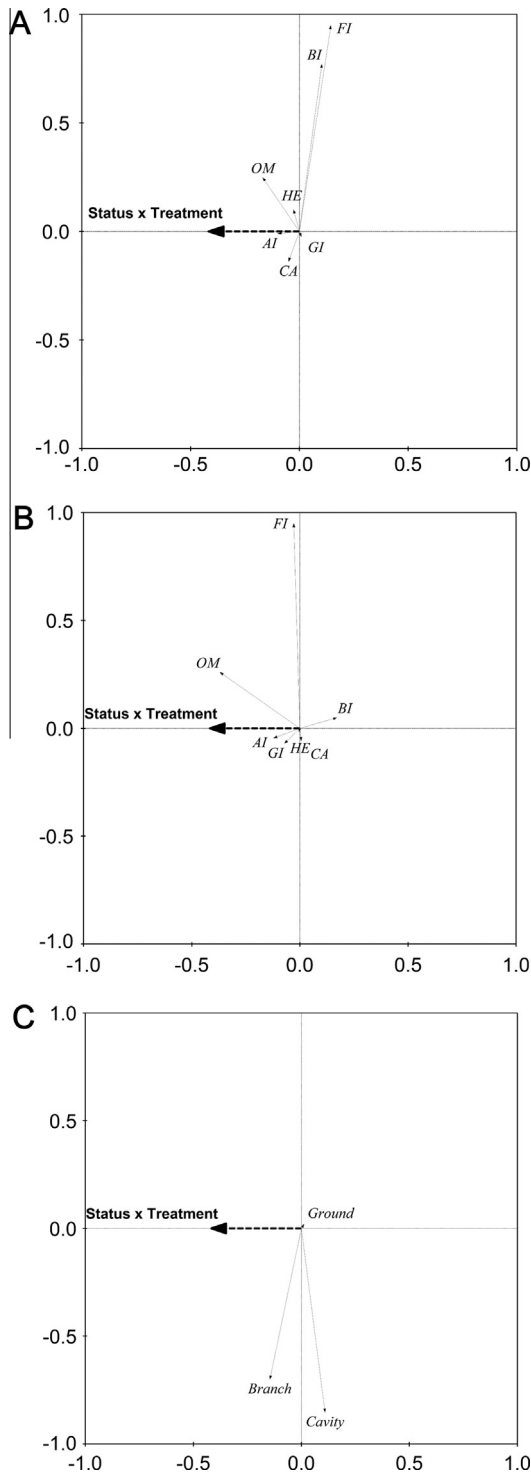


Fig. 3. Results from Redundancy Analyses (RDA) used to investigate guild-level summer and winter resident bird community changes on the Griffith League Ranch (GLR), Bastrop County, Texas, USA, following high severity wildfires that occurred in September and October 2011. For these analyses species were grouped into foraging guilds in summer (A) and winter (B), and nesting guilds in summer (C). Guilds pointing towards the Status \times Treatment interaction arrow were positively associated with burned habitat, guilds pointing away from the interaction arrow were negatively associated with burned habitat, and guilds pointing at approximate right angles to the interaction arrow had little association with burned habitat. A significant interaction effect was only found in the winter foraging guild analysis. Foraging guild acronyms include aerial insectivore (AI), bark insectivore (BI), carnivore (CA), foliage insectivore (FI), ground insectivore (GI), herbivore (HE), and omnivore (OM).

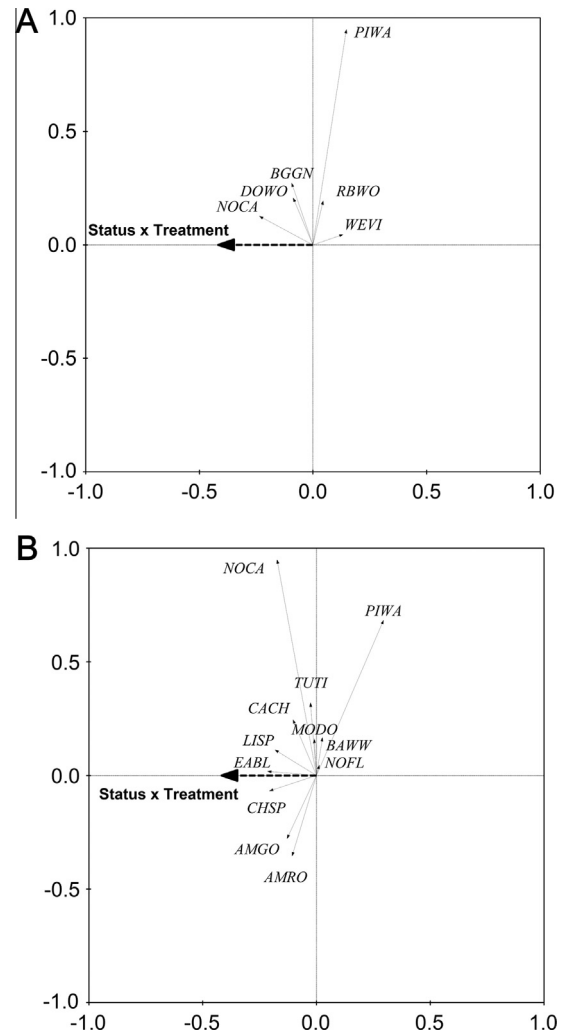


Fig. 4. Results from Redundancy Analyses (RDA) used to investigate species-level summer (A) and winter (B) resident bird community changes on the Griffith League Ranch (GLR), Bastrop County, Texas, USA, following high severity wildfires that occurred in September and October 2011. Species pointing towards the Status \times Treatment interaction arrow were positively associated with burned habitat, species pointing away from the interaction arrow were negatively associated with burned habitat, and species pointing at approximate right angles to the interaction arrow had little association with burned habitat. A significant interaction effect was found for both the summer and winter bird communities. Only species with at least 5% of variation explained by the interaction effect were included in the figure for ease of interpretation. Species acronyms are provided here; full species names can be found in [Appendix A](#).

4. Discussion

In this study, we found that high severity wildfires resulted in summer and winter resident bird community composition shifts in a southern USA mixed pine/hardwood forest. In concordance with our prediction, aerial insectivores, ground insectivores, and omnivores were positively associated with burned habitat, while bark insectivores were negatively associated with burned habitat. However, foliage insectivores did not show a strong response to the wildfire, despite previous research that indicated this could be the most negatively impacted foraging guild (Saab and Powell, 2005). In addition, a strong foraging guild response was only apparent for the winter bird community, although the directions of foraging guild responses were similar in winter and summer (see Fig. 3). We note that although our choice to use a BACI experimental design in this study was optimal for causative inference,

the sample size was inherently small, and thus we might have failed to detect real changes in the bird community that were not as large as the changes we detected.

We did not detect a strong shift in nesting guild structure. Fire can benefit cavity nesters by creating snags that support new nesting cavities, but fire can also damage and destroy pre-existing nesting cavities (Conner, 1979; Conner et al., 1991; Wiebe, 2014). In addition, species that are both cavity nesters and bark insectivores (e.g., woodpeckers) could be positively impacted by increased nesting site availability, but negatively impacted in the short-term by reduced food resource availability. Wood-boring beetles often increase dramatically within 1–2 years following forest fires (Kaynas and Gürkan, 2005; Boulanger and Sirois, 2007), and thus abundance of cavity nesters and bark insectivores could increase substantially in the next few years. Bagne and Purcell (2011) found that bark insectivores began to increase three to six years after prescribed burning in California. Similarly, Saab et al. (2007) determined that nest densities for cavity nesters increased with time-since-wildfire in Idaho.

Of the species that exhibited the strongest positive and negative responses to the post-wildfire habitat, all were omnivores except the white-eyed vireo, which is primarily a foliage insectivore. Species-specific results indicated that forest structural preferences had a major influence on community changes, a result consistent with our hypothesis and other studies that examined bird responses to fire (e.g., Hutto, 1995; Seavy and Alexander, 2014). For example, pine warblers, which prefer mature pine-dominated forests with a sparse understory (Conner et al., 1983; Annand and Thompson, 1997), were negatively associated with burned habitat. Alternately, eastern bluebirds and chipping sparrows, species that prefer open woodlands (Gowaty and Plissner, 1998; Sallabanks et al., 2006; Atwell et al., 2008), were positively associated with burned habitat.

The local-scale study we conducted indicated that many species responded positively to the forest habitat created by the high severity wildfires. Thus, at the landscape-scale, the increase in habitat heterogeneity caused by the fires could increase overall bird community diversity (Roberts et al., 2012; Barton et al., 2014; Sitters et al., 2014). We expect bird community composition will be dynamic in the Lost Pines ecoregion over the next few decades as the burned habitat continues to change through natural successional processes and post-fire management actions (Barber et al., 2001; Perry and Thill, 2013). Of particular interest are the

broad-scale post-fire clearcutting operations and loblolly pine planting initiatives that were initiated after the wildfires (Brown et al., 2014b). Clearcutting began on the GLR shortly after this study was completed, with over 500-ha on the property clearcut as of December 2014, a portion of which has been replanted with loblolly pine. The study presented here will serve as a valuable baseline for additional studies that investigate management action impacts and long-term bird community changes in the Lost Pines.

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Appendix A

Bird species detected and point count survey results for this before-after, control-impact (BACI) study assessing the response of summer and winter resident birds to a high severity wildfire in the Lost Pines ecoregion of Texas. Control and wildfire areas included 4 and 15 point count stations, respectively, with each station surveyed 3 times in summer and winter of 2002 to 2003 and 2012 to 2013. Because sample sizes differed for control and wildfire point count stations, we present the mean number of individuals observed (i.e., total observations/point count stations). Nesting and foraging guild classifications were derived from Saab and Powell (2005) and Poole (2005). The nesting habitats of 3 species did not fit easily within our classification system (shown as blanks below), and thus these species were excluded from the nesting guild analysis. In addition, 15 meadowlark (*Sturnella* sp.) counts were included only in an investigation of changes in total number of detected individuals, and 4 detected migratory species (Eastern Whip-poor-will [*Antrostomus vociferus*], Nashville Warbler [*Oreothlypis ruficapilla*], Olive-sided Flycatcher [*Contopus cooperi*], and Willet [*Tringa Semipalmata*]) were excluded from the study. Acronyms are given for species included in Fig. 4.

Species	Foraging Guild	Nesting Guild	Residence Period	Summer				Winter			
				Control		Wildfire		Control		Wildfire	
				Pre	Post	Pre	Post	Pre	Post	Pre	Post
American crow (<i>Corvus brachyrhynchos</i>)	Omnivore	Cavity	Annual	3.5	1.3	2.0	1.6	0.3	0.5	1.4	2.1
American goldfinch (AMGO; <i>Spinus tristis</i>)	Omnivore	Branch	Winter	0	0	0	0	4.5	0	2.2	0.2
American kestrel (<i>Falco sparverius</i>)	Carnivore	Cavity	Winter	0	0	0	0	0	0.3	0	0
American robin (AMRO; <i>Turdus migratorius</i>)	Ground insectivore	Cavity	Annual	0	0	0	0	4.3	0	2.5	0
Barn swallow (<i>Hirundo rustica</i>)	Aerial insectivore		Summer	0	0	0	0.1	0	0	0	0
Bewick's wren (<i>Thryomanes bewickii</i>)	Foliage insectivore	Cavity	Annual	0	0	0	0	0	0	0.1	0
Black-and-white warbler (BAWW; <i>Mniotilta varia</i>)	Bark insectivore	Ground	Summer	0.3	0.3	0.2	0.1	0.5	0.3	0.7	0
Black vulture (<i>Coragyps atratus</i>)	Carnivore	Ground	Annual	0	0	0	0.1	0	0.3	0.1	0.4
Blue jay (<i>Cyanocitta cristata</i>)	Omnivore	Cavity	Annual	0.8	1.3	0.9	1.7	0	0	0.5	0
Blue-gray gnatcatcher (BGGN; <i>Poliptila</i>)	Foliage	Cavity	Summer	2.3	0	1.1	0.3	0	0	0	0

Appendix A (continued)

Species	Foraging Guild	Nesting Guild	Residence Period	Summer				Winter			
				Control		Wildfire		Control		Wildfire	
				Pre	Post	Pre	Post	Pre	Post	Pre	Post
<i>caerulea</i>)	insectivore										
Brown-headed cowbird (<i>Molothrus ater</i>)	Omnivore		Annual	0.3	0	0.2	0.1	0.3	0	0.1	0
Carolina chickadee (CACH; <i>Poecile carolinensis</i>)	Foliage	Cavity	Annual	2.5	4.0	2.0	1.9	2.3	1.5	1.5	2.1
Carolina wren (<i>Thryothorus ludovicianus</i>)	Ground	Cavity	Annual	2.8	0.5	3.7	1.4	0.8	0.5	1.9	1.2
Cattle egret (<i>Bubulcus ibis</i>)	insectivore										
	Ground	Cavity	Summer	0	0.3	0	0	0	0	0	0
	insectivore										
Chipping sparrow (CHSP; <i>Spizella passerina</i>)	Omnivore	Branch	Winter	0	0	0	0	4.3	0	0	1.7
Common ground-dove (<i>Columbina passerina</i>)	Herbivore		Annual	0	0	0.2	0	0	0	0.1	0
Dark-eyed junco (<i>Junco hyemalis</i>)	Omnivore	Ground	Winter	0	0	0	0	0	0	0	1.5
Downy woodpecker (DOWO; <i>Picoides pubescens</i>)	Bark	Cavity	Annual	1.8	1.8	0.8	1.9	1.5	0.8	0.9	1.3
	insectivore										
Eastern bluebird (EABL; <i>Sialia sialis</i>)	Aerial	Cavity	Annual	0	0	0	0.9	0	0.3	0	3.5
	insectivore										
Eastern phoebe (<i>Sayornis phoebe</i>)	Aerial	Cavity	Annual	0.3	0	0	0	0	0.8	0.2	0.9
	insectivore										
Eastern towhee (<i>Pipilo erythrophthalmus</i>)	Omnivore	Ground	Winter	0	0	0	0	0	0	0	0.1
Eastern wood-pewee (<i>Contopus virens</i>)	Aerial	Cavity	Summer	0	0	0	0.1	0	0	0	0
	insectivore										
Field sparrow (<i>Spizella pusilla</i>)	Omnivore	Ground	Winter	0	0	0	0	0	0	0	0.1
Gadwall (<i>Anas strepera</i>)	Omnivore	Ground	Winter	0	0	0	0	0	0	0.1	0
Golden-crowned kinglet (<i>Regulus satrapa</i>)	Foliage	Cavity	Winter	0	0	0	0	0	0	0.1	0
	insectivore										
Golden-fronted woodpecker (<i>Melanerpes aurifrons</i>)	Omnivore	Cavity	Annual	0	1.0	0.1	0.5	0	0	0	0
Grasshopper sparrow (<i>Ammodramus savannarum</i>)	Omnivore	Ground	Annual	0	0	0	0	0	0	0	0.1
Great blue heron (<i>Ardea herodias</i>)	Carnivore	Cavity	Annual	0	0	0.1	0.1	0	0	0	0
Great crested flycatcher (<i>Myiarchus crinitus</i>)	Aerial	Cavity	Summer	0	0	0.1	0	0	0	0	0
	insectivore										
Green heron (<i>Butorides virescens</i>)	Carnivore	Cavity	Summer	0.5	0	0.1	0	0	0	0	0
Hairy woodpecker (<i>Picoides villosus</i>)	Bark	Cavity	Annual	0	0	0.1	0	0	0	0	0
	insectivore										
Hermit thrush (<i>Catharus guttatus</i>)	Ground	Branch	Winter	0	0	0	0	0	0	0.1	0
	insectivore										
Hooded merganser (<i>Lophodytes cucullatus</i>)	Carnivore	Cavity	Winter	0	0	0	0	0	0	0	0.5
House finch (<i>Haemorhous mexicanus</i>)	Omnivore	Cavity	Annual	0	0	0	0	0	0	0	0.1
Inca dove (<i>Columbina inca</i>)	Herbivore	Cavity	Summer	0	0	0	0.4	0	0	0	0
Lesser goldfinch (<i>Spinus psaltria</i>)	Herbivore	Cavity	Summer	0	0	0	0.1	0	0	0	0.2
Lincoln's sparrow (LISP; <i>Melospiza lincolnii</i>)	Omnivore	Ground	Winter	0	0	0	0	0	0	0	2.8
Mallard (<i>Anas platyrhynchos</i>)	Omnivore	Ground	Winter	0	0	0	0	0	0	0.3	0.3
Mourning dove (MODO; <i>Zenaidura macroura</i>)	Omnivore	Branch	Annual	0.8	2.0	1.4	1.1	0.3	0	0.7	0.6
Northern cardinal (NOCA; <i>Cardinalis cardinalis</i>)	Omnivore	Branch	Annual	10.3	2.8	7.7	5.1	5.8	2.3	5.8	4.5
Northern flicker (NOFL; <i>Colaptes auratus</i>)	Omnivore	Cavity	Winter	0	0	0	0	0	0	0.1	0
Northern mockingbird (<i>Mimus polyglottos</i>)	Ground	Branch	Annual	0	0	0	0.2	0	0	0	0.1
	insectivore										
Northern parula (<i>Setophaga americana</i>)	Foliage	Cavity	Summer	0	0	0.2	0.3	0	0	0.1	0
	insectivore										
Orange-crowned warbler (<i>Oreothlypis celata</i>)	Foliage	Ground	Winter	0	0	0	0	0	0	0	0.1
	insectivore										
Painted bunting (<i>Passerina ciris</i>)	Omnivore	Cavity	Summer	1.5	0	0.5	0.3	0	0	0	0
Pileated woodpecker (<i>Dryocopus pileatus</i>)	Omnivore	Cavity	Annual	1.5	0.8	0.8	0.4	0.5	0.5	0.7	0.7
Pine warbler (PIWA; <i>Setophaga pinus</i>)	Bark	Cavity	Annual	1.8	3.5	4.1	3.8	3.0	13.3	4.4	7.7
	insectivore										
Purple gallinule (<i>Porphyrio martinicus</i>)	Omnivore	Ground	Summer	0	0	0	0	0	0	0.1	0
Purple martin (<i>Progne subis</i>)	Aerial	Cavity	Summer	0	0	0	0	0	0	0.1	0

(continued on next page)

Appendix A (continued)

Species	Foraging Guild	Nesting Guild	Residence Period	Summer				Winter			
				Control		Wildfire		Control		Wildfire	
				Pre	Post	Pre	Post	Pre	Post	Pre	Post
Red-bellied woodpecker (RBWO; <i>Melanerpes carolinus</i>)	insectivore Bark	Cavity	Annual	0.5	0.8	1.7	1.7	0.8	1.8	0.9	1.3
Red-eyed vireo (<i>Vireo olivaceus</i>)	insectivore Foliage	Cavity	Summer	0	0	0.3	0.1	0	0	0.1	0
Red-headed woodpecker (<i>Melanerpes erythrocephalus</i>)	insectivore Aerial	Cavity	Winter	0	0	0	0.1	0	0	0	0.1
Red-shouldered hawk (<i>Buteo lineatus</i>)	insectivore Carnivore	Cavity	Annual	0	0	0.4	0.3	0	0	0.2	0.1
Red-tailed hawk (<i>Buteo jamaicensis</i>)	Carnivore	Cavity	Annual	0	0	0	0	0	0.3	0	0.3
Ruby-crowned kinglet (<i>Regulus calendula</i>)	Foliage insectivore	Cavity	Winter	0	0	0	0	1.8	0	0.9	0.3
Song sparrow (<i>Melospiza melodia</i>)	Omnivore	Branch	Winter	0	0	0	0	0	0	0	0.9
Summer tanager (<i>Piranga rubra</i>)	Foliage insectivore	Cavity	Summer	1.5	1.0	1.8	1.4	0	0	0	0
Tufted titmouse (TUTI; <i>Baeolophus bicolor</i>)	Foliage insectivore	Cavity	Annual	1.0	0.8	1.3	1.1	3.5	0	3.8	0.3
White-eyed vireo (WEVI; <i>Vireo griseus</i>)	Foliage insectivore	Branch	Annual	0	1.8	0.1	0.1	0	0	0.5	0.1
White-tailed kite (<i>Elanus leucurus</i>)	Carnivore	Cavity	Annual	0	0	0	0	0	0.3	0	0
White-winged dove (<i>Zenaidura macroura</i>)	Omnivore	Branch	Annual	0	0.8	0	0.9	0	0	0	0.5
Wild turkey (<i>Meleagris gallopavo</i>)	Omnivore	Ground	Annual	0	0	0	0	0.5	0	0.1	0
Winter wren (<i>Troglodytes hiemalis</i>)	Ground insectivore	Cavity	Winter	0	0	0	0	0	0	0.1	0.1
Yellow-bellied sapsucker (<i>Sphyrapicus varius</i>)	Omnivore	Cavity	Winter	0	0	0	0	0	0	0.1	0.1
Yellow-billed cuckoo (<i>Coccyzus americanus</i>)	Foliage insectivore	Branch	Summer	0.5	0	0.4	0.1	0	0	0	0
Yellow-rumped warbler (<i>Setophaga coronata</i>)	Foliage insectivore	Cavity	Winter	0	0	0.1	0	1.0	6.3	0.1	4.8
Yellow-throated warbler (<i>Setophaga dominica</i>)	Bark insectivore	Cavity	Summer	0	0	0	0.1	0	0	0	0

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